An assessment of the potential biodiversity impacts from biofuel production in South Africa

by

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Declaration

By submitting this dissertation electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the owner of the copyright thereof (unless to the extent explicitly otherwise stated) and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

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R. Blanchard

Date: ....

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Summary

Biofuels are being promoted as a global necessity to meet climate change targets through the replacement of fossil fuels. Many countries have identified biofuels as a potential mechanism to meet these challenges, with policy directives driving biofuel production. The South African government has proposed that biofuels form part of the country's future renewable energy and has proposed a draft biofuel strategy. This study aims to investigate appropriate approaches to determine potential biodiversity impacts from biofuel production.

Since biofuels are not currently grown to any large extent in South Africa, impact was modelled using future scenarios of converting available land within the Eastern Cape Province of South Africa. Suitable species were identified using the species distribution modelling programme MaxEnt. Some of the proposed biofuel crops were considered as invasive (i.e. they spread from sites where they are cultivated) or are very likely to be invasive in South Africa. This study also highlighted the considerable overlap between suitable growing areas and areas considered important for future biodiversity conservation.

The biodiversity intactness index (BII), a broad based biodiversity indicator, was used to assess the biodiversity implications of transforming available land to biofuels. The BII indicates losses of biodiversity between 17.6% and 42.1% for the land use scenarios identified. An important finding was that excluding important biodiversity areas that occur outside of protected areas can reduce biodiversity losses by as much as 13% and maintain an overall intactness of ~70%. Currently the BII does not account for fragmentation or landscape configuration. This was addressed by developing a revised biodiversity intactness index (R-BII) which included the effect of patch-size and habitat fragmentation on biodiversity intactness. This study found that although the original BII reported on the biodiversity trends of large-scale shifts in land-use across multiple scales it could not detect changes in landscape configuration which was reflected by the R-BII.

Land-use change can impact on ecosystem processes that underpin the provisioning of ecosystem services by changing the combinations of species and the plant functional

traits within communities. The impacts of cultivating potential biofuel species (*Acacia mearnsii*, *Sorghum halepense* and *Eucalyptus* species) were investigated using a plant functional traits approach. These species were shown to affect the leaf nitrogen content, leaf phosphorous content and leaf dry matter content associated with important ecosystem functions within an ecosystem service hotspot in the Eastern Cape. A decline in functional diversity was reported for all transformed land-uses by as much as ~40%. These shifts may be used to identify potential changes to ecosystem services associated with natural vegetation.

The methods used in this thesis highlight the overall relevance of this work and its importance to minimising biodiversity resulting from biofuel production. Some of the key findings address resolving spatial conflict, using biodiversity indicators, assessing impacts of potential invasive species and planning for ecosystem services. New drivers of change to land-use, such as biofuel production, are a major challenge to conservation biologists and planners and the insights derived in from this study can be successfully applied to guide biofuel production.

Opsomming

Biobrandstof word internasionaal beskou as 'n noodsaaklike komponent in die bereiking van klimaatsverandering doelwitte deur fossielbrandstowwe daarmee te vervang. Daarom word biobrandstof deur verskeie lande geïmplementeer as 'n potensiële meganisme om aan hierdie uitdaging te voldoen. Die Suid-Afrikaanse regering het voorgestel dat biobrandstof deel vorm van die land se hernubare energie toekoms en het daarom 'n konsep biobrandstofstrategie voorgestel. Die aanvaarding van so 'n strategie sal waarskynlik ‘n aantal verreikende gevolge inhou. Hierdie studie gebruik verskeie benaderings ten einde die impak van biobrandstof produksie op biodiversiteit te bepaal.

Aangesien biobrandstof nie tans ‘n beduidende bydra maak tot tradisionele brandstofproduksie in Suid-Afrika nie, word die impak daarvan geskoei op die omskakeling van beskikbare grond. Die Oos-Kaap provinsie van Suid-Afrika speel a sleutelrol in hierdie opsig en vorm daarom die fokus van hierdie analise. Geskikte spesies is geïdentifiseer deur die sagtewareprogram, MaxEnt, waardeur spesiesverspreiding gemodelleer word.

Hierdie studie beklemtoon die aansienlike oorvleueling wat daar bestaan tussen geskikte aanplantingsgebiede en belangrike biodiversiteitsareas wat nie tans formeel bewaar word nie. Sommige van die voorgestelde biobrandstofgewasse is tans indringers, of het die potensiaal om indringerplante te word en daarom is daar toenemende kommer oor die kweek van biobrandstof gewasse in Suid-Afrika. Die “Biodiversity Intactness Index” (BII), 'n algemene biodiversiteitsaanwyser, is gebruik om die implikasies van grondomskakeling na biobrandstof op biodiversiteit te evalueer. Die BII dui op verliese van tussen 17,6% en 42,1% vir die grondgebruikscenario's wat geïdentifiseer is. 'n Belangrike bevinding was dat die uitsluiting van belangrike biodiversiteitsareas buite beskermde gebiede die verlies van biodiversiteit met soveel as 13% kan verminder en biodiversiteit eenheid van ~ 70% kan behou. Die BII maak egter nie tans voorsiening vir landskap fragmentasie nie. ‘n “Revised-Biodiversity Intactness Index” (R-BII) is ontwikkel wat die effek van kol-grootte en habitat op biodiversiteit eenheid insluit. Hierdie studie het bevind dat alhoewel die oorspronklike

BII grootskaalse verandering in die grondgebruik op verskeie skale aandui, dit egter nie verandering in landskapsamestelling kon opspoor soos die R-BII nie.

Ten slotte, die impak van die aanplanting van potensiële biobrandstofspesies (*Acacia mearnsii, Sorghum halepense* en *Eucalyptus spesies*) op biodiversiteit is ondersoek deur ‘n plant funksionele eienskappe benadering te gebruik. Daar is bevind dat hierdie spesies die stikstof, fosfor en droë materiaal inhoud van blare verander wat geassosieer word met belangrike ekosisteem funksies binne 'n biodiversiteit brandpunt in die Oos-Kaap. ‘n Vermindering van funksionele diversiteit van soveel as ~ 40% is binne alle omgeskakelde grondgebruike gevind. Hierdie skuiwe kan gebruik word om potensiële veranderinge van ekosisteemdienste te identifiseer en benadruk ook die potensiële impak van uitheemse spesies.

Die metodes wat gebruik word in hierdie studie beklemtoon die relevansie van die werk asook die belangrikheid daarvan om die nadelige uitwerking van biobrandstofproduksie op biodiversiteit te minimaliseer. Verskeie benaderings tot die oplossing van ruimtelike konflik, die gebruik van biodiversiteitaanwysers, die beoordeling van die impak van die potensiële indringerspesies en die beplanning vir ekosisteemdienste. Nuwe dryfvere van grondgebruikverandering soos biobrandstof is 'n groot uitdaging en die insigte wat uit hierdie studie verkry is dra by tot die vermindering van die potensiële impak van biobrandstofproduksie op biodiversiteit.

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Chapter 1: General Introduction

1.1. Background

1.1.1. Drivers of global biofuel production

The Intergovernmental Panel on Climate Change (IPCC) predicts a large uptake in renewable energy sources of which solar, wind, and bioenergy (including biofuels) are likely to dominate the future energy mix (Tollefson, 2011). Many countries need to import fossil fuels and increasing oil prices coupled with the uncertainty of supply pose threats to economic growth and development (Amigun *et al.*, 2008). Alternate fuel sources, such as biofuels[1], offer a possible solution to internalise energy production, especially if land resources are available to grow biofuel crops (Von Maltitz & Brent, 2008). Biofuels aim to supplement the fossil transportation fuels (Connor & Hernandez, 2009). They may also reduce the effects of climate change, if grown and managed responsibly (Junginger *et al.*, 2006, Tilman *et al.*, 2009).

Biofuels feature on the agendas of both developing and developed countries (European Commission, 2006, Tollefson, 2011). However the drivers of biofuel production differ across regions. Developed countries seek to increase fuel security and advance the reduction in greenhouse gas emissions. For developing countries, biofuels also provide a means with which to generate jobs, reduce poverty and increase investment opportunities (Von Maltitz & Brent, 2008). In many cases the production of biofuels in one country is meant for export to another (Dauvergne & Neville, 2009). For example, Brazil, an emerging economy, exports 52% of its bioethanol (USDA FAS, 2008), whereas Indonesia exports approximately 50-70% of crude palm oil produced in the country for biodiesel production (USDA FAS, 2013). This demand for biofuels is partially driven by mandated blending targets which requires that biofuels be mixed with petroleum fuels (Eggers *et al.*, 2009). Although crops that drive this global trade in bioethanol and biodiesel include food crops such as sugar cane, corn, soya, canola, and palm oil (Pimentel *et al.*, 2009), there is much progress towards the production of non-food

[1] In this thesis I use "biofuels" to describe the liquid fuel substitute derived from plants to produce either bioethanol or biodiesel

biofuel crops (Hoogwijk *et al.*, 2005, Lynd *et al.*, 2003). Currently, food crops are cultivated to produce starch, sugars, and oils needed to produce biofuels using well-established chemical processing mechanisms which are commonly referred to as first-generation technologies (see section *Feedstocks* on page 21 for more detail). Efficient new cellulosic technologies, known as second generation technologies (see section feedstocks on page 21 for more detail), aim to increase the range of feedstocks used to produce biofuel, thereby reducing competition and resources needed for food production (Slade *et al.*, 2011, Tilman *et al.*, 2009). However, these technologies are not currently commercially viable. As yet, few statistics are available for many of these proposed crops, making estimates of potential production locations and yields difficult to determine (Lapola *et al.*, 2009).

1.1.2. Biofuels in South Africa

In South Africa, biofuel production is likely to depend strongly on both local and global policy directives. For example, the mandatory blending targets in the European Union have stimulated the use and production, and importation, of biofuels (European Commission, 2006). The South African biofuel strategy outlines the development of the biofuel industry and proposes how biofuels should be produced, managed and supported where possible (Department of Minerals and Energy, 2007). There are currently no mandatory blending targets for liquid biofuels in South Africa. Should this change in the near future, then biofuel production will initially be dependent on well-established first generation technologies (Department of Minerals and Energy, 2007). However, the development of second generation non-food crops will also be considered to avoid competition with the production of food (Tilman *et al.*, 2009, Von Maltitz & Brent, 2008). Recently, initiatives have explored the growing of biofuels in South Africa for export (e.g. www.phytoenergy.co.za). Should the infrastructure not be in place to facilitate blending targets for domestic fuel use, the strategy may capitalise on the potential to export biofuels, therefore still fulfilling its mandate of creating jobs. The IPCC has indicated that while many countries are committed to reduce the effects of climate change, various barriers and challenges need to be overcome before widespread biofuel production takes place.

The biofuel strategy acts as a guide to encourage the growing of biofuels in South Africa. However, no guidance is provided for approaching or dealing with the changes in land-use that are inevitable or the potential conflicts (i.e. conflicts with biodiversity or the use of species that have conflict of interest e.g. potentially invasive species) that are sure to emerge on many fronts. Clearly, much further investigation is required to understand the potential impacts on biodiversity. The biofuel strategy is discussed in more detail in Chapter 2.

1.1.3. Impacts of biofuels on biodiversity

Biofuels and biomass energy rely heavily on the cultivation of agricultural or forestry (including short rotation woody crops) species, increasing the concern for land-use change and potential threats to biodiversity (Firbank, 2008, Koh *et al.*, 2009). For example, in 2012 sugarcane (needed for bioethanol) production in Brazil occupied approximately 9.7 million ha and approximately 6.5 million ha was occupied to produce palm oil in Indonesia (FAOSTAT, 2014). In both Brazil and Indonesia, the cultivation of biofuel crops acts as a driver of land cover change, especially the loss of tropical forest cover (Koh & Wilcove, 2007, Lapola *et al.*, 2009). Land-use change is recognised as a global threat for natural ecosystems and is a major driver of biodiversity loss (Foley *et al.*, 2005). Biofuel production can be placed at the nexus of multiple disciplines, such as climate change (Junginger *et al.*, 2006), land-use change (Lapola *et al.*, 2009), biodiversity loss (Dauber *et al.*, 2010, Immerzeel *et al.*, 2014), and economic development (Von Maltitz *et al.*, 2009), with varying impacts attributed to each (Mathews, 2008, Tilman *et al.*, 2009). As a result, there is a need to provide information to decision-makers and for policies that minimise potential tradeoffs between the natural environment and the need to meet increasing human development. While decisions for sustainable biofuel production should include social and economic considerations, this dissertation focuses solely on impacts on biodiversity.

1.1.4. The importance of biodiversity and factors affecting its loss

Biodiversity is considered an important contributor to ecosystem functioning (Hooper *et al.*, 2005) and ecosystem resilience (Cardinale *et al.*, 2012), and underpins the provisioning of ecosystem services (Balvanera *et al.*, 2006, Millennium Ecosystem Assessment, 2005). Biodiversity is a complex term (Noss, 1990) and to capture all of the intricate components (i.e. genetic, species and ecosystem diversity) requires different indicators focussed at each level of biodiversity. Nonetheless, biodiversity is being lost at an alarming rate and the trajectory of loss has not slowed despite considerable efforts in this direction (Stokstad, 2010). The loss of biodiversity and degradation of natural ecosystems presents a serious challenge to governments and human societies around the world (Cardinale *et al.*, 2012). These impacts arise from the need to provide food, fibre and fuel for growing human populations which have greatly transformed the world's terrestrial surface (Hooke *et al.*, 2012). While countries have agreed to slow the rates of biodiversity loss, tools that assist biodiversity monitoring are needed to facilitate land-use planning at the earliest possible stage (Gibbons *et al.*, 2009). Biodiversity indicators that may be most useful to decision-makers should therefore be sensitive to land-use changes and incorporate a spatial aspect. The simplest way to reduce biodiversity loss is to maintain intact and healthy ecosystems. Alternatively, recent research into understanding biodiversity and ecosystem functioning (BEF) may help to sustain healthy ecosystems by identifying and maintaining the key components within functional landscapes (Diaz *et al.*, 2006, Lavorel *et al.*, 2011).

1.2. Assessing impact across multiple scales of biofuel production

The impacts of land-use change can be assessed at multiple scales ranging from broader landscape scales to smaller field scales (Firbank *et al.*, 2008). In attempting to assess the impact of biofuels, I draw on examples from the discipline of invasion biology. Many proposed biofuel crops are known to be invasive (i.e. they spread from sites where they are cultivated) or are very likely to be invasive (Raghu, 2006) if introduced to new regions and cultivated in large numbers (Richardson and Blanchard, 2011). Plant invasions associated with the cultivation of alien plants have a much longer history and have resulted in informed approaches to manage and reduce potential problems which could be successfully applied to guide biofuel production. Conceptual frameworks that address the impacts of introduced species have received much attention in recent years (Thomsen *et al.*, 2011). These examples address either the economic (Pimentel *et al.*, 2005), social (Kull *et al.*, 2007) or ecological impacts (Levine *et al.*, 2003, Parker *et al.*, 1999) of invasive alien species (Dodet & Collet, 2012).

For this research, I adopt an existing framework that was developed to assess the ecological impacts of plant invasions to identify important factors that should be considered when addressing ecological impacts that act across different scales. The framework by Parker *et al.* (1999) defines impact as the product of three factors, namely: range, abundance, and the per capita effect. This framework, which suggests that the role of each factor must be understood to appreciate overall impacts, is used as the foundation for exploring potential impacts of biofuel production on biodiversity. This framework is closely aligned with that of Firbank (2008) who suggests that impacts on biodiversity need to be considered at multiple scales, ranging from regional to field scales, to more accurately identify scale appropriate responses to reduce adverse pressures. Impacts of biofuel production could therefore be assessed by considering the extent of land area to be occupied (range), the density at which species are planted (abundance), and the effect of the species and their traits on ecosystems and ecosystem functioning (per capita effect). These conceptualizations of impact were used to compartmentalize the approaches used in this dissertation for exploring impacts on biodiversity across different scales (see Figure 1.1 on page 10).

In this study land-use is the key variable that is used to link the three components of impact. By focusing on land-use, it is possible to link scenarios of changes in land-use, either spatially or functionally, to determine the effects on biodiversity and ecosystem services (IEEP *et al.*, 2009). Recent studies that have addressed the impacts of biofuels as a land-use option have focused on modelling patterns land-use change (Li *et al.*, 2012) or identifying potential areas of biofuel suitability (Evans *et al.*, 2010, Geyer *et al.*, 2010b, Stoms *et al.*, 2011). The challenge exists in linking the effects of land-use change on biodiversity (Geyer *et al.*, 2010a, Koh *et al.*, 2010, O' Connor & Kuyler, 2009). Biodiversity indicators designed to function with changing land-uses may hold promise as a tool for future planning (A review of potential indicators are located in the

Appendix A). I address these challenges using the framework described above and focus at multiple levels to understand potential tradeoffs between meeting human needs and maintaining the capacity of ecosystems to provide goods and services into the future.

1.3. Problem statement

South Africa has proposed that biofuels should be used to supplement the liquid petroleum fuel supply, forming part of the country's renewable energy policy (Department of Minerals and Energy, 2007). The South African biofuel strategy outlines the manner in which biofuels will be developed and where government support is likely to be focused (Department of Minerals and Energy, 2007). Biofuels do not currently contribute to the national fuel supply and development of this industry will require substantial investment in infrastructure for both the production of biofuels and the processing thereof. The implementation of this biofuel strategy has been slow, providing opportunities for assessing the potential impacts that this industry may cause. While there have been some studies that have focused on the technological aspects (Lynd *et al.*, 2003), the social issues (Amigun *et al.*, 2011), or economic implications (Funke *et al.*, 2009, Meyer *et al.*, 2009) of biofuels production in South Africa, uncertainties remain regarding the potential impacts on biodiversity. These concerns are based on global evidence that biofuels production can greatly increase the total area under cultivation thereby increasing rates of habitat and biodiversity loss (Tilman *et al.*, 2009, Wiens *et al.*, 2011). South Africa is biologically diverse and any large-scale changes to land-cover will have negative impacts on biodiversity and ecosystem services. Research is therefore needed to determine the potential impacts on biodiversity by assessing the issues, locations, and potential conflicts related to biofuel production in South Africa.

1.4. Rationale

Global trends indicate that the establishment of biofuel industries will exacerbate the major drivers of biodiversity loss, including habitat loss and the movement of potentially invasive species around the world (Fischer *et al.*, 2010a, Raghu *et al.*, 2006). For example, the expansion in production of existing food crops such as corn, sugar cane, soya bean and oil palm for use in biofuel production has increased dramatically in the last decade (Connor & Hernandez, 2009, Fischer *et al.*, 2009, Sala *et al.*, 2009). Furthermore, unregulated cultivation in many tropical regions (Indonesia, Malaysia and Madagascar) has raised questions regarding the sustainability of biofuels development, especially high biodiversity areas (Fitzherbert *et al.*, 2008, Koh, 2007a, Koh, 2007b). The demand for biofuels has led to an increase in research on feedstocks other than food crops, and many of the species that have been targeted have worryingly similar traits to current invasive species (Low & Booth, 2007, Raghu *et al.*, 2006). Species will likely be selected on the basis of their ability to grow fast and grow without major nutrient inputs. The risk of some introduced biofuel species becoming invasive is of real concern (Barney & DiTomaso, 2010, Richardson & Blanchard, 2011) and needs to be considered in any assessment of the potential impacts of biofuel production.

With the projected increase in global populations, the need to provide sufficient food for approximately 8-10 billion people by 2050 (Lutz & KC, 2010) coupled with the rising consumption of biofuels will place increasing pressure on the remaining natural areas in South Africa and around the world (Foley *et al.*, 2005, Reyers, 2004). This presents a particular challenge for land-use planning to find a balance between the production of food or fuel, and biodiversity conservation (Foley *et al.*, 2005, Tilman *et al.*, 2009). Whilst protected areas aim to conserve important biodiversity, these reserves only protect approximately 12% of the earth's surface (Jenkins & Joppa, 2009). The remaining 88% is potentially 'available' to conversion and subject to local legislation. However, an estimated 54% of the world's terrestrial surface has already been altered to some extent by humans (Hooke *et al.*, 2012), making the remaining natural habitat more valuable from a variety of perspectives. These remaining areas contain a substantial proportion of the world's biodiversity and also deliver vital ecosystem services (Reyers, 2004, Reyers, 2013). Some of these areas have also been identified as having vast potential for biofuel and biomass production, and many occur in developing countries (Beringer *et al.*, 2011).

In South Africa, approximately 82% of the land area is considered to be natural or semi-natural, yet only 6.5% of the landscape is under formal protection (Driver *et al.*, 2012). This means that many ecosystems are not adequately conserved by the protected area network (Driver *et al.*, 2012) and future development could threaten areas with high biodiversity that occur outside of protected areas. Although a relatively large proportion of the country remains in a natural or semi-natural state, the rates of transformation are not evenly spread across the many ecosystems that occur in South Africa. Some ecosystems are more threatened or likely to be threatened based on their potential for transformation to land-uses such as cultivation agriculture and biofuel production (Mucina & Rutherford, 2006, O' Connor & Kuyler, 2009). Alien plant invasions, many of them resulting from the spread of commercially important species, are also a major and growing threat to biodiversity and livelihoods (Richardson & van Wilgen, 2004, van Wilgen & Richardson, 2009), resulting in large national programs tasked with clearing existing alien infestations and to reduce the threat of future alien plants from becoming invasive.

These issues, which are briefly outlined in this section, will be addressed by focusing on a potential biofuel producing region in South Africa. The Eastern Cape Province of South Africa has been identified as having a large potential for increased agricultural development (Amigun *et al.*, 2011, Estes *et al.*, 2013, Government of South Africa, 2008). Current estimates of approximately 3 million ha of land are available for cultivation in this province (Von Maltitz *et al.*, 2010). Establishing a biofuels industry in this region requires that the potential risks to biodiversity resulting from future land-use changes or species introductions be further investigated.

Managing tradeoffs between biodiversity resulting from increased development presents a serious challenge for conservation scientists around the world. In this dissertation important challenges for assessing biodiversity impacts are addressed. These challenges include how to deal with all aspects of biodiversity in a manner that is easily understandable and meaningful for decision-makers, reflecting on issues of scale, and estimating the potential impacts of land-use change. Potential areas for biofuel production are identified and biodiversity indicators are used to determine impacts of potential land-use change. In predicting which ecosystems or natural areas are most at risk from future land-use change, likely impacts on biodiversity can be anticipated to provide alternative actions to minimise potential losses.

1.5. Objectives and research questions

The major topics investigated by this research are biodiversity loss through land-use change, the effects of fragmentation on biodiversity, and the effect of introduced species on the functional characteristics of ecosystems. The questions central to this research are as follows:

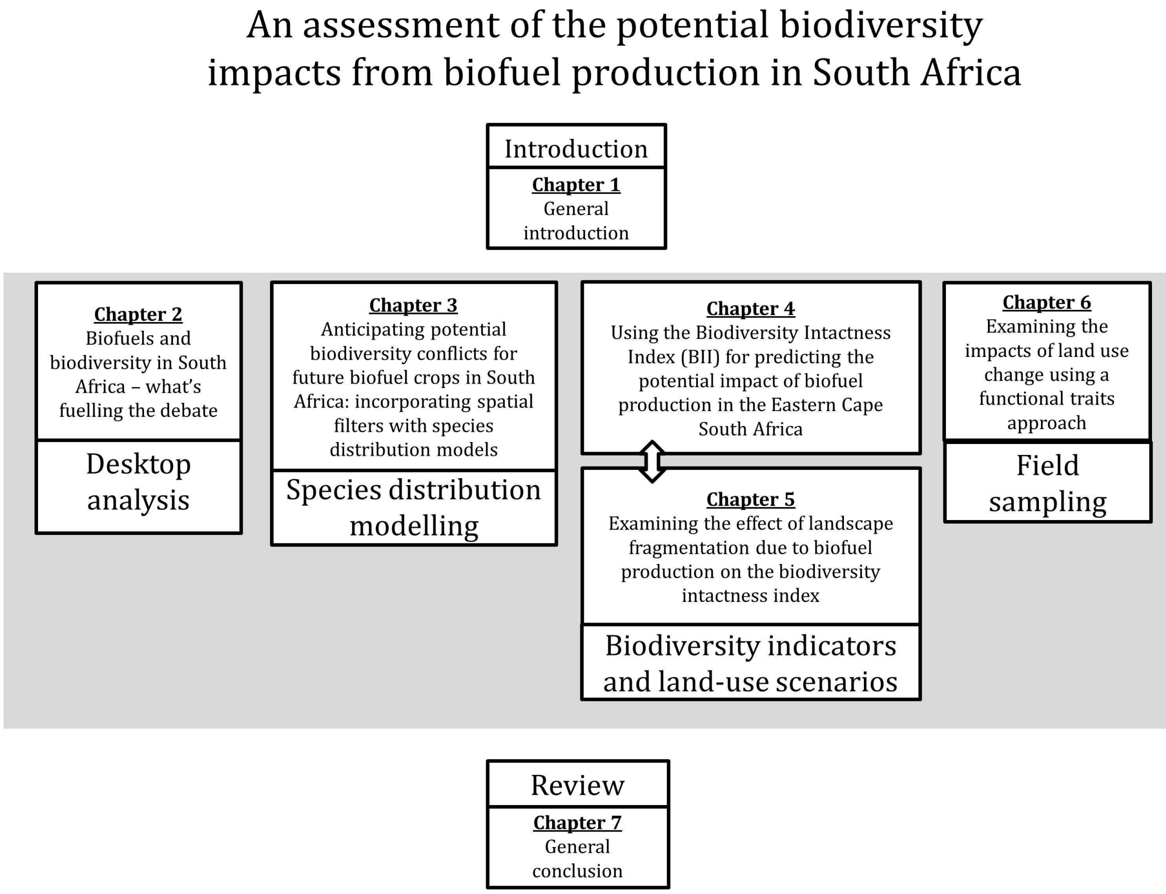
1) What are the potential impacts of adopting a biofuel strategy on land-use change and its associated impacts on biodiversity?

2) What methods are available to assess biodiversity and how can these be improved or modified to better address questions relating to biofuel production more directly?

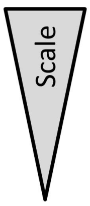
3) How can the components of impact (range, abundance, and per capita effect) be assessed to address biodiversity and ecosystem services across multiple scales?

The studies presented in this dissertation aim to address each of these questions and to provide examples of the use of existing tools needed to anticipate and measure impact. The multidisciplinary approach used to understand biodiversity requires a similar approach to determine impacts. This research therefore has multiple objectives which are outlined in the section below. These objectives form part of an integrative assessment of impact as outlined by the Parker framework (Parker *et al.*, 1999) and focus at different scales of analysis.

a)



b)



Components of impact	Chapter 3	Chapter 4	Chapter 5	Chapter 6
- Range	X	X	X	
- Abundance				X
- Per capita effect		X	X	X

Figure 1.1: A schematic overview of the chapters and methods applied in this dissertation a) the chapter outline and methods used and b) where the components of impact are addressed in different chapters. The main dissertation topic was evaluated according to three components of a framework (b) used to understand the impacts of invasive alien plants developed by Parker et al. (1999). The impact framework is discussed in detail in Chapter 2. The components (range, abundance and, per capita effect) can be used to assess impact at different scales. The research aims to assess potential impacts on biodiversity resulting from land-use change for the production of biofuels in South Africa.

1.6. Structure

To answer the main research questions, five stand-alone studies were carried out, each with its own research objectives. These chapters are also interrelated since they deal with the same topic (i.e. potential impacts of biofuel production), focus on the same area (i.e. the Eastern Cape, South Africa), and all address one or more of the main research questions. Chapter 1 presents a general overview of the research topics. Chapters 2 to 6 are stand-alone papers. In chapter 3, 4 and 5 I use land-use scenarios to determine potential biodiversity conflicts, firstly using GIS (chapter 3) and secondly by applying biodiversity indicators (chapter 4 and 5). In chapter 6, impact is assessed at the field scale based on plant functional traits. Chapter 7 presents the conclusions, challenges faced, and recommendations for future research. I use the terms "I" in chapters that are unpublished (Chapters 4-6) and "we" in co-authored published manuscripts (Chapters 2-3).

The stand-alone chapters, objectives including a description my contribution to each chapter is described below:

i) **Chapter 2: Biofuels and biodiversity in South Africa – the issues.**
Main objective: To determine the key issues regarding the production of biofuels and potential impacts on biodiversity.
   - This chapter discusses the South African biofuel strategy and its potential impacts on biodiversity. Biodiversity impacts were conceptualised using the framework for assessing the impact of invasive alien species developed by Parker *et al*. (1999).

Author contributions: I conducted the review and wrote the manuscript.

ii) **Chapter 3: Species distribution models.**
Main objective: To determine potential biofuel production areas in the Eastern Cape and to assess biodiversity conflicts in this region.
   - The species distribution modelling programme MaxEnt was used to identify the potential location of biofuel species based on the matching of climate variables. Climate models are combined with land-use to identify spatial filters that limit the extent of potential biofuel production.

Author contribution: I conducted the review, ran the analysis and wrote the manuscript.

iii) **Chapter 4: Using the Biodiversity Intactness Index for predicting the potential impact of biofuel production on biodiversity in the Eastern Cape, South Africa.**

Main objective: To determine the use of the biodiversity intactness index to assess biodiversity impacts.

- The Biodiversity Intactness Index (BII) was used to estimate biodiversity losses based on the scenarios for land-use change described in chapter 3. The BII function requires an estimate on area per land-use, species richness per habitat type or biome, and an impact factor of land-use on species abundance.

Author contribution: I conducted the review, ran the analysis and wrote the manuscript.

iv) **Chapter 5: Examining the effect of landscape fragmentation due to biofuel production on the biodiversity intactness index.**

Main objective: To determine relationships between the BII and fragmentation indicators and the extent to which the effect of fragmentation are captured when applying the BII.

- The BII is known to be insensitive to the effects of fragmentation. Relationships between the BII and fragmentation indicators were assessed resulting in modification of the BII.

Author contribution: I conceived the idea, ran the analysis and wrote the manuscript.

v) **Chapter 6: Examining the impacts of changing land-use on ecosystem services using a functional traits approach.**

Main objective: To determine the effect of introduced plant species on community functional structure and potential impacts on ecosystem services.

- Plant functional traits have been identified as the primary biological mechanism that influences ecological functions and processes. Field sites were sampled to characterise the functional composition of land-uses in the grasslands of the Eastern Cape. The functional traits of non-native key species associated with different land-uses were assessed in relation to native grasslands.

Author contribution: I conducted the reviews of traits, undertook the field work, ran the analysis and wrote the manuscript.

Part I:
Issues and potential scope of biofuels

Chapter 2: Biofuels and biodiversity in South Africa – What's fuelling the debate?[2]

2.1. Abstract

The South African government, as part of its efforts to mitigate the effects of the ongoing energy crisis, has proposed that biofuels form an important part of the country's energy supply. Biofuels currently do not contribute to the national fuel supply, but were expected to contribute at least 2% of liquid fuels by 2013. The Biofuels Industrial Strategy of the Republic of South Africa of 2007 outlines key incentives for promoting biofuels development. This paper discusses issues relating to the strategy as well as key drivers in biofuel processing with reference to potential impacts on South Africa's rich biological heritage.

Our understanding of many of the broader aspects of biofuel needs to be enhanced and we identify key areas where challenges exist, such as the link between technology, conversion processes and feedstock selection. The available and proposed processing technologies have important implications for land-use as well as the use of non-native plant species as desired feedstocks. South Africa has a long history of planting non-native plant species for commercial purposes, notably for commercial forestry. We propose that lessons can be learnt from this experience to mitigate against potential impacts by considering plausible scenarios and the appropriate management framework and policies. We conceptualize key issues embodied in the biofuels strategy, adapting a framework developed for assessing and quantifying impacts of invasive alien species. This provides guidelines for minimizing potential impacts of biofuel projects on biodiversity.

Key Words: Biofuels, biodiversity, biological invasions, impact

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2.2. Introduction

The uncertainty of long-term fossil fuel supply and volatile fuel prices, which recently reached record levels, as well as increasing CO_2 emissions has generated much interest in alternative fuel sources, especially "biofuels" which are defined as solid, liquid or gaseous fuels obtained from biological material (Fischer *et al.*, 2009). The production of biofuels is being widely promoted as a renewable and environmentally friendly way of reducing the use of fossil fuel (Connor & Hernandez, 2009). Biofuels have garnered much support and feature on political agendas in both developed and developing countries (European Commission, 2006). In the developed world, biofuels offer a potential means of reducing greenhouse gas (GHG) emissions and increasing fuel security. In the developing world, besides the issue of fuel security the main drivers behind the biofuel industry are the need to facilitate rural and national development, provide jobs, and improve trade balances. These factors are all influenced by the establishment and demand from international markets for biofuels, such as those in Europe (Junginger *et al.*, 2006, Lynd *et al.*, 2003, Von Maltitz & Brent, 2008).

Much of the global debate on biofuels has focused on policy, economics, social issues such as competition with food crops, and the overall implications of pursuing a biofuel strategy for GHG emissions (Lynd *et al.*, 2003, Von Maltitz & Brent, 2008). However, recent studies are showing that the expansion of biofuel cultivation to produce corn, sugar cane, soya bean and oil palm pose significant threats to ecosystems and biodiversity (Fischer *et al.*, 2009, Sala *et al.*, 2009). Examples of this are the unregulated cultivation in many tropical regions (Indonesia, Malaysia and Madagascar) which has resulted in reduced forest cover and massive habitat transformation.

Before being considered a viable alternative, biofuels must demonstrate an overall net energy gain (Chakauya *et al.*, 2009) and, among other things, provide some environmental benefits, despite being produced in large quantities and therefore requiring large areas of land. These issues are normally assessed using the life-cycle assessment (LCA) process which considers all resource and energy inputs, wastes and ecological impacts associated with the final product. The LCA process has identified key issues relating to land-use change and the carbon debt associated with the destruction of vulnerable ecosystems such as tropical forests and peat lands to make way for biofuel plantations (Tilman *et al.*, 2009). The growing negative perception regarding biofuel planting in developing countries is likely to influence public acceptance of this product (Phalan, 2009), reducing demand, despite the increasing investment opportunities (Dauvergne & Neville, 2009). The lack of appropriate environmental standards and criteria for biofuel cultivation has not stopped governments pursuing biofuel strategies that promise certain economic and fuel-security incentives (Hill *et al.*, 2006).

South Africa can be considered a world leader in conservation planning (Balmford, 2003) where researchers are progressing the science (Cumming & Child, 2009) leading to a range of initiatives and interventions aimed at protecting its high levels of biodiversity (Driver *et al*., 2012). The main threats to biodiversity in South Africa are habitat degradation/transformation and a range of impacts due to invasive alien species, especially plants (Rouget *et al.*, 2004). Rapid climate change also poses a major threat. Fourteen percent the country's land surface area is already under some form of cultivation or afforestation (Biggs & Scholes, 2002) and there is growing concern that the existing network of protected areas still does not conserve a representative sample of our biodiversity, or fully include key ecological processes (Rouget *et al.*, 2004).

A number of alien plant species, introduced in the past for forestry, ornamental use, sand dune stabilization and other purposes have become invasive, altering ecosystem function and affecting the capacity of ecosystems to deliver goods and services (Richardson & van Wilgen, 2004, van Wilgen *et al.*, 2008). Biofuel initiatives have the potential to add substantially to these existing threats to biodiversity in two main ways: directly, through habitat conversion, and indirectly, through impacts due to the invasive spread of feedstock species out of areas set aside for biofuel production (many of the most suitable feedstock species are either known to be invasive somewhere in the world or are likely to be invasive) (Buddenhagen *et al.*, 2009, Sala *et al.*, 2009).

There is no denying the potentially positive benefits of biofuels (i.e. job creation and development of rural economies – See section 2.3 for more information) to the South African economy (Chakauya *et al.*, 2009), but failure to consider all possible outcomes from biofuel production could have unforeseen and substantial costs. Biofuels currently do not contribute to the national fuel supply, but government initiatives are underway (see below - South African Biofuels Industrial Strategy and Box 1.1) to establish a biofuel industry that is capable of improving the country's fuel security and driving rural development objectives. Despite the rapid global expansion of biofuel plantations, little information is available on the potential ecosystem impacts of such land-use (Junginger *et al.*, 2006). Given South Africa's globally significant biodiversity, such information is critical for developing appropriate strategies to minimize impacts before any major production initiatives are launched. The South African forestry, timber, pulp and paper industry has a long history dating back to the 1800s (Richardson *et al.*, 2003). The emergence of this industry also relied on the use of alien plants, many of which are invasive, resulting in large-scale transformation of habitats, and many conflicts with other potential land-uses such as conservation. The history of forestry in South Africa is therefore informative when considering strategies for the use of biofuels. Besides the many policy and legal frameworks that regulate the forestry industry, considerable research has also been undertaken on the impacts of forestry species on ecosystems. South Africa also has a long history of problems with invasive plant species, and of devising innovative approaches for managing such problems. Research insights from invasion biology should also be useful for predicting and preventing or reducing additional problems from invasive species that could result from the specific pathways created by new introductions and dissemination patterns that will be required to launch a biofuel industry in the country.

This paper examines key issues relating to the potential for the sustainable production of biofuels in South Africa while minimizing the impact of the region's biodiversity. We discuss the emerging local biofuel industry with reference to the Biofuels Industrial Strategy of the Republic of South Africa (Department of Minerals and Energy, 2007). Although there are many aspects of growing biomass for bioenergy, we restrict this review to the production of liquid fuels. We discuss the role of existing and future technologies in relation to feedstock selection and associated ecological impacts. Measuring the impacts of biofuels is also addressed in relation to proposed frameworks for assessing the impact of invasive species. Finally, we highlight the role of industry as a mechanism for facilitating the introduction and dissemination of potentially invasive species, and identify ways of reducing impacts.

2.3. The South African Industrial Biofuels Strategy

South Africa is the only southern African country that has a formal biofuels strategy – the Biofuels Industrial Strategy of the Republic of South Africa of 2007 (Department of Minerals and Energy, 2007) hereafter "the strategy". The Department of Minerals and Energy (DME) envisages biomass energy (mainly liquid biofuels) contributing 35% to the national targets for renewable energy as set by The White Paper on Renewable Energy (Department of Minerals and Energy, 2003) by 2013; the rest will come from solar and wind projects. The strategy outlines mechanisms to undertake a five-year pilot programme to supplement a cautionary initial biofuel target of 2% of liquid fuels, with a decision on whether to increase this proportion to be made at the end of the pilot phase (Department of Minerals and Energy, 2007).

The strategy aims to achieve economic and social development in rural areas, particularly the former homeland areas. Other objectives of the strategy include the promotion of agricultural development, adding to the renewable energy pool, and improvement of the country's fuel security (Funke *et al.*, 2009). The creation of jobs and improving the development imbalance between informal and small-scale farming areas and commercial farming areas are key components of the biofuel supply chain (Lynd *et al.*, 2003). These socio-economic goals have resulted in government support for the strategy being confined to regions that are likely to benefit most, such as the former homeland areas in the Eastern Cape (Department of Minerals and Energy, 2007). Incentives for locally-based processing plants are based on the fact that feedstocks should be acquired via contractual agreements from small-scale farmers in the region (Funke *et al.*, 2009). This is intended to stimulate demand and incentivise farmers to optimise longer-term yields while increasing land productivity (Department of Minerals and Energy, 2007).

Current feedstock options for biofuel processing in South Africa consist of agricultural food crops, as no dedicated fuel crops have been approved by government for biofuels production (Department of Minerals and Energy, 2007, Funke *et al.*, 2009). However, the strategy recognises that food security should not be compromised and maize has initially been excluded. Approved crops for bioethanol production are sugar cane and sugar beet, and for biodiesel sunflower, canola and soya beans. The suitability of these species has been mapped for South Africa according to both rainfed and irrigated options, including information on grain sorghum for bioethanol (Van der Walt & Schoeman, 2006). The strategy also recognises potential problems of introducing dedicated energy crops. Consequently, the Department of Agriculture has placed a moratorium on a potential energy crop, *Jatropha curcas* (See Box 2.2), because of concerns regarding its potential invasiveness. However, there continue to be new developments that promote the introduction of different dedicated energy crops that already are considered invasive or could threaten to become invasive in the future (Buddenhagen *et al.*, 2009). However, many of the promises for dedicated energy crops, especially jatropha, are proving to be largely unfounded (Achten *et al.*, 2010), emphasising that the use of crops with no history of cultivation can result in unpredictable and inconsistent yields (Connor & Hernandez, 2009, Von Maltitz & Brent, 2008). Consequently, based on South Africa's biofuel feedstock selection, the production of existing crops need to be scaled-up considerably to meet the proposed minimum target of 2% biofuel penetration into the liquid fuels market.

Box 1.1. Biofuels in South Africa

South Africa's history of using biomass as an energy source dates back to the 1920s when ethanol derived from sugar cane was mixed with petrol (Von Maltitz & Brent, 2008). Between the 1970s and early 1990s South African involvement in the development of alternative fuel sources was largely in response to sanctions placed on the apartheid government (Lynd *et al.*, 2003). As a consequence, South Africa developed the capacity to convert both coal and natural gas to petroleum using the Fischer-Tropsch process. Currently about 23% of liquid fuel used in South Africa is derived from coal, 5% from natural gas, and 72% from imported crude oil (Winkler, 2006). From the perspective of greenhouse gas emissions, coal-derived liquid fuels are about twice as polluting as oil-derived fossil fuels. Research projects in this area were scaled down after the democratic elections in 1994, as improving services and opportunities for previous-disadvantaged people was seen as a more immediate need (Lynd *et al.*, 2003). There is currently renewed public and political interest in biomass energy and the agricultural practices through which biofuel can be produced is considered to have potential to fulfil both social and energy mandates in many countries, including South Africa (Department of Minerals and Energy, 2007). A challenge for the South African government is the considerable investment and infrastructure required to guarantee continued supply of appropriate feedstocks, and the need for efficient biomass conversion techniques (Von Maltitz & Brent, 2008). Globally, biofuel production is a relatively a new industry and more research is needed on technologies, agricultural practices, and the potential environmental and social impacts. Initial growth of biofuel enterprises will depend on first-generation technologies. Recent technological advancements that allow for increased feedstock selection (e.g. hardwoods, agricultural and municipal wastes) are constrained by the non-viability of commercial applications since the conversion technologies of cellulosic biomass are still in development and may be commercially available within the next two decades (Fischer *et al.*, 2009, Lynd *et al.*, 2003). Nevertheless, projected technological advancements could expand options feedstock species and create more efficient conversion processes for lowering CO_2 emissions.

2.4. Technology as a potential driver of impact

The demand for biofuels, like other commodities, is driven by the needs of human societies, and is influenced by available processing technologies. These will determine the success and the extent to which the biofuel industry will be developed in South Africa. While the strategy aims to use biofuel expansion as a vehicle for development, little attention is given to the implications of current and future technologies for the environment. In the following section we discuss the role of technology in biofuel feedstock selection and the implications for the environment.

2.4.1. Feedstocks

Feedstock options are limited by available processing technologies, appropriate equipment, research on yields, and the ability to utilise the genetic diversity to produce different varieties well suited to local growing conditions (Cheesman, 2004). There is no right or wrong feedstock; the best option is determined by profitability, mediated by climatic and environmental factors. Until now, biofuel production has focussed mainly on food and feed crops that rely on ***first-generation*** conversion pathways, using agricultural mechanisms to produce sugar, starch, or vegetable oil components for biofuel processing. However, since feed crops have a low land-use efficiency, their use in commercial biofuel production is expected to be extremely demanding on land resources (Fischer *et al.*, 2009). Also, the use of food and feed crops for fuel purposes raises major ethical and nutritional concerns as resources such as energy, water, fertilizers and land may be allocated to produce feedstock for fuels over the provision food in poorer communities (Tilman *et al.*, 2009).

A key challenge surrounding the utilization of food crops will be to maximise existing agricultural output in order to produce a surplus for biofuel production, without affecting the pricing and availability of food. Historically, increasing food demands have been met by initially increasing the area under cultivation, and later through the development of new technologies (use of higher yield cultivars, pesticides and inorganic fertilizers) increasing yields (Biggs & Scholes, 2002). However, recent attempts to further increase plant potential via genetic modification have in most cases failed to fulfil promised potential (Sala *et al.*, 2009). Novel farming practises such as introduction of new crops like sugar beet and sweet sorghum to supplement the need for sugars and starches as well as use of dedicated energy crops are likely options that need further exploration. Furthermore, the economic viability of many first-generation crops is dependent on the ability to derive value from both biofuel and by converting wastes into useful by-products (Chakauya *et al.*, 2009, Fischer *et al.*, 2009). Successful utilization of by-products includes valuable livestock feed (e.g. rapeseed cake, soybean meal), biomass fuels (straw, husks and bagasse), and materials for industrial use (such as glycerine).

For many (see Tilman *et al*., 2009 and Fischer *et al*., 2009) the future of biofuels lies in the ability to commercialise advanced conversion technologies, such as the Fischer-Tropsch Synthesis process, and biochemical pathways capable of converting lignocellulosic material to produce ethanol or liquid hydrocarbons (Fischer *et al.*, 2009). These advanced processing methods, termed **second-generation** technologies, are currently in various stages of development and are expected to be commercially available in the next 10-20 years (Fischer *et al.*, 2009). The role of second–generation processing technologies will favour the production of perennial crops such as fast-growing trees (e.g. short-rotation woody crops) and grasses, and could utilise waste products generated from non-biofuel production systems (i.e. crop and forest residues) (Junginger *et al.*, 2006, Tilman *et al.*, 2009).

Box 1.2. *Jatropha curcas* - the solution for Africa?

Jatropha curcas (see Figure 2.1), Euphorbiaceae, (hereafter jatropha) has been widely promoted as a drought-resistant non-edible perennial oil crop for producing biodiesel which can grow in low–rainfall and otherwise marginal areas (Achten *et al.*, 2010, Achten *et al.*, 2008). The oil content of the seed is around 30−35%; this can be used as a fuel prior to trans-esterification to biodiesel. It is potentially suitable for rural village electrification as demonstrated in pilot projects in India and Mali. It has also been recommended for reclaiming marginal and degraded lands (Achten *et al.*, 2010) by improving soil quality and reducing erosion (Ogunwole *et al.*, 2008). Jatropha plantations were expected to increase to 5 Mha in extent by 2010 and although widely considered by most countries in the Southern African Development Community (SADC) as a viable biodiesel crop, there is a moratorium on planting in South Africa, mainly because of concerns about its invasive potential (Department of Minerals and Energy, 2007). It is currently only grown in a few trial plantations and hedgerows in South Africa (Gush, 2008). Yield claims for jatropha range between 0.4−12 t ha^{-1} y^{-1} indicating that the undomesticated nature and limited physiological data (Achten *et al.*, 2008) makes crop productivity poorly predictable. Whereas recent research has demonstrated that jatropha is unlikely to use more water than indigenous vegetation in the study area of Kwa-Zulu Natal (Gush, 2008), these findings need to be scaled up to determine impacts of plantations on watersheds (Maes *et al.*, 2009). Jatropha, like other agricultural crops, requires water and good soils to realise the large yields that initially raised the species to prominence as a biofuel crop. Notions of growing this still 'wild' plant in monoculture seem to be shifting towards using it in small-scale farming so that the potential of the plant can benefit the rural poor by using the oil to fuel stoves, lamps and diversify income (Achten *et al.*, 2010).

a)

b)

c)

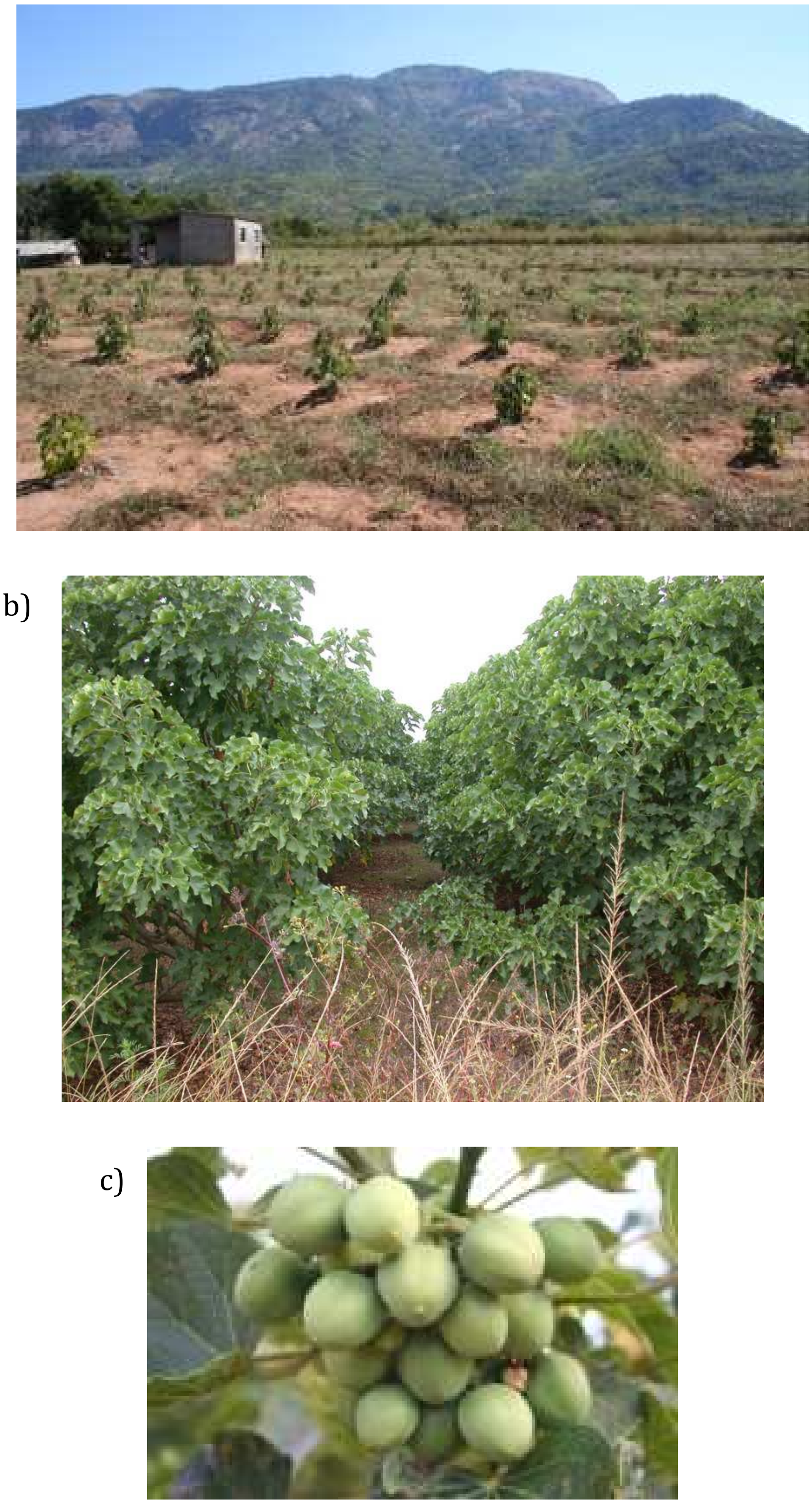

Figure 2.1: *Jatropha curcas* is a perennial oil crop that is planted across much of southern Africa and used to produce biodiesel: (a) a young jatropha plantation in Mozambique indicating the spacing and extent of various planting schemes, (Photo credit: K. Setzkorn) (b) a 4-year-old trial plantation of 2-m-tall jatropha plants in Pietermaritzburg (Photo credit: R. Blanchard) and (c) jatropha fruits which contain the seed nuts that are harvested for their oils (location unknown) (Photo credit: K. Setzkorn)

2.4.2. Ecological implications

Current agricultural management systems are struggling to improve production, minimise water and nutrient use, conserve soils, control weeds, and reduce the impacts of pesticides (Cheesman, 2004). Many of these problems will only be exacerbated by the use of new crops with which farmers have little familiarity (Connor & Hernandez, 2009). Accommodating biofuel production (first- or second generation approaches) within the emerging bioeconomy will place strain on sustainable use of different land types as the demand for land is likely to increase (Junginger *et al.*, 2006), placing the agricultural and conservation sectors under pressure. This will be especially important for areas without formal protection status but which play important roles in regional and global biodiversity conservation (Rouget *et al.*, 2004, Sala *et al.*, 2009).

Experience in the forestry and agricultural sectors in South Africa has shown that formal policies and legal instruments are crucial for limiting negative impacts to resources (Gush *et al.*, 2002) (e.g. water) and biodiversity (Scholes, 2002). Such insights are useful when considering likely trajectories of expansion and scenarios of impact on biodiversity for the biofuels industry in South Africa. At the smallest scale, attempts to increase the yield potential of bioenergy crops, and the repeated introduction of new plant varieties, may affect the gene pools of wild relatives, thereby placing the genetic variation within local populations at risk (Firbank, 2008). In South Africa this concern has been highlighted as the introduction of genetically-modified (GM) canola (*Brassica napus*, a favoured biofuel feedstock) poses a risk of hybridization with closely related species within regions in the fynbos biome (McGeoch *et al.*, 2009). However, unlike GM crops where the risks are triggered by plant breeding, impacts from biofuel crops are likely to result in larger environmental impacts due to the location, cultivation practise and the choice of species (Firbank, 2008).

New technologies and the use of dedicated energy crops could persuade farmers and landowners to shift to biomass cultivation in favour of drought-tolerant shrubs, fast-growing trees or perennial grasses, often in monocultures. These shifts in production focus are likely to cause major structural changes in vegetation over large areas, affecting vegetation cover and biomass, albedo, phenology, water use, micro-climates, fire hazard, habitat for other biota, and many other factors (Firbank, 2008, Richardson & van Wilgen, 2004). Apart from such impacts on landscape structure, composition and function, such transformations will also create a new pathway for the introduction and dissemination of potentially invasive species to and within the country (Richardson *et al.*, 2003). The role of invasive species in biofuel production is discussed in the following section, but it is important to mention here that feedstocks that are known to be invasive and which are *guaranteed* to cause problems in this regard if used for biofuels in South Africa include perennial grasses: *Sorghum halepense, Miscanthus* spp., *Arundo donax* and trees, including *Millettia pinnata* and species of *Acacia*, *Eucalyptus* and *Populus*. We are only beginning to understand the links between alien plant invasions and the delivery of ecosystem goods and services and although headway is being made, there is a need for more detailed studies (Richardson & van Wilgen, 2004).

Further ecological impacts are likely due to changes in the management of resources. For example, crop and forestry waste materials/residues left *in situ* contribute to nutrient-cycling processes that are important for maintaining soil quality and increasing the carbon organic matter returned to the soil (Lal & Pimentel, 2007). Therefore using residues for biofuel may increase the risk of soil erosion and deplete soil organic matter, potentially requiring excessive use of fertilizers and herbicides to maintain crop yields. In a similar fashion water is a limiting factor for development (Le Maitre *et al.*, 2002) and the redirection or increased demand may place a further strain on dependent terrestrial and aquatic ecosystems (Lorentzen, 2009). Changes in management may also relate to the abandonment of many first generation or species-specific crops in favour of more efficient species or methods to produce alternative fuels. The current pursuit of biofuels following existing technologies is acknowledged as the 'first wave' and is likely to increase in efficiency as technology evolves (Mathews, 2008). The possibility of existing production methods becoming redundant is therefore a reality and the abandonment of crops could act as an invasion foci, for example, if left unmanaged.

2.5. Measuring the impact of biofuels

To understand the potential biodiversity implications of introducing and cultivating alien plant species for biofuel production in South Africa one needs to consider the impact as the product of the extent, abundance and local-scale effect (Firbank, 2008), as summarized in the equation: $I = R \times A \times E.$ (Parker *et al.*, 1999). Impact (*I*) is hereby described as the product of the (potential) geographical range of the introduced/invasive species (*R*), the (potential) abundance or density (*A*), and the effect (*E*) per individual or per biomass unit of the invader (Parker et al., 1999). Although originally proposed for quantifying the impact of invasive species (Parker *et al.*, 1999) (see also Richardson and van Wilgen 2004) this approach allows us to identify the individual and combined dimensions of areas of conflict between agricultural expansion and biodiversity conservation and also the classify production scenarios according to various scales of impact (Figure 2.2).

2.5.1. Geographical range / land- use and conservation

It is difficult to predict the likely geographical footprint of the biofuels industry in South Africa, for the reasons discussed above. However, the strategy outlines that government support is currently restricted to the rural areas of the Eastern Cape, mainly because of the need for social upliftment in that region (Funke *et al.*, 2009), with refineries planned for the towns of East London and Coega, near Port Elizabeth (http://www.asgisa-ec.co.za). The optimal size for different kinds of biomass processing plants have yet to determined but they are expected to depend upon the nature of biomass processed and the kind of processes employed (Wright & Brown, 2007). According to a recent study (Von Maltitz & Brent, 2008), utilizing roughly 10% of an available 3 million ha in the Eastern Cape should be sufficient to meet the 2% blending ratio based on average yields of sugar cane and *Jatropha curcas* (currently banned from further planting in South Africa but nonetheless frequently used in projections). These land estimates, however, ignore the possibilities of growing biomass for the export market (e.g. www.Phytoenergy.org) driven by considerable demand from overseas markets for biodiesel and biomass production in developing and emerging countries such as South Africa (Dauvergne & Neville, 2009, Fischer *et al.*, 2009). Nor do they consider the role that private land holders could play in increasing the amount of land under biofuel production. What is clearly needed is a review of all stakeholder involvement and commitments to determine the likely direction that biofuels will be headed. This will in turn refine scenario projections as well as provide some understanding regarding land-use requirements.

2.5.2. Abundance and density

A much debated topic is the manner in which biofuel crops should be cultivated (Tilman *et al.*, 2009, Von Maltitz & Brent, 2008). Firstly, the cultivation system relates to issues of scale and the size of area to be planted. The implementation of large-scale plantations or numerous small-scale rural outgrowing schemes is influenced by the need to deliver a large and reliable supply of new agricultural feedstock to nearby refineries (Wright & Brown, 2007). The strategy is pushing for small growers as the main producers. The choice of feedstock will also affect the nature of planting (Worster & Mundt, 2007). For example, sugar cane is often planted in monocultures, often over large areas; sugar beet can be used as a rotational crop; *Jatropha curcas* (or other perennial trees and shrubs) can be planted as hedges in an agroforestry system or in monocultures (Achten *et al.*, 2007, Cheesman, 2004). These decisions may be influenced by the land owners themselves, their willingness to buy into biofuel schemes, and the different approaches to land management such as the land-sparing versus various wildlife-friendly farming approaches (Koh *et al.*, 2009).

Such decisions will affect landscape heterogeneity where large-scale plantings may act to create a more uniform landscape compared to small-scale schemes that may create a more diverse setting and increase fragmentation (Firbank, 2008). In most instances, land will have to be released from its current land-use (Sala *et al.*, 2009) and this could potentially influence the structure of the farm at the lowest level of organisation.

From the perspective of problems with invasive biofuel species, the layout of biofuel planting will play a significant role in the dispersal of propagules to new environments. For example, numerous small scale plantings (with a large edge to total area ratio) provide favourable conditions to initiate invasions (numerous propagule-source foci, with a large area of contact with potentially invasible habitat).

2.5.3. Effects of individual species

The effect of biofuel crops on ecosystems and biodiversity depends to a large extent on functional attributes of the species. Whether native or introduced, the ability they have to utilise resources, add resources, promote or suppress disturbance regimes is important in determining the impact of an individual species (Richardson & van Wilgen, 2004).

Generally there is a lack of quantitative studies to determine the potential effects of biofuel species on ecosystems (Koh, 2007a, Sala *et al.*, 2009). It is uncertain to what extent, if any, these species are capable of altering important processes within the receiving environment. Much dubious information is distributed in this regard, for example proclaiming that jatropha has the capacity to improve or “reclaim” degraded areas by “improving” soil quality and reduce erosion via the input of organic materials (Ogunwole *et al.*, 2008). What is needed is an objective framework for assessing the effect and the resulting biodiversity impacts from specific biofuel cultivation systems to evaluate future risks (Butler *et al.*, 2009). Modifying the Biodiversity Intactness Index (Scholes & Biggs, 2005) or developing a case-by–case, evidenced-based approach to fully determine the impact of biofuel feedstocks on recipient communities are options that need to be explored (Barney & DiTomaso, 2008).

In most instances, there are likely to be considerable impacts on biodiversity during the initial set-up phase for biofuel plantations (Achten *et al.*, 2008). The type and extent of such impacts will depend on the ecosystem and on the previous land-use (Achten *et al.*, 2007). Greatest changes are likely where relatively natural areas are converted to monocultures of biofuel species. The greater the difference in overall structure between the biofuel plantation and the original (natural) vegetation at a given site, the greater the likely overall impact on ecosystem functioning, services, and biodiversity (Gaertner *et al.*, 2009). Where possible, impacts should be limited to the area undergoing cropping, thereby minimising disturbances beyond the farmed area. Possible ways of minimising effects of individual species include: i) avoiding gene flow to wild relatives in centres of diversity; ii) preventing invasion by the crop into other habitats; iii) avoiding degradation of sensitive habitats and a loss of species within local landscapes; iv) not increasing the risk of loss of primary habitat (Firbank, 2008).

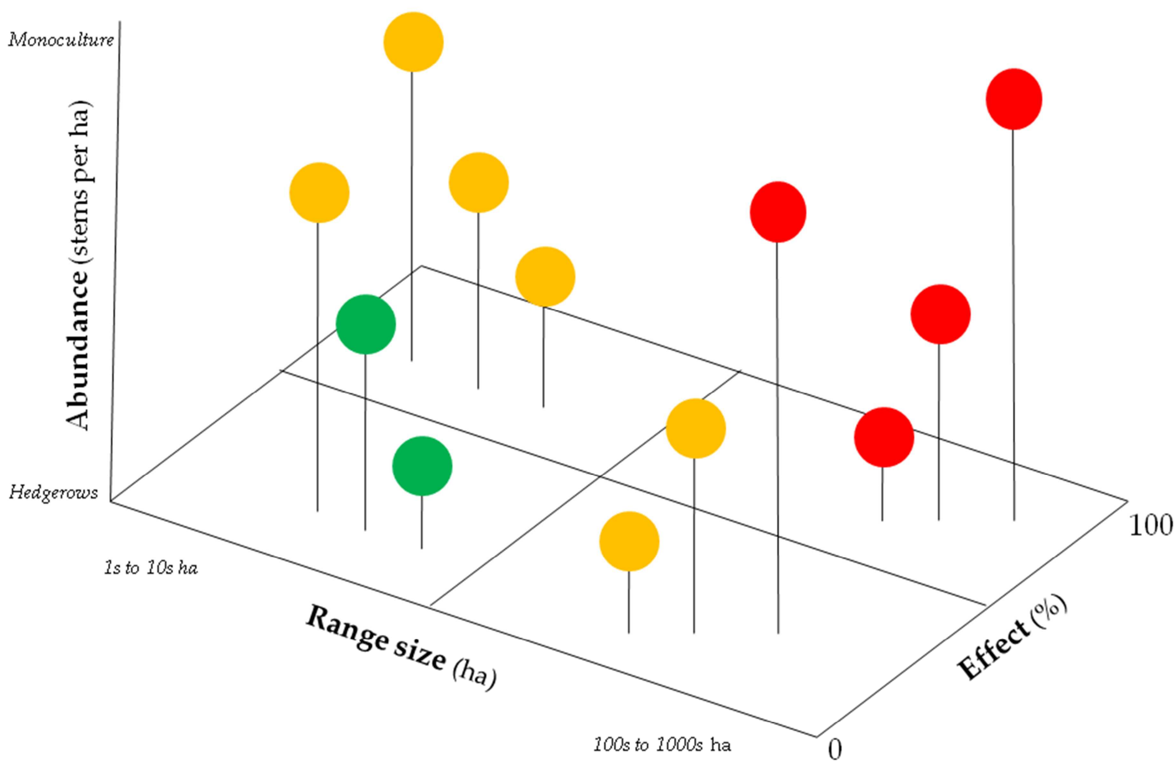

Figure 2.2: Applying the impact framework ($I = R \times A \times E$) to a biofuel classification system. Impact (I) is defined as the product of the (potential) geographical range (R), the (potential) abundance or density (A) and the effect (E) of an individual species or the measurable impacts at the smallest spatial scale. A range of scenarios can be depicted, depending on the range, abundance and effect of biofuel species. Together these depict the overall biofuel footprint, the role of planting configurations and effects at the local scale. The diagram incorporates the role of small-scale growers and large-scale commercial plantations at various abundances (e.g. from hedgerows to larger scale monocultures). The diagram also allows for various types of feedstocks as depicted by the per capita (local-scale) impacts on the receiving environment. The filled circles depict the overall impact (green = low; yellow = medium; red = high).

2.6. Industry as a pathway for the introduction and dissemination of invasive alien plant species

Non-native plant species are widely used for commercial forestry, agro-forestry, agriculture and horticulture in South Africa (Richardson *et al.*, 2003). Although economically important, wide-scale plantings have had considerable negative impacts on biodiversity and ecosystem services (Richardson *et al.*, 2003, van Wilgen *et al.*, 2008). Escapes from plantations (e.g. wattles, pines and eucalypts) have become important invasive species in adjacent landscapes (Richardson, 1998). Many of the problems we face today with invasive trees are the result of plantings incentivised by government schemes focusing on short-term economic or social incentives (Louw, 2004). It is estimated that 10 million ha of South Africa has been invaded to some degree and if condensed to adjust the cover to 100%, then this amounts to 1.7 million ha which is greater than the extent of commercial forestry totalling 1.5 million in 1996/7 (Le Maitre & Chapman, 2000).

Despite the contribution of alien trees and shrubs to economies and livelihoods, there are many unexpected consequences and subsequent costs, both environmental and economic, which often outweigh the benefits of introduction (Le Maitre *et al.*, 2002, Richardson, 1998). Nevertheless the potential economic gains may be too great to prevent the widespread introduction of alien species for biofuel purposes (Hill *et al.*, 2006).

Biofuel production using alien plant species provides an additional pathway whereby invasive species may be disseminated over large distances (Wilson *et al.*, 2009). A recent survey of potential biofuel feedstock species proposed for cultivation in Hawaii revealed that 70% of these have a high risk of becoming invasive (Buddenhagen *et al.*, 2009). The reasons for choosing alien plants over indigenous species for commercial forestry has been discussed by Richardson *et al.* (2008), and the situation is similar for the selection for biofuel species whereby species: a) need to grow fast with minimum tending; b) are easy to manage; c) produce large quantities of seeds/biomass; d) are marketable and profitable (Barney & DiTomaso, 2008). Many of the dedicated energy crops will be non-native to the region of planting and will probably be introduced multiple times as new varieties are developed for improved yield or resistance to pests and diseases (with propagules probably originating from different regions), further compounding the risk of future invasion (van Wilgen & Richardson, 2009). The risk of private investors looking to profit from overseas demand for biomass could see many plant species entering the country before effective legislation and planning guidelines are in place.

2.7. Existing measures for control

International regulatory bodies such as the Forest Stewardship Council (FSC) and environmental management systems such as the International Standards Organisation 14001 have made it possible to mainstream biodiversity issues into the everyday management of South African plantations (Louw, 2004, Louw, 2006). For example South Africa has ninth largest FSC certified area in the world, as well as the largest area of certified exotic plantations (Louw, 2006). Despite these environmental and major economic contributions, plantation forestry has been accused of impacting negatively on biodiversity and water resources, and contributing significantly to the current invasive species problem (Richardson, 1998, van Wilgen & Richardson, 2009). As far as we know, no similar environmental management systems are in place for the biofuels industry, although there are plans to develop such systems (see Roundtable on Sustainable Biofuels; http://cgse.epfl.ch/page65660.html), their importance in informing policy, which is essential for mitigating biodiversity loss (Diaz *et al.*, 2006), cannot be overstated.

The spatial scale of the potential biofuel industry could be its single biggest threat and national level priorities could affect regional and local level commitments to biodiversity plans. A major aim for sustainable biofuel production to reduce impacts to biodiversity can be achieved from the outset by following appropriate site selection criteria and efficient land-use planning. Whereas previously the use of suitability mapping (combining species requirements with soil and climate variables) was sufficient in industries such as forestry, current approaches to conservation and natural resource management require that operational, social and environmental factors be considered for sustainable practises to be recognised (Richardson *et al.*, 2009). In the South African context this means balancing development goals with conservation concerns while maintaining large investor interest and opportunities. The importance of available mapping data in determining potential conflict and "no go" areas (which recognises the importance of maintaining ecosystem function and connectivity) is crucial in this regards.

According to Richardson *et al.* (2003) the South African laws that regulate invasive alien organisms could, until recently, only be used indirectly to tackle invasive species issues, as the focus of national controls has been on the protection and conservation of natural and agricultural resources (e.g. Conservation of Agricultural Resources Act (CARA); National Water Act). The promulgation of the draft National Environmental Management: Biodiversity Act (Act 10 of 2004) (NEMBA) provides a more focused and direct approach for dealing with potential impacts to biodiversity and invasive species and it is anticipated that these will include regulations pertaining to biofuel production. Also, there is currently no record of the number of new species being introduced into the country (van Wilgen & Richardson, 2009) and the lack of an adequate screening and risk-assessment system is likely to allow many additional invasive plants to be introduced and planted.

Given the extent of the current invasive plant problem and the measures put in place to control existing alien species (van Wilgen *et al.*, 1998), South Africa can ill afford to allow the widespread dissemination of additional potential invaders. To this end, the identification of emerging invaders (Nel *et al.*, 2004) and the establishment of an Early Detection Rapid Response programme are testament to the importance attached to reducing future threats (www.sanbi.org). However it should be recognised that legislation alone will not be enough to prevent the introduction of harmful species. Building relationships with key stakeholders is crucial for reducing the use of potentially invasive plants and for reducing negative impacts. Tools, such as weed risk assessment protocols, are available for assessing the potential of introduced species becoming invasive. However, our ability to conduct risk assessments is affected by the multitude of interactions that can occur before negative impacts are recognized but nonetheless, they are useful for flagging potential hazards based on species biology, ecology, climatic requirements, history and biogeography in relation to the target region (Barney & DiTomaso, 2008). These challenges could be exacerbated by climate change as alterations to species distribution patterns could see new areas being invaded, as well as new species becoming invasive (Richardson & van Wilgen, 2004). Furthermore, the calculation of the risks involved becomes more challenging as the interactions between alien and native biota become increasingly difficult to predict (Hellmann *et al.*, 2008).

If the planting of dedicated energy crops proceeds and given institutional limitations, minimising biodiversity impacts will be best addressed by implementing effective farm design and managing crops to prevent invasion into surrounding areas. The co-introduction of appropriate biocontrol species to reduce propagule pressure should be considered from the outset, and planning in this regard should commence before bioenergy programmes are implemented.

2.8. Finding a balance

The delay in implementing South Africa's biofuels strategy gives us an opportunity to carefully consider the full range of potential impacts posed by this emerging industry, and to plan accordingly. The impacts and threats posed by this industry can be markedly exacerbated by the various technological aspects of biofuel production. The selection of feedstock options, the intensity of planting, and the eventual geographic distribution of plantings across South Africa are crucial factors that will influence the impact of the industry on biodiversity. As technology and plant breeding advance we could see multiple new species or varieties of species emerging as potential biofuel feedstocks. Therefore, what is needed is a framework in which existing and future feedstocks are scrutinized to carefully evaluate the risks, costs and benefits. By considering possible development scenarios we are better placed to move beyond reactive responses towards the integration of sound guidelines, policies and legislation to ensure sustainability and accountability from the outset. It is also important that short-term incentives to promote the biofuels industry do not overlook the possibility that changing technologies could result in crop abandonment or downscaling in the near future.

Despite this call for a cautionary approach, the urgent and escalating need for rural upliftment and poverty alleviation may tip the balance in favour of increasing development mechanisms such as biofuels expansion, reducing environmental concerns to a lower level. Globally the biofuels industry is expected to undergo rapid growth and

South Africa should be well positioned to accommodate investments in this area. Policy is needed that will anticipate future developments and encourage projects that can contribute positively to rural development as well as reducing greenhouse gas emissions and preventing biodiversity loss. Although the issue remains complex, the way forward requires collaboration between disciplines and the transfer of lessons learnt from similar industries to develop appropriate solutions to avoid unforeseen problems.

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Chapter 3: Anticipating potential biodiversity conflicts for future biofuel crops in South Africa: incorporating spatial filters with species distribution models[3]

3.1. Abstract

Liquid biofuel production will likely have its greatest impact through the large-scale changes to land-use that will be required to meet the production of this energy source. In this study, we develop a framework which integrates species distribution models, land cover, land capability and various biodiversity conservation data to identify natural areas with (1) a potentially high risk of transformation for biofuel production and (2) potential biodiversity conflicts. The framework was tested in the Eastern Cape of South Africa, a region which has been earmarked for the cultivation of biofuels. We expressly highlight the importance of biodiversity conservation data that enhances the Protected Area Network to limit potential losses by comparing the overlap of areas likely to become cultivated with (1) protected areas; (2) biodiversity hotspots not currently protected; and (3) "ecological corridors" (areas deemed important for the migration of species and linkages between important biodiversity areas). Results indicate that the introduction of spatial filters reduced land available for biofuels from 54% to 45% of the Eastern Cape. Prioritising biodiversity conservation by including all biodiversity scenarios reduced available land to 15% of the Eastern Cape. The assumption that agriculturally marginal land offers a unique opportunity to be converted to biofuel crops does not consider the biodiversity value attached to these areas. We highlight that decisions relating to large-scale transformation and changes in land cover need to take account of broader ecological processes. Determining the spatial extent of threats to biodiversity facilitates the analysis of spatial conflict. This paper demonstrates a proactive approach for anticipating likely habitat transformation and provides an objective means of mitigating potential conflict with existing land-use and biodiversity.

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Keywords: Bioenergy crops, Land suitability, Biodiversity, Spatial analysis, MaxEnt, Conflict, Agricultural land, Spatial filters

3.2. Introduction

Almost all scenarios for energy provision into the future include some focus on the emergence of a bioeconomy that includes large-scale bioenergy and biofuel production that offers lower greenhouse gas emissions than fossil fuels (Alkemade *et al.*, 2009, Slade *et al.*, 2011, Tilman *et al.*, 2009). There is a strong focus on bioenergy crops that can be grown on lands that will not directly compete with existing agricultural resources. Plant biomass, including traditional wood use, is currently the largest contributor to renewable energy (Tollefson, 2011). Projections indicate increasing demand for biomass fuel sources which are seen as crucial for a low-carbon future (Fischer *et al.*, 2009). The emergence of this new economic sector will entail radical and extensive changes in land-use and land cover (Wiens *et al.*, 2011). To help meet this demand, dedicated energy-crop cultivation is expected to follow large-scale and diversified practises similar to that of agriculture and forestry (Firbank, 2008, Koh *et al.*, 2009, Richardson & Blanchard, 2011). However, regions with suitable soil and climatic conditions which are currently considered marginal for conventional agriculture are likely to be targeted as potential production areas (Hoogwijk *et al.*, 2003, Wicke *et al.*, 2011). This potential increase in land conversion is likely to have severe consequences for biodiversity (Evans *et al.*, 2010, Wilcove *et al.*, 2000), as a wider range of land types can be brought into production when compared to conventional agricultural areas (Beringer *et al.*, 2011, Field *et al.*, 2007, Righelato & Spracklen, 2007). One of the challenges is to find suitable land to grow bioenergy crops in a manner that does not threaten biodiversity.

Among the innovative ways of selecting suitable land for bioenergy are methods that involve spatial planning (Li *et al.*, 2012). To avoid biodiversity losses the designation of biodiversity areas have been linked to protected areas or areas of high biodiversity conservation value. However, judging from recent literature, there is little consensus as to which biodiversity information should be included. For example, Beringer *et al.* (2011) rely on the overlapping of global biodiversity datasets to inform land-use restrictions. More importantly, Wicke *et al.* (2011) highlights the fact the biodiversity data is under-represented for some regions within global datasets. In this paper we aim to illustrate that assumptions regarding areas of biodiversity importance are crucial for identifying areas that are suitable for biofuel production. Despite the many examples of innovative frameworks adopting a spatial approach to anticipate and reduce land-use conflicts (Nelson *et al.*, 2009, O' Farrell *et al.*, 2012, Schweers *et al.*, 2011, Stoms *et al.*, 2011), none of these have focused solely on biodiversity and the value of data availability to the overall impact analysis.

Attempts at estimating the extent to which biofuels can contribute to global energy supplies has produced informative global estimates that include the spatial distribution of potential biofuel producing areas (Fischer *et al.*, 2007, Smeets *et al.*, 2004). To accomplish this either mechanistic models have been calibrated with established crop species or broad-scale vegetation models have been adapted to indicate areas with the greatest potential for energy production (Beringer *et al.*, 2011, Hoogwijk *et al.*, 2005, Lapola *et al.*, 2010, Smeets *et al.*, 2004, van Vuuren *et al.*, 2009). The focus of this work has often been at a global scale, typically overestimating potential biomass supply, returning estimates regarded as being in the upper range of biomass potentials (Beringer *et al.*, 2011, Lapola *et al.*, 2009, Slade *et al.*, 2011, van Vuuren *et al.*, 2009). The need to generalise model parameters stem from the large pool of potential energy crops for which little physiological information exists making the individual calibration of these models difficult (Lapola *et al.*, 2009). This is often addressed as a limitation of mechanistic models (Estes *et al.*, 2013, Fischer *et al.*, 2010b, Smith *et al.*, 2010).

The recent comparison of mechanistic and empirical models has positioned the latter as useful tool to determine potential distribution of certain agricultural species (Estes *et al.*, 2013). In particular, current species distribution modelling (SDM) techniques that rely on presence-only records have been shown to provide a useful screening tool to determine suitable climatic environments for potential dedicated energy crops (Evans *et al.*, 2010). The recent use of SDMs in determining suitable areas for biofuel feedstock production demonstrates the potential for estimating the broad climatic suitability for species with limited known physiological data (Barney & DiTomaso, 2011, Evans *et al.*, 2010, Trabucco *et al.*, 2010). For example, the modelling tool MaxEnt has been shown to perform well when compared with other SDMs (Edgerton, 2009, Elith *et al.*, 2006, Elith *et al.*, 2011, Evans *et al.*, 2010, Phillips *et al.*, 2006) and more recently mechanistic models themselves (Estes *et al.*, 2013). Since many countries are seeking to adopt and establish renewable energy strategies, the matching of suitable feedstocks to available areas is likely to become increasingly prominent in the literature. SDMs may therefore have the potential to act as a first-cut analysis to determine the broad climatic suitability of dedicated energy crops that rely on a rain-fed water supply. Dedicated energy crops are a potential solution to the challenge of producing sufficient biomass for biofuel production, without competing for similar resources or affecting the pricing and availability of food (Fischer *et al.*, 2009).

To fully address potential impacts of biofuel production on biodiversity (Barney & DiTomaso, 2011, Dauber *et al.*, 2010, Groom *et al.*, 2008, Wiens *et al.*, 2011) there is a need to include limiting factors which act as spatial filters that ultimately constrain the location of bioenergy cultivation in the landscape (Beringer *et al.*, 2011). However, the quality of information used as limiting factors could potentially underestimate future impacts (Smith *et al.*, 2010, Tilman *et al.*, 2009). We focus on biodiversity as an example of one such spatial filter that has important implications for limiting potential future land-uses (Beringer *et al.*, 2011, Schweers *et al.*, 2011, Slade *et al.*, 2011, van Vuuren *et al.*, 2009, Wicke *et al.*, 2011). There are multiple biodiversity datasets available, often generated at global scales, and there is little consensus on which datasets to include in modelling scenarios (Beringer *et al.*, 2011, Brooks *et al.*, 2006). Consequently, biodiversity is usually accounted for through the identification and exclusion of formal protected areas. Although this can avoid critical biodiversity losses, the question of whether this approach is adequate for biofuel production has not yet been addressed in the literature. Assessing the vulnerability of untransformed land that has no formal protection, yet is easily accessible, is a worthy conservation objective (Reyers, 2004, Wessels *et al.*, 2000).

Although protected area networks aim to safeguard existing biodiversity for future generations, the location and configuration of these areas often arose haphazardly, rather than following decisions based on rigorous science (Wicke *et al.*, 2011). Conservation areas are often in areas with poor agricultural potential. Consequently, tradeoffs with agriculture or other potential land-uses have mostly been avoided until now (Gabriel *et al.*, 2009). Whilst these areas may be relatively high in diversity, they may not adequately conserve the required regional taxa or important ecosystem functions that drive evolutionary change in landscapes (Berliner & Desmet, 2007). For example, in South Africa, the need to increase the Protected Area Network has resulted in the identification of additional areas needed to meet conservation goals (Government of South Africa, 2008). However, the management and procurement costs limit the total inclusion of all suitable areas (Gallo *et al.*, 2009). To avoid future tradeoffs with food and feed production, biofuel production strategies have typically highlighted these marginal areas as key production sites (Romijn, 2011, Wicke *et al.*, 2011). Research interest in dedicated energy crops that may fill this potential niche is increasing, increasing the potential for future land transformation in these areas. Where conventional biofuel crops may be required to occupy arable areas, the diversification of the industry may need marginal areas to be brought into production as well. This provides an excellent opportunity to test a framework regarding biodiversity as a spatial limiting factor. Given that land-use has a severe impact on biodiversity integrity, it would be useful to understand potential impacts that biofuels, as a land-use option, present (O' Connor & Kuyler, 2009).

In this paper, we present a framework that combines the outputs of global scale species distribution models with a localised land suitability analysis, to identify areas with a potentially high risk of transformation for biofuel production. To demonstrate the effect of biodiversity as a spatial filter for bioenergy suitability we use the Eastern Cape province of South Africa. The framework aims to simplify the complex issues surrounding land-use planning that are likely to be typical for developing world scenarios. We use biofuel production as one proxy for agricultural expansion which is a known driver of habitat loss. Additional spatial layers and socio-economic variables can be added to the framework to further increase the resolution of conflict between biodiversity and biofuel production. More specifically, we illustrate that spatial filters could prove useful in model predictions which are aggregated on broad scale climate data. These provide a much more realistic estimate of available land and potential conflict. This proactive approach anticipates likely habitat transformation and provides an objective way of mitigating potential conflict with existing land-use and biodiversity (Lindborg *et al.*, 2009, Wessels *et al.*, 2003).

In summary, our objectives were to: 1) determine the potential spatial extent of land available; 2) identify potential biofuel crops based on species distribution models; and

3) test a biodiversity-impact framework aimed at highlighting the importance of inclusive biodiversity data.

3.3. Material and methods

3.3.1. Study area

The Eastern Cape province of South Africa (Figure 3.1) was chosen as our study area because it is earmarked to undergo large-scale changes in land-use as a result of national developmental policies, which include possible biofuel production (Berliner & Desmet, 2007, Blanchard *et al.*, 2011). This region is also recognised as a biodiversity hotspot that is threatened by a long history of cultural and politically enforced land-use practices (Critical Ecosystem Partnership Fund, 2010, Evans *et al.*, 1997). As a result, the dichotomy of development pressures and conservation are prevalent in this region.

South Africa's biofuel policy forms part of its Renewable Energy portfolio which includes wind and solar energy production (Department of Minerals and Energy, 2003). Concurrently, biofuel production is meant to contribute to enterprise development and ongoing job creation programmes. Biofuels, which are as yet an untested industry in South Africa, are therefore likely to compete with alternative land-use options for reducing poverty. The expansion of conventional agricultural practices or increased livestock farming is among alternative potential land-use options. However, the Government has declared support for biofuel production within the former "homeland" areas of South Africa, to facilitate job creation and the improvement to the socio-economic status of informal, small-scale or enterprising farmers in the region (Department of Minerals and Energy, 2003, Department of Minerals and Energy, 2007). Ongoing research into biofuel viability is currently underway in the Eastern Cape with projects currently in the establishment phase (Musango *et al.*, 2010). A stable market for biofuels would not exclude the commercial farming sector, which has the capacity to increase production of candidate crops should prices allow for it (Von Maltitz & Brent, 2008). The expected potential for agriculture, forestry and agro-processing initiatives in the former homeland areas are considered to be large, but currently unrealised (Lynd *et al.*, 2003). Reasons include a strong traditional focus on livestock farming and a land tenure system based on tribal or communal land ownerships (Hoffman & Ashwell, 2001). The current trend of rural de-agrarianisation may also contribute to the recent increase in abandoned land, as well the slow uptake of new farming activities (Andrew & Fox, 2004, Davis *et al.*, 2008). Both commercial and subsistence farming are practised in the Eastern Cape, with the latter achieving significantly lower yields in some areas (Shackleton *et al.*, 2001). It is anticipated that biofuel production could supply the needed investments to increase yields in some regions through the supply of much needed technical knowledge and infrastructural investments within former homeland areas (Biggs & Scholes, 2002).

The Eastern Cape is renowned for its biological diversity containing five of the seven biomes in South Africa, and includes the Maputaland-Pondoland-Albany biodiversity hotspot (Critical Ecosystem Partnership Fund, 2010, Driver *et al.*, 2012, Mucina & Rutherford, 2006). Large areas of grassland and savanna ecosystems are strongly underrepresented in the province's formal protected area network and are at risk of current and future land transformation (Driver *et al.*, 2012, O' Connor & Kuyler, 2009). The lack of formal protection and extensive land-use practices have led to some vegetation types in the grassland biome being proclaimed vulnerable or critically endangered (Mucina & Rutherford, 2006). The expansion of forestry, agriculture and urbanisation of rural areas are among the key threats to biodiversity. Furthermore overgrazing, alien plants and poor management of agricultural lands have resulted in degraded and transformed areas (Evans *et al.*, 1997, Hoffman & Ashwell, 2001). Despite this only 5% of the area is protected within 190 nationally declared Protected Areas (0.69 Mha) and 79 informal conservation areas (0.25 Mha) that gives responsibility of conservation to landowners operating private game or nature reserves.

The dynamic setting of the Eastern Cape provides a unique opportunity to validate a conceptual framework taking advantage of a large biodiversity network and the potential impacts of land-use change represented by biofuel production. The inclusion of biofuels as a possible land-use option raises additional awareness of potential biodiversity threats. Species outlined in the biofuel strategy include traditional agricultural crops such as soya or canola, which are expected to be grown on fertile soils, to achieve maximum yields. In this study we model biofuel crops which are meant to be grown with fewer inputs than conventional agricultural crops. These species are considered suitable for degraded or marginal areas with the potential to offer greater benefits to farmers in such landscapes. Although there is much uncertainty regarding the viability of these crops (Achten *et al.*, 2010) or the willingness to cultivate such crops (Amigun *et al.*, 2011), the potential land resources may exist in Eastern Cape.

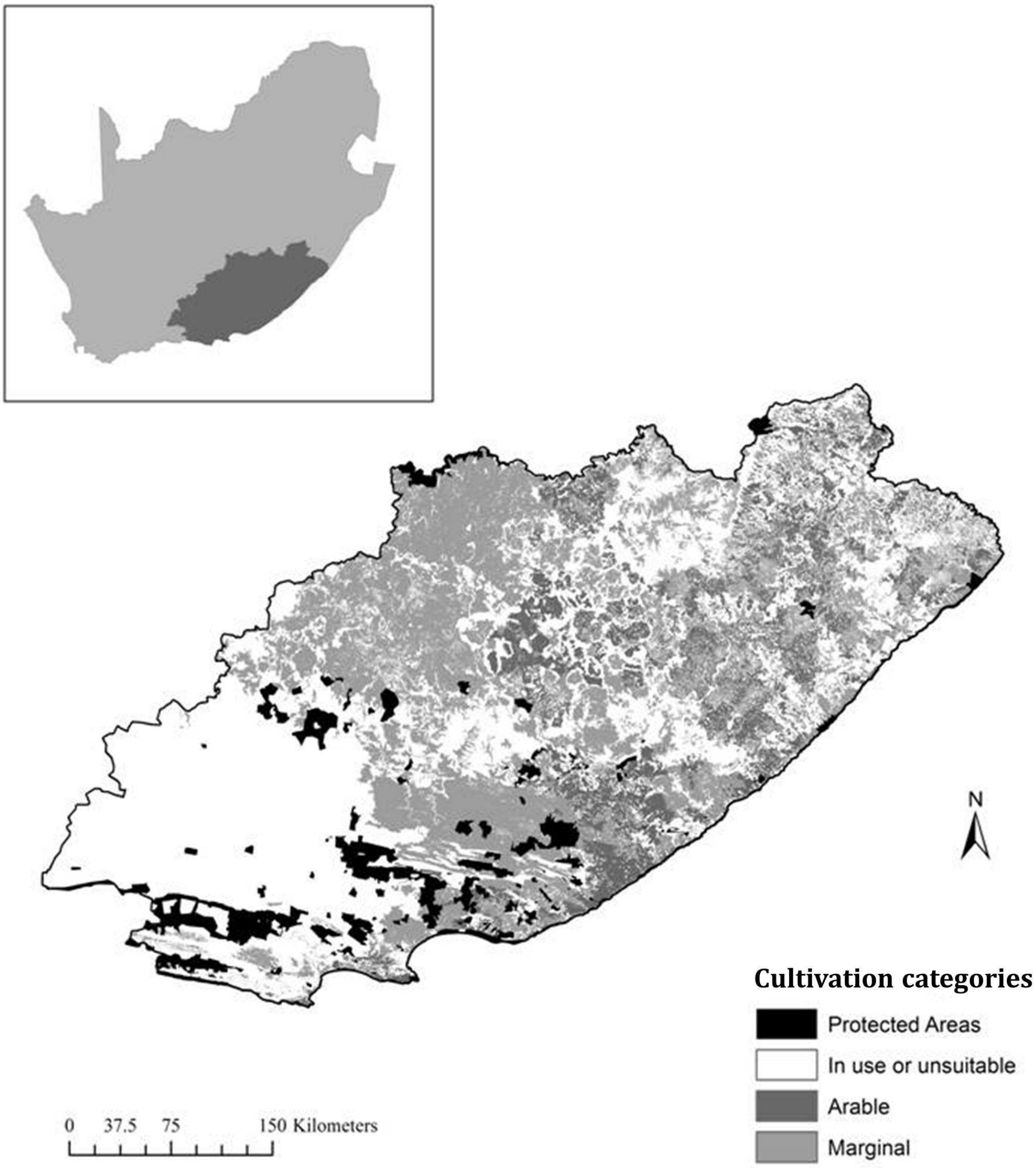

Figure 3.1: The location of the Eastern Cape Province, South Africa (inset), indicating broad categories of cultivation potential. Protected Areas (black) indicate locations of the formal and informal conservation network, which are automatically excluded from land availability assessments.

3.3.2. *Description of the modelling framework*

We propose the framework presented in Figure 3.2 which provides a schematic outline of the methodology used in this study. The framework builds on existing methodologies used to determine land availability (Fiorese & Guariso, 2010) and includes the use of species distribution models to provide a potential biofuel layer with which to investigate biodiversity conflicts. The framework also highlights the use of localised spatial filters to analyse conflict. Unfortunately, we are not able to capture the full complexity of land tenure and other socio-political issues in the region as explained above but rather focus on a limited set of issues. The framework presents a simplified approach to this complexity which has the capacity to incorporate more complexities should the need arise. We summarise these logical components of the framework in more detail below:

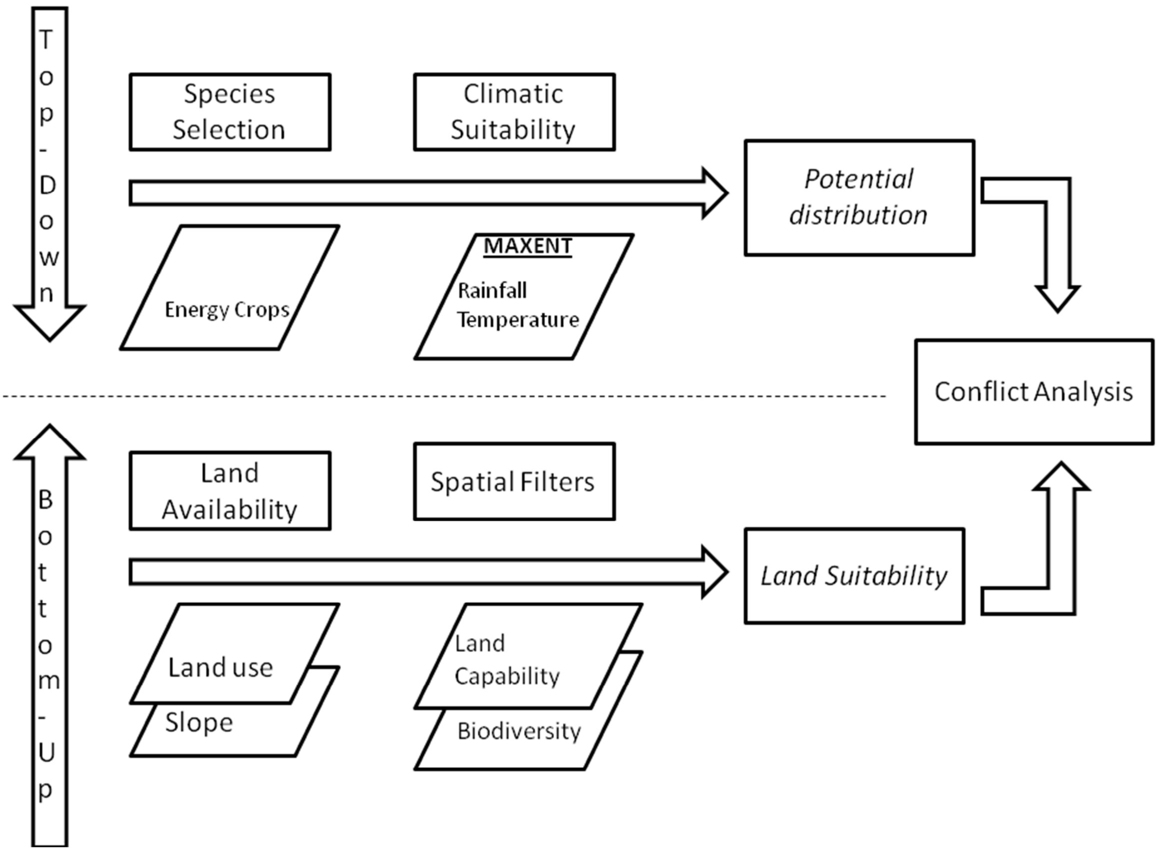

Figure 3.2: The methodological framework adopted for this analysis and the related databases.

3.3.3. Species selection and data preparation

South Africa's biofuel strategy aims to produce bioethanol and biodiesel but excludes the use of staple food crops, such as maize, for biofuel production. Recognised species are conventional agricultural crops like sugar cane, sugar beet, sunflower, canola and soya bean, intended for production on unutilised arable land (Von Maltitz & Brent, 2008). Our species choice therefore focuses on likely alternative energy crops based on international interest as gauged by a literature search on the ISI Web of Science. These species are anticipated not to compete with conventional agricultural crops for resources intended for food and feed production. The keywords, 'biofuel', 'biomass' and 'bioenergy', were used to determine the most common crop candidates as found in searches of articles, titles or abstracts. Characteristics that make some energy crops attractive as biofuel feedstocks include a wide environmental tolerance, rapid growth, ease of establishment, low water demand and the potential to generate a high biomass or prolific seed production. We included current plants listed as invasive in South Africa, as these may also provide a source for biomass production. The plants were: *Acacia mearnsii*, *Sorghum halepense* and *Arundo donax.* Suitable locations for selected biofuel species not currently cultivated in South Africa were modelled using MaxEnt ver. 3.3.3 (Phillips *et al.*, 2006). We followed a number of steps to minimise spatial sorting bias, known to inflate AUC values during model evaluation methods (Hijmans, 2012).To reduce the possibility of sampling bias, we used location records from many online global data sets to estimate the potential global range. The online databases used include: the Global Biodiversity Information Forum (GBIF, www.GBIF.org); the Australian Virtual Herbarium (AVH, www.ersa.edu.au/avh); The National Commission for Knowledge and Use of Biodiversity (CONABIO, www.conabio.gob.mx) and the Southern African Plant Invaders Atlas (SAPIA, www.agis.agric.za, (Henderson, 2007)). Downloaded data were screened for geo-referenced records only and where possible erroneous records and duplicate localities were removed from the dataset following analysis in a GIS (ARCGIS 9.3). To further reduce sampling bias, records were regularised to the 5-minute WorldClim environmental data, resulting in one record per grid cell using the ENMT Tools package version 1.3 (Warren & Seifert, 2011).

3.3.4. Modelling methodology and calibration

Our decision to use MaxEnt as our single species distribution model is based on the evidence that MaxEnt can model the relative suitability of a species (including some agricultural crops), to accurately predict the potential spatial distribution (Estes *et al.*, 2013, Evans *et al.*, 2010). MaxEnt determines the environmental requirements of a species by matching globally available temperature and rainfall variables to the closest empirical average of the species habitat provided (Phillips *et al.*, 2006). The outputs are indicated as relative suitability within the region modelled, indicative of the climatic suitability for a particular species. The full set of nineteen bioclimatic variables, downloaded from the WorldClim database (http://www.worldclim.org; (Hijmans *et al.*, 2005)), were used to train the models and to determine the most important environmental variables. The relative performance of each variable was firstly determined by MaxEnt by means of 'training gain', which is the improved predictability of MaxEnt based on the incorporation of a particular variable (Phillips *et al.*, 2006, Trabucco *et al.*, 2010). Following this we reduced the overall number of explanatory variables to a limited set of more significant and less correlated variables to increase the transferability of model results (moving from the realized to the fundamental niche). The use of correlated environmental variables can result in model overfitting (model being too constrained) which can be exacerbated in areas outside of the training range (Elith & Leathwick, 2009, Phillips *et al.*, 2006, Trabucco *et al.*, 2010). Important variables were selected following a correlation analysis using Pearson's correlation with a cut-off of >0.8 (Blach-Overgaard *et al.*, 2010). In addition to climate variables, we included soil variables obtained from the Harmonised World Soil Database (FAO, 2012), if it was shown to be important and provided a better model fit.

The area where MaxEnt draws climate samples from is known as the background; the choice of this area has a major influence on the outcome of the model (Elith *et al.*, 2011, Vanderwal *et al.*, 2009). We chose the global Köppen-Geiger climate classification system, as this provides a uniform background layer and is widely used to determine agronomic potential of plant species (Trabucco *et al.*, 2010, Webber *et al.*, 2011). The Köppen-Geiger classifications, as applied to the 5-minute resolution WorldClim global climatology (www.worldclim.org), were downloaded from the CliMond set of climate data products (www.climond.org, (Kriticos *et al.*, 2011)). Backgrounds were produced by intersecting occurrence records for each of the different biofuel species with the Köppen-Geiger polygon layers in a GIS (ARC-GIS 9.3). Following Webber *et al*. (2011), Köppen-Geiger polygons were included in the background if they contained one or more records of the biofuel species. This inclusive approach allows for the full ecological range of the species to be used. This reduces the need for extrapolation to areas unsampled that might cause the model to be ecologically questionable.

The modelling procedure followed that of Elith *et al.* (2011) using only hinge features with default regularization parameters. Final models were tested using 20% of the dataset whereas variation in the environmental variables was tested using 5-fold cross validation. Model outputs were tested for goodness of fit with training data using the threshold independent Area Under the receiver operating characteristics Curve (AUC), which provides a measure of model accuracy commonly used in predictive distribution models. Where a value of 0.5 indicates that the model is no better than random, a more accurate model value are >0.75 (Phillips & Dudík, 2008). As a measure of model suitability, threshold indicators were evaluated using Fischer's exact 1-tailed binomial test (see below) as applied to model prevalence and sensitivity to verify the model (Thompson *et al.*, 2011, Webber *et al.*, 2011). This method tests for the sensitivity of the model using the proportion of the model background estimated to be climatically suitable (Webber *et al.*, 2011).

3.3.5. Suitability

For the purpose of this study, thresholds were used to convert the continuous output of MaxEnt model predictions to indicate suitable and unsuitable areas. The choice of threshold affects the mapped results and could significantly affect perceived implications of environmental impacts of modelled biofuels. For example, increasing this threshold value has the negative effect of reducing the predicted suitable area as the criteria for suitability increases (Evans *et al.*, 2010). There is currently no dominant method for choosing a threshold value and current options are either based on subjective or objective methods depending on the research question (Liu *et al.*, 2005, Pearson, 2007). For example should the potential range of a species need to be calculated, an inclusive measure such as the lowest presence threshold (LPT) would be appropriate. This approach maximises sensitivity, whereby all presence points are included in the model prediction. If relative suitability was to be maximised, then we may opt for a higher threshold value or balancing presence point omissions and sensitivity. For this study, we choose threshold values that indicate suitable locations with a higher relative suitability, which we assumed to be a requirement for indicating agricultural potential. To illustrate uncertainty in determining suitability, suitable areas were calculated for threshold values associated with the LPT, cut-offs at 95% and 90% of presence points and where sensitivity equals sensitivity. The use of thresholds was evaluated using the binomial test (Pearson, 2007). More conservative threshold values exclude the lowest probability cells. Subsequently, all areas that fell below these threshold values were excluded from further analyses.

3.3.6. Spatial filter – Available and suitable land

Land availability was determined by current land-use patterns (derived from land-cover) and limited to include natural areas with a slope less than 16 degrees (Equation 3.1). Land-cover classes were re-classified in ARC-GIS 9.3 to represent either natural (reclassified as 1) or non-natural (reclassified as 0) habitats using the South African National Land Cover database (Fairbanks *et al.*, 2000). Slope, in degrees, was calculated using a 90 m digital elevation model and reclassified to represent terrain suitable for cultivation (i.e. <16 degrees reclassified as 1) or not suitable for cultivation (i.e. >16 degrees reclassified as 0). Excluding steep slopes retains areas which are suitable for conventional cultivation and plantation forestry offering lower production risks and costs (Fischer *et al.*, 2007). Land-cover classes representing potential food or production areas (rain-fed and irrigated croplands, forestry plantations) and areas totally unsuitable for biofuel production (water bodies, urban and mining areas) were excluded from further analysis.

Maximising the economic viability of biofuel production requires landscapes to have some potential for plant growth (Achten *et al.*, 2010). To determine land suitability, a measure of economic viability, we limited our analysis to likely agro-ecosystems using the Land Capability Classification for South Africa (Schoeman *et al.*, 2000) (Equation 3.2). Land capability class units act as a third spatial filter to indicate the technical potential of the available land as well as to identify current or future land transformation threats. Land capability classification identifies eight classes associated with decreasing levels of agricultural potential. Each class represents similar production potential and physical limitations (i.e. soils risk of erosion, physical terrain constraints and climate). Three classes were derived and reclassified in ARCGIC as the following, Arable (Class 1-4 reclassified as 1), Marginal (Class 5 and 6 reclassified as 2) and Excluded (Class 7 and 8 reclassified as 0).

The calculations were carried out using raster grids in ARC-GIS 9.3:

Availability$_i$ = Land-use x Slope **Equation 3.1**

Suitability$_i$ = Availability$_i$ x Land Capability$_i$ **Equation 3.2**

Where *i* is the grid cell that spatial filters such as land-use, slope and land capability are applied to derive an estimation of suitability, indicating natural areas with high potential for cultivation based on soil and land-use characteristics. These GIS data layers represent coded grid cells to determine the presence of favourable or unfavourable conditions to the location of suitable or conflict areas. The GIS layers indicate the spatial distribution of landscape properties and the areas of each category were determined using Spatial Analysis Tools in ARGIS and were reported in Million hectares (Mha).

Another form of land-use in the region is commercial livestock farming, carried out over large areas. Whilst potential livestock carrying capacities have been mapped in the Eastern Cape (Scholes, 1998), the locations of ranches are not available and we exclude this land-use from our analysis. However, accounting for this land-use will further reduce land availability.

3.3.7. Spatial filter - Biodiversity

South Africa has large tracts of untransformed land, much of it suitable for cultivation of crops or for some forms of forestry (Reyers, 2004). Our approach is based on the assumption that intact habitat is indicative of higher habitat quality, translating to greater ecosystem health. Any changes to land cover through cultivation, reduces the habitat quality and in turn results in biodiversity losses. Usually areas of high biodiversity, indicated by the location of protected areas, are excluded from land availability assessments.

We used three synergistic data sources for identifying and capturing biodiversity features: 1) the formal Protected Area network (PA), 2) the National Protected Area Expansion Strategy (NPAES) and 3) a region-based systematic conservation plan, The Eastern Cape Biodiversity Conservation Plan (ECBCP) (Berliner & Desmet, 2007). The data were extracted from an online database supplied by the South African National Biodiversity Institute online geographic information database (www.BGIS.co.za).These datasets provided the necessary information to produce three biodiversity scenarios (Table 3.1) used as spatial filters for biodiversity.

There is a recognised need to expand the existing network of protected areas in South Africa, so as to account for complementarity (being representative of distinctive features in the landscape), irreplaceability (a measure of conservation option lost in a landscape) and to allow for habitat shifts under future climate projections. The NPAES indicates areas of highest priority for future conservation needed to meet representative biodiversity targets as well as protect areas under future climate change (Government of South Africa, 2008). The ECBCP is based on the systematic conservation planning approach of identifying areas needed to maintain corridors and ecological processes (Driver *et al.*, 2005, Margules & Pressey, 2000). This plan identifies critical biodiversity areas and important *ecological corridors* (areas deemed important for migration and linkages between important biodiversity areas). For this analysis we defined *important biodiversity areas* by combining the critical biodiversity areas of the ECBCP with the NPAES to create a single biodiversity priority map.

3.3.8. Analysis of conflict

Two measures of threat status are shown 1) *Vulnerability* - determined as the total overlap of each biodiversity scenario with agricultural potential (Equation 3.3) and 2) *Conflict* - calculated as the spatial overlap of modelled suitability of energy crops with vulnerable areas (Equation 3.4). Each model was converted into a binary (0=feature absent, 1=feature present) surface layer and used to indicate positive interactions with vulnerable grid cells. All SDM outputs (derived from above) were re-sampled to the coarsest resolution used in the land availability assessment (i.e. 90m of the SRTM DEM). Model results provide a measure of suitability at the scale of the input variables, which in this case is 5 minute data. The assumption that all land within a suitable cell is available contributes to the overestimation of land availability (Evans *et al.*, 2010).

$$\textbf{Vulnerability}_{b} = \textbf{Suitability}_{i} \times \textbf{Biodiversity}_{b} \qquad \textbf{Equation 3.3}$$

$$\textbf{Conflict}_{species} = \textbf{SDM}_{output} \times \textbf{Vulnerability}_{b} \qquad \textbf{Equation 3.4}$$

Where *b* represents biodiversity scenario.

Table 3.1: Three biodiversity spatial filters used to indicate biodiversity conservation scenarios utilised in this analysis. All data were extracted from an online database (www.bgis.sanbi.org).

Biodiversity scenarios	Description of biodiversity layers
Protected area	Protected Areas are indicative of the minimum data available for biodiversity conservation. These layers indicate areas that are excluded from land availability assessments. In this assessment informal protected areas (private nature reserves, game farms) are included here.
Important biodiversity area	This scenario identifies areas of high biodiversity that occur outside of protected areas. Two biodiversity databases were used to compile this spatial filter, The National Protected Area Expansion Strategy (NPAES) and Critical Biodiversity Areas taken from the Eastern Cape Biodiversity Conservation Plan (ECBCP). These areas are not formally conserved, and have been identified to contain high biodiversity value.
Ecological corridor	Ecological corridors enhance the connectivity between important biodiversity areas and reduce vulnerability of intact patches in the landscape. These areas are known to contribute to the provision of ecosystem services.

3.4. Results

3.4.1. Model evaluation and prediction of suitability

The potential distributions of the nine biofuel species are presented in Figure 3.3. The MaxEnt models performed adequately, with AUC values ranging between 0.78 and 0.92 for training data, based on a 5-fold cross validation (Table 3.2). Perfect models produce an AUC value close to 1, whereas models with a value less than 0.5 are considered random. All models were statistically significant using the exact binomial test for the threshold values indicated (Table 3.2).

Matching plant species to novel climates requires careful consideration especially when training and prediction areas do not overlap. The multivariate environmental suitability surface (MESS) map is a feature included in MaxEnt that allows the user to identify areas where environmental variables fall outside the training range, thus indicating caution during model evaluation (Elith *et al.*, 2010). However, the modelled environmental variables for each species matched those within the Eastern Cape and were within accepted limitations according to the MESS maps.

Suitability maps were produced using the threshold model values associated with the LPT, 95%, 90% and where sensitivity was equal to specificity for display purposes. These values indicate an increasingly stricter threshold that can affect the area displayed as suitable or unsuitable. Increasing the threshold value for predictions of relative suitability results in a decrease in the area projected to be suitable (Figure 3.4). Values at the LPT incorporate all presence points resulting in large overlaps within the study region for all species. The species with the largest suitable climatic range within the Eastern Cape are locally present such as *Arundo donax*, *Acacia mearnsii* and *Sorghum halepense* (Table 3.2). These results are likely to be explained by the high percentage of presence points occurring in the region. Other species with international interest have among the smallest ranges such as *Camelina sativa* and *Panicum virgatum.*

Table 3.2: Summary statistics for nine biofuel species based on MaxEnt projections to the Eastern Cape. Suitability in millions of hectares (Mha) is indicated for four threshold values, namely: LPT (lowest minimum threshold), sensitivity at 95% and 90% of presence points and where sensitivity equals specificity. Model accuracy was assessed using the Area Under the receiver operating characteristics Curve (AUC).

				LPT		95%		90%		Equal sensitivity and specificity	
Fuel type	Species	AUC	Std dev.	Value	Area	Value	Area	Value	Area	Value	Area
Bioenergy	*Acacia mearnsii***	0.92	0.005	0.003	16.87	0.169	14.25	0.370	10.42	0.426	9.37
Ethanol	*Arundo donax***	0.91	0.006	0.004	16.87	0.092	16.87	0.224	16.76	0.374	14.97
Ethanol	*Beta vulgaris**	0.87	0.005	0.003	16.87	0.196	1.28	0.366	0.76	0.473	0.00
Biodiesel	*Camelina sativa*	0.90	0.005	0.009	16.87	0.102	1.64	0.219	0.13	0.423	0.00
Biodiesel	*Jatropha curcas***	0.78	0.034	0.005	15.96	0.103	4.71	0.162	3.45	0.343	1.64
Biodiesel	*Miscanthus sinensis*	0.90	0.018	0.014	14.33	0.100	0.69	0.185	0.16	0.257	0.02
Bioethanol	*Sorghum halepense***	0.80	0.004	0.010	16.87	0.159	16.86	0.277	14.72	0.481	1.00
Bioethanol	*Panicum virgatum*	0.81	0.007	0.013	16.70	0.147	1.92	0.311	0.01	0.480	0.00
Biodiesel	*Ricinus communis**	0.84	0.012	0.013	16.87	0.138	16.87	0.225	16.87	0.381	15.62

*present in South Africa

**declared an invasive alien plant in South Africa

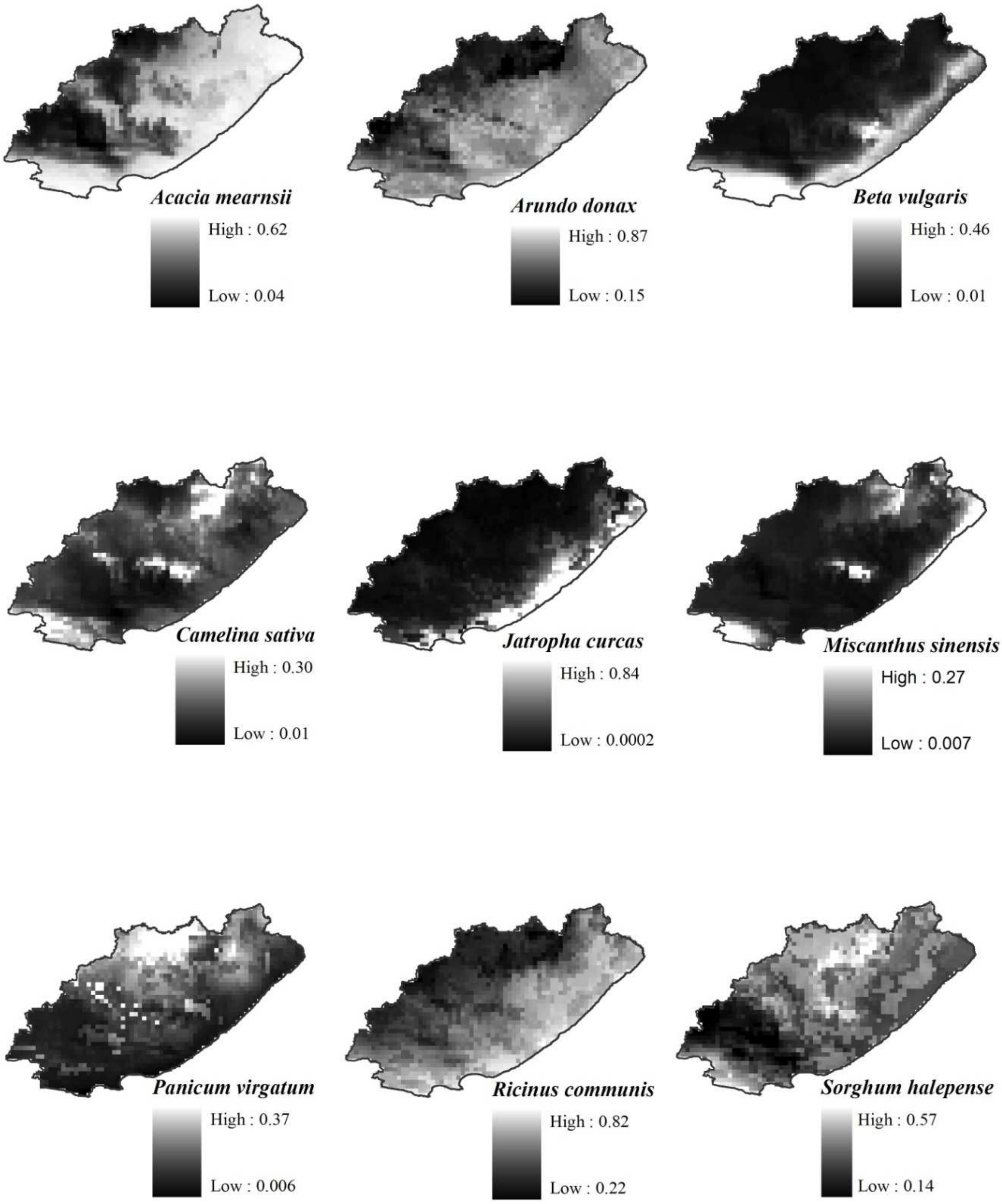

Figure 3.3: Suitability estimates for nine potential biofuel species modelled for the Eastern Cape province using the species distribution model MaxEnt. The figure is in grayscale and indicates areas of high suitability in white and low suitability in black.

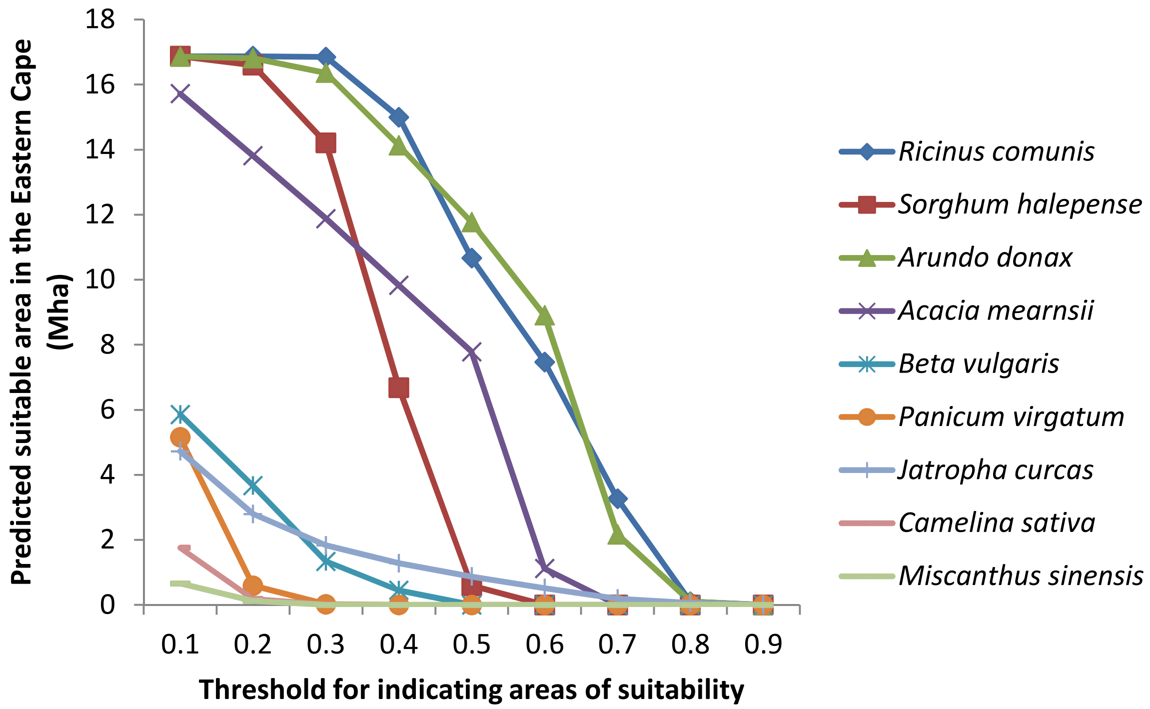

Figure 3.4: The effect of threshold choice on the predicted area (in millions of hectares) of nine biofuel species.

3.4.2. Land availability

A large portion of the study area is untransformed with natural areas accounting for ~82% of the province (Table 3.3). Of the remaining area, ~16% is transformed or degraded (Figure 3.1). Arable areas cover ~18% of the Eastern Cape, with ~5% currently in use following the selection criteria described (Figure 3.2). These arable areas are scattered throughout the eastern half of the province (Figure 3.1). Despite the perceived condition of marginal areas which covers ~38% of the Eastern Cape, ~40% of cultivation is indicated to occur here (Table 3.3). For this reason, we include marginal areas within the current analysis. Excluding steep slopes and accounting for the technical ability of the land reduced available land from ~54% to ~46% of the Eastern Cape province. The resulting spatial filter that can be applied to modelled outputs account for ~18% of arable land and ~41% of marginal land. The remaining area has been characterised as excluded, with limited potential for future land-use transformation.

Table 3.3: The total area and percentage of land-use occupied within land capability classes (Arable, Marginal and Excluded) in the Eastern Cape.

Land-use classes	Arable Mha (%)	Marginal Mha (%)	Excluded Mha (%)	Total Mha (%)
Forestry	0.06 (51.9)	0.02 (18.4)	0.04 (29.6)	0.12 (0.74)
Cultivation	0.32 (47.1)	0.28 (40.6)	0.09 (12.4)	0.69 (4.09)
Other	0.40 (13.4)	0.66 (22.2)	1.91 (64.3)	2.97 (17.6)
Natural*	2.32 (17.7)	5.39 (41.2)	5.36 (41.1)	13.1 (77.6)
Total	3.10 (18.4)	6.35 (37.7)	7.40 (43.9)	16.86 (100)

* as indicated in the National Land Cover Database 2000

3.4.3. Biodiversity scenarios

The three biodiversity spatial layers used to indicate conservation scenarios revealed sizeable differences to the overall area considered important for biodiversity conservation (Table 3.4). The majority of Protected Areas (including informal protected areas) are found in the south-western half of the region and account for ~6% of the province. These Protected Areas have low cultivation potential and are distributed across marginal and excluded areas. Important biodiversity areas, represented by merging the NPAES with Critical Biodiversity areas of the ECBCP, account for ~25% of the province. Approximately 39% of IBA's are considered either arable or marginal representing increased vulnerability to future land-use transformation. Recognised ecological corridors identify a further ~41% of the land area contributing to important functions needed for biodiversity conservation, approximately half of which are potentially vulnerable to future land-use transformation. Accounting for all biodiversity scenarios highlights ~72% of the Eastern Cape as contributing to biodiversity conservation, as compared to 5% if only Protected Areas were to be considered. Figure 3.5 shows the increasing vulnerability of suitable land as biodiversity scenarios are included in the land availability assessment. Should all biodiversity scenarios be accounted for in the suitability analysis then potential available land is reduced from 7.6 Mha to 2.6 Mha. The remaining arable or marginal areas have that no recognised biodiversity features account for ~15% of the province, of which marginal areas make up the largest proportion.

Table 3.4: The area and percentage overlap of biodiversity scenarios with land capability classes (Arable, Marginal and Excluded) in the Eastern Cape. Areas with no recorded biodiversity value are also indicated.

Biodiversity Scenarios	Arable Mha (%)	Marginal Mha (%)	Excluded Mha (%)	Sum Mha (%)
Protected Areas	0.04 (4.0)	0.23 (24.8)	0.66 (71.2)	0.93 (5.5)
Important Biodiversity areas	0.51 (12.0)	1.13 (26.8)	2.59 (61.9)	4.23 (25.1)
Ecological corridors	1.02 (14.8)	2.22 (32.3)	3.65 (52.9)	6.89 (40.9)
Total	**1.56 (12.9)**	**3.59 (29.8)**	**6.90 (57.3)**	**12.05 (71.5)**
Non Biodiversity Areas	0.75 (15.6)	1.80 (37.4)	2.26 (46.9)	4.81 (28.6)
Total all	**2.32 (13.7)**	**5.39 (31.9)**	**9.16 (54.3)**	**16.86 (100)**

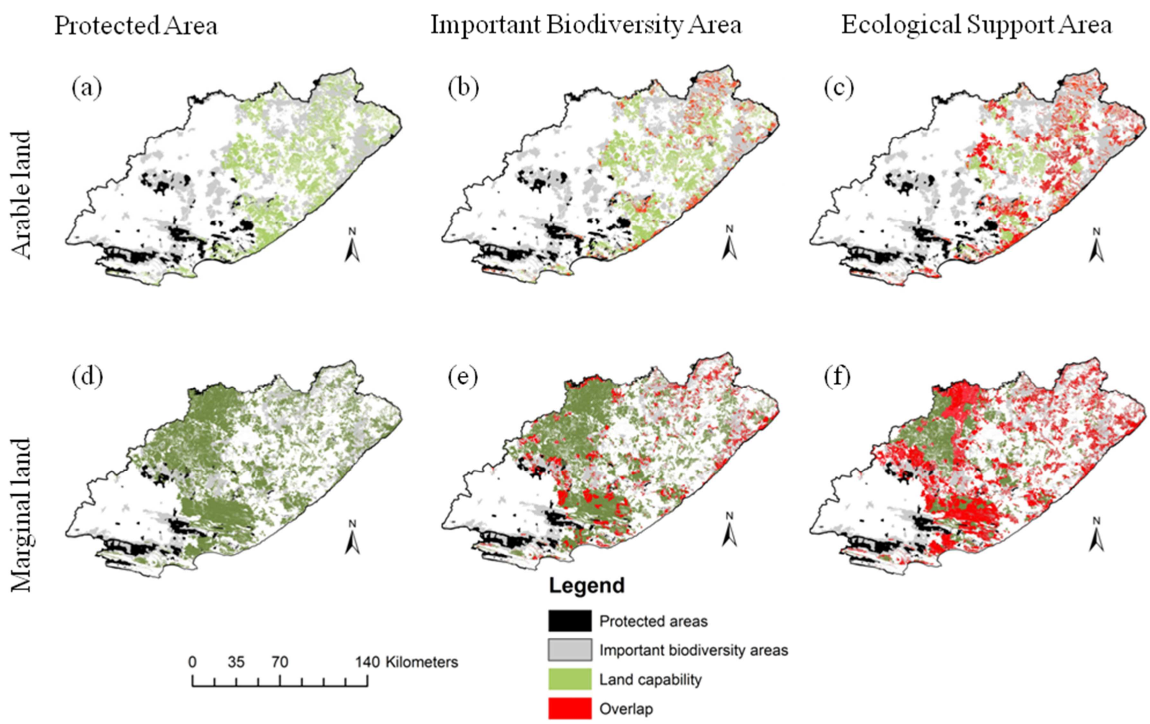

Figure 3.5: Maps indicating increased vulnerability as biodiversity scenarios are introduced to land availability assessment for both optimal (a-c) and marginal (d-f) areas.

3.4.4. Biofuel conflict analysis

In order to match climatically suitable areas with available land the spatial filters described above were applied to each MaxEnt model projection. The climatic projections were reduced to coincide with available land, excluding climatically suitable areas where commercial cultivation may be unfeasible. The range of biofuel species projections that overlap with available areas and in particular vulnerable areas are presented in (Table 3.5). The overlap analysis showed that, depending on the species chosen, between 0-98% of arable areas and remaining marginal areas are predicted as climatically suitable for the biofuel species chosen. Similarly, IBA's and EC's provide climatically suitable habitat for the biofuel species modelled, resulting in significant potential conflict with biodiversity conservation areas.

The difference between arable and marginal areas is reflected as threshold values are increased to indicate higher relative suitability. The level of potential transformation within arable areas remains higher than marginal areas. This can be related to more favourable climatic conditions within the arable classes used to determine land capability. However marginal areas account for a larger proportion of the Eastern Cape that reflect climatic suitability for biofuel cultivation. These areas coincide with EC's and IBA's that are not protected under the formal conservation network.

Table 3.5: The range in percentage overlap of model projections as applied to suitable areas within the Eastern Cape. Overlaps with biodiversity scenarios are also indicated for Protected Areas, Important Biodiversity Areas (IBA) and Ecological corridors (EC).

		Arable Area (Mha)			Total arable overlap	Marginal Area (Mha)			Total marginal overlap	No bio-diversity overlap
		PA	IBA	EC		PA	IBA	EC		
Area (Mha)		(0.04)	(0.51)	(1.02)	(1.56)	(0.23)	(1.13)	(2.22)	(3.59)	(2.56)
Species	Threshold									
Acacia mearnsii	LPT*	95.7	96.5	99.0	98.1	95.7	97.1	99.0	98.2	99.3
	95	95.7	94.7	97.8	96.7	92.2	92.1	86.3	88.5	53.7
	90	94.1	90.5	84.9	86.9	51.6	72.0	54.0	59.5	84.5
	sens=spec**	86.7	88.2	82.4	84.4	38.1	62.7	46.2	50.9	45.7
Arundo donax	LPT	95.7	96.5	99.0	98.1	95.7	97.0	99.0	98.1	99.3
	95	95.7	96.5	99.0	98.1	95.7	97.0	99.0	98.1	98.6
	90	95.7	96.3	98.2	97.5	95.7	95.6	98.7	97.5	99.3
	sens=spec	95.1	92.4	93.2	93.0	93.3	87.0	88.8	88.5	87.4
Beta vulgaris	LPT	61.9	27.5	35.5	33.5	18.1	20.4	17.7	18.6	15.2
	95	61.9	27.5	35.5	33.5	18.1	20.4	17.7	18.6	1.2
	90	19.3	2.5	2.5	2.9	1.5	2.0	1.3	1.5	15.2
	sens=spec	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.1	0.0
Camelina sativa	LPT	95.7	96.5	99.0	98.1	95.7	97.1	99.0	98.2	99.3
	95	61.9	27.5	35.5	33.5	18.1	20.4	17.7	18.6	0.2
	90	0.0	0.5	0.4	0.4	0.0	1.8	0.6	1.0	15.2
	sens=spec	95.1	92.4	93.2	93.0	93.3	87.0	88.8	88.5	87.4
Jatropha curcas	LPT	95.7	95.2	98.7	97.5	94.4	96.3	98.5	97.5	98.1
	95	67.7	39.1	51.7	48.0	54.0	30.8	30.4	32.0	17.0
	90	54.3	33.0	41.1	38.8	30.4	25.3	21.2	23.1	24.0
	sens=spec	38.4	18.5	22.4	21.5	10.9	14.9	8.5	10.7	7.5
Miscanthus sinensis	LPT	89.9	89.5	82.2	84.7	49.9	85.6	79.2	79.3	81.3
	95	15.9	7.7	2.7	4.7	4.1	8.1	2.4	4.3	0.2
	90	12.0	1.9	0.3	1.1	2.0	2.0	0.2	0.9	1.4
	sens=spec	0.0	0.3	0.0	0.1	0.0	0.5	0.0	0.1	0.2

Panicum virgatum	LPT	83.3	94.4	97.9	96.4	92.6	96.0	98.2	97.2	99.0
	95	8.6	12.3	10.4	11.0	2.4	10.7	23.8	18.3	0.0
	90	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	15.1
	sens=spec	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ricinus communis	LPT	95.7	96.5	99.0	98.1	94.4	97.0	99.0	98.1	99.3
	95	95.7	96.5	99.0	98.1	94.4	97.0	99.0	98.1	99.3
	90	95.7	96.5	99.0	98.1	94.4	97.0	99.0	98.1	99.3
	sens=spec	95.1	94.9	96.0	95.6	92.8	87.3	92.9	91.1	89.1
Sorghum halepense	LPT	95.7	96.5	99.0	98.1	95.7	97.1	99.0	98.2	99.3
	95	95.7	96.5	99.0	98.1	95.7	97.1	99.0	98.2	99.3
	90	95.7	96.5	99.0	98.1	95.7	97.1	99.0	98.2	99.3
	sens=spec	14.5	2.7	3.2	3.3	1.4	3.8	5.8	4.9	6.5

*LPT: Lowest presence threshold; **sens=spec: Equal sensitivity and specificity

3.5. Discussion

3.5.1. Outcomes of the modified framework

A framework incorporating species distribution models and land suitability analysis was tested to determine biodiversity conflict in a region of South Africa where the production of biofuel is being considered. This approach demonstrates the importance of spatial filters as applied to species distribution model estimates. It is important to note that while MaxEnt provides an overall climatic niche for a species the application of spatial filters can identify areas with the most likelihood of being converted. However, these results do not infer the potential to reach high abundance or in this case high yield and environmental factors that achieve this goal are outside the scope of this study. The framework presented allows for the spatial extent of potential biofuel crops to be visualised and placed within a localised land-use context. More importantly, we highlight the importance of biodiversity elements as spatial filters to reduce potential impacts of biofuel production on biodiversity.

Our aim in highlighting the need for data that is inclusive of ecological processes has been achieved, and the increased potential conflict with future land-use, demonstrated. The large body of evidence that points to inadequate reserve selection based on land-use opportunities does not facilitate conservation within productive landscapes (Knight & Cowling, 2007). As a result, the likelihood of not accounting for ecological processes or other important biodiversity areas that occur outside of protected areas may lead to an inflated estimation of available land resources. Biodiversity is often in conflict with developmental requirements and the former is often given low priority by governments (Wilson *et al.*, 2010), with natural habitat acting as maintenance areas often being overlooked within managed landscapes.

Significant biodiversity-development conflicts can only be avoided if sufficient information is included in the spatial analysis. The additional biodiversity information available for the Eastern Cape is not representative of other developing countries, where the best available global data may lack sufficient resolution. In areas where biodiversity information is lacking, the spatial filters approach allows proxy data such as carbon content to be incorporated into the analysis framework (e.g. Schweers *et al.*, 2011) .

Although a standardised method for determining land availability is needed, the framework proposed in this study emphasizes the importance of using available local and fine-scale data. We argue that to avoid important biodiversity losses, some measure of biodiversity occurring outside of Protected Areas should be incorporated. Where this information is lacking expert opinion (O' Connor & Kuyler, 2009) or modelled scenarios (Esselman & Allan, 2011) should be used to provide additional insight into biodiversity conflicts.

Admittedly the framework indicates that land-use issues have been simplified within the Eastern Cape and ignores the complex tenure arrangements within rural land areas (Von Maltitz and Brent, 2008). For example, the available land calculated, does not necessarily indicate the willingness to cultivate these areas. Amigun *et al.* (2011) have shown that stakeholder engagement is a key factor to the success of large bioenergy projects and in realising any projected future land-use transformation or conflict estimates. Similarly, in reality, the proportion of excluded areas, as calculated above, may decrease, as potentially available land could exist in the form of abandoned or slightly degraded lands currently identified as cultivated. Biggs and Scholes (2002) showed that agricultural demand has been met by increasing yields per unit area corresponding with a contraction of farming areas. The abandonment of crop land in the 1990s as well as the de-agrarianisation of rural areas has yet to be captured in land-use maps.

3.5.2. Observation on energy crops and model predictions

Previous studies have positioned MaxEnt as an empirical model capable of capturing the distribution of agricultural crops (Estes *et al.*, 2013, Evans *et al.*, 1997). Although it is recommended that more than one model be used to determine suitability of a species (Araujo & New, 2007), the outputs provided by MaxEnt were considered robust enough for the goals of this study. Similarly, estimating the climatic potential of as yet undomesticated species and the likelihood of occurrence, we feel that the use of applying a climatic niche approach to potential crop species was justified. Recent reviews have indicated that the relative probability of occurrence should not be interpreted as an absolute probability of occurrence but rather that the areas indicated as suitable have a higher likelihood of accommodating the modelled species. Similarly, Hijmans (2012) argument based on spatial sorting bias, cautions against the direct comparison and selection of the most suitable species based on the AUC values alone. Not fully accounting for spatial sorting bias may influence direct species comparisons as a result of inflated AUC values. New introductions will likely require the establishment of test sites (Pattison & Mack, 2008) to determine economic viability of species cultivation and to overcome the numerous challenges associated with cultivation. For similar reasons, this modelling procedure does not lend itself to yield predictions despite some innovative attempts that have used MaxEnt for this purpose (Trabucco *et al.*, 2010). The likelihood of yield estimates could be potentially simulated through the selection of high-abundance locations from presence data (Estes *et al.*, 2013), when such information is available.

Our results indicate that the Eastern Cape has potentially suitable areas for the production of biofuel crops that are of global interest. The selected crops have a wide climatic range of which many appear to be potentially suitable within and beyond the borders of the Eastern Cape (not shown here). It was observed that the species chosen for this analysis highlight the dominance of temperate species in biofuel research, with few arid and moderate climate species receiving attention in the literature (e.g. *Jatropha curcas*).

A major source of uncertainty is the presence points used in the model prediction. Using multiple online databases to extract presence records results in species backgrounds that are broader than the native habitat from which they are found (Wolmarans *et al.*, 2010). The resulting model outputs may therefore represent a shift in the niche background as compared to the native background, especially when records are obtained from managed populations found outside their natural range (Wolmarans *et al.*, 2010). The results can also be used to indicate potential risk of newly introduced and planted species becoming invasive, which is a major global concern (Barney & DiTomaso, 2011, Raghu *et al.*, 2006, Richardson & Blanchard, 2011). The most promising global energy crops are known to be invasive in some regions (Barney & DiTomaso, 2008). There are many plant species that have escaped beyond their regions of introduction due to inadequate consideration of the other potential impacts that these plants might pose (Simberloff, 2008). Assuming that such risks can be mitigated, lands with soil and climatic conditions that are marginal for conventional agriculture are likely to be targeted as potential production areas.

3.5.3. Biodiversity and implications for conflict

Using a spatial approach to identify areas of potential threat is of real interest to both the conservation community and local authorities as scenarios can be developed to conserve biodiversity based on the spatial arrangement of new and existing farms (Gabriel *et al.*, 2009). One of the key challenges, however, is to account for all available factors within a spatial framework. Land-use in the Eastern Cape is dynamic. Commercial game farms and cultural choices are strong drivers of land-use patterns. These drivers are set to continue into the future and may contribute to the preservation of biodiversity or act as ongoing threats to it. It is not practical to designate all lands for biodiversity conservation, especially when development is linked to goals such as poverty alleviation, and this increases the need for multifunctional landscapes (Koh *et al.*, 2009). Biofuels are likely to account for a small proportion of land-use within the coming decades. However this could change with increasing demands for alternative fuel sources. It is prudent to acknowledge this sector in order to mitigate against extensive losses of important biodiversity areas to productive landscapes, and this stimulates the need for innovative approaches for the future design of productive landscapes (Koh *et al.*, 2009). Similarly, climate change is likely to be a major driver of shifting agricultural landscapes (Bradley. *et al.*, 2012). The projected loss of climatic suitability of current agricultural crops is likely to shift cultivation into as yet uncultivated areas where biodiversity conservation areas coincide (i.e. increased overlap with NPAES areas). Minimising potential conflict through the implementation of farming practises that maintain biodiversity at plot, region and landscape levels is of increasing importance to both current and future biodiversity conservation (Firbank, 2008, Scherr & McNeely, 2008).

Gabriel *et al.* (2009) suggest that farming on slightly poorer agricultural quality areas is linked with more extensive practices compared to intensive farming on arable lands. Although, extensive farming spreads the risks over a larger area and has a potentially lower impact on biodiversity, this depends on the crop and the farming practice adopted (Anderson-Teixeira *et al.*, 2012). It is also recognised that marginal land not used for conventional crops will contribute to biodiversity benefits. However, the financial benefits of crop diversification may drive expansion into these marginal areas (Bryan *et al.*, 2010). A further consideration is that the potential for energy crops may seem favourable in areas where water demands can only be met by natural rainfed sources. Highlighting these areas could narrow the scope of biodiversity conflicts. Irrigation into the future will most likely be limited since 98% of water in South Africa is already allocated and a proportion of the population still requires improved access to water (Blignaut *et al.*, 2009).

While we have focused on the biodiversity conflict associated with potential land-use change at a regional level, it would be useful to contrast these findings with studies undertaken using internationally available data. The conservation sector recognises the importance of ecological support areas, especially for providing corridors and migration routes, yet global estimates of biofuel production cannot adequately include these areas.

The broader impacts of biofuels are likely to impact on ecosystem services in a similar fashion given their direct links to ecological processes (Gasparatos *et al.*, 2011). The potential use of ecosystem service maps should be integrated into future analysis (Freudenberger *et al.*, 2012). Apart from serving as a proxy for the broader landscape processes, this will capture the utilitarian value of biodiversity which is lacking and therefore left out of models.

The need for globally recognised frameworks and standards to guide potential land-use changes should be recognised. Being consistent in accounting for conservation actions which address land-use, biodiversity and ecological support areas will reduce future impacts associated with land-use change. Where global datasets are not available, our results show that enhancing land suitability assessments with available local and fine-scale data can assist in providing a realistic estimation of potentials and conflicts. Similarly, land suitability methods that focus on areas with increased production potential can narrow the scope for estimating threats to biodiversity (Stoms *et al.*, 2011, Wessels *et al.*, 2003). This proactive approach anticipates likely habitat transformation and provides an objective way of mitigating potential conflict with existing land-use and biodiversity.

3.6. Acknowledgements

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Part II:

Biodiversity indicators

Chapter 4: Using the Biodiversity Intactness Index (BII) to predict the potential impact of biofuel production in the Eastern Cape South Africa

4.1. Abstract

Biofuel production will likely increase the rates of land-use change and habitat loss. This is very likely to have substantial negative impacts on biological diversity. Countries have agreed to minimise biodiversity losses by monitoring the status of biodiversity using appropriate biodiversity indicators. These indicators should inform planning and policy development. There are numerous biodiversity indicators that have been proposed although none have been agreed upon as official indicators as yet. In this paper I test the applicability of the biodiversity intactness index (BII) as a potential tool to monitor potential impacts of biofuel production on biodiversity. The BII has proven to be a cost effective indicator that adopts a broad view of biodiversity. The BII was used to assess impacts from future biofuel production following the hypothetical conversion of available (Degraded and Moderate use) and suitable (Arable and Marginal) land in the Eastern Cape province of South Africa. In addition conservation data that enhances the protected area network was included in the analysis. The effect of excluding these areas from cultivation is assessed using the BII to quantify their importance to biodiversity intactness. These results demonstrated the potential for substantial biodiversity losses (between 0.5 - 25%) as compared to current levels across the different scenarios developed. I found that important biodiversity areas contribute between 0.2 and 12.2% to biodiversity intactness. Disaggregating the indicator shows greater declines at local municipality and biome scales, highlighting specific locations at risk of biodiversity losses. Conservative biofuel production targets can have low impacts on biodiversity. However, the species choice and extent of planting will determine whether important biodiversity areas without formal protection are at risk of being converted. The BII provides a scientifically acceptable approach to determine the effect of biofuel production on biodiversity as a result of land-use change. I highlight usefulness of the indicator and provide recommendations to improve its applicability.

Keywords: biodiversity indicators, land-use change, protected areas

4.2. Introduction

There is a growing need for alternate fuel sources to reduce the recognised contribution of fossil fuels to climate change (Slade *et al.*, 2011, Tilman *et al.*, 2009). This has resulted in policy-driven targets for biofuel production to supplement existing energy demands (Fischer *et al.*, 2010b). It is expected that large-scale production of biofuels will increase rates of land-use change and habitat loss (Firbank, 2008, Koh *et al.*, 2009), driving further pressure to transform unutilised arable land around the world. Furthermore, technological advancements have resulted in the potential for biofuel crops to be planted across a wider range of land types, targeting areas that may otherwise have been excluded from production estimates (Sala *et al.*, 2009, Slade *et al.*, 2011, Wicke *et al.*, 2011). Land-use change and habitat loss are among the greatest threats to biodiversity (Foley *et al.*, 2005, Sala *et al.*, 2000). In particular, an increase in agricultural and plantation landscapes is responsible for large changes to the earth's terrestrial surface (Chapin *et al.*, 2000, Foley *et al.*, 2005).

Measuring the loss of biodiversity and the degradation of ecosystems has been a major goal of conservation scientists (Mace & Baillie, 2007). This is especially important as intact habitats are recognized to provide important ecosystem services (Cardinale *et al.*, 2012, Reyers, 2013) as well as to provide increased resilience against natural disasters (Nel *et al.*, 2014). Currently, while screening for suitable locations for biofuel production, biodiversity areas are best accounted for through the identification of protected areas (e.g.Beringer *et al.*, 2011). However, conserving biodiversity outside of protected areas is recognised as a critical step to minimising global biodiversity losses (Reyers, 2013) and also for maintaining the ecological processes that sustain biodiversity (Bennett *et al.*, 2009).

Previous studies have shown that potential conflicts between areas of high biodiversity and unutilised arable land do occur (Rouget *et al.*, 2003, Wessels *et al.*, 2003). However, while these studies aim to pre-empt potential conflicts between land-use change and biodiversity, it is difficult to translate these results to the overall impact on biodiversity (Lamb *et al.*, 2009).

Measuring and accounting for biodiversity is not an easy task, as various levels (genetic, species, ecosystem) and aspects (composition, structure and function) of biodiversity are recognised for which there are many competing views (de Bello *et al.*, 2010b, Duelli & Obrist, 2003, Feld *et al.*, 2010, Noss, 1990). To address this, a variety of indicators have been developed to meet different objectives (Biggs *et al.*, 2007, Feld *et al.*, 2010, Vačkář *et al.*, 2012). Still, there is much debate over which indicators are appropriate for monitoring biodiversity (de Bello *et al.*, 2010b, Pereira *et al.*, 2013, Vačkář *et al.*, 2012) and how these can be used to inform decision makers (Nicholson *et al.*, 2012). To date, the application of biodiversity indicators to assess potential impacts of biofuel production has received little attention (Hellmann & Verburg, 2010). Where indicators are suggested, Nicholson *et al.* (2012) recommends that these be demonstrated to assess the applicability of the indicator to the task required.

It is important to consider which indicators may be useful to address the potential impacts on biodiversity as a result of changes in land-use. This implies the need for biodiversity indicators that can be linked to land-use change or habitat loss. To do so effectively would require a “broad” approach to biodiversity that acts at multiple scales across different ecosystems and is inclusive of human interactions (Biggs *et al.*, 2008). While a few indicators have been developed that addresses biodiversity in this manner (Alkemade *et al.*, 2009, Ten Brink, 2006), I use the Biodiversity Intactness Index (BII) which reports the magnitude of change relative to a reference state (Scholes & Biggs, 2005).

The BII (see Box 4.1) was developed in accordance with requirements of the Convention on Biological Diversity (CBD, 2003) and aims to provide the means for summarizing the status of biodiversity in very general terms (Scholes & Biggs, 2005), as opposed to assessing various components thereof (Duelli & Obrist, 2003). One advantage of the BII is that empirical biodiversity data can be supplemented with expert opinion, as detailed information is often lacking for many regions (Scholes *et al.*, 2012). While a demonstration of the BII has already been presented (Biggs *et al.*, 2008), this general approach to biodiversity assessment has not been applied in determining potential impacts of biofuel production. The BII aims to provide a synthetic overview of biodiversity for policy makers, as it provides a well-informed link between changes in land-use and the effects on biodiversity (Biggs *et al.*, 2008).

In this study I use the BII to evaluate consequences for terrestrial biodiversity under potential levels of landscape transformation for biofuel production, using one region of South Africa as a test case. I explore the scalability of the index to determine how different impacts are captured at multiple scales. First, the BII was used to calculate the impact on biodiversity following the conversion of available and suitable land. Next, the importance of excluding important biodiversity areas that are not protected from cultivation was explored in seeking to maintain biodiversity intactness. The paper provides insight into the application of the BII and provides recommendations for enhancing decision-making.

BOX 4.1 - The Biodiversity Intactness Index (BII)

Rationale – The Biodiversity Intactness Index was developed to measure the status of biodiversity as required by the Convention on Biological Diversity (CBD) (Scholes & Biggs, 2005). The output of the BII is an indication of the state of diversity within a given geographical area (Biggs *et al.*, 2006). The indicator is an estimation of the average population size of a wide range of organisms relative to their baseline populations. The method uses existing biome or vegetation-type classifications, existing distribution information for major taxonomic groups, and expert opinions about the relative reduction in average abundance of species (across different taxa), as a consequence of various mapped land-uses. It is possible to derive both an overall score of the integrity of biodiversity within a region (aggregated) or can it focus on a particular functional type within various land-use activities (disaggregated). The format of the BII provides an intuitive measure compared to traditional diversity indicators (i.e. Shannon index, Simpson index), by reporting on the magnitude of change in relation to a reference condition (Lamb *et al.*, 2009).

Strengths – A major advantage of the BII is that it differentiates between impacts of different land-uses on terrestrial biodiversity which can be used to identify the impact of biofuel as a land-use practice. The BII can complement indicators such as the Natural Capital Index (NCI) in data sparse areas as expert opinion can be consulted to determine the levels of biodiversity loss (Ten Brink, 2006). The Mean Species Abundance (MSA) index utilises a similar approach of incorporating abundance-fractions for different land-uses (Alkemade *et al.*, 2009). The BII has the same meaning at all spatial scales and can be calculated nationally, by province, municipality or for any spatial unit (Biggs *et al.*, 2006). The versatility of the BII has allowed it to be calibrated to estimate past changes as well as to project into the future under various scenarios (Biggs *et al.*, 2006, Biggs *et al.*, 2008).

Weaknesses - The BII has been criticised because it is based on expert opinion rather than field data (Mace & Baillie, 2007) and incorrect measures of land-use can provide the wrong sentiment regarding the integrity of biodiversity (Rouget *et al.*, 2006). The BII has also been noted to be insensitive to species identity and changing abundance at the community level (Ewers *et al.*, 2009, Faith *et al.*, 2008). Another view of the BII is that it is really an estimation of abundance and does not indicate the impact on diversity which is an important aspect of the biodiversity definition (Faith *et al.*, 2008). Faith *et al.* (2008) have reviewed this and suggested that species area curves be included in intactness measures to overcome the lack of diversity representation, however this requires more detailed species information.

Uses – The BII has been used as a primary indicator within South Africa to show current, past and future impacts. The BII was used to assess the impact of invasive alien plants (van Wilgen *et al.*, 2008) and model the positive benefits of biological control on invasive alien plants (de Lange & Van Wilgen, 2010). Where detailed species data are available, the BII may be superseded by alternate indicators (Ewers *et al.*, 2009, Faith *et al.*, 2008). However abundance-fraction indicators such as the MSA are utilised and where little data to no exists on the BII presents a credible approach to estimating the integrity of biodiversity (Scholes & Biggs, 2005).

4.3. Material and methods

4.3.1. National biofuel strategy

South Africa's biofuel policy forms part of its Renewable Energy portfolio which includes wind and solar energy production (Department of Minerals and Energy, 2003). The Department of Minerals and Energy (DME) envisaged biomass energy (mainly liquid biofuels) contributing 35% to national targets for renewable energy by 2013 (Department of Minerals and Energy, 2003). A provisional strategy for biofuel (hereafter "the strategy") was outlined in the Biofuels Industrial Strategy of the Republic of South Africa of 2007 (Department of Minerals and Energy, 2007). The strategy outlines a cautionary initial biofuel target of 5% which was revised to 2% of liquid fuels, with a decision on whether to increase this proportion to be made at the end of the pilot phase (Department of Minerals and Energy, 2007). Although these targets were to be met by 2010, no biofuels have been produced yet.

The strategy aims to achieve economic and social development in rural areas, such as the former "homeland" areas in the Eastern Cape (Department of Minerals and Energy, 2007). Homeland areas in South Africa have a long history of social, political and environmental neglect since colonial times. The creation of jobs and improving the development imbalance between informal and small-scale farming areas and commercial farming areas are key components of the biofuel supply chain (Lynd *et al.*, 2003, Musango *et al.*, 2010). Incentives for locally-based processing plants assume that feedstocks will be acquired via contractual agreements from small-scale farmers (Funke *et al.*, 2009). This philosophical underpinning is intended to stimulate demand and incentivise farmers to optimise longer-term yields while increasing land productivity (Department of Minerals and Energy, 2007).

4.3.2. Study area

The Eastern Cape Province of South Africa (Figure 4.1) was chosen as the study area because it has been identified as a focal area for biofuel production. Historically the region has undergone considerable investment in infrastructure to alleviate poverty and to empower rural farmers to meet national developmental policies, but much additional effort is needed to meet developmental goals. Existing poverty alleviation programmes include the expansion of conventional agricultural practices or increased livestock farming. Current farming patterns in the province follow a combination of commercial and subsistence farming, with the latter achieving significantly lower yields in some areas (Shackleton *et al.*, 2001). Biofuel production is intended to contribute to enterprise development and provide increased agricultural investment targeted at small-scale or emerging commercial farmers. Viability assessments are currently underway in the Eastern Cape with several projects at planning and establishment phases (Musango *et al.*, 2010).

The expected potential for agriculture, forestry and agro-processing initiatives in the former homeland areas is considered to be large, but have yet to be realized (Lynd *et al.*, 2003). Some reasons for underdevelopment include a strong traditional focus on livestock farming and a land-tenure system based on tribal or communal land ownerships (Hoffman & Ashwell, 2001). The current trend of rural de-agrarianisation has also contributed to the recent increase in abandoned land and the slow uptake of new farming activities (Andrew & Fox, 2004, Davis *et al.*, 2008). The resulting development pressures pave the way for policy intervention that could facilitate a rapid increase in biofuel production.

The Eastern Cape has high levels of biological diversity. The average mean annual rainfall in is 538 mm, although a rainfall gradient across the province results in dryer conditions (100-200 mm) in the west and wetter conditions (1000-1200 mm) the east (Schulze & Lynch, 2007). Five of the seven terrestrial biomes of South Africa and the Maputaland-Pondoland-Albany biodiversity hotspot occur in the province (Critical Ecosystem Partnership Fund, 2010, Driver *et al.*, 2012, Mucina & Rutherford, 2006). Large areas of grassland and savanna ecosystems are strongly underrepresented in the province's network of formal protected areas and are at risk of transformation (Driver *et al.*, 2012, O' Connor & Kuyler, 2009). These conservation priorities also form part of the national biodiversity objectives (Government of South Africa, 2008). The expansion of forestry, agriculture and urbanisation of rural areas are among the key threats to biodiversity in this region (Berliner & Desmet, 2007). Overgrazing, invasive alien plants and poor management of agricultural lands have also resulted in the degradation and transformation of many areas (Evans *et al.*, 1997, Hoffman & Ashwell, 2001). The dynamic setting of the Eastern Cape provides a unique opportunity to assess potential impacts on biodiversity from biofuel production. The inclusion of biofuels as a possible land-use option raises important questions regarding availability and suitability of remaining land and the spatial distribution across the province. This will ultimately determine the potential impacts on biodiversity.

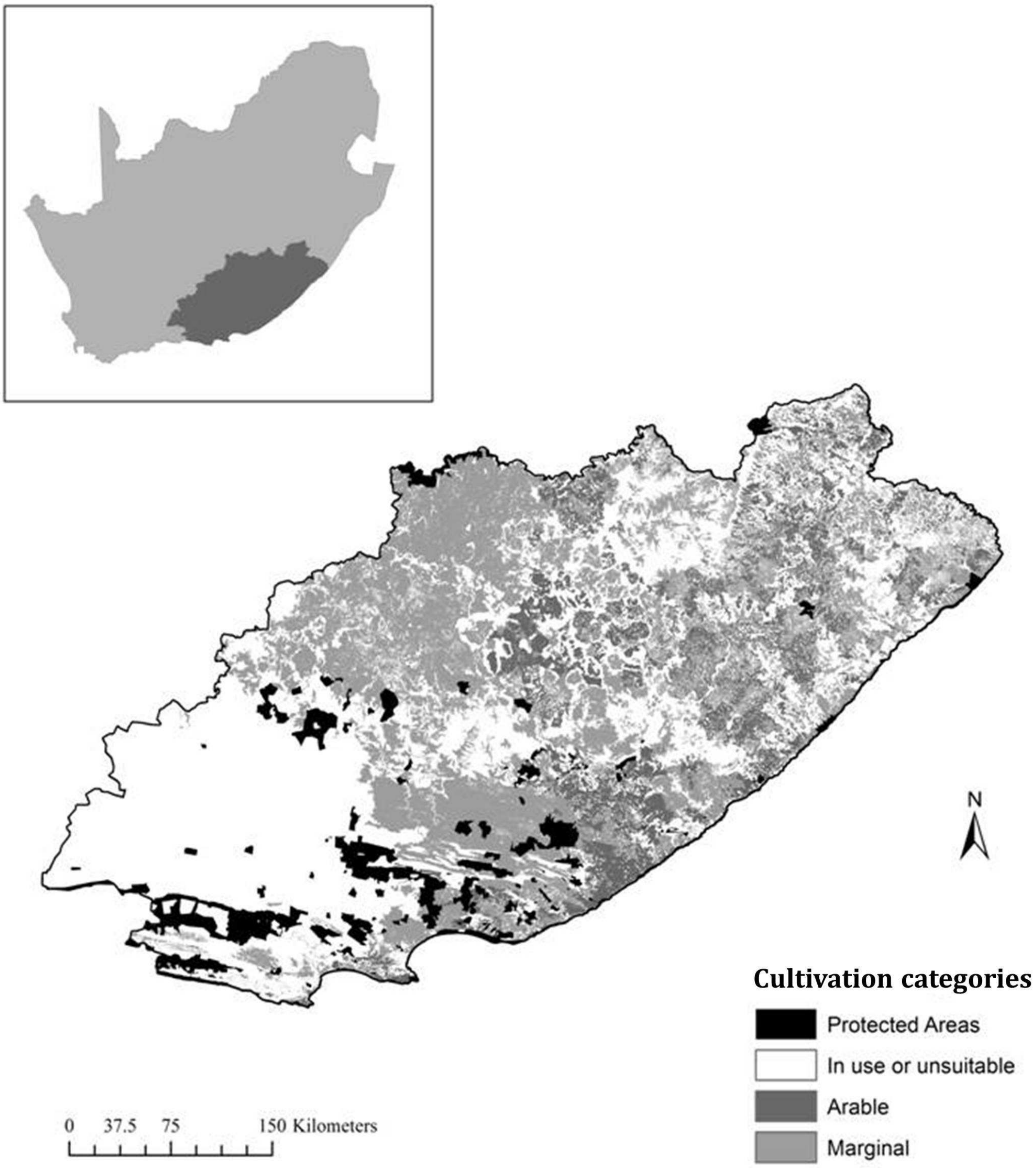


Figure 4.1: The distribution of arable and marginal land in the Eastern Cape Province of South Africa.

4.3.3. GIS Data layers

4.3.3.1. **Determining land availability and suitability**

To determine land availability, six land-use classes were derived from the National Land Cover 2000 for South Africa (CSIR, 2003) namely: *Protected, Urban, Cultivated, Plantation, Degraded* and *Moderate Use*. These classes were used to identify natural (Protected, Moderate-use and Degraded) and non-natural (Urban, Cultivated and Plantation) land. For this study, I assumed that both *Moderate Use* and *Degraded* land would be available for biofuel production despite a variety of extractive land-use practices (e.g. grazing, wild-harvesting) that occur here (Von Maltitz & Brent, 2008).

Land suitability was based on agricultural potential using the land capability classification for South Africa (Schoeman *et al.*, 2000). The land capability map identifies similar land units based on potential production capability while also considering the limitations and hazards within a particular region (i.e. soils, risk of erosion, physical terrain constraints and climate) (Wessels *et al.*, 2003). Land capability classes were reclassified as follows: Classes 1-4 were designated as *arable,* Classes 5-6 were designated as *marginal,* while Classes 7-8 were characterised as *excluded*. In addition to *arable* land, *marginal* land was considered for agricultural potential based on the growing demand for biofuels to be allocated to this land type to reduce direct competition with food production (Pimentel *et al.*, 2009, Plieninger & Gaertner, 2011).

In addition, areas with steep slopes were excluded from the analysis, due to increased environmental and economic costs associated with cultivation on steep slopes. A 90 m digital elevation model was used to identify and exclude areas with a slope greater than sixteen degrees (Fischer *et al.*, 2007).

All GIS layers were combined to identify available and suitable areas with agricultural potential (Table 4.1).

4.3.3.2. Biodiversity layers

Only ~5% of the area in the Eastern Cape is under some form of protection. Additional biodiversity spatial layers that identify important biodiversity areas occurring outside of protected areas which aimed to complement the formal protected area network were sourced from an online database supplied by the South African National Biodiversity Institute's online geographic information database (www.BGIS.co.za). Two independent data sources for identifying and capturing biodiversity features were used, namely: (i) the National Protected Area Expansion Strategy (NPAES) (Government of South Africa, 2008) and (ii) a region-based systematic conservation plan, The Eastern Cape Biodiversity Conservation Plan (ECBCP) (Berliner & Desmet, 2007). The NPAES indicates areas of highest priority for conservation needed to meet representative biodiversity targets and to protect areas in the face of climate change. The ECBCP is based on the systematic conservation planning approach of identifying areas needed to maintain corridors and ecological processes (Driver *et al.*, 2012, Margules & Pressey, 2000). These GIS layers were used to identify 1) important biodiversity areas (IBA) and 2) ecological corridors occurring outside of protected areas. They were used as additional spatial filters to identify potential areas that should be excluded from being transformed and to assess the importance natural habitat outside of protected areas to conservation.

4.3.3.3. Mapping potential supply areas

There are large areas of remaining natural land of which 14% is arable and 33% is characterised as marginal land (Table 4.1). Future land-use change scenarios were developed based on the agricultural potential of available land (Table 4.1). Nine classes of available land were used to identify potential areas suitable for the supply of biofuels. These were: 1) degraded-arable; 2) degraded-marginal; 3) moderate use – arable; 4) moderate use – marginal; 5) all arable; 6) all marginal; 7) all degraded; 8) all moderate use; 9) all available land. Each class was transformed to *Cultivated* in a GIS. This extreme view of land-use change is in line with the long-term projections outlined by Biggs *et al.* (2008) for the southern African region. Furthermore, these maps highlight the areas at risk of conversion, as well as the maximum potential for change based on natural rain-fed agricultural conditions. The scenarios exclude the expansion of urban or infrastructural developments. Results from the scenarios were compared to a baseline using the National Land Cover of 2000 (CSIR, 2003). The viability of crops was not considered in this study, but could include suitable biofuel species or other commercial agricultural crops such as canola or sunflower (Blanchard *et al.*, 2011, Von Maltitz *et al.*, 2009).

Table 4.1: Areas of arable and marginal land within "degraded" and "moderate use" land-use categories in the Eastern Cape, South Africa. Only the potential available area is displayed within the defined categories. Percentage area (%) was calculated in reference to the land area within the Eastern Cape.

Land availability	Land suitability (Mha)		
	Arable (%)	Marginal (%)	Both (%)
Degraded	0.43 (2.8)	0.38 (2.8)	0.81 (5.6)
Moderate use	1.89 (11.3)	5.01 (30.4)	6.90 (41.7)
Degraded and Moderate use	2.32 (14.1)	5.39 (33.2)	7.71 (47.3)

4.3.3.4. The biodiversity intactness index (BII)

The methodology for the BII (Box 4.1) has been extensively discussed in previous publications (Biggs *et al.*, 2006, Biggs & Scholes, 2002, Scholes & Biggs, 2005) and only the most important aspects of the algorithm are highlighted below.

For a specific biome, the BII is calculated as:

$$BII = \frac{\left(\sum_i \sum_j \sum_k R_{ij} A_{jk} I_{ijk}\right)}{\left(\sum_i \sum_j \sum_k R_{ij} A_{jk}\right)} \qquad \textbf{Equation 4.1}$$

where R_i is the richness of species group i, A_{jk} the area of land-use j and I_{ijk} the population impact on species group i of land-use j. The BII provides a measure of population integrity and is the average impact across all available taxa. The species richness values (R) used to calibrate the original BII for South Africa was used in this analysis. This information was available for well-known taxa (plants, mammals, birds, frogs and reptiles) across all of the WWF Ecoregions (Olson *et al.*, 2001) occurring in the Eastern Cape. Expert estimates of land-use impacts on populations within Ecoregions were obtained from the original BII application (Scholes & Biggs, 2005) and summed within each land-use category. Impact factors were derived relative to pristine population estimates within large protected areas (Appendix B Figure B1).

4.4. Data analysis

The BII was calculated for each scenario to determine how changes in land-use affect biodiversity intactness. For each scenario impact factors had to be recalibrated to include biofuel as a land-use. Originally, impact factors were derived from expert opinion for taxa examined relative to populations within large protected areas (Figure A1) (Scholes & Biggs 2005). Since specific biofuel related species impacts are not readily available, I used the existing impact factors associated with *Cultivation* within this analysis. For a comparison of the sensitivity of the BII, alternate impact factors associated with biofuels based on *Cultivation*, *Plantation*, and *Degraded* are presented in (Appendix B2 Table B1).

In addition, land-use maps were recalibrated to include biodiversity spatial filters where important biodiversity areas and corridors were excluded from being transformed. All spatial layers were analysed using ARCGIS models created using model builder to analyse raster grids. Data for species richness and impact factors were provided by Dr Reinette Biggs, developer of the original BII.

BII calculations were summarised to three local government levels and to the biome scale based on the WWF Ecoregion boundaries (Olson *et al.*, 2001). The local government levels used are the Provincial, District and Local municipalities which reflect where management decisions are made.

To illustrate the effect of a biofuel strategy, I calculated the potential area required to meet the initial fuel targets as well as future targets. Species yield data were extracted from Von Maltitz and Brent (2008) and are presented in Appendix B3.

4.5. Results

4.5.1. Potential biodiversity impacts of land-use change

The current estimate of the BII for the Eastern Cape is ~83%. The reduction in intactness, accounts for the conversion of natural land to agriculture, forestry or urban areas. This represents a ~17% decline in the average abundance across all plant, mammal, bird, reptile, and amphibian species relative to their precolonial populations.

Table 4.1 indicates that untransformed arable and marginal land accounts for 14% and 33% of the Eastern Cape respectively. A large portion of arable and marginal land is classified as moderate use, which can be considered to have a high intactness value. Including additional biodiversity spatial filters reduces that amount of available arable and marginal land to only ~5% and ~11%, respectively (Table 4.2). The excluded areas overlap with either important biodiversity areas or ecological corridor areas.

Transforming available land within the nine classes result in BII scores that range from ~75% for the conversion of all arable land, to ~66% for the conversion of all marginal land, and ~58% for the conversion of both arable and marginal land (Table 4.3). Including the additional biodiversity spatial filters reduces BII losses between 0.1% and 6.5% for important biodiversity areas, and between 0.2% and 12.5% for the inclusion of both important biodiversity areas and ecological corridors (Table 4.3). The conversion of both arable and marginal land allows the most extreme land-use change scenario to be visualised and indicates the potential maximum decline in the BII should all available land, including marginal land, be transformed.

The estimated effect of land-use changes on taxa are shown in Figure 4.2. Similar trends are observed to overall BII scores where an increase in land converted to cultivation reduces the abundance values of remaining taxa groups. Plant abundance drops dramatically under the Protected Area only scenario, whereas the inclusion of corridors and IBA's reduces this effect. The effect on birds is estimated to be less than mammals or plants especially at higher levels of transformation.

Table 4.2: The area and percentage overlap of biodiversity scenarios with land capability classes indicating Arable, Marginal and Excluded areas in the Eastern Cape. Excluded areas comprise formal and informal conservation areas as well as areas unsuitable for cultivation. Non-biodiversity areas with no recorded biodiversity value are also indicated.

Biodiversity scenarios	Arable Mha (%)	Marginal Mha (%)	Excluded Mha (%)	Total Mha (%)
Protected areas	0.04 (4.0)	0.23 (24.8)	0.66 (71.2)	0.93 (5.5)
Important biodiversity areas	0.51 (12.0)	1.13 (26.8)	2.59 (61.9)	4.23 (25.1)
Ecological corridors	1.02 (14.8)	2.22 (32.3)	3.65 (52.9)	6.89 (40.9)
Total	**1.56 (12.9)**	**3.59 (29.8)**	**6.90 (57.3)**	**12.05 (71.5)**
Non biodiversity areas	0.75 (15.6)	1.80 (37.4)	2.26 (46.9)	4.81 (28.6)
Total all	**2.32 (13.7)**	**5.39 (31.9)**	**9.16 (54.3)**	**16.86 (100)**

Table 4.3: Summary results of the BII scores for land-use scenarios following the exclusion of protected areas (PA), important biodiversity areas (IBA) and Corridors from transformation. Both importance biodiversity areas and corridors occur outside of protected areas.

Agricultural potential	Land-use	PA	IBA	Corridors
	Present land-use	83.1	83.1	83.1
Arable	Degraded	82.4	82.4	82.6
	Moderate use	75.4	77.2	79.2
	Degraded and Moderate use	74.7	76.6	78.9
Marginal	Degraded	82.5	82.6	82.8
	Moderate use	66.9	70.6	75.5
	Degraded and Moderate use	66.3	70.1	75.2
Arable and Marginal	Degraded	81.8	81.9	82.3
	Moderate use	59.2	65.7	71.7
	Degraded and Moderate use	57.9	63.6	70.1

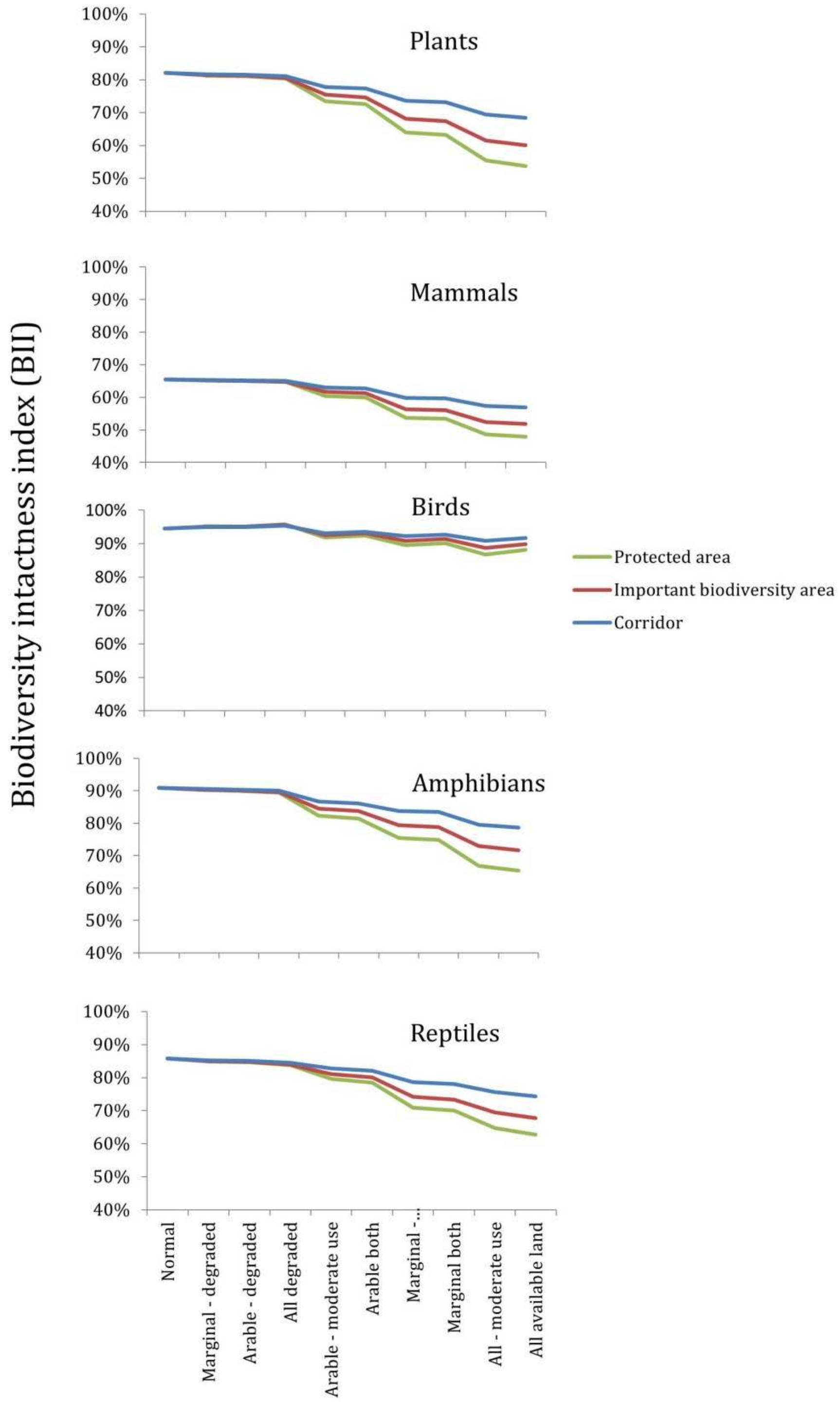


Figure 4.2: The effect of excluding different biodiversity layers on taxa for a) protected areas, b) important biodiversity areas, and c) ecological corridors from being converted to biofuel production and its effect on the BII scores for the taxa measured.

The effect of fuel scenarios on land requirements are shown in Figure 4.3. A simple equation of available land and yield per hectare illustrates a sharp decline in potential fuel production should all land with biodiversity importance be excluded from potential production. Also shown is the potential overlap with biodiversity important areas. This was based on the Calculations for fuel yield are presented in Appendix B3.

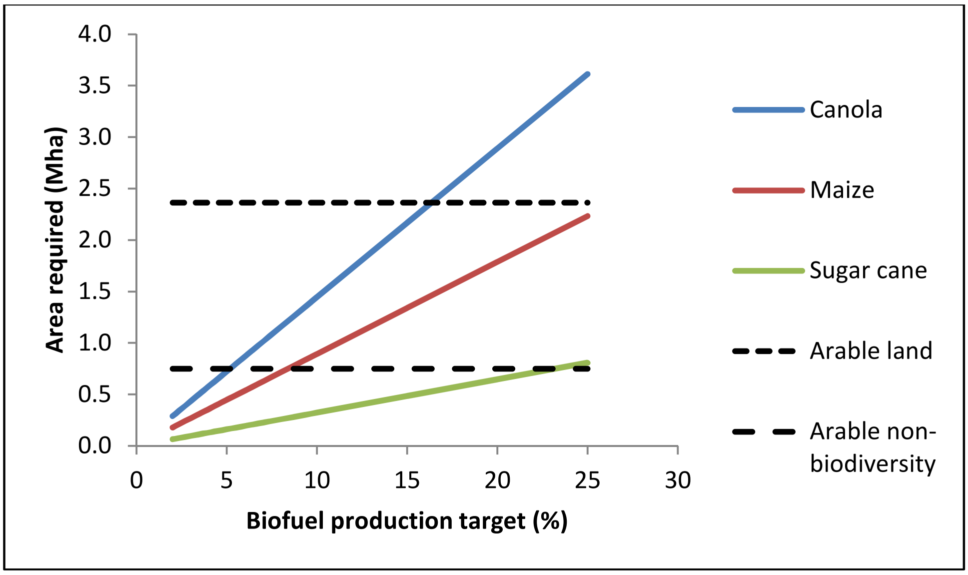


Figure 4.3: An illustration of biofuel production targets and the area required to meet these using the biofuel feedstocks canola, maize and sugar cane examples. The dashed lines indicate arable land with no biodiversity conflicts (long dash) and total areas (short dash).

4.5.2. BII at different scales of analysis

The BII is an area weighted index and the effect of increasing the proportion of cultivated areas within each management unit results in a similar trend across multiple scales (Figure 4.4). The trend indicates a linear decline in BII; at finer spatial scales (i.e. local municipality) lower correlations can be attributed to the effect of other land-uses, such as urban areas or the presence of protected areas.

Table 4.4 shows the range of disaggregating the BII scores to the local government and biome scale. The range in BII scores is indicative of the disproportionate distribution of land resources across the province. It is evident that some municipalities are not affected by the criteria for land-use change used in this study nor would they be affected by future biofuel policies. Disaggregating the BII across multiple scales indicates levels of spatial heterogeneity and identifies those local municipalities with the largest risk to biodiversity resources. For example, the conversion of arable land identifies the District Municipalities of O. R Tambo and Amathole, whereas the conversion of marginal land identifies Joe Gqabi and Chris Hani municipalities (Appendix B).

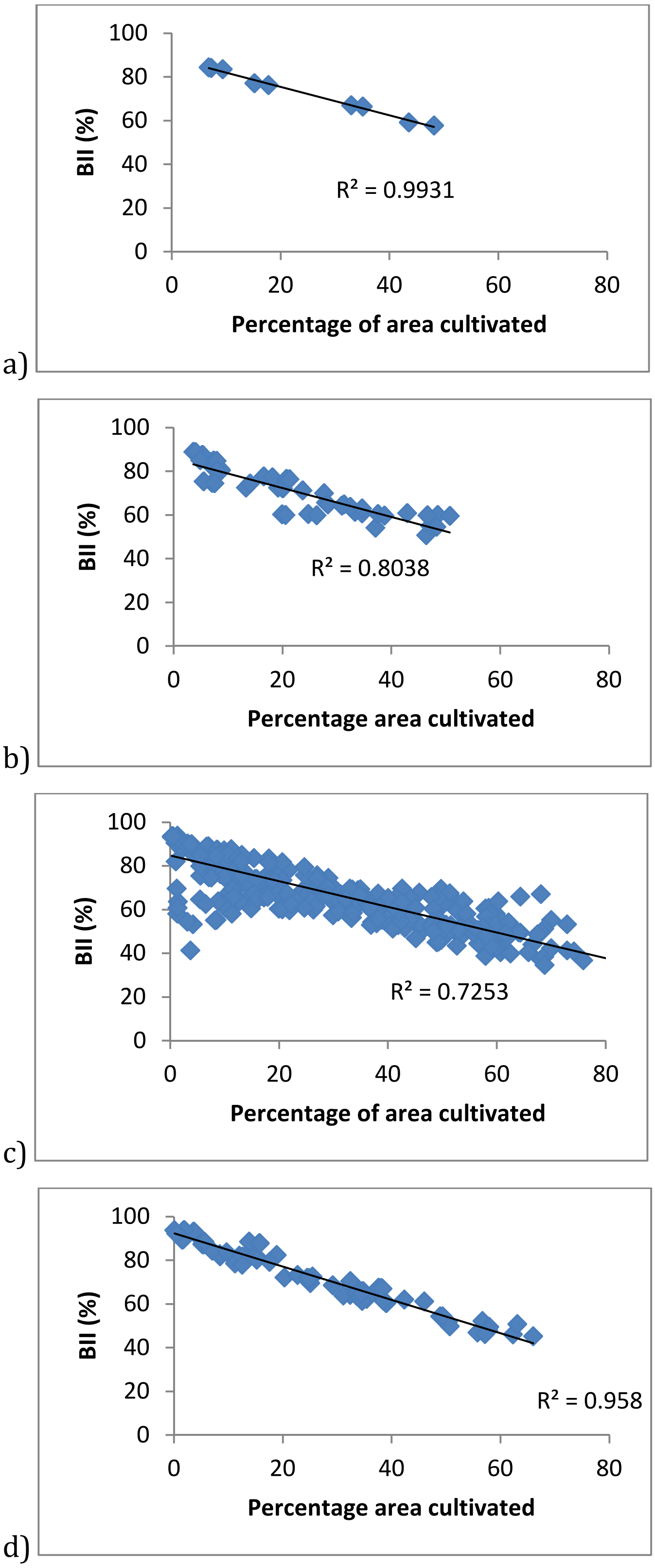


Figure 4.4: The linear relationship between BII and the percentage of area cultivated areas across multiple scales for the: a) Eastern Cape and disaggregated to b) district municipality, c) local municipality, and d) ecoregion. Similar patterns occur for disaggregated results and reflect the ability of the BII to report across difference scales. The range of values increases as the scale of analysis decreases.

Table 4.4: Summaries of the range of BII scores as analysed at two municipal scales and at the biome scale, indicated by the WWF ecoregions for the land-use scenarios. This shows the response of the BII to the conversion of available land to cultivation. Complete tables listing individual district municipalities, local municipalities and ecoregion names can be found in Appendix B4.

Agricultural potential	Land-use	EC Province	District municipality (N=8)		Local municipality (N=39)		Ecoregion (N=12)	
			Low	High	Low	High	Low	High
	Present land-use		73.2	89.0	61.9	93.7	72.0	93.9
Arable	*Degraded*	84.1	72.2	89.0	60.8	93.7	71.9	93.8
	Moderate use	77.0	54.2	87.5	49.1	93.6	64.1	93.1
	Degraded and Moderate use	76.2	50.7	87.5	45.5	93.6	63.5	93.1
Marginal	*Degraded*	84.3	72.3	88.8	61.3	93.7	71.9	93.8
	Moderate use	67.0	55.0	76.6	41.2	93.4	60.4	92.1
	Degraded and Moderate use	66.4	54.7	76.4	41.0	93.4	60.4	92.1
Arable and Marginal	*Degraded*	83.5	71.3	88.8	41.2	93.4	71.8	93.8
	Moderate use	59.1	38.6	75.0	37.1	93.4	46.1	92.1
	Degraded and Moderate use	57.8	34.6	74.8	31.1	93.4	45.2	92.1

N=number of management units

4.6. Discussion

Impacts on biodiversity resulting from the potential conversion of areas suitable for biofuel production were identified using the BII. Potential areas were converted to biofuels production within a GIS emphasising the spatial adaptation of the BII and its ability to estimate impacts resulting land-use change.

4.6.1. BII and land-use change

This analysis shows that large areas of land remain within the Eastern Cape that have varying degrees of agricultural potential, which if converted could significantly reduce biodiversity. By modelling the conversion of available land to biofuels I demonstrated the range of potential impacts on biodiversity in the Eastern Cape. A distinction in future land-use can be made between arable and marginal land, the latter accounts for a much larger area and may be more important for the production of second generation biofuel crops. Considering that conventional crops will be utilised in the initial phase of the biofuels industry (e.g. canola, sunflower, sugarcane etc.), arable land with favourable agricultural potential would be targeted instead of marginal land. Cultivation in arable areas is likely to produce better yields and require fewer nutrient inputs compared to marginal lands (Achten *et al.*, 2010, Slade *et al.*, 2011). However, marginal areas may be favoured for the production of biofuels to reduce potential conflict with future food and animal feed produced on arable land (Plieninger & Gaertner, 2011, Wicke *et al.*, 2011). For example, in India, biofuel production is mostly promoted on marginal or degraded areas (Ravindranath *et al.*, 2011).

Several studies indicate that remaining arable land in the Eastern Cape may be suitable for increased cultivation, including biofuel production (Department of Minerals and Energy, 2007, Estes *et al.*, 2013, Ghosh *et al.*, 2007, Von Maltitz *et al.*, 2010). Previous estimates of available land within the Eastern Cape ignored the potential biodiversity importance of areas not currently protected. The conversion of all remaining arable land reduces the BII by a further ~8% compared to current levels. However, including additional biodiversity spatial filters that identify important biodiversity areas reduced the amount of available land area from ~2.3 Mha to ~0.75 Mha. The subsequent change in BII was only ~4%. This finding emphasises the potential conflict between areas of high biodiversity occurring outside of protected areas and unutilised arable land. These trends are also reflected in marginal areas, where the exclusion of biodiversity areas greatly reduces the effect on the BII. Moreover, the large potential for conservation within marginal areas that are not normally suitable for cultivation may require active conservation activities (i.e. buying of land, land owner stewardship contracts etc.) to protect biodiversity (Gallo *et al.*, 2009).

While the threat of conversion presents a large risk to biodiversity, there are some factors limiting the widespread conversion of land for biofuel production that need to be mentioned here. These relate to complex tenure arrangements within communal and rural land management which means that not all mapped land is necessarily available for biofuels (Von Maltitz & Brent, 2008). Similarly most of the marginal areas mapped in this study are also considered important rangeland areas, needed to maintain livestock numbers (Biggs *et al.*, 2008). Furthermore, the income generated from eco-tourism in some areas may act as a deterrent to produce biofuels, and therefore contributes to the maintenance of natural landscapes (Driver *et al.*, 2012).

Current experience shows that average yield per hectare is likely to vary spatially and that linear relationships between area and yield may not be entirely accurate (Geyer *et al.*, 2010b). Whilst anticipating the spatial variation of projected yields was beyond the scope of this study, estimates of areas required to meet biofuel production provides some insight into the effect of crop choice, land management and the potential effect of fuel production scenarios. Reducing the amount of available land effectively limits the amount of biofuels that can be produced. Furthermore, it is likely that not all selected species will be suitable for the same locations, affecting the potential to accurately determine yields and production targets (Stoms *et al.*, 2011). These challenges may be overcome with the use of more sophisticated crop modelling (e.g. Bryan *et al.*, 2010). Despite these limitations, this study provides a baseline of converting areas with broad agricultural suitability.

4.6.2. BII and Biodiversity tradeoffs

One of the important challenges facing decision makers is the scale at which biodiversity loss should be assessed (Gabriel *et al.*, 2010). For example, the uneven distribution of biodiversity can result in contrasting spatial trends, which may be masked by aggregated indicators and reduce the importance of localised effects (Galewski *et al.*, 2011). Disaggregating the BII provides some unique insights that might be masked at higher scales. For example, the current overall BII score indicates an approximate 17% decline in population abundance compared to pre-colonial times which may be considered acceptable (see Biggs *et al*., 2008), given that approximately 5% of the area is formally protected within the region (Driver *et al*., 2012). However, disaggregating the BII to smaller scales reveals much greater declines at both local and ecological scales. The state of biodiversity and reported impacts are closely linked to the selection of meaningful boundaries which should reflect the scale at which decisions are made. For example converting just 3% of degraded arable land reduces the BII reported at the provincial scale by <1% while at smaller scales, impacts are much larger.

The state of biodiversity and reported impacts are closely linked to the selection of meaningful boundaries which should reflect the scale at which decisions are made (Galewski *et al.*, 2011). For the BII to detect impacts at the aggregated scale (e.g. for the whole province) requires the conversion of fairly large areas of land at smaller scales. These challenges are evident for other aggregated indicators such as the widely used Living Planet Index (Galewski *et al.*, 2011), which highlights some of the potential shortfalls of using compound indicators to report on biodiversity.

It is important to understand that regional differences are likely to exist and that using an aggregated indicator may mask these. In a region as diverse as the Eastern Cape, biodiversity loss should not be traded-off against areas where no biodiversity losses are likely to occur therefore requiring that local management be tuned in to local conditions (e.g. Henry *et al.*, 2008). This requires that regional thresholds are identified and that constant monitoring across different scales be included in BII reporting. However, there is no consensus on what these thresholds should be or how the decline in intactness relates to species richness.

When reporting the BII, biological meaning should only be ascribed at ecological scales, such as biome scale, whereas administration boundaries, such municipalities, can be used to convey the state of the biodiversity within a particular region. Decisions on acceptable trade-offs should be debated and assessed in relation to socio-political goals.

Perhaps, an acceptable future state of biodiversity should be linked to values that indicate viable populations. Or perhaps in line with the CDB targets a value of 10% below current terms must not be reached. Thresholds linked to the BII should be debated and set in accordance with the overarching goal of biodiversity conservation.

The large contribution of natural and semi-natural habitats (i.e. moderate-use areas) to the overall BII score requires that these conservation strategies be included in development planning (Chazdon *et al.*, 2009). These response factors relate to the protection of natural habitat outside of formally protected areas. There are areas that are widely available but also important for habitat conservation and ecological process should be considered in relation with biodiversity (Bennett *et al.*, 2009).

4.6.3. Taxa

These results show that the impacts of land-use change as a result of increased cultivation does affect different taxa in different ways. For example, mammals, plants and amphibians are sensitive to increased cultivation whereas the impact on birds is minimal – this despite birds being widely used as indicators for biodiversity change (Biggs *et al.*, 2008, Butler *et al.*, 2007). The decline in plant abundance is linked to the large amount of land converted to cultivation. These relationships are also a factor of expert estimates of population impacts (I_{ijk}) as a result of past land-use changes, which may not necessarily apply under biofuel production scenarios.

4.6.4. Limitations

One important aspect of the BII is the ability to link land-use impacts to species population data. Although derived from expert opinion, this approach is considered appropriate (at least for broad-scale planning), and greatly reduces the amount of work required. However, two important lessons are drawn from this research. The first is that the impact factors used in this study are based on current agricultural understanding and could be inaccurate when applied to biofuels (Hui *et al.*, 2008), especially since there are many different approaches to producing biofuels (Slade *et al.*, 2011). Differences in management regarding fertilizer use, tilling, burning regimes and harvesting of products could result in differences to the current impact factors used in this study (Firbank, 2008, Lal & Pimentel, 2007). The second is that the BII is acknowledged to be insensitive to detect changes to landscape patterns, such as increasing fragmentation or changing patch size (Scholes & Biggs, 2005). This is important as the size of remaining habitat and habitat pattern are closely linked to biodiversity conservation (Lindenmayer & Fischer, 2006). This is an important avenue of research and will be addressed in Chapter 5.

The BII has been criticised for addressing all species as being equal (Ewers *et al*., 2009). Endemic or phylogenetically unique species should be identified and possibly weighted differently within the study area. It is recommended that this be considered for future research. As an example, where sufficient data for important focal species exists, innovative spatial modelling allows for more transparent and robust understanding of potential biodiversity impacts (Overmars *et al*., 2014). Overmars *et al*., (2014) suggest that more work is needed in this field.

The BII provides insight into the conversion of natural land on biodiversity. However, other important factors are likely to impact on biodiversity. For example, many biofuel plant species are very likely to be invasive and this could substantially increase their overall impact on biodiversity (Barney & DiTomaso, 2008, Raghu *et al.*, 2006). Biofuels could be an avenue for the introduction of genetically modified organisms (e.g. maize, sugar beets or canola) or dedicated energy crops that have little history of domestication (Barney & DiTomaso, 2008, Firbank, 2008). Changes to the functional diversity, associated with the intensification of land-use (Flynn *et al.*, 2009) and the introduction of novel species could further affect interactions between ecosystem processes and biodiversity (de Bello *et al.*, 2010b). These impacts may be beyond the simplified screening indicators such as the BII, requiring more in depth consideration for the type of biofuels cultivated (Firbank, 2008, McGeoch *et al.*, 2009).

4.6.5. Conclusion

Estimating the impact of biofuels on biodiversity has proven to be rather challenging, and assessments are usually the product of time-consuming field surveys (e.g.Dauber *et* *al.*, 2010). The BII provides an overall index which can be used to simplify the various components of biodiversity. There is strong evidence to suggest that rapid development will erode natural resources and any method that can quickly communicate potential effects to policy makers should be welcomed. A distinct advantage of the BII is that in the face of limited biodiversity knowledge it possible to engage with policy to include biodiversity in decision making processes. The ability to disaggregate the index shows importance of local level impacts should be assessed at either administrative or ecological scales to provide insight on taxa and biome level impacts.

These results show that transforming remaining available land with agricultural potential within the Eastern Cape could have substantial impacts on biodiversity. The BII also captured the effect of conserving habitat outside of protected areas. More importantly, the effects of these actions could be realized at the provincial scale and at smaller administrative scales. Overall, the BII provides a useful method for screening land and biodiversity resources to facilitate planning and to reduce unnecessary biodiversity losses.

Chapter 5: Examining the effect of landscape fragmentation associated with biofuel production using the Biodiversity Intactness Index

5.1. Abstract

In the Eastern Cape Province of South Africa, large areas of arable and marginal land have been declared suitable for biofuel production. The Biodiversity Intactness Index was used to determine potential biodiversity impacts of converting natural land to biofuel production. However, the BII is known to be insensitive to the effects of fragmentation, which is also considered as a large driver of biodiversity loss. First I analysed the relationships between the BII and fragmentation indicators derived from the landscape fragmentation package, FRAGSTATS. These indicators were number of patches, patch size, percentage of landscape, and largest patch index. The fragmentation indicators revealed an increase in the number of patches as arable and marginal land was converted. This was associated with a decrease in the BII as remaining percentage of land and largest patch index declined following conversion. A Revised-Biodiversity Intactness Index (R-BII) was developed and tested. This includes the effect of fragmentation on species populations by incorporating the size of remaining habitat patches using published data. Applying the R-BII to the land-use change scenarios resulted in a further decrease in BII scores between 7-10%. Results between correlations of R-BII and fragmentation indicators increased significantly compared to the BII. These results suggest that the potential increase in land-use change to produce biofuels will also increase fragmentation of natural habitats, the effect of which is captured by the R-BII. These results show that while the BII is an effective biodiversity indicator that tracks large scale changes in land-use and its effects on biodiversity, it does not adequately address the effects of habitat fragmentation at smaller scales. The R-BII is based on the assumptions of the original BII and requires available or easily obtainable data to report on the status of biodiversity. The R-BII complements the original BII and provides an additional measure to biodiversity assessment.

Keywords: Fragmentation, Biodiversity Intactness Index, Indicators, Land-use

5.2. Introduction

Biodiversity is being lost at an alarming rate. The Convention on Biological Diversity (CBD, 2003) has facilitated the development of monitoring and assessment indicators that are able to track changes in biodiversity over time. A major cause of biodiversity loss is habitat loss, accompanied by increasing levels of fragmentation of remaining natural habitats (Lindenmayer & Fischer, 2006, Sala *et al.*, 2000). One of the major concerns of this decade is the potential expansion of proposed biofuel production and the effect this will have on increasing rates of land-use change and habitat loss (Sala *et al.*, 2009, Turner *et al.*, 2008). This is recognised as a large potential threat to biodiversity. Questions need to be asked whether existing biodiversity monitoring indicators can adequately identify the impacts of biofuels as they relate to habitat loss and increasing levels of fragmentation. This will allow for effective responses to reduce biodiversity loss.

Few studies have considered the integrated effects of fragmentation on biodiversity in bioenergy studies (Immerzeel *et al.*, 2014). Habitat fragmentation occurs when large contiguous areas are divided into smaller units that are separated by the introduction of different forms of land-use that replace original vegetation (Bennett & Saunders, 2010, Fahrig, 2003). Specifically habitat fragmentation relates to the subdivision of natural vegetation which is measured by the number and the size of remaining habitat patches (Lindenmayer & Fischer, 2006). Increased fragmentation can lead to the sub-division of patches and changes to the species composition and population sizes of remnant patches (Ewers *et al.*, 2009). The size of the patch can therefore act as a good surrogate for the effects on ecological function (e.g. dispersal or reproduction rates) and impacts on species composition (Alkemade *et al.*, 2009, MacArthur & Wilson, 1967, Millennium Ecosystem Assessment, 2005).

There have been many studies to determine the effects of fragmentation on biodiversity. The general consensus is that large and connected areas are likely to support larger populations than smaller fragmented areas (Fahrig, 2003). However, the broad range of taxa studied has resulted in fragmentation having both negative (on most species) and positive effects (on fewer generalist species) (Ewers *et al.*, 2010). For example, transformed areas can negatively affect the movement of some native species, but at the same time facilitate the spread of invasive species (Le Maitre *et al.*, 2004).

The Biodiversity Intactness Index (BII) is a popular biodiversity indicator originally developed to report biodiversity information at a high level for decision making purposes (Scholes & Biggs, 2005) and has proven itself useful for estimating the effects of potential land-use change scenarios on biodiversity (Biggs *et al.*, 2008). The indicator can be applied in a GIS to address the spatial changes in land-use (Nickless & Scholes, 2009). It was developed and used in southern Africa (Biggs *et al.*, 2006) where a lack of data on population abundance is supplemented by expert knowledge on specific taxa. This aspect of the indicator unlocks the potential for many countries without sufficient information on biodiversity to track and monitor biodiversity. This is especially important as many developing countries (e.g. India, Brazil, and countries in Africa) are being suggested as locations for biofuel production (Lapola *et al.*, 2009, Von Maltitz & Brent, 2008). This provides the opportunity for biodiversity to be included in national policy decisions. The BII presents a framework for a general cause and effect relationship to be applied to the effect of land-use change on various taxa (Scholes & Biggs, 2005).

Previous work on the effect of habitat fragmentation on biodiversity can be grouped according to two major effects (Fahrig, 2003). These two effects are habitat loss and fragmentation *per se*. While the BII intuitively accounts for habitat loss, a recognised disadvantage of the indicator is the potential insensitivity to the effect of fragmentation *per se* (Scholes & Biggs, 2005). While this has not been tested, it presents a particularly challenging problem for which advances are needed.

The BII could be a useful tool to assess changes to biodiversity as a result of increased biofuel production and could be used to report on the potential impacts at early stages of development to inform planning. Identifying the potential areas for biofuel production could benefit from estimating the resulting changes in biodiversity, should these areas be converted, to facilitate adequate planning (Koh & Ghazoul, 2010). The BII provides an adequate link to land-use change and biodiversity that is easy to interpret and allows for biodiversity goals to be monitored.

Like many governments worldwide, the South African government is planning to grow biofuels (Department of Minerals and Energy, 2007). Future landscape changes and land-use transitions are likely to be non-random as the best remaining habitat will be selected for cultivation. This suggests that a spatial approach can be used to assess potential biodiversity impacts as well to determine the effects of changes in landscape patterns. There have been many examples of how excessive growth of biofuel production can reduce biodiversity integrity. For example, in Malaysia large-scale palm plantations have reduced the extent of tropical forest cover threatening biodiversity (Fitzherbert *et al.*, 2008, Koh, 2007b). Palm oil plantations have lower biodiversity value and in some cases act as barriers preventing connectivity to natural forests, increasing the isolation of habitat patches (Koh *et al.*, 2009). Methods that allow for impacts to be adequately assessed may prevent unnecessary losses.

In this paper I address the limitation of the BII and utilise hypothetical land-use change scenarios based on realising the agricultural potential of the Eastern Cape province of South Africa. Recognising the limitations to the BII, I set out to 1) determine the landscape pattern and fragmentation indicators for different land-use scenarios, 2) assess correlation between the BII and fragmentation indicators, 3) incorporate the effect of patch size within the BII, and finally, 4) determine the effect of fragmentation on the BII results.

5.3. Materials and Methods

5.3.1. Study Area

The study area was the Eastern Cape province of South Africa. This region is relatively poor and is expected to undergo large-scale changes in the future as a result of increased cultivation and development which aims to reduce poverty and increase human well-being (Department of Minerals and Energy, 2007). Biofuels are expected to be grown in this region to meet South Africa's growing fuel demands as well as for export purposes (Department of Minerals and Energy, 2003). The Eastern Cape Province is biologically diverse and includes five of South Africa's seven terrestrial biomes (Mucina & Rutherford, 2006).

5.3.2. Scenario definition and land-use change predictions

Land-use change scenarios were developed based on the agricultural potential and involved converting all available arable and marginal land to cultivation. The suitability of cultivated crops was not considered in this paper but could include suitable biofuel species, cereals or other commercial agricultural crops. Marginal land was also considered for its agricultural potential based on the increasing demand for biofuels to be grown here and to reduce direct competition with food production (Pimentel *et al.*, 2009, Plieninger & Gaertner, 2011).

Environmental information needed to determine available land within a GIS, included information on slope, land-use (including roads) and land capability. Slope was derived from a 90 m digital elevation model and areas with a slope greater than sixteen degrees were excluded (Fischer *et al.*, 2007). Six land-use classes were derived from the National Land Cover 2000 for South Africa (CSIR, 2003) namely: *Protected, Urban, Cultivated, Plantation, Degraded* and *Moderate Use*. The land capability map of South Africa identifies similar land units based on the potential production capability while also considering the limitations and hazards within a particular region (i.e. soils, risk of erosion, physical terrain constraints, climate) (Schoeman *et al.*, 2000). Land capability classes were reclassified as follows: Classes 1-4 were designated as *arable,* Classes 5-6 were designated as *marginal* while Classes 7-8 were characterised as *excluded*. These classes represent land productivity and can be related to potential cultivation success (Wessels *et al.*, 2003). All maps were resampled to a cell size of 100 m.

All three GIS layers were combined to identify available areas with agricultural potential. For the purpose of this study, suitable land identified as either arable or marginal was selected from available *Degraded* or *Moderate Use* land-use classes. The scenarios consisted of converting incremental units by reclassing them as *Cultivated* within a GIS based on the following rules: 1) degraded-arable; 2) degraded-marginal; 3) moderate use – arable; 4) moderate use – marginal; 5) all arable; 6) all marginal; 7) all degraded; 8) all moderate use; 9) all available land. This resulted in nine future land-use scenarios representing increasing land requirements for cultivation. These maps also highlight the maximum potential for change based on natural rain-fed agricultural conditions. The scenarios exclude the expansion of urban or infrastructural developments. Results from the scenarios were compared to a baseline using the National Land Cover of 2000 (CSIR, 2003) as a reference condition.

5.3.3. BII calculation

The methodology for calculating the BII has been extensively discussed in previous publications (Biggs *et al.*, 2007, Biggs *et al.*, 2008, Scholes & Biggs, 2005) and only the most important aspects of the algorithm are highlighted below. For a specific biome, the BII is calculated as:

$$\boldsymbol{B}\mathbf{II} = \frac{\left(\sum_{\mathbf{i}}\sum_{\mathbf{j}}\sum_{\boldsymbol{k}} \boldsymbol{R}_{\mathbf{ij}}\mathbf{A}_{\mathbf{jk}}\mathbf{I}_{\mathbf{ijk}}\right)}{\left(\sum_{\mathbf{i}}\sum_{\mathbf{j}}\sum_{\boldsymbol{k}} \boldsymbol{R}_{\mathbf{ij}}\mathbf{A}_{\mathbf{jk}}\right)} \qquad \textbf{Equation 5.1}$$

where R_{ij} is the richness of species group *i* in ecosystem *j*, A_{jk} the area of land-use *k* in ecosystem *j* and I_{ijk} the population impact on species group *i* of land-use j. The data for the impact scores for the species groups mammals, birds, reptiles frogs and plants were associated with six land-use classes (*Protected, Urban, Cultivated, Plantation, Degradation* and *Moderate Use*) and were taken from the original BII calculation (Scholes & Biggs, 2005). The BII provides a measure of population integrity and is the average impact across all available taxa. The BII was calculated for each scenario and was based on changes to the area within the land-use classes.

5.3.4. Fragmentation assessment

The National Land Cover 2000 (CSIR, 2003) map was used to derive the extent of natural (*Protected, Degraded, Moderate Use*) and non-natural (*Urban* - including roads, *Cultivated, Plantation*) areas for each of the scenarios. The following commonly used pattern indices were calculated; *patch area, number of patches*, *percentage of landscape* and *largest patch index* using the landscape fragmentation package, FRAGSTATS 4.1 (Mcgarigal *et al.*, 2012). The metric number of patches, summarised to the landscape scale, was used to analyse the effect of increased cultivation on fragmentation within remaining natural areas. A relative increase in the number of patches is considered unfavourable as the landscape becomes more fragmented. This represents an adequate measure of fragmentation, as the landscape size is kept constant for each scenario (McGarigal et al 2002). Fragmentation was expected to increase within natural areas at the landscape level as more land was converted to cultivation.

5.3.5. BII and fragmentation analysis

Fragmentation maps were characterised by patch size based on the concept of minimum viable area needed to maintain viable species population (Table 5.1). To demonstrate the effect of fragmentation on biodiversity intactness values derived by (Alkemade *et al.*, 2009) were used, who determined the relationship between patch size and mean species abundance based on a review of published literature. The averaged values were extracted from an existing global biodiversity model, GLOBIO 3 (Alkemade *et al.*, 2009) and used in subsequent calculations. Following Table 5.1, patch area of remaining natural fragments was assigned a proportionate value between 1 and 0.2 that was multiplied using the BII algorithm to produce a *Revised-BII* (R-BII) score for same cell within ARCGIS (Equation 5.2). This equation was applied in ARCGIS and followed the raster method for calculating the BII (Nickless and Scholes, 2009). The modified BII equation (R-BII) is an area sensitive indicator that includes the effects of patch size in the assessment of biodiversity intactness. Fragment size within non-natural areas was ignored in the calculation.

$$\textbf{R-BII} = \frac{\left(\sum_i \sum_j \sum_k R_{ij} \text{ x } (A_{jk} \text{x Fragemtation Index}) \text{ x } I_{ijk}\right)}{\left(\sum_i \sum_j \sum_k R_{ij} A_{jk}\right)} \qquad \textbf{Equation 5.2}$$

The null hypothesis of no relationship between the BII and fragmentation indicators was tested at two spatial scales. At the provincial scale correlations were tested based on the summary statistics of BII and fragmentation indicators. At the local scale correlations between BII, R-BII and patch area (see Table 5.1) were based on a 10000 point sample for each scenario. The spearman rank correlation coefficient was used to determine relationships between the two independent indicators. Multiple linear regression analysis was used to determine the importance of fragmentation indicators in predicting the response of either the BII or the R-BII. The predictors used were Number of Patches (NP), Percentage of Landscape (PLAND) and Largest Patch Index (LPI).

Table 5.1: The relationship between area and corresponding fraction of species assumed to meet the minimal area requirement developed by Alkemade *et al.* (2009) for the GLOBIO3 Framework.

Patch size (Km^2)	Fraction of species that will persist over time*
<1	0.2
<10	0.6
<100	0.7
<1000	0.9
<10,000	0.95
>10,000	1

*This was used to represent the fragmentation index in the R-BII calculations

5.4. Results

5.4.1. Fragmentation assessment

Fragmentation indicators for each scenario along with associated BII scores are shown in Table 5.2. Overall, decreasing the remaining percentage of natural land resulted in an increase in fragmentation with respect to an increase in the number of patches. The largest patch index also declined from 2.4% to 1.8%. The current configuration of natural areas represents the least fragmented landscape while converting all available *Arable* and *Marginal* land increased the number of patches by 890% within natural areas. The conversion of *Marginal-Moderate Use* land increased the number of patches by 2000% (Table 5.2).

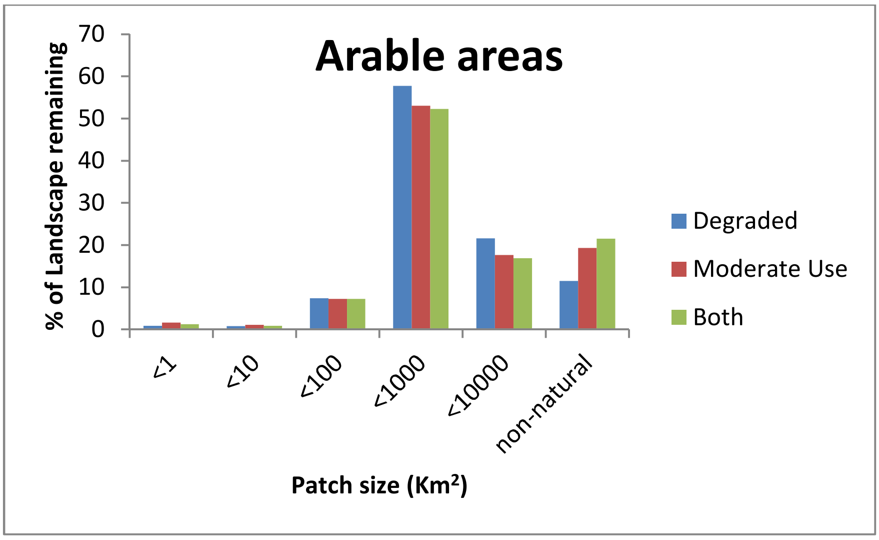
Biophysical characteristics (i.e. soil, slope or rainfall) results in large parts of the Eastern Cape that are excluded from having agricultural potential. As a result 50% of the natural areas remain unaffected from the threat of cultivation. Similarly, the number of patches per scenario is dependent on the physical landscape configuration, as well as the total available area per scenario. For example, converting degraded land or arable land, which have smaller areas, results in fewer fragments than converting *Moderate Use* or *Marginal* areas, which have significantly larger areas Table (5.2).

Figure 5.1 shows the percentage area remaining grouped by patch size for each scenario (Table 5.1). This shows that the proportion of large patches is reduced, while the proportion of smaller patches increases in frequency under extreme land conversion scenarios. Most of the landscape comprises of patches between 100 – 1000 Km^2, of which a large proportion are marginal or not suitable for conversion. There were no patches larger than 10000 Km^2 across all scenarios (Figure 5.1). Large natural areas were reduced as these were bisected by the national road network.

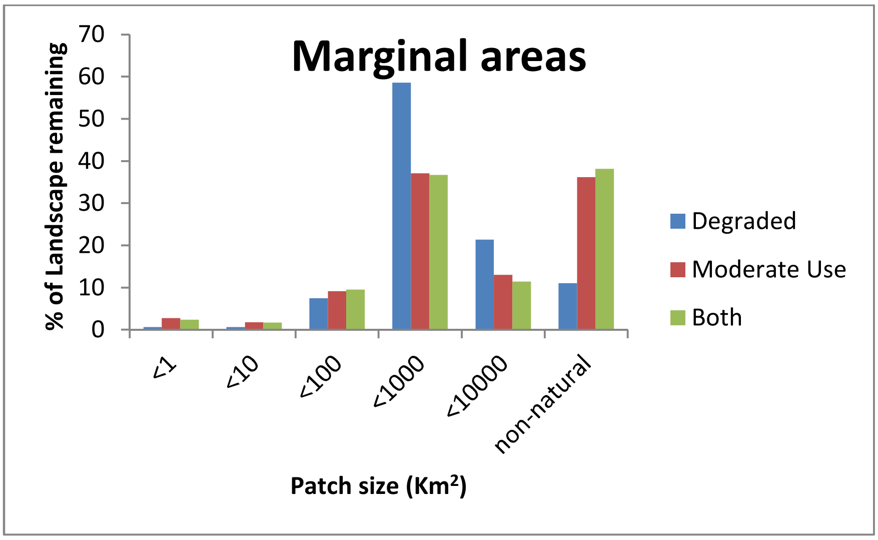
Correlations between fragmentation indicators shows that the *number of patches* was negatively correlated with *percentage natural land* (r = -0.97) and *largest patch index* (r = -0.84) whereas *percentage natural land* is positively correlated with *largest patch index* (r = 0.93).

Table 5.2: Results of fragmentation indicators for remaining natural areas following the conversion of available land to cultivation according to the scenarios developed in this study. The Biodiversity Intactness Index scores are also presented.

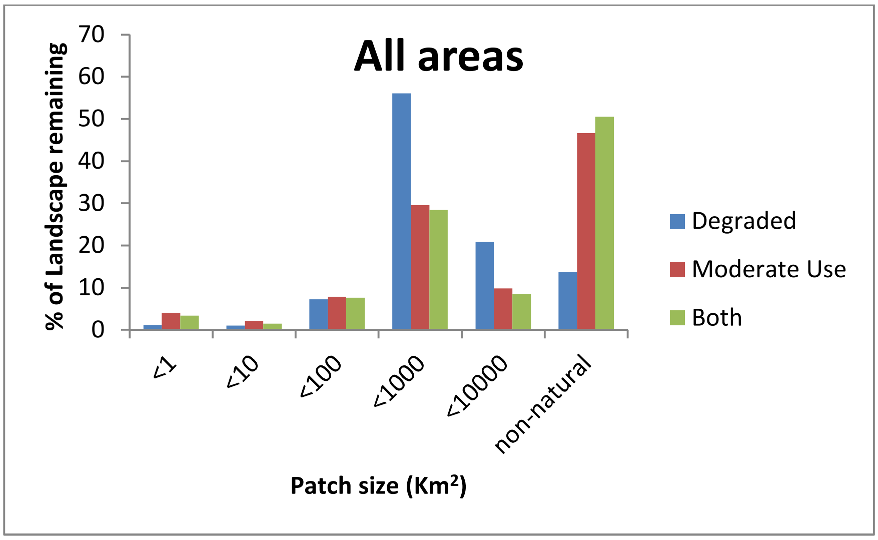
Agricultural Potential	Land-use scenario	Number of patches	Percentage natural land	Largest patch index	BII
	Present land-use	6682	90.9	2.4	83.1
Arable	Degraded	18419	88.5	2.4	82.4
	Moderate use	48749	80.9	2.4	75.4
	Degraded and Moderate use	59372	78.5	2.4	74.7
Marginal	Degraded	12891	88.9	2.4	82.5
	Moderate use	72121	63.8	1.8	66.9
	Degraded and Moderate use	77405	61.9	1.8	66.3
Arable and Marginal	Degraded	25552	86.5	2.4	81.8
	Moderate use	119049	53.9	1.8	59.2
	Degraded and Moderate use	134867	49.5	1.8	57.9



a)



b)



c)

Figure 5.1: The percentage of natural area remaining grouped according to patch size (Km^2) for each of the land suitability classes a) arable b) marginal and c) both arable and marginal.

5.4.2. Difference between normal BII and R-BII

Accounting for fragmentation and patch size within R-BII resulted in the provincial BII scores decreasing between 6% and 9% for the scenarios evaluated (Table 5.3). The current BII score of ~83% was adjusted to ~76% using the R-BII.

Table 5.3: The difference and percentage decrease in the reported intactness between the Biodiversity Intactness Index (BII) and the revised biodiversity intactness index (R-BII). The R-BII accounts for the effect of fragmentation within the BII algorithm.

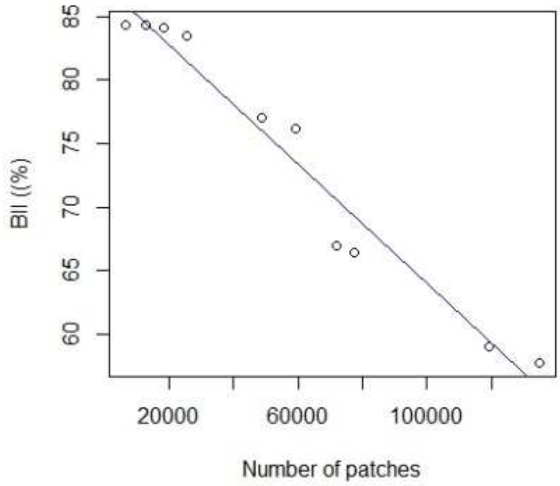
Land capability	Land-use	BII	R-BII	% decrease
	Present land-use	83.1	75.5	7.6
Arable	Degraded	82.4	75.3	7.1
	Moderate use	75.4	68.8	6.6
	Degraded and Moderate use	74.7	68.5	6.2
Marginal	Degraded	82.5	75.6	6.9
	Moderate use	66.9	57.9	9
	Degraded and Moderate use	66.3	57.6	8.7
Arable and Marginal	Degraded	81.8	74.1	7.7
	Moderate use	59.2	51.1	8.1
	Degraded and Moderate use	57.9	50.9	7

5.4.3. *Relationship between the BII and fragmentation indicators*

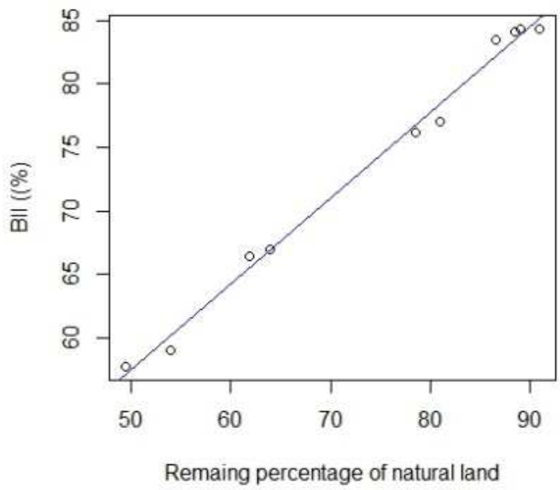
At the Provincial scale, the BII scores were negatively correlated with *number of patches* (r = -0.97, P<0.001) and positively correlated with both *percentage natural land* (r = 0.99, P<0.001) and *largest patch index* (r = 0.95, P<0.001) (Figure 5.2). All variables are highly correlated with each other resulting in effects of multicollinearity as a both BII and fragmentation metrics are sensitive to changes in area. At this large scale, the effects of habitat loss are captured by both indicators.

The results of the linear regression analysis indicates that the percentage of remaining natural land to be a better variable to explain BII than NP or LPI (Table 5.4).

At the landscape scale, Spearman rank correlations with patch area were higher for R-BII than BII alone (Table 5.5). The R-BII resulted in a reduction in BII scores that had the effect of reducing the variation within the cells sampled. The effect of including patch size within the BII algorithm resulted in an appropriately scaled relationship, whereby large patches maintained high BII scores while the BII scores of small patches were reduced.



a)



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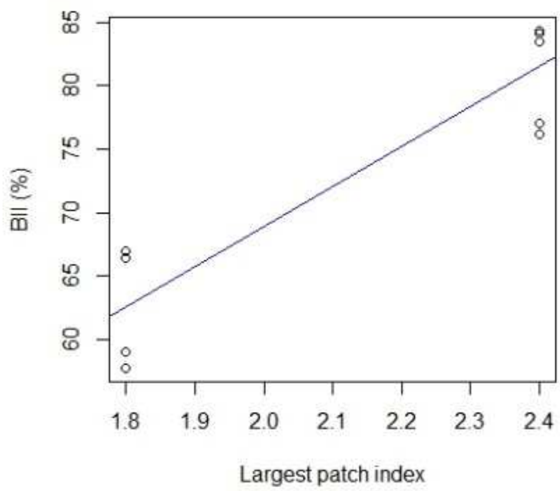


c)

Figure 5.2: Relationships between BII and fragmentation indicators for a) number of patches; b) percentage natural land and c) largest patch index

Table 5.4: Results of the linear regression analysis for the BII and fragmentation indicators.

Model	Factors	coefficient	R^2	NP	PLAND	LPI
BII	NP + PLAND*	33.4***	0.99	0.31	<0.001	-
	LPI + PLAND*	26.02***	0.99	-	<0.001	0.377
	NP+PLAND+LPI	53.5***	0.99	0.48	0.91	0.59
R-BII	NP + PLAND	1.3	0.99	0.65	<0.001	-
	LPI+PLAND	4.3	0.99	<0.001	-	<0.001
	NP+PLAND+LPI	47.2	0.99	0.41	0.90	0.34

*** P<0.001

Table 5.5: Results of Spearman Rank correlations between the BII, R-BII and patch size as characterised by a Fragmentation Index**.**

		BII			R-BII		
	Scenario	r	df	p	r	df	p
Arable and Marginal	Degraded	0.13	8460	<0.001	0.82	8460	<0.001
	Moderate use	0.51	5267	<0.001	0.69	5267	<0.001
	Degraded and Moderate Use	0.54	4814	<0.001	0.91	4814	<0.001
Arable	Degraded	0.06	8658	<0.001	0.72	8658	<0.001
	Moderate use	0.32	7920	<0.001	0.73	7920	<0.001
	Degraded and Moderate Use	0.31	7645	<0.001	0.78	7645	<0.001
Marginal	Degraded	0.1	8707	<0.001	0.25	8707	<0.001
	Moderate use	0.37	6286	<0.001	0.82	6286	<0.001
	Degraded and Moderate Use	0.37	6088	<0.001	0.81	6088	<0.001

5.5. Discussion

The effects of land-use change on biodiversity have been well documented (Bennett & Saunders, 2010, Lindenmayer & Fischer, 2006). Both habitat loss and habitat fragmentation are significant contributors to the loss of biodiversity (Alkemade *et al.*, 2009, Fahrig, 2003, Millennium Ecosystem Assessment, 2005). The aim of this study was to explore the effects of transforming habitat for biofuel production on landscape pattern and the associated impacts on biodiversity. The BII was chosen as an appropriate indicator to monitor changes in biodiversity using a spatially explicit approach. In particular the BII was examined to determine its efficacy in assessing the impact associated with landscape and habitat fragmentation.

From this analysis I find that the conversion on untransformed land results in large habitat loss, as well increasing habitat fragmentation. The BII was shown to be insensitive to habitat fragmentation, however the R-BII does account for fragmentation of landscapes. These findings are discussed in more detail below.

5.5.1. Relationship between fragmentation indicators and BII

A simple approach to modelling land-use change was used in this study, but the principles are based on those that guide more complicated land-use models (Trisurat *et al.*, 2010). An increase in cultivation was examined from a biophysical perspective. This resulted in habitat fragmentation being caused by the spatial arrangement of environmental variables such as slope, soil fertility and rainfall (Carvalho *et al.*, 2009, Li *et al.*, 2012). However, social factors also play an important part in determining the location of cultivation. As expected, the BII was correlated with significant reductions in remaining natural land. Consequently, the fragmentation indicators *number of patches, percentage natural land* and *largest patch index* were highly correlated with the BII when all scenarios were considered together. Both the BII and fragmentation indicators track the loss of large areas of natural habitat. These indicators are identified as measures of landscape condition and place a higher weighting on the presence of natural areas and finding correlations between these indicators was anticipated.

5.5.2. Consequences of fragmentation for biodiversity intactness

While the BII is an effective biodiversity indicator aimed to communicate biodiversity information to policy makers, the results show that it does not adequately address the effects of habitat fragmentation at local scales. In a recent comparison Ewers *et al.* (2009) found the BII to under-report community change in a fragmented landscape by approximately 50%. This is because the BII is calculated on the total remaining habitat area and ignores the patch size or number of patches. The authors of the BII acknowledged this limitation to the index (Scholes & Biggs, 2005).

Fahrig (2003) hypothesised that that biological impacts of habitat loss outweigh the impacts of habitat configuration. The BII functions on this approach and only when the amount of land-use change at smaller scales are of large enough extent, are the effects noticed within the overall BII at larger scales. The land-use change scenarios depict this phenomenon at the provincial scale which supports Fahrig's hypothesis. Even at the local scale, BII decreased in relation to patch size but the relationships were weak.

However, the inability of the BII to account for processes that are related to patch area was clearly expressed when R-BII appeared to explain more than 60% of the variation for all scenarios apart from one. The R-BII incorporates the effects of increased fragmentation particularly if patch size decreases. By scaling smaller patches to have lower BII scores the index accounts for the minimum viable area needed to maintain species populations. This approach also accounts for the relatively large effect that fragmentation has on species populations (Koper *et al.*, 2007). Not accounting for the fragmentation effect ignores the spatial aspects of habitat configuration resulting in high BII scores assigned to any patch size.

R-BII scores were also lower for each of the scenarios developed. For example, the baseline of the current landscape pattern reduced by 10%, indicating that the effects of fragmentation are inherent in the landscape. This suggests that patch size has been reduced by existing non-natural landscape features (i.e. road, urban, cultivated areas). Not accounting for these features provides an inflated estimation of biodiversity intactness.

5.5.3. Implications for assessing the impacts of biofuels

Estimating the impacts of biofuel production on biodiversity requires indicators that include a spatial component and that are sensitive to the effects of fragmentation and remaining patch size of natural areas (Koh & Ghazoul, 2010, Overmars *et al.*, 2014). Given that biofuel production systems range from small holder farming (1-10 ha) to large scale commercial farming (100-1000's ha) (Blanchard *et al.*, 2011, von Maltitz *et al.*, 2012), the impact of landscape fragmentation will vary. The advantage of the R_BII will be in detecting potential differences between production systems. The consequences of many small scale farms are likely to increase fragmentation of habitats whereas large scale plantations will likely increase habitat loss. The R-BII allows for the effects of patch size to be incorporated into local scale assessments, for providing a more realistic assessment of biodiversity impact. This is also an important requirement of compound indicator such as the BII, for local scales assessments are normally considered a limitation (Czúcz *et al.*, 2012).

5.5.4. Priorities for future research

The following avenues for future research are suggested:

To illustrate the effect of fragmentation on the BII, the minimum area values were extracted from the GLOBIO-3 framework (Alkemade *et al.*, 2009). Although these values were extracted from a wide literature review, the use of these values might not apply under all conditions. According to Scholes and Biggs (2005), using the best available data provides a baseline from which to work with. There is potential to increase the accuracy of the technique by applying fragmentation values that may be better suited to the habitat type analysed. Furthermore, minimum viable areas were equally assigned to all taxa. It might be more valid to account for fragmentation in a taxa specific manner, especially since species react in different ways to habitat loss and fragmentation (Bennett & Saunders, 2010). However this information is likely to exist for only a few well studied species, resulting in the exclusion of many taxa in future studies (e.g.Stoms *et al.*, 2011). Just like the BII, these values could be calibrated using expert opinion.

If the R-BII is to be implemented then further research should be directed at the benefits of a single indicator vs. a comprehensive set of biodiversity indicators. Recently indicator approaches that make use of multiple, but complementing indicators have been identified to convey meaningful information on biodiversity (de Bello *et al.*, 2010b, Vačkář *et al.*, 2012). Comparisons between the indicator approaches could provide information on time, costs and the ability to inform decisions. Doing so would benefit the identification of the strengths and weaknesses of different approaches, which are currently lacking.

5.6. Conclusion

Biofuel production poses a significant threat to biodiversity as the area under cultivation is likely to increase, driving habitat loss and increased levels of fragmentation of natural habitats. There is a need for biodiversity indicators that can track the drivers of biodiversity loss to provide responses to minimise the losses of future land-use changes. The BII is an easy to use indicator that provides clear information on the effects of land-use change on biodiversity. However, the effect of landscape pattern is not adequately captured within index. A revised BII was derived that integrates habitat fragmentation and reports more accurately the level of intactness for a region.

Combining techniques of spatial evaluation with biodiversity and fragmentation analyses provides a means to simulate the extent and locations of potential land-use change (Liu *et al.*, 2014) as well as assess potential changes in biodiversity (Wessels *et al.*, 2003). This approach can contribute to the monitoring of landscapes and provide environmental decision-makers with information needed to maintain landscape configuration that are important for biodiversity conservation. Adapting this approach to biodiversity assessment and coupling it with the more rigorous approach of systematic conservation planning, may help distinguish unique management efforts needed to address the effects of habitat loss and habitat fragmentation under different circumstances (Koper *et al.*, 2007). Landscape planning should seek to design future landscapes that can accommodate multi-functional systems to find a balance between development and biodiversity objectives. The R-BII could be used as a bench mark for which landscape configurations are assessed. This revised index can facilitate decision making on land-use change by providing biodiversity indicators that can incorporate an understanding of the spatial arrangement of habitats.

Part III:

Plant functional traits

Chapter 6: Examining the impacts of introduced species for the production of biofuels on functional diversity and ecosystem processes

6.1. Abstract

Biofuel production and its use are expected to increase in the future, potentially driving large scale land-use changes. Land-use change results in changes to plant diversity and the functional composition of communities that underpin the various ecosystem processes and ecosystem services. However, it is uncertain how the introduction of non-native species associated with biofuel production will affect ecosystem processes. I used a recently developed conceptual framework that links ecosystem functioning and plant traits in determining the effects of land-use change. This was carried out in an area previously identified as an ecosystem service hotspot in the grasslands of the Eastern Cape, South Africa. Leaf traits (leaf dry matter content, leaf nitrogen content and leaf phosphorous content) were used to characterise the functional composition of the landscape and to illustrate the difference between native vegetation and potential biofuel land-uses. Functional trait diversity was found to be greatly reduced when natural systems were converted to different land-uses. Differences between native and non-native traits could be ascribed to individual species as well as the abundance weighted mean value of traits within communities. There was large overlap between non-native and native trait values however, each biofuel species occupied a different location in multidimensional space. The results show that the conversion of unutilised arable land will result in large shifts in functional diversity that may reduce the capacity of an ecosystem service hotspot to deliver provisioning services associated with natural vegetation. This approach provides insight into the many challenges in integrating ecosystem services into planning and management.

Keywords: land-use, plant functional traits, ecosystem services, functional diversity indices

6.2. Introduction

Biofuel production and its use are expected to increase in the future (OECD-FAO, 2014). The benefits of biofuel production are attributed to potential climate regulation through the reduction in fossil fuel use (Hill *et al.*, 2006). However, there are numerous concerns relating to the negative impacts of biofuels on biodiversity (Fitzherbert *et al.*, 2008) and ecosystem services (Gasparatos *et al.*, 2011). Most of these concerns are centred on the added pressure to transform land needed to meet fuel production targets (Eggers *et al.*, 2009). Land transformation is a major threat to remaining biodiversity as agricultural production needs to keep pace with a growing human population and as mandatory biofuel targets around the world become increasingly popular (Eggers *et al.*, 2009, Lapola *et al.*, 2009, Li *et al.*, 2012).

There is significant evidence that the conversion of natural habitats to croplands usually results in losses to biodiversity, changing the functional characteristics of ecosystems and eroding their capacity to deliver essential regulating and supporting ecosystem services required for human well-being (Cardinale *et al.*, 2012, Naeem, 2009, Olden *et al.*, 2004, Vitousek *et al.*, 1997). However, little is known how biofuels will impact on ecosystem processes and ecosystem services (Gasparatos *et al.*, 2011). This requires better understanding of the impacts of land-use change on biodiversity, ecosystem processes, and ecosystem services.

First generation biofuel production requires the cultivation of crops from which sugar, starches and oils are used to make biodiesel and bioethanol (IEA 2010). More recently, the production of second generation biofuels requires cellulosic materials, derived from woody biomass, which are utilised in modern biofuel conversion techniques (Slade *et al.*, 2011). Producing biofuels therefore requires changing the structure of landscapes to grow feedstocks which include coarse grains, oilseeds and cellulosic materials typically in the form of woody biomass.

The impacts of land transformation can be assessed at multiple scales ranging from the broader landscape to smaller field scales (Firbank *et al.*, 2008). A previously developed framework by Parker *et al.* (1999), that integrates these scales, defines impact as the product of three factors, namely: range, abundance, and the per capita effect. This framework, suggests that the role of each factor must be understood to appreciate overall impacts. Calculating impacts at broader spatial scales (i.e. landscape) are easy enough with the use of a GIS or land-use change models (Blanchard *et al.*, 2014, Hellmann & Verburg, 2010, Lapola *et al.*, 2009). However, at smaller scales, such as the field scale, some of these impacts are not well understood (Dick *et al.*, 2013, Parker *et al.*, 1999). At this scale, the functional characteristics of communities may be affected by factors of abundance or per capita effect of introduced species, which need to be considered. Methods and theory are slowly being developed for this purpose (Lavorel *et al.*, 2011, Thiele *et al.*, 2010). This is especially important considering that the functional characteristics of species within communities underpin ecological processes and ecosystem services (Balvanera *et al.*, 2005, Diaz *et al.*, 2007, Kremen, 2005, Luck *et al.*, 2009).

This has been the focus of biodiversity-ecosystem functioning (BEF) research over the last few years and has yielded valuable insight into the functioning of ecosystems (Cardinale *et al.*, 2012)**.** For plants, one way to measure these fine scale interactions is to focus on mechanisms that drive ecological processes, such as the functional traits of species (de Bello *et al.*, 2010a, Kremen, 2005). Species functional traits have been identified as the biological mechanism having the largest impact on ecosystem functioning (Diaz *et al.*, 2004). Functional traits are defined as components of organisms with any measurable morphological, physiological, phenological feature that influence ecosystem level processes (Violle *et al.*, 2007, Weithoff, 2003), while measures of functional diversity refer to the value and range of the traits measured (Tilman *et al.*, 1997).

Species functional traits have been linked to important ecosystem processes such as nutrient cycling (Lavorel & Garnier, 2002), litter decomposition rates (Garnier *et al.*, 2004), and biomass production (Lavorel *et al.*, 2011). These ecosystem processes are important for providing regulating and supporting ecosystem services such as biomass production and grazing potential (Lavorel *et al.*, 2011). Changes to the functional trait diversity (including the relative abundance of plant functional traits) within a community can have wide reaching implications for ecosystem processes (Diaz *et al.*, 2007, Hooper *et al.*, 2005). The mass ratio hypothesis (Grime, 1998) considers traits of the most abundant species to largely determine ecosystem processes. Following this hypothesis, the replacement of the dominant plant community, as happens when a crop replaces a native plant assemblage, is likely to have significant impacts on ecosystem processes and the ecosystem services derived for a parcel of land (Garnier *et al.*, 2007, Olden *et al.*, 2004). The functional characteristics of land-uses play an important part in maintaining ecosystem processes.

This understanding of ecosystem functioning provides a novel approach to assess the impacts of biofuels on ecosystems and the services provided. Biofuels production involves the movement and cultivation of new species over large areas and within many different ecological communities around the world (de Vries *et al.*, 2007, Simberloff, 2008). While impacts of biofuels have been assessed at broader scales, based on potential changes to land-use, the impacts of introducing species at a functional level are yet to be explored.

In this paper I show that the impacts of biofuels can be assessed using a functional traits approach. I begin by characterising the functional traits, as indicated by the dominant species, within the natural vegetation in a known ecosystem service providing hotspot in the grasslands of the Eastern Cape province of South Africa (Egoh *et al.*, 2008). Three existing land-uses (Cultivation, formal Woodlots and informal woodlots), defined by the presence of a specific non-native species which have the potential to be used as biofuel feedstocks, were assessed in terms of their functional trait composition. It was hypothesised that transformed plots would have altered functional trait values compared with natural vegetation, and that non-native species associated with each land-use type would be responsible for the change in functional trait values. If correct, then this method could be used to demonstrate the potential impact of biofuel production on ecosystem processes in an informed manner. These insights are needed to direct new approaches in predicting potential impacts to ecosystems and to understand the mechanisms that drive tradeoffs in ecosystem services.

6.3. Methods

The conceptual framework used in this study is outlined in Figure 6.1. It is understood that ecosystem services can be derived from a particular land-use and the functional characteristics associated with the species therein. Changes to land-use through the introduction of non-native species to increase a particular service will result in shifts of the functional characteristics that maintain the system as a whole. Existing land-uses, that contain potential biofuel species are identified within the grasslands of the Eastern Cape, are used to explore differences between functional characteristics of communities.

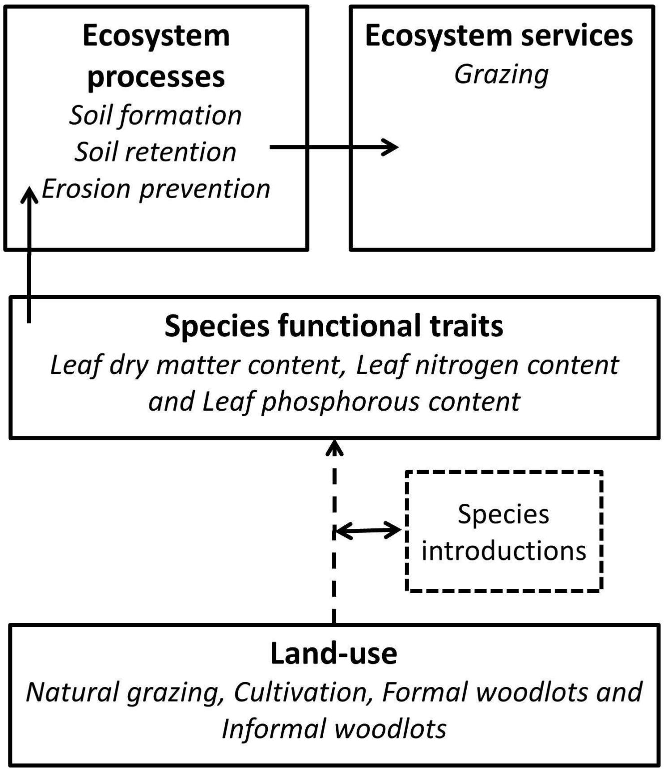


Figure 6.1: The conceptual framework used in this study showing the links between land-use and functional traits that underpin the ecosystem processes and ecosystem services. The analysis aims to identify the effect of land-use and the introduction of non-native species on community-level traits to estimate impact on ecosystem properties. The land-uses of Cultivation, Woodlots, and informal woodlots (consisting of *Acacia mearnsii*) were compared to natural grazing areas. Natural grazing is a key service provided by untransformed grassland vegetation types. This service is impacted on through the reduction of grassland communities by, areas of cultivation, plantations or by invaded areas (which have become informal woodlots). Figure adapted from (Lavorel *et al.*, 2011) and (Luck *et al.*, 2009).

6.3.1. Study area

The study area was located in the grassland biome of the Eastern Cape Province of South Africa (Figure 6.2). Study sites were selected to coincide with an ecosystem service hotspot based on previously mapped ecosystem services by Egoh *et al.* (2008). For this study, service hotspots identify areas where the services of soil-formation and soil-retention occur, two important regulating services that have major implications for above ground biomass, erosion control and the provision of grazing (Egoh *et al.*, 2008). These ecosystem services were originally identified by Egoh *et al*., (2008) in a GIS using broad scale biophysical data (e.g. vegetation cover, soil depth etc.) as a proxy for service provision. As land-use was not included in the identification of services, these services may be diminished through land transformation.

Six locations within the service provisioning area were sampled in order to characterise the functional composition of the four selected land-use classes (Figure 6.2). The sites were located within traditional rural communities where ecosystem services derived from the natural landscape are important for livelihoods and well-being. Access to the sites was facilitated by staff at Dohne agricultural research institute and the Department of Rural Development and Land Reform as approval by village chiefs and herdsman were needed prior to sampling. All sites were grazed year round by free-ranging cattle, sheep and goats.

The natural vegetation here falls within the Dohne sour veld grassland vegetation type (Acocks, 1988). The community is characterized by the structural dominance by a small number of grass species of which *Elionurus muticus, Heteropogon contortus, Sporobolus africanus, Themeda triandra* and *Tristachya leucothrix* are the most important (Du Toit & Aucamp, 1985). Woody species only occur in specialised areas and are not a feature of this landscape (Mucina & Rutherford, 2006). However, overgrazing, land degradation, fire and climate change are thought to contribute to bush encroachment within grassland communities.

The land-use classes used in the study were *free-range grazing*, *woodlots*, *cultivated areas* and *informal woodlots*. Free-range grazing areas were the largest of the land-use classes comprising of native grassland vegetation. Cultivated areas were identified by the presence of *Sorghum* species, Woodlots by the presence of *Eucalyptus* species and informal woodlots by the presence of *Acacia mearnsii*. These land-use classes were identified by the presence of key non-native species. These are discussed below.

Cultivated areas contained *Sorghum* which performs well as potential a bioethanol feedstock (de Vries *et al.*, 2010). On average, *Sorghum* yields are considered favourable and have been compared to maize and other bioethanol feedstocks (Ngepah, 2010). Sorghum has the added advantage of being planted in small-holder farms or large scale plantations and could be used to produce food or animal feed (Ngepah, 2010). Pilot projects to determine yield and viability have been established in the study area.

Woodlots containing *Eucalyptus* species are currently used by local communities to provide fuelwood and timber products such as poles (Ham, 2000). Eucalyptus species are also grown by commercial plantation owners in the region. The potential for *Eucalyptus* as a biofuel or biomass crop has been explored in Australia (Farine *et al.*, 2012). Where the wood quality is not of sufficient quality for the timber or paper industry, products could be used as cellulosic crops suited to biofuel or bioenergy production.

Informal woodlots were determined by areas invaded by *Acacia mearnsii.* These areas provide important timber resources to rural communities (Shackleton *et al.*, 2007). In South Africa *Acacia mearnsii* and other wattle species are estimated to cover a condensed area of more than 0.49 million ha (Le Maitre *et al.*, 2013). This is clearly a vast resource for biofuel production. Informal woodlots were divided into two categories to separate densely invaded areas from lightly invaded areas.

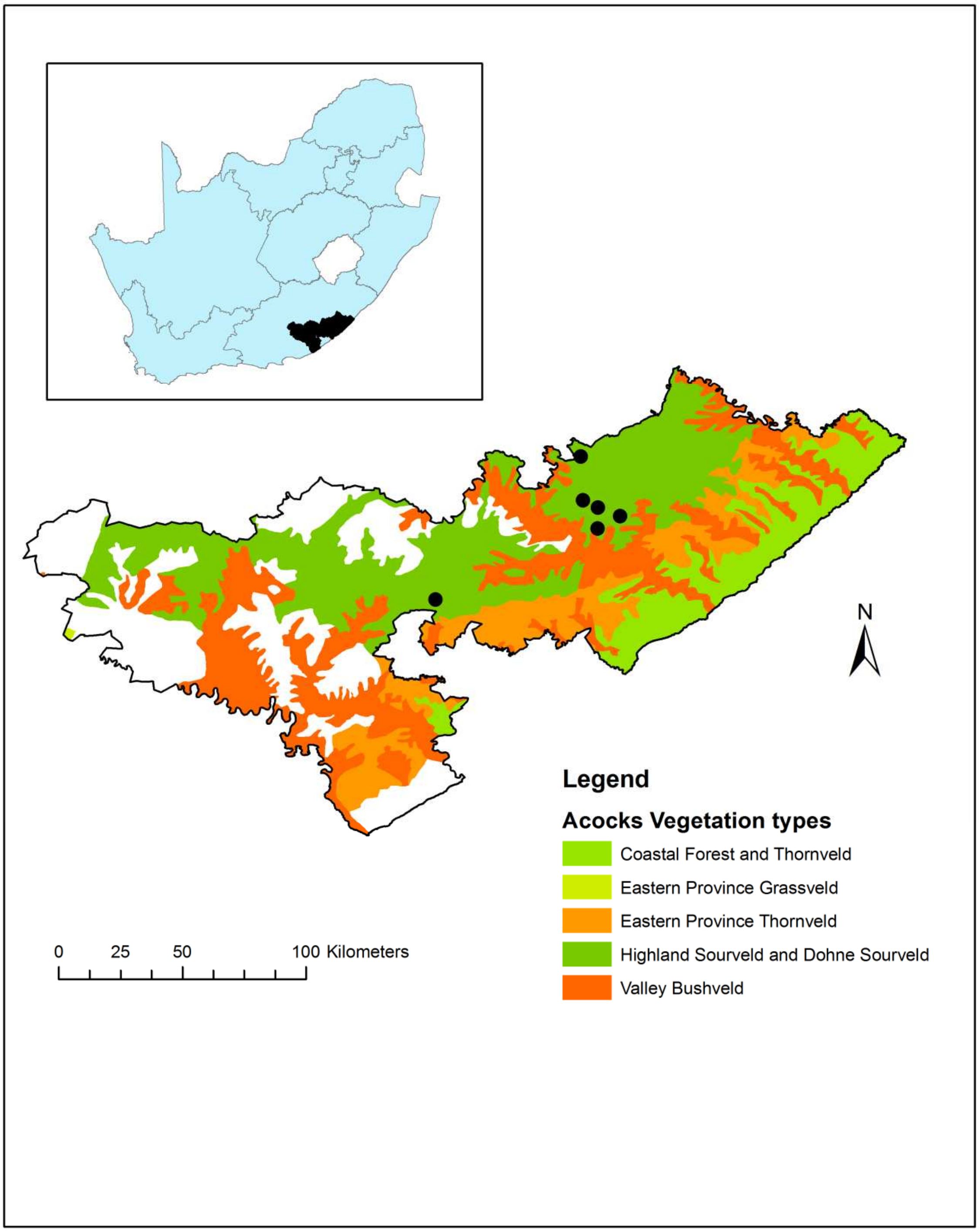


Figure 6.2: The location of the six study sites within the Dohne Sourveld grassland vegetation type located within the Amathole district municipality in the Eastern Cape Province. These areas also located within an ecosystem service hotspot identified by Egoh et al (2008). Veldtypes were taken from Acocks (1988).

6.3.2. Vegetation survey

Field sampling took place during February – March of 2012. Dominant plant species contributing to 80% cumulative cover was sampled for forty-one sampling locations distributed across the six sites (Figure 6.2). At each sampling location an average of three 1x1 m plots were surveyed. A plot size of 1x1 m was shown to be representative of larger scale plots for Eastern Cape grasslands (O' Connor & Kuyler, 2009). Additional measurements included plant average and maximum height, percentage bare ground, litter cover and growth form composition within plots. Plant specimens were identified using field guides and expert knowledge from colleagues at the Department of Agriculture situated at the Dohne Research Institute in Stutterheim. Both native and alien species were recorded within each plot.

6.3.3. Functional trait sampling

Ecosystem processes were controlled for by sampling within a single vegetation type, within the service providing area. The assumption was that ecosystem processes and ecosystem services would be uniformly spread throughout the study area. Table 6.1 outlines the links between leaf traits and ecosystem services selected for this study. The plant functional trait selection follows that of Lavorel *et al.* (2011) and de Bello *et al.* (2010a). The plant leaf traits sampled include: leaf dry matter content, leaf nitrogen content and leaf phosphorous content. These traits have been linked to important ecosystem properties related to biogeochemical cycling of nutrients (Table 6.1), as well as resource acquisition rates of plants (de Bello *et al.*, 2010a, Lavorel *et al.*, 2011, Quetier *et al.*, 2007). In particular, leaf dry matter content can be used as an indicator of resource conservation, and leaf nitrogen content and leaf phosphorous have been linked to rapid acquisition of resources (Freschet *et al.*, 2010). These traits have been used to indicate and differentiate plant strategies that are important for ecosystem functioning. Leaf sampling protocols followed (Cornelissen *et al.*, 2003) which identifies the sampling procedure, storage and amount of material required to obtain functional trait values. Leaf samples were taken from all dominant species within each plot and stored in a cool bag by day during sampling. Samples were weighed for wet mass in a lab following each sampling session. Each sample was then dried at 60 degrees for 48 hours and weighed to obtain a dry mass (mg). Dried leaves were then analysed for leaf nitrogen content and leaf phosphorous content by Bemlab in Somerset West, South

Africa. To avoid intraspecific variation within species as a result of environmental variation, a mean analysis for each trait was obtained where multiple values were reported.

Table 6.1: Definitions and the proxy data used to determine ecosystem services identified by Egoh et al (2008) and the links to plant functional traits.

Ecosystem service derived from natural vegetation	Supporting ecosystem service and processes	Definition of ecosystem service	Links to functional traits	Trait effect from literature
Grazing	Soil formation/ litter generation	Ability of vegetation to maintain soils and generate litter	Canopy cover was a major indicator for litter production. Leaf traits are known to be important contributors to soil properties.	Leaf traits (de Bello *et al.*, 2010a)
Grazing	Soil stability/ soil retention Erosion prevention	Depth of soil to determine erosion factor and leaf litter quantity to stabilise soil	Leaf attributes influence soil nutrient quality. The percentage cover of vegetation is important for soil properties	Canopy traits/cover Leaf traits (de Bello *et al.*, 2010a)

6.3.4. Data Analysis

Two approaches are commonly used to analyse species traits within communities (Kleyer *et al.*, 2012). One is to analyse traits at the species level where the species themselves are the statistical units (Tecco *et al.*, 2010). A second approach is to assess community level trait values which includes either an abundance-weighted measure or a measure of the functional diversity within the community (Garnier *et al.*, 2007; Lavorel *et al.*, 2011).

For species level analysis, species traits were firstly tested for correlations using the Pearson correlation coefficient. To determine the distribution of traits associated with native and non-native species, data were arranged in a *species* x *trait* matrix to perform a Principal Components Analysis (PCA). This analysis groups species with similar trait values closer together than those which are considered to be more different (Weithoff, 2003). Following Tecco *et al.* (2010) the scores of native and non-native species were compared along axis 1 and 2 using a *t*-test. This tested for any differences in trait values between native and non-native species associated with the variation described by the PCA. Pearson correlations between leaf traits (leaf nitrogen, leaf phosphorous and leaf dry matter content) and PCA axes were performed to evaluate levels of association of leaf traits to PCA axes 1 and 2.

At the community level, the community-weighted mean value (CWM) and the variance of traits, measured by FD_{var} (Mason *et al.*, 2003), for all plots were calculated using an Excel macro freely available at http://botanika.bf.jcu.cz/suspa/FunctDiv.php (Lepš *et al.*, 2006). The CWM is calculated for each plot as the mean trait value within a community (Garnier *et al.*, 2004, Violle *et al.*, 2007). These are weighted by relative abundance of the species carrying each value (Diaz *et al.*, 2007, Vandewalle *et al.*, 2010). The CWM value represents the dominant trait value in a community, which can be related to Grime's (1998) mass ratio hypothesis. The FD_{var} of traits within a community (Hejda & Bello, 2013) expresses the distribution of traits to determine the degree of overlap or dissimilarity (Mason *et al.*, 2003, Mason *et al.*, 2005). This index is independent of number of species (Petchey & Gaston, 2002, Weithoff, 2003). The FD_{var} (variance of traits within communities) is interpreted so that lower values depict more similarity among species and higher values depict more dissimilarity among species. I also used show Petchey's FD which calculates overall diversity within a community and is based on the total branch length of a functional dendrogram (see Appendix C Figure C1) (Petchey & Gaston, 2002). This measure indicates the functional relationships among species and can be used to express overall difference between land-uses.

Data was analysed within land-use categories using Kruskal-Wallis non-parametric test (Flynn *et al.*, 2009). Post-hoc tests were performed using a pairwise Wilcoxon test with Bonferroni correction. Analysis was performed on log transformed data for all traits. Note that only two plots of woodlots and cultivated areas were sampled. These were included in the analysis, to show the range of trait values under different land-use types.

To determine relationships between the traits measured and environmental variables the community weighted mean values of traits were used within a redundancy analysis (Kleyer *et al.*, 2012). The redundancy analysis was constrained by the environmental variable derived from GIS datasets used by Egoh *et al.* (2008). This analysis is usually used to determine whether average trait expressions within transformed communities respond differently to environmental variables than intact communities. This analysis was carried out for all plots and plots excluding non-native species.

All analyses were conducted in R 3.02 (R Foundation for Statistical Computing, Vienna, Austria), including the vegan, ade4, FD and psych libraries.

Table 6.2: Species list of the dominant species comprising 80% vegetation cover across all study sites. Both native and non-native species trait values for leaf nitrogen content, leaf phosphorous content and leaf dry matter content are displayed.

	Species	Leaf nitrogen content	Leaf phosphorous content	Leaf dry matter content
Native grasslands	*Alloteropsis semialata*	1.53	0.11	39.67
	Cymbopogon plurinodis	1.54	0.12	32.91
	Centella affinis	1.98	0.1	26.93
	Cynodon dactylon	1.98	0.18	38.89
	Cyperus rotundus	1.23	0.21	31.18
	Digitaria sp.	1.93	0.17	18.05
	Digitaria eriantha	1.38	0.09	20.95
	Diheteropogon amplectens	1.43	0.12	28.36
	Eragrostis capensis	1.41	0.14	32.27
	Eragrostis curvula	1.67	0.14	44.57
	Eragrostis plana	1.17	0.12	37.24
	Forb 1	1.13	0.12	20.16
	Forb 2	1.51	0.11	21.01
	Forb 3	2.01	0.26	28.57
	Helictotrichon turgidulum	1.59	0.16	34.2
	Heteropogon contortus	1.26	0.1	26.25
	Hyparrhenia hirta	1.54	0.12	23.69
	Hyparrhenia filipendula	1.44	0.14	28.55
	Paspalum notatum	1.45	0.12	28.66
	Sporobolus africana	1.38	0.14	33.32
	Sporobolus pyramidalis	1.15	0.09	39.08
	Themeda triandra	1.35	0.11	41.24
Non-native species	*Acacia mearnsii*	2.77	0.08	46.67
	Eucalyptus sp.	1.28	0.08	30.68
	Paspalum dilatatum	1.81	0.19	24.58
	Riccardia brasiliensis	1.16	0.11	16.77
	Sorghum sp.	1.96	0.26	27.84

6.4. Results

A total of 28 species contributed to the dominant species of which 22 were native and 6 were non-native (Table 6.2). Average trait values are displayed. Correlations between the leaf traits showed that leaf nitrogen content has a positive but weak relationship with leaf phosphorous content ($r = 0.31$; Pearson correlation test) and leaf dry matter content ($r = 0.23$). Whereas leaf phosphorous content and leaf dry matter content ($r = -0.11$) had a weak negative correlation. All correlations were not significant.

6.4.1. Comparison between native and non-native species

The PCA (Figure 6.3) accounted for 84% of the variance (axis 1 = 50% and axis 2 = 34%). Axis 1 was negatively correlated with leaf phosphorous content ($r = -0.98$) and leaf nitrogen content ($r = -0.46$). Axis 2 was positively correlated with leaf dry matter content ($r = 0.96$) and leaf nitrogen content ($r = 0.43$). Native and non-native species expressed a range of trait values as indicated by the dispersion across ordination space. Differences between native and non-native species could only be distinguished along axis 2 ($t = -2.5$, $df = 26$, $p < 0.05$) which can be linked to leaf nitrogen and leaf dry matter content. Naturalised herbaceous alien species such as *Riccardia brasiliensis* and *Paspalum dilatatum* occur throughout native vegetation and are separated from each other in multidimensional space. Trait values for non-native key species were distributed across the range of native species however, the woody tree *Acacia mearnsii*, was separated the most from native species.

The biofuel species occupy different locations in multidimensional space. *Acacia mearnsii* has elevated leaf nitrogen content and leaf dry matter content, *Sorghum* has elevated leaf phosphorous content and *Eucalyptus* species have elevated leaf nitrogen content, compared to native vegetation. This suggests that the potentially different impacts on ecosystems may be expected for each of these species.

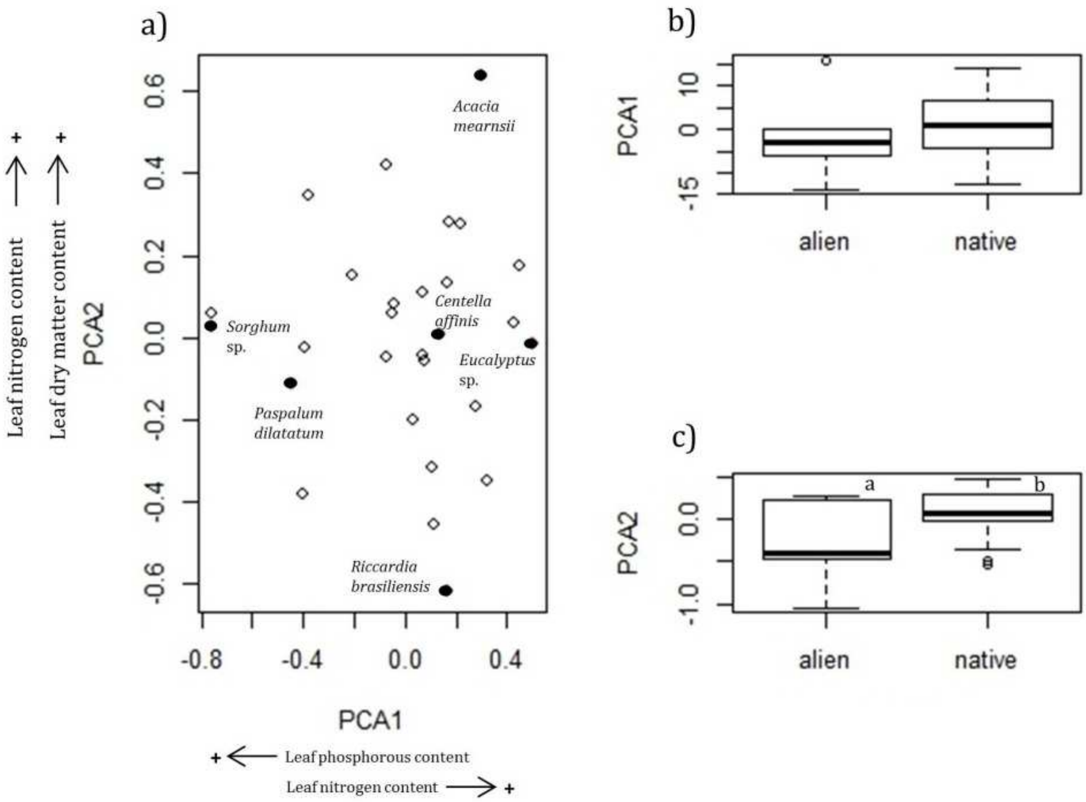


Figure 6.3: PCA ordination based on a traits x species matrix . a) PCA ordination of the dominant plant species from grassland sites based on the leaf traits of leaf nitrogen content, leaf phosphorous content, and leaf dry matter content. The first two axes explain 84% of the variance; axis 1 is correlated with leaf phosphorus content (r =-0.98) and axis 2 is correlated with leaf dry matter content (0.96). Open circles are native species and black circles are non-native species. Box plots indicate the distribution of PCA scores for native and non-native species, b) along axis 1 and c) and axis 2. Significant difference occur on axis 2 only (t-test; p<0.05).

6.4.2. Comparison of community level traits under different land-uses

The differences between land-uses were calculated based on the community weighted mean of trait values and functional diversity (FD_{var}). The results are displayed in Figure 6.4 and are discussed below. Each of these analyses was calculated with alien species and without.

6.4.2.1. Community weighted mean traits

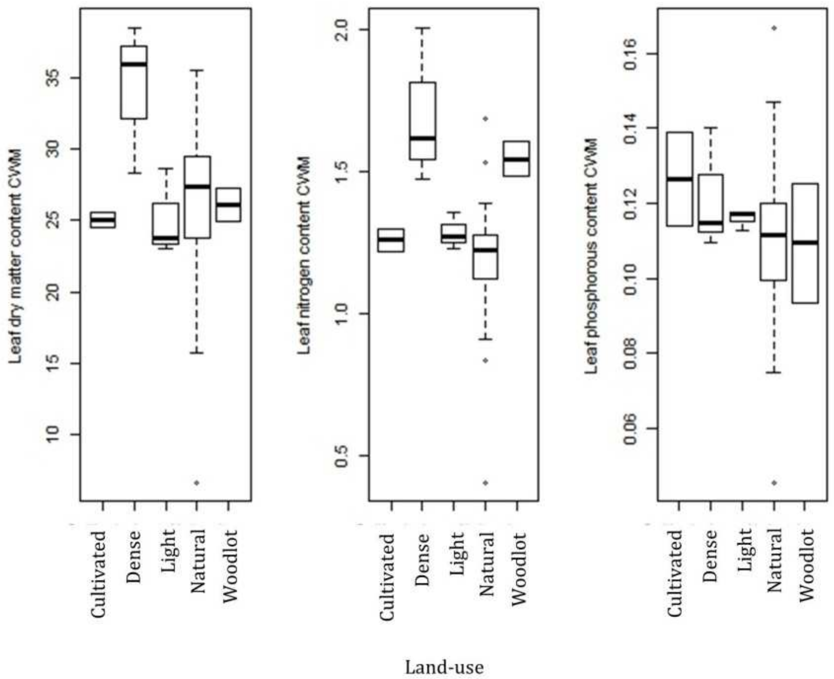
The range of traits values weighted by abundance is displayed in Figure 6.4. There was significant difference among land-uses for both leaf nitrogen content ($p < 0.01$, Kruskal Wallis test) and leaf phosphorous content ($p < 0.05$) (Table 6.3). The addition of key non-native species increases the mean value for leaf nitrogen content and leaf phosphorous content. Densely invaded areas have significantly higher leaf nitrogen content values than native vegetation, whereas cultivated areas result in increased leaf phosphorous content values when compared to densely invaded areas. There was no difference in leaf dry matter content across land-uses.

6.4.2.2. Community functional diversity

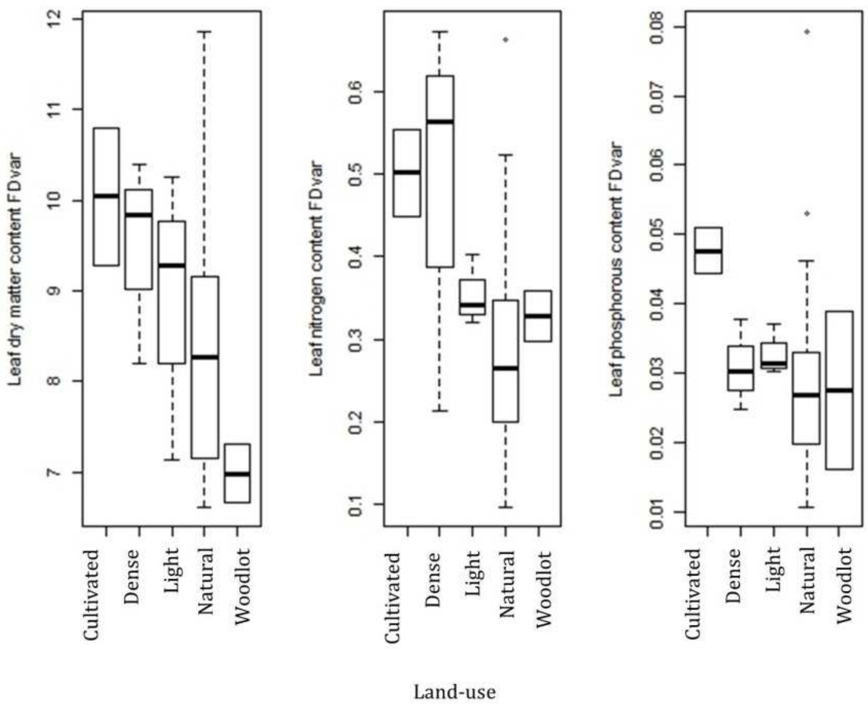
Despite the considerable variation in traits between land-uses only leaf nitrogen content was significantly different ($p < 0.05$) (Figure 6.4). In particular, high FD_{var} values for leaf nitrogen content indicate that species are more dissimilar to each other under dense invasion by *Acacia mearnsii* ($p < 0.05$, Wilcoxon test) than native vegetation. Under light invasion, leaf nitrogen content approximates that of natural vegetation, although FD_{var} is slightly lower, meaning more species have similar trait values to each other than under dense *Acacia mearnsii* invasion. Natural vegetation accounts for a wide trait range in trait values, resulting in an overlap with cultivated areas for both leaf dry matter content and leaf phosphorous content which have higher average values. Removing the key non-native species from each land-use resulted in no difference for FD_{var} values for all traits measured (results not displayed). Petchey's FD shows that natural plots have a wide variety of traits, whereas woodlots, cultivated and informal woodlots have much reduced functional diversity (Table 6.4). These findings show a decline in functional diversity was reported for all transformed land-uses by as much as ~40% across all transformed land-uses compared to natural vegetation.

6.4.1. Relationship between traits and environmental variables

The variance explained by the redundancy analysis for natural plots was 35% compared to 15.5% for all plots (Figure 6.5). Only the environmental variables for erosion, depth and litter were significantly correlated with trait variables. Axis 1 and axis 2 of the natural plots (Figure 6.5a) explained 72% and 21% respectively whereas axis 1 and axis 2 explained 80% and 12% (Figure 6.5b) respectively for all plots. The addition of key non-native species to the plots reduces the correlation between leaf nitrogen content and leaf phosphorous content as trait values are altered, suggesting that changes between environmental variables and ecosystem processes are likely to occur.



a)



b)

Figure 6.4: Trait values per land-use as expressed by a) community weighted mean values and b) functional diversity (FD_{var}) for the traits leaf dry matter content, leaf nitrogen content and leaf phosphorous content. Community weighted mean trait values are the average trait values weighted by the relative abundance within the community. The functional diversity measure FD_{var}, indicates the overall dissimilarity of species traits within a community.

Table 6.3: Trait mean and standard deviation (±SD) of the four land-use classes within an ecosystem service hotspot in the Eastern Cape, South Africa. The Informal woodlot land-use class was analysed as either lightly density or high density indicated by the presence of the invasive alien tree, *Acacia mearnsii*. Statistically significant differences were calculated using Kruskal-Wallis non-parametric test.

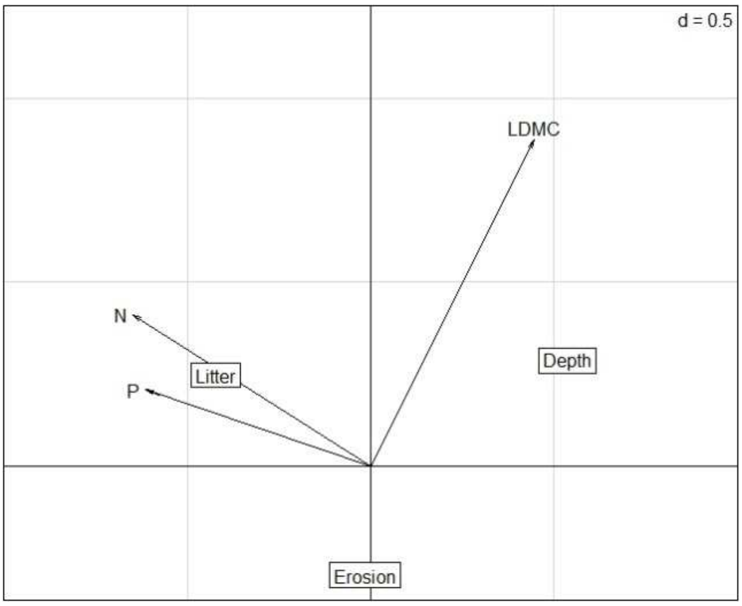
	Cultivated	Woodlot	Informal woodlots		Natural vegetation	Chi square	P
			High density	*Low density*			
N	2	2	3	3	31		
Leaf nitrogen content	1.26 ± 0.05	1.54 ± 0.09	1.7± 0.28	1.29 ± (0.06)	1.19 ± (0.22)	14.51	**0.006**
Leaf phosphorous content	0.13 ± 0.02	0.11 ± 0.02	0.12 ± 0.02	0.12 ± 0	0.11 ± 0.02	10.61	**0.031**
Leaf dry matter content	25.05 ± 0.69	26.11 ± 1.68	34.28 ± 5.29	25.11 ± 3.06	26.47 ± 5.56	9.41	0.051

*Values displayed are the raw untransformed values. Significance is calculated based on log values of traits.

Table 6.4: Petchey's measure of functional diversity (FD) for all species within different land-uses. These values are based on the sum of the branch lengths within a dendrogram and indicate the overall diversity within each land-use.

	Cultivated	Woodlot	High density informal woodlots	Low density informal woodlots	Natural vegetation
FD	52	52	51	53	80

a) Natural plots



b) All plots including non-native species

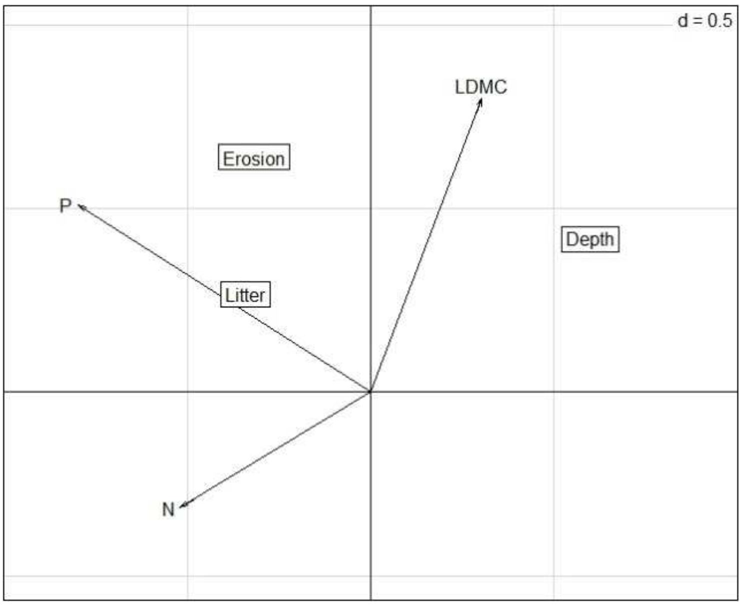


Figure 6.5: Projections of traits and environmental variables in a redundancy analysis based on the community weighted mean trait values for a) natural plots only b) all plots. Relationships between environmental variables and functional traits are altered when non-native species are included in the analysis.

6.5. Discussion

The potential impacts of increasing biofuel production in the study area indicate that the functional traits associated with identified ecosystem services are likely to change. These changes can be attributed to the introduction of non-native species as well as changes in functional diversity under different land-uses. While the traits of biofuel species may overlap with native species, the overall functional diversity under different land-uses decreases when compared with native vegetation. These findings are discussed below.

6.5.1. Differences in functional diversity at species and community levels

Overall the results show that detecting differences between native and introduced biofuel species using functional traits are dynamic. For example, while differences between species and land-uses were detected, not all species introductions were different to natural vegetation and not all land-uses resulted in significant differences. The introduced species traits and the associated abundance are partly responsible for any changes detected. For example, only dense informal woodlots, containing the invasive alien tree *Acacia mearnsii,* were found to be significantly different to natural vegetation. This finding provides further evidence that the abundance, in combination with the per capita effect, of the non-native species contribute to overall impact (Parker *et al.*, 1999).

These findings are also consistent with previous studies which show that the addition of species to communities can either increase or decrease functional similarity within communities (Flynn *et al.*, 2009, Hejda & Bello, 2013). This was shown in the PCA where large variation occurred between both native and non-native species, with each of the biofuel species occupying different locations within multidimensional space. Pyšek *et al.* (2012) describes the challenges with attributing impact to a single species and recognised that native communities are diverse and might either be similar or dissimilar to the introduced species. However, this study showed that by using functional traits and combining species and community type assessments, the effects of a single species can be detected.

The two functional diversity measures used in this study provided an understanding of community dynamics. Firstly, despite the large variation among trait values displayed within natural vegetation, the average FD_{var} suggests that species within grassland communities have similar traits. Previous studies have shown that grazing pressure on natural vegetation could select for functionally similar plants (Hodgson *et al.*, 2005). Transformed land-uses resulted in increased trait dissimilarities, for some traits, which could be related to the introduced species themselves. The second functional diversity measure, Petchey's FD, showed that transformed landscapes have much lower diversity compared to native vegetation. One of the main reasons was the decreased species richness that contributed to the dominant vegetation within each land-use (Flynn *et al.*, 2009) (see Appendix C1). While, the dominant grass species were present throughout most natural and lightly invaded sites, these were absent under woodlot, cultivation and dense informal woodlot land-uses.

6.5.2. Implications of changes in trait assemblages and ecosystem processes

There was little evidence to suggest reductions in the delivery of ecosystem services measured by Egoh *et al.* (2008). However, the analysis between environmental variables and functional traits did reflect some variation between the association of natural plots and transformed plots. This could suggest possible changes in the trajectory of ecosystems under different vegetation composition. To measure these changes more effectively requires information that was not captured in this study, some of which are best acquired through long-term study plots (Lavorel *et al.*, 2011). Although these rates were not measured, there is enough evidence to link changes in traits values to potential shift in ecosystem structure. For example, increased nitrogen and phosphorous levels are signs of species with rapid acquisition rates of resources such as *Acacia mearnsii* or *Sorghum* species which were introduced for productive reasons.

In general, less intensive land-uses have higher leaf dry matter content values which can also be related to slower decomposition rates, for which the converse is also true (Garnier *et al.*, 2004, Garnier *et al.*, 2007). Leaf dry matter content was the only trait that did not display any difference across land-uses, possibly due to the wide range of values, reflecting the diversity within grasslands and plant responses to grazing. Leaf dry matter content has also been an important indicator of above ground biomass which can affect litter accumulation rates (Garnier *et al.*, 2007).

6.5.3. Functional trait selection and measures of impact

The ability of functional traits to act as a possible measure of impact is increasing as novel methods of analysis are being developed (Kleyer *et al.*, 2012). However, this is a new area of research which requires standardized methods to understand the importance of functional traits to ecosystem service provision (de Bello *et al.*, 2010a) and to provide ways to assess possible impacts as a result of change (Dick *et al.*, 2013). Doing so will provide an important approach to potentially scale impacts from individual species through to landscape scale impacts (Lavorel & Grigulis, 2012).

The functional trait selection will play an important part in determining key components of ecosystems for which change can be measured (Petchey & Gaston, 2006). It is possible to include a wider range of traits that may account for more variation than those traits used in this study, however they will need to be appropriately linked to corresponding ecosystem processes. In this study height was excluded as both woody and herbaceous taxa were included in the same analysis and this may have resulted in biased results for land-uses containing woody species.

The importance of functional trait research is reflected in recent studies showing differences in invaded and natural communities (Hejda & Bello, 2013) and in guiding restoration activities (e.g.Kyle & Leishman, 2009). However, there are still many potential challenges that need to be overcome before functional traits can be used to adequately predict impact of introduced or invading species on natural ecosystems or ecosystem services. Two main challenges were encountered during this research. Firstly, knowing whether natural vegetation displays convergent or divergent trait selection (Tecco *et al.*, 2010) will facilitate the determination of whether an introduced species is more likely to be dissimilar or similar to the native vegetation. Knowing this information beforehand, along with measures of abundance, will enable rapid assessments using trait data obtainable from online sources (e.g. TRY network, (Kattge *et al.*, 2011)). Secondly, the availability of data on ecosystem services are not that common and proxies for ecosystem services, may not be available at the scale required for local scale analysis (Eigenbrod *et al.*, 2010).

6.5.4. *Impacts of biofuel production*

The methodology used in this study aims to improve our understanding of impacts on ecosystem processes to complement well-established spatial assessments. The species associated with biofuel production, and the accompanying land-uses practices, have the ability to change the functional attributes which can be detected across small and large scales. Lin *et al.* (2011) has demonstrated that detecting a reduction in trait diversity across large scales, such as the regional scale, using the relative abundance of cultivated areas, highlights vulnerability of these systems to environmental change (Folke *et al.*, 2004).

The potential to increase cultivation within the study area highlights possible changes that can occur at the landscape scale. Reduced diversity and changes in trait values may erode the ecosystems ability to maintain the provision of ecosystem services within the service providing area (Cardinale *et al.*, 2012, Lavorel & Garnier, 2002). Furthermore, increasing evidence suggests that intact systems with higher functional diversity may have greater resilience under environmental change.

A potential goal would be to use functional traits to facilitate landscape management and allow species to be selected based on functional requirements that maintain important ecosystem services (Kontogianni *et al.*, 2010). Costly experiments to determine a species risk of becoming invasive (e.g. Davis *et al*., 2011, Flory *et al*., 2012) may be complemented by assessing potential impacts based on the introduced species functional traits and those within the recipient community. However, functional replaceability should not undermine the conservation efforts of indigenous biodiversity (Luck *et al.*, 2009).

6.6. Conclusion

The difference between functional traits of potential biofuel species and native vegetation can be attributed to the change in species and the management intensity therein. From an ecological viewpoint, understanding how functional diversity links to ecosystem processes provides a potential predictive mechanism that link patterns of community trait assemblages and ecosystem functioning (Ricotta & Moretti, 2011). Here, I explored the opportunity to test a theory that changing functional traits could affect ecosystems properties by altering the functional structure therein. Using potential biofuel species, differences between native and transformed land-uses could be detected. Furthermore, the functional characteristics of a service providing hotspot where shown to be altered under increasing land-use change. The widespread reduction in functional diversity and altered trait values may alter ecosystem processes and ultimately the services that they deliver. The ability to detect changes at the functional scale provides further insight into impacts of land-use change that may not be anticipated when focussed only at broader scales. The capacity to anticipate these possible changes will enhance our ability to plan and maintain multifunctional landscapes.

6.7. Acknowledgements

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Part IV:

Synthesis

Chapter 7: Conclusion

The South African government has proposed that a biofuel strategy should form part of the renewable energy mix in South Africa. Globally, the production of biofuels is expected to increase into the future as a demand for alternate fuels to fossil fuels is likely to increase. However, there are numerous concerns relating to the negative impacts of biofuels on biodiversity (Fitzherbert *et al.*, 2008, Wiens *et al.*, 2011). Documented impacts include habitat loss, a reduction in biodiversity, increased landscape fragmentation and changes to the functional diversity of landscapes needed to maintain critical ecosystem processes (Foley *et al.*, 2005). The main challenge addressed in this dissertation was how best to measure the potential impacts of biofuel production at a scale that could provide adequate information for use in regional or local planning. The research investigates the drivers, impacts and trade-offs of potential biofuel production and its impact on biodiversity at multiple scales in the South African context.

The conceptual framework developed by Parker et al (1999) elucidated the components of impact and has been utilised here to unpack the outcomes of biofuel production. Parker et al's (1999) scheme lists these components as range, abundance and per capita effect. These components clearly link to the various methods used in each chapter (such as computer-based modelling, biodiversity indicators, and field sampling techniques), and this demonstrated the need for both broad- and local-scale assessments to fully understand the effects and impacts of land-use change associated with biofuel production.

Chapter 1 outlined the specific objectives of the this research and detailed the contribution of each chapter in determining: 1) the key issues regarding the production of biofuels and potential impacts on biodiversity; 2) the potential biofuel production areas in the Eastern Cape and the biodiversity conflicts that arise*;* 3) the use of the Biodiversity Intactness Index to assess likely impacts of biofuel production on biodiversity; 4) the relationships between the Biodiversity Intactness Index and fragmentation indicators; and 5) the effect that introduced plants have on community functional structures and the potential impacts on ecosystem services that arise. In the following section, the main questions presented in Chapter 1 are revisited and discussed in relation to the findings within the individual chapters.

7.1. Summary of findings

1) *What are the potential impacts of adopting a biofuel strategy on land-use change and its associated impacts on biodiversity?*

The research identified five main impacts that should be considered when adopting a biofuel strategy. These include: potential conflicts between areas of biodiversity and areas with production potential, the extent of conversion of natural land to biofuels production, a reduction of biodiversity associated with large scale land-use changes, a potential increase in habitat fragmentation, and the potential decrease in functional diversity within ecological communities.

The South African biofuel strategy was reviewed and discussed in relation to the growing literature on biofuels to highlight potential impacts on biodiversity (Chapter 2). From this review a framework was produced which highlights the different impacts associated with the many different biofuel production systems. Differences here relate to the scale of production (i.e. small holder farming or large scale biofuel plantations), the configuration of plantings as well as the species used or cultivated. Despite these differences the main impact can be related to the extent of land area required to meet production targets. The potential impacts of converting available land, either arable or marginal, was shown in Chapter 3 (using overlaps of arable and marginal land with important biodiversity areas) and in Chapter 4 (by applying the Biodiversity Intactness Index).
Available arable land accounts for approximately 14% of the Eastern Cape. Using spatial overlaps with important biodiversity areas revealed that only 5% of arable areas have no biodiversity conflicts. The Biodiversity Intactness Index (BII) showed that converting arable land results in a decline of the BII from 83% to 75% and that excluding the important biodiversity areas reduces BII to only 79%. The effect of fragmentation, as

captured by the revised-BII resulted in the BII decreasing to 69% following the conversion of all arable land.

Applying the BII to the Eastern Cape under different scenarios of biofuel production demonstrated that while relatively minor impacts are reported at the aggregated provincial scale, disaggregating the BII shows that specific municipalities will experience great reductions in biodiversity intactness. The impact of land-use changes on biodiversity is greatest when natural or near-natural land (i.e. areas classified as moderate use) is converted to cultivation. Using degraded land could reduce biodiversity impacts as this land-type is regarded to have lower habitat quality and is associated with lower species abundance.

In Chapter 6, the effect of land-use on functional diversity was explored using a functional traits approach. The results showed that functional diversity was reduced by approximately 40% as a function of land-use change. More importantly, different impacts of proposed biofuel species could be detected. This suggests that impacts of biofuels should not be considered to have the same effect and these might affect different components of ecosystem processes.

This analysis could not capture the effect of different biofuel production scales (i.e. small-holder farming vs. large-scale plantations) on biodiversity. In part, this is because there are no estimates of the minimum viable area required for biofuels farmers, regardless of the crops that are utilised. For example, assessing the potential for small-holder farming with *Jatropha* requires information relating to effective management of farms and the overall viability of this industry as a whole (von Maltitz *et al.*, 2012). For that reason, only the cumulative area needed to meet biofuel production targets or the potential available land was used in this analysis.

2) *What methods are available to assess biodiversity impacts and how can these be improved or modified to better address questions relating to biofuel production more directly?*

The specific tools used to measure biodiversity impacts in this study vary in complexity. I identified potential biodiversity impacts using overlaps between potential biofuel producing areas and areas considered important for biodiversity. This was dependent on the availability of additional biodiversity information (i.e. the National Protected Area Expansion Strategy and Eastern Cape Biodiversity Conservation Plan). Next, the BII was used to estimate the overall change in regional biodiversity based on changes to the spatial extent of land-uses. Finally, I used a functional traits approach to determine the potential changes to ecosystem processes as a result of changing the dominant natural vegetation to produce biofuels. This was in response to the growing interest in biodiversity and ecosystem function research (Cardinale *et al.*, 2012).

These available methods were improved in several ways. Firstly, in Chapter 3, a conceptual framework merged the outputs of species distribution models with land suitability analysis within a GIS to indicate conflicts between biodiversity and potential areas suitable for biofuel production. The framework presents a novel approach for integrating different biodiversity spatial filters to better identify areas at risk of conversion. While this type of analysis identified potential spatial conflicts, it did not provide an overall measure of potential impacts on biodiversity – such information is required by planners for making decisions regarding permitting and zoning of land-use.

To address this requirement, the BII was used to estimate the impact of transforming available land to biofuel production in Chapter 4. The limitations of the BII, with regards to fragmentation, were assessed in Chapter 5. The BII provides an alternative to costly field-based assessments and presents a robust estimate of biodiversity intactness. The BII was particularly useful in assessing changes in land-use as this index can be incorporated into, and calculated within, a GIS, providing spatial information on potential biodiversity impacts. However, the aggregated nature of the BII algorithm results in the effects of habitat fragmentation being overlooked. A revised-BII (R-BII) was developed using known impacts of patch size on the abundance of species. The R-

BII accounts for fragmentation and provides a realistic measure of intactness within a landscape.

The impacts of land-use change were assessed using a plant functional traits approach. It was important to assess the species and the community level effects of changing functional values. The potential for biofuel species to change the functional characteristics within each land-use was dynamic. The abundance and the per capita effects were important factors to needed to assess potential changes. The species used for biofuels, *Sorghum, Eucalyptus* and *Acacia mearnsii*, each had different effect on trait values. However, overall functional diversity declined native vegetation, as well as to change the community weight mean values of important functional traits such as leaf nitrogen content.

3) *How can the components of impact (range, abundance, and per capita effect) be integrated to address impacts to biodiversity and ecosystem services across multiple scales?*

Synthesising the work from the independent chapters to provide an overall assessment of impact into a single index was not possible. This is not to say that it cannot be done, but that much additional work is required to make this possible. Some of the key uncertainties identified for future biofuel production include: species selection, location of plantings, extent of plantings, and the effect that introduced species may have on the surrounding landscape.

Although the factors of impact are currently assessed separately there are indications that they interact in complex ways. For example the abundance and per capita effect as discussed using functional traits was shown to be important for determining differences between native vegetation and assemblages dominated by introduced species. Scaling these up to landscape and regional scales is a research goal of global interest.

Cross-scale assessments provide further insights into separation of pressures, impacts and potential solutions. Furthermore, this work highlights the importance of undertaking assessments at multiple scales to capture a fuller range of impacts and to shed light on the magnitude of impacts. This information is of key importance to those tasked with planning and shaping energy strategies at both the national and the international level.

7.2. Overall insights

Biofuel production is a global phenomenon and is likely to have considerable impacts in developing countries such India, Brazil and South Africa. The drivers of biofuels for developing countries include, among others, energy security and rural development (Gasparatos & Stromberg, 2012). Biofuel strategies are therefore expected to have positive impacts on the socioeconomic status of biofuel growers and to possible increase international trade. The need to meet these goals might overlook the potential negative impacts on biodiversity and ecosystem services. This is especially concerning as detailed biodiversity are often lacking for many of the regions considered for biofuel production.

The methods and results presented here address important questions that are currently of interest to conservation biologists and to decision makers. The methods can be applied in a developing world context and ensure that biodiversity is represented within decision making processes.

This work is also applicable to wider audience, despite the focus on a regional scale. For example, the modelling approach for integrating species distribution models with spatial filters adapts integrative modelling techniques to better determine suitable locations for biofuels (Evans *et al.*, 2010, Trabucco *et al.*, 2010). The use of the BII increases the evidence for the application of biodiversity indicators to assess the impacts of biofuels and to anticipate future impacts of different policy actions (Polasky *et al.*, 2011). The work also focuses on further developing and testing indicator theory (Vačkář *et al.*, 2012). Furthermore, adopting an ecosystem service approach to resource management provides a useful tool for assisting in complex land-use decisions. The methods for linking functional traits to ecosystem services requires further development and additional empirical research, but pave the way for designing landscapes capable of meeting multiple objectives. This aim is in line with the global targets as established by the CoP10 (2010). For example, Pereira *et al.* (2013) has highlighted the importance of providing measures on biodiversity that address components on species populations, species traits and ecosystem structure and function that are all needed to reduce biodiversity losses.

This study has highlighted the potential impacts of increasing cultivation for biofuel production in the Eastern Cape. Although I focus on this one region, the lessons learnt of global significance. For example, the challenges to conserve areas that are considered important, from biodiversity and ecosystem service perspectives are a growing concern globally (Cardinale *et al.*, 2012). Despite an existing strategy directed at expanding the protected area network in South Africa (Government of South Africa, 2008), such expansion is likely to be limited due to financial constraints (Gallo *et al.*, 2009) and competing alternative land-uses. This is emphasised by recent studies focusing on the Eastern Cape (Bradley. *et al.*, 2012, Estes *et al.*, 2013) that have indicated that future global changes could shift agricultural climate patterns into areas that are currently indicated as having biodiversity importance. Clear approaches need to be devised to guide the increasing pressure to convert untransformed land into productive landscapes. Methods such as these presented here can facilitate the management of conflicts between human development and biodiversity conservation.

South Africa's Biofuel Strategy emphasizes that human development should be a major goal of biofuel production and that stimulating the agricultural sector is one way of meeting the urgent need to provide jobs in rural areas. However, the lessons from plantation forestry dictate that the process of species selection, plantation establishment and management should be well informed. For example, the lack of information and poor management in the past has resulted in *Acacia mearnsii*, a commercial tree species, becoming a hugely problematic invasive species. Estimating the potential risk of introduced species becoming invasive and having negative impacts can be strengthened by adopting a functional traits approach to assess how vegetation dominated by native species vegetation differs from assemblages dominated by various types of non-native species, such as plants grown for biofuel production.

The need to mainstream biodiversity within development processes has received much attention in recent years (O' Connor & Kuyler, 2009, Reyers, 2004, Wessels *et al.*, 2003), yet more work is needed to reduce threats to resources outside protected areas (Reyers, 2013). This must include initiatives to conserve processes that occur at the landscape scale. To meet these objectives, planning should be informed by a strategic approach to biofuel production focusing on multiple objectives such as food, energy, human wellbeing, biodiversity conservation and the provisioning of ecosystem services. I trust that the results from this research provide important insights in this regard. If political decisions dictate that biofuel production should proceed, planning should seek to ensure the maintenance of ecological corridors, minimise the potential for fragmentation, and maximise regional connectivity to reduce negative environmental consequences and increasing loss of ecosystem services. These recommendations are equally important in a developing world context where much of the world biodiversity currently resides and this work provides some insight into the requirements to assess potential impacts on biodiversity in data sparse areas.

The main conclusions that can be drawn from this work include the following: 1) The combination of climatic suitability for many biofuel species and suitable land suggests that there is large potential for biofuels to be produced in some regions of South Africa. 2) Areas that may be suitable for biofuel production may also occur in areas where land is considered important for biodiversity conservation, yet is currently not protected. 3) Biodiversity indicators present a useful way of communicating potential impacts and therefore for proposing locations for biofuel plantations where impacts are smallest. 4) The introduction of biofuel species may shift the functional characteristics of ecosystems through a shift in functional traits that differ from natural systems.

This dissertation expands on the complexity associated with large-scale land-use changes. In doing so, I have gained much experience in understanding the broader implications of biofuel production and the potential conflict between the need to meet development goals and conservation objectives. Through the investigation of potential biodiversity losses, I aim to contribute to the safeguarding of biodiversity and ecosystem services for future generations. The findings are the outcomes of many hours of learning and adapting new approaches, and based on these observations, I suggest some priorities for future research.

7.3. Future research

Future research should focus on two major areas addressed in this research. Firstly, more work is needed to enhance the BII and its ability to inform policy makers. There is currently little evidence that indicators like this are being used to inform policy. Impact factors associated with specific approaches to biofuel production (i.e. large scale vs. smallholder farming) need to be explored in more detail and may distinguish the effects associated with different biofuel production systems in more accurately.

Another area of research that justifies much more work is the link between functional traits and land-use interactions. The main challenge here involves linking the functional traits with the production functions that underpin ecosystem services that are relevant to policy in particular situations. Advances in this area would allow the direction and magnitude of changes to be better assessed following the replacement of the natural communities or a portion thereof, by non-native species. This would facilitate much better landscape planning and design to maintain the integrity of ecosystems and the services provided.

Chapter 8: References

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Chapter 9: Appendices

These appendices include:

1) **Appendix A: supporting information for Chapter 1**
    a. **A review of biodiversity indicators**

2) **Appendix B: supporting information for Chapter 4**
    a. **Display of BII impact factors**
    b. **Comparison of impact factors**
    c. **Biofuel scenarios**
    d. **Disaggregated BII tables**

3) **Appendix C: supporting information for Chapter 6**
    a. **Functional trait dendrograms**

Appendix A: Review of biodiversity indicators

Approaches to monitoring and conserving biodiversity

To mitigate the loss of biodiversity, effective land-use and conservation planning would require areas of high biodiversity value to be identified and adequately incorporated into management at a regional level. Both the *Key Biodiversity Areas* and the *High Conservation Area* approaches have been developed as an assessment tool to identify local scale priority conservation areas ((Erken *et al.*, 2004); www.hcvnetwork.org). Whereas the Key Biodiversity Area approach mainly focuses on areas of high biodiversity important for reserve site selection, the High Conservation Value approach identifies six different criteria requiring appropriate management. Both these approaches would highlight areas that should be excluded from intense biofuel production.

Key Biodiversity Areas

This approach is used as a rapid assessment tool to identify local scale priority conservation areas (Knight & Cowling, 2007) and is based on data obtained from existing approaches (e.g. the IUCN Red List and BirdLife International's Important Bird Area program). The KBA program identifies globally important populations of key species that have been classified as vulnerable, endangered or critically endangered by the IUCN Red List. KBA's look to protect both range and biome restricted species and important sites required during the life histories of such (Erken *et al.*, 2004). Although having gained international recognition there is a general lack of data to expand the approach to new areas. Where successfully applied, the availability of extensive datasets, availability of knowledgeable local experts, funding and strong stakeholder support were readily accessible.

High Conservation Value approach

Areas that have exceptional biodiversity, rare species or have cultural significance can be declared areas of High Conservation Value (Table A1). According to the HCV Resource Network, "the key to using the HCV approach is the identification of the six High Conservation Values (HCVs)". These are intended to cover the range of conservation priorities and both include social ecological values. The presence these values within identified areas are important and need to be protected.

The HCV area approach has typically formed part of the FSC certification process yet it is continually applied to broader certification standards (Hennenberg *et al.*, 2010). The identification of HCV areas can assist in prioritising land-use planning and conservation plans. The value of this approach is that areas of significant biodiversity and cultural value should be identified and managed accordingly, ideally buffered from further biofuel cultivation pressures (Hennenberg *et al.*, 2010). Areas that fall outside of these classification systems will most likely be subject to pressures from biofuel production. This requires the detailed information be collected within potential biofuel producing areas.

Table A1: The definitions of the six High Conservation Value classes

HCV1. Areas containing globally, regionally, or nationally significant concentrations of biodiversity values (e.g., endemism, endangered species, refugia). For example, the presence of several globally threatened bird species within a Kenyan montane forest.
HCV2. Globally, regionally, or nationally significant large landscape-level areas where viable populations of most if not all naturally occurring species exist in natural patterns of distribution and abundance (e.g., a large tract of Mesoamerican flooded grasslands and gallery forests with healthy populations of Hyacinth Macaw, jaguar, maned wolf, and giant otter, and most smaller species).
HCV3. Areas that are in or contain rare, threatened, or endangered ecosystems (e.g., patches of a regionally rare type of freshwater swamp in an Australian coastal district).
HCV4. Areas that provide basic ecosystem services in critical situations (e.g., watershed protection, erosion control) (e.g., forest on steep slopes with avalanche risk above a town in the European Alps).
HCV5. Areas fundamental to meeting basic needs of local communities (e.g., subsistence, health) (e.g., key hunting or foraging areas for communities living at subsistence level in a Cambodian lowland forest mosaic).
HCV6. Areas critical to local communities' traditional cultural identity (areas of cultural, ecological, economic, or religious significance identified in cooperation with such local communities) (e.g., sacred burial grounds within a forest-management area in Canada).

Source Network 2005–7, http://www.hcvnetwork.org/site-info/The%20high-conservation-values-folder

Biodiversity Indicators

An indicator is able to summarise complex data into simple standardised and communicable figures (Postnote, 2008). There are numerous agencies working on indicators for monitoring biodiversity (e.g. Biggs *et al.*, 2007, Ten Brink, 2006). The problem is that some indicators are quite complicated and much of the information required to complete the indicator are lacking or not available, especially for developing countries. Indicators can be based on simple measures (such as number of species) or aggregated to produce composite indicators. Recently many more indices have been developed to assess the intactness or state of the environment (these are discussed below).

The most widely used composite indicators to date include: i) Natural Capital Index – NCI; ii) Mean Species Abundance – MSA iii) Living Planet Index – LPI; iv) Biodiversity Intactness Index – BII. These indicators are similar in that they measure current species richness in relation to a previous state. The proportion of remaining biodiversity is the value of the index often expressed as a percentage. These composite indicators have been reviewed by Ten Brink (2006) and only a summary is presented here, however the BII is expanded upon due to usefulness in assessing biodiversity impacts from land-use change. Species area curves are also included as a means to assess biodiversity loss.

a) **Natural Capital Index (NCI)** –The NCI is calculated as the product of the remaining ecosystem area (quantity) and the mean species abundance in the remaining ecosystem (quality). It provides an assessment of human impact and returns a value of the naturalness/intactness of remaining ecosystem. Examples of this approach can be found in (Reidsma *et al.*, 2006). The NCI has been divided to assess naturalness and agricultural changes separately. The NCI recognises the value of cultural landscapes such as historically cultivated areas and provides a separate assessment for agricultural areas. Data for the index is derived from land cover and land-use monitoring, whereas species abundance is limited to available data or retrieved from complex distribution models.

b) Mean Species Abundance (MSA) – The MSA was developed in response to the data intensive needs of the NCI (Alkemade *et al.*, 2009). The MSA calculates the abundance of original species relative to natural or undisturbed state at ecosystem level. It is presented as a value from 0% to 100%. It is similar to NCI in that ecosystem quantity and quality are needed to calculate the index. The main difference is that land-use and relative species abundance data were based on cause-effect relationships derived from published data. The MSA has also made provision for drivers of biodiversity loss by including effects from land-use, nitrogen deposition, infrastructure, fragmentation and climate change. MSA has been considered as a measure for the tracking trends in abundance of selected species.

c) Living planet index (LPI) - The LPI measures global vertebrate abundance trends within ecosystems over time relative to a baseline (Loh *et al.*, 2010, Loh *et al.*, 2005). The LPI is calculated on the mean species abundance using a database that consists of 7950 populations of over 2500 species of mammal, bird, reptile amphibian and fish (Loh *et al.*, 2010). The populations are tracked through time to provide a measure of abundance trends. The change in the number of individuals within a population can infer changes to ecosystems. as a result of habitat change. LPI has been applied in various WWF reports (e.g. Deinet *et al.*, 2010) and used in the 2nd Global Biodiversity Outlook.

d) The Biodiversity Intactness Index (BII) - The Biodiversity Intactness Index provides an estimation of the average population size of a wide range of organisms relative to their baseline populations for a given area (Biggs *et al.*, 2006, Scholes & Biggs, 2005). The method uses existing biome or vegetation-type classifications, existing distribution information for major taxonomic groups, and expert opinions about the relative reduction in average abundance of species (in different groups), as a consequence of various mapped land-uses (Faith *et al.*, 2008). It is possible to derive both an overall score of the biodiversity of a region (aggregated) or can it focus on a particular functional type within various land-use activities (disaggregated). The versatility of the BII allows it to be calibrated to estimate past changes as well as to project into the future under various situations (Scholes & Biggs, 2005). Expert opinion is consulted to determine the levels of biodiversity loss for different land-uses. The BII can complement indicators such as the NCI in data sparse areas. The BII does not make distinctions between species types and does account for beta diversity across the landscape. This has been criticised by some (Ewers *et al.*, 2009) due to the inability of the index to distinguish between an increase in generalist specie s vs a more sensitive species.

e) **Species Area Curves** - Biofuel production is likely to influence the overall land cover as a result of changing land-use. It is understood that the number of species increases with increasing land area (Sala *et al.*, 2005) and that reducing the land area available will similarly impact on species abundance and diversity. The species area relationship was utilised by the Millennium Assessment (Sala *et al.*, 2005) and is likely to be applicable as a method of estimating losses of species in particular regions as a result of land-use change due to the increase in biofuel cultivation (Sala *et al.*, 2009). The distribution of species is not uniform across landscapes or latitudes, and knowing likely locations of biofuel production will greatly increase the efficacy of the MA approach. Focusing on specific regions could yield results on a number of species likely to be impacted on because of habitat loss, and a ranking of the most vulnerable vegetation types. Species area curves can indicate the increase in diversity with sampling area. Different land-uses can have varying effects on species accumulation over increasing area. Species area curves can also be used to estimate diversity patterns at different spatial scales (see de Bello *et al.*, 2010b). The species-area relationship can be modified firstly to represent a fractional loss between old and new values and secondly to represent an intactness index (Faith *et al.*, 2008). As an intactness index the species area relationship can represent the losses of area, as well as provide an assessment of the condition of the habitat area. Despite the likely decrease in species richness as a result of reduced habitat, it may be difficult to generate generic rules for critical change points for vegetation or habitat cover (e.g., 10%, or 30%, or 70%) that can be applied broadly across different landscapes and different biotic groups (Lindenmayer *et al.*, 2005).

Information required for effectively establishing biodiversity estimates are lacking in many parts of the world and many efforts rely on the coarse resolution of global datasets. These data are important to develop effective indicators, and where data are lacking the BII which derives important information from experts provides an opportunity to assess impacts where information is lacking.

Appendix B: Supporting information for Chapter 4

B.1 Impact factors associated with the BII

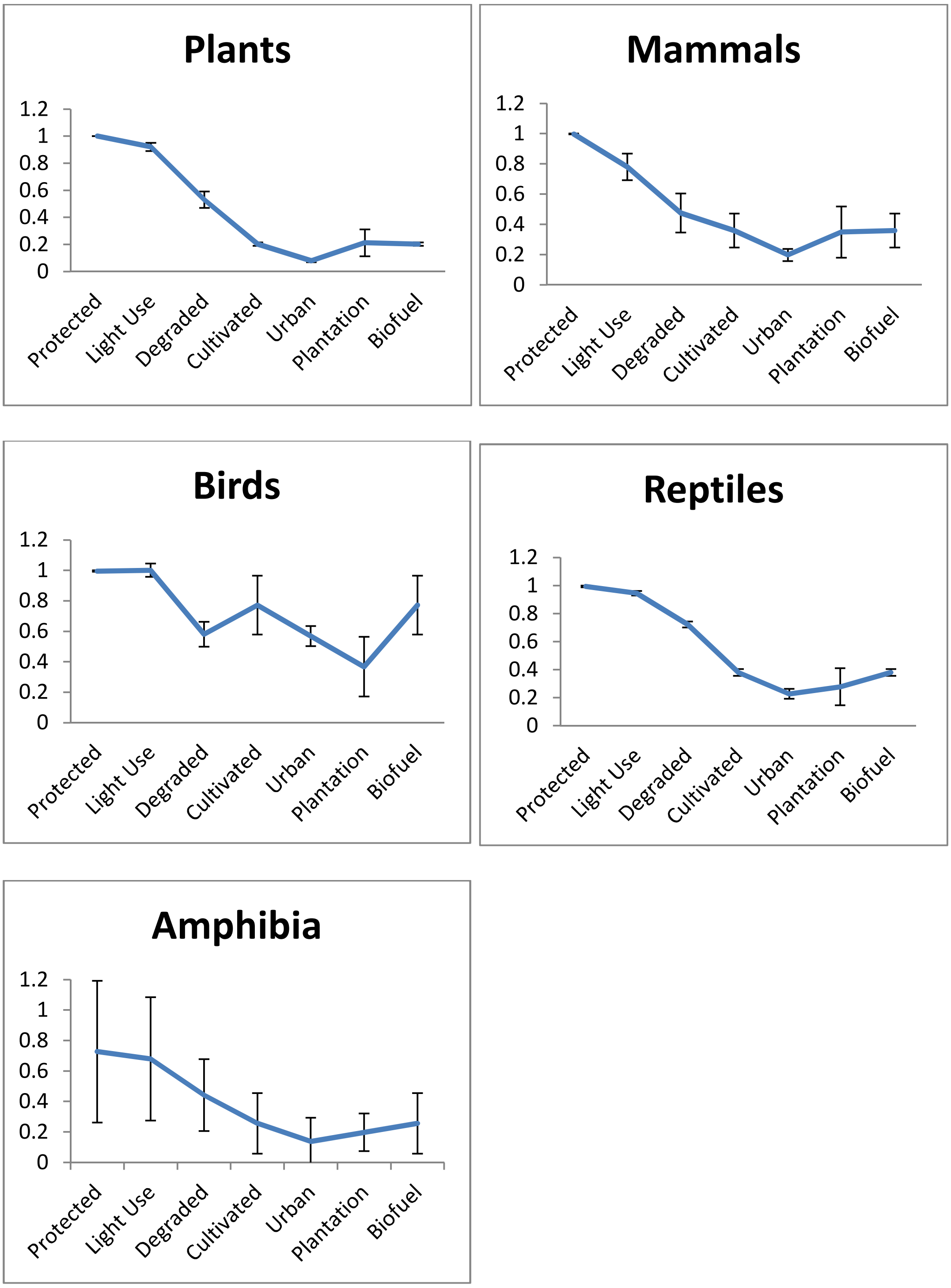

Figure B1: Impact factors derived from Scholes and Biggs (2005). The impact factors associated with biofuels are based on cultivation. Impact factors relate to the proportion of biodiversity left after disturbance as a function of land-use change in relation to the protected areas. Error bars indicate the range of impacts associated with taxa.

B.2 Comparison of impact factors

Impact factors have not been assigned to biofuel plantations. In this analysis, four different impact factors were assigned to biofuel production (Table B1). The impact factor with the lowest effect on BII is associated with degradation. This value is approximately half the impact to biodiversity as moderate use areas and twice as much as cultivation or plantation. Cultivation and Plantation impact factors have the greatest effect on BII with Plantation factors resulting in the greatest impact to biodiversity.

Table B1: The summary table of sensitivity analysis for BII following the conversion of arable and marginal land within Degraded and moderate use land-use classes. Results are shown for four different impact factors attributed to biofuel cultivation**.**

Area converted		Proportion of E. C	Impact factors			
Land capability	Land-use		Cultivation	Plantation	Average (cult + plant)	Degraded
Arable	Degraded	2.8	83.6	83.6	83.6	84.4
	Moderate use	11.3	75.8	75.7	75.8	78.7
	Both	14.1	75.1	75.0	75.0	78.7
Marginal	Degraded	2.8	83.7	83.6	83.7	84.4
	Moderate use	30.4	65.5	63.7	65.4	71.8
	Both	33.2	64.9	62.9	64.8	71.8
Arable and marginal	Degraded	5.6	83.0	82.8	83.0	84.4
	Moderate use	41.7	57.0	55.1	56.9	66.1
	Both	47.3	55.6	53.5	55.5	66.1

B.3 BII applied to biofuel strategy fuel scenarios

Applied example: The BII was applied to basic land-use scenarios to determine impact from biofuel production.

The impact of biofuel production within the Eastern Cape Province of South Africa was tested based on the land area needed to produce the initial 2% and 5% up to 25% of biofuel production targets (Table B2). These scenarios were guided by the biofuel strategy and are based on targets needed to meet blending requirements of both biodiesel and bioethanol. Biofuel species used to illustrate these targets are sugar cane, canola and maize.

Fuel estimates from biofuel production based on average yield in metric tons per ha were used to calibrate the spatial requirements of both biodiesel and bioethanol production (Table B2). Fuel production values for South Africa were taken from Von Maltitz and Brent (2008). Land area requirements were based on extracted values. Species to achieve targets are sugar cane, canola and maize.

Table B2: Land area (Km^2) requirements needed to meet blended percentages for fuel based on 2005 fuel figures**.

Species	Fuel type	2%	5%	10%	15%	20%	25%
Canola	Biodiesel	0.29	0.72	1.45	2.17	2.89	3.61
Maize	Bioethanol	0.18	0.45	0.89	1.34	1.79	2.23
Sugar cane	Bioethanol	0.06	0.16	0.32	0.48	0.65	0.81

*Fuel use per year 2005 (1000 000l): Petrol -7987 and Diesel – 10289

**Sugar cane 65t/ha 70l/t 4500l/ha (70%conversion); Canola (588l/ha) conversion 0.94)

B4: Disaggregated BII for district municipalities, local municipalities and ecoregions

This shows the BII scores as they are disaggregated to various level management. The local and district municipalities are included. The ecoregions are also included to show the effect of land conversion at an ecological scale.

Table B4-1: Disaggregated BII scores for District municipalities

Municipality	Normal	arable degraded	Light	Both	marginal degraded	marginal Light	marginal Both	all degraded	all Light	all Both
Buffalo City	76.0	72.5	54.2	50.7	75.4	60.4	59.8	71.9	38.6	34.6
Cacadu	89.0	89.0	87.5	87.5	88.8	76.6	76.4	88.8	75.0	74.8
Amathole	85.7	84.4	71.3	70.0	85.0	63.7	63.1	83.7	49.3	47.3
Chris Hani	82.8	81.1	74.3	72.6	81.8	60.9	59.9	80.0	52.4	49.6
Joe Gqabi	85.3	84.9	77.6	77.2	84.8	60.1	59.6	84.4	52.4	51.4
O.R.Tambo	81.1	80.3	65.7	64.9	80.5	64.8	64.3	79.7	49.4	48.1
Alfred Nzo	73.2	72.2	61.5	60.5	72.3	60.7	59.8	71.3	49.0	47.1
Nelson Mandela Bay	74.9	74.7	60.4	60.1	74.6	55.0	54.7	74.3	40.4	39.8

Table B4-1: Disaggregated BII scores for District municipalities

		Arable			Marginal			Both arable and marginal		
	BII current area	Degraded	Light use	Both	Degraded	Light use	Both	Degraded	Light use	Both
Buffalo City	76.0	72.5	54.2	50.7	75.4	60.4	59.8	60.4	38.6	34.6
Camdeboo	90.5	90.5	90.5	90.5	90.5	82.0	82.0	82.0	82.0	82.0

Blue Crane Route	92.2	92.2	92.1	92.1	92.2	63.6	63.6	63.6	63.4	63.4
Ikwezi	93.1	93.2	93.2	93.2	93.2	93.2	93.2	93.2	93.2	93.2
Makana	93.7	93.7	88.8	88.8	93.7	58.1	58.1	58.1	53.2	53.2
Ndlambe	87.3	87.3	74.4	74.4	87.3	55.2	55.2	55.2	42.3	42.3
Sundays River Valley	89.3	88.9	87.4	87.0	87.6	83.3	81.7	83.3	81.5	79.4
Baviaans	93.6	93.6	93.6	93.6	93.6	93.4	93.4	93.4	93.4	93.4
Kouga	69.3	69.2	66.9	66.8	69.2	61.5	61.5	61.5	59.1	59.0
Kou-Kamma	85.0	85.0	83.5	83.5	85.0	75.7	75.7	75.7	74.2	74.2
Mbhashe	83.4	82.4	68.9	68.0	82.8	67.7	67.2	67.7	53.3	51.7
Mnquma	77.0	76.1	53.7	52.7	76.4	69.4	68.7	69.4	46.0	44.4
Great Kei	90.2	90.1	78.1	78.1	89.9	53.3	52.9	53.3	41.2	40.8
Amahlathi	87.8	87.5	76.1	75.8	86.6	64.6	63.5	64.6	53.0	51.6
Ngqushwa	80.6	75.4	50.7	45.5	79.8	67.0	66.2	67.0	37.1	31.1
Nkonkobe	87.1	84.8	73.4	71.1	86.2	60.7	59.9	60.7	47.1	44.0
Nxuba	92.4	92.4	92.4	92.4	92.4	60.7	60.7	60.7	60.7	60.7
Inxuba Yethemba	89.2	89.2	89.2	89.2	88.9	54.3	54.1	54.3	54.3	54.1
Tsolwana	84.4	82.8	78.9	77.3	83.1	58.0	56.8	58.0	52.5	49.7
Inkwanca	85.5	82.7	72.3	69.5	84.9	66.0	65.4	66.0	52.8	49.4
Lukanji	83.6	81.9	73.6	71.9	81.9	65.5	63.7	65.5	55.4	52.0
Intsika Yethu	71.4	69.2	57.4	55.2	69.6	60.8	59.0	60.8	46.8	42.8
Emalahleni	78.7	73.9	67.0	62.3	76.8	67.3	65.4	67.3	55.6	49.0
Engcobo	72.6	70.9	56.3	54.5	71.4	59.6	58.4	59.6	43.3	40.4
Sakhisizwe	81.4	80.2	66.6	65.4	80.7	63.7	63.0	63.7	48.8	46.9
Elundini	78.9	78.1	61.5	60.7	78.1	67.5	66.7	67.5	50.1	48.5
Senqu	85.4	84.9	82.6	82.1	84.9	69.3	68.7	69.3	66.4	65.3
Maletswai	85.6	85.4	69.1	68.9	85.1	65.8	65.4	65.8	49.3	48.6
Gariep	89.8	89.8	89.2	89.2	89.6	41.2	41.0	41.2	40.7	40.4
Ngquza Hill	84.7	84.6	70.3	70.2	84.5	63.7	63.5	63.7	49.3	49.0
Port St Johns	92.3	92.3	86.1	86.1	92.3	69.6	69.6	69.6	63.4	63.4
Nyandeni	85.0	84.7	73.6	73.3	84.6	62.4	62.0	62.4	51.0	50.3

Mhlontlo	73.9	72.4	53.0	51.5	72.8	65.9	64.8	65.9	45.0	42.4
King Sabata Dalindyebo	75.6	74.1	56.1	54.6	74.8	64.8	64.0	64.8	45.3	43.0
Matatiele	78.1	77.2	64.9	63.9	77.2	65.8	64.8	65.8	52.5	50.7
Umzimvubu	74.9	73.9	64.7	63.6	73.8	60.2	59.2	60.2	50.0	47.9
Mbizana	61.9	60.8	49.1	47.9	61.3	51.2	50.6	51.2	38.4	36.7
Ntabankulu	74.2	73.5	66.5	65.8	72.9	62.3	61.0	62.3	54.6	52.6
Nelson Mandela Bay	74.9	74.7	60.4	60.1	74.6	55.0	54.7	55.0	40.4	39.8

Table B4-3: Disaggregated BII scores within each ecoregion following the conversion of available land to cultivation land-use for the nine land-use scenarios used in this study.

		Arable			Marginal			All		
	Normal	degraded	Light	Both	degraded	Light	Both	degraded	Light	Both
Albany Thickets	89.4	89.3	87.3	87.2	88.6	82.3	81.4	88.5	80.2	79.3
Lowland Fynbos And Renosterveld	72.0	71.9	65.9	65.8	71.9	60.4	60.4	71.8	54.4	54.3
Montane Fynbos And Renosterveld	93.9	93.8	93.1	93.1	93.8	88.4	88.4	93.8	87.7	87.7
Drakensberg Montane Grasslands, Woodlands And Forests	82.6	81.2	72.1	70.7	81.8	62.2	61.4	80.4	51.7	49.5
Southern Africa Mangroves	93.7	93.7	81.7	81.7	93.7	82.4	82.4	93.7	70.5	70.5
Highveld Grasslands	82.9	82.3	73.2	72.6	82.1	62.0	61.2	81.5	52.2	50.8
Nama Karoo	89.8	89.8	89.5	89.5	89.6	67.4	67.3	89.6	67.2	67.1
Knysna-Amatole Montane Forests	79.6	78.0	64.1	63.5	78.6	61.4	61.4	78.0	46.9	46.3
Kwazulu-Cape Coastal Forest Mosaic	84.6	83.9	65.8	65.1	84.3	65.0	64.7	83.7	46.1	45.2
Succulent Karoo	92.8	92.1	92.1	92.1	92.1	92.1	92.1	92.1	92.1	92.1

Appendix C: Supporting information for Chapter 6

Petchey's FD (Petchey & Gaston, 2002) was used to calculate the functional diversity of communities within each land-use class. The functional trait dendrograms are presented in Figure C1. This shows the reduced species richness values under compared to native vegetation. Species names are abbreviations and refer to species indicated in Table 6.2.

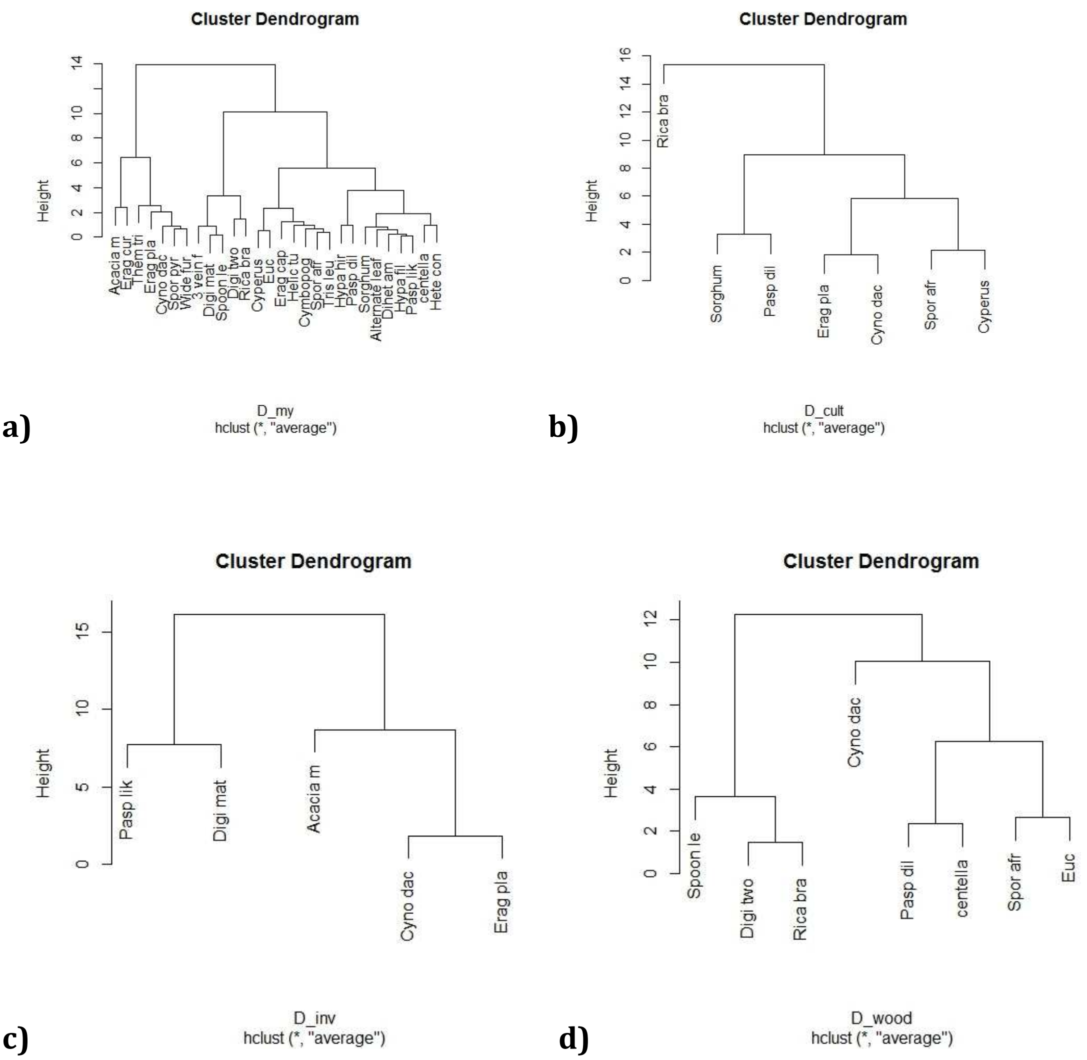

Figure C1: Functional trait dendrograms as calculated by Petchy's FD for a) native vegetation, b) cultivated c) *Acacia mearnsii* woodlots d) *Eucalyptus* woodlots