Quantifying the effectiveness of private land conservation areas in preventing losses of natural land cover and biodiversity intactness across South Africa

by

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Abstract

Global biodiversity conservation targets cannot be achieved by relying on state-owned protected areas (PAs) alone. Private land conservation areas (PLCAs) are one potential complementary conservation strategy. However, despite their increasing extent and recognition, little is known about their effectiveness in conserving biodiversity, or how different environmental and social-ecological factors influence their effectiveness. In South Africa, a long history of conservation on PLCAs and the diverse PLCA models provide an interesting case study to address this knowledge gap. The effectiveness of PLCAs across South Africa, and factors influencing their effectiveness, were thus quantified using losses in natural land cover (NLC) and the biodiversity intactness index (BII) as proxies. NLC was based on 1990 and 2013 national land cover maps, while BII represented a measure of the percentage of major taxa that can persist in an area given different land use scenarios. Points within PLCAs were matched with unprotected control points to test the prediction that if PLCAs offer effective protection, losses in NLC and BII would be significantly lower within their boundaries in comparison to unprotected controls exposed to similar conditions. NLC and BII losses were then compared across different types of PLCAs, with the hypothesis that legally protected PLCAs would be more effective than the informal ones. Of particular interest was also how different factors influenced the effectiveness of PLCAs in preventing losses of NLC and BII. In that regard accessibility (distance to road, distance to town, elevation and slope), rainfall, age and size of PLCAs were considered as explanatory variables. There were significant differences in losses in NLC and BII between PLCAs and matched unprotected areas. PLCAs lost 3% NLC and 2% BII between 1990 and 2013, while unprotected areas lost 6% NLC and 4% BII. These findings indicate the relative effectiveness of PLCAs, and provide insight into the implications of NLC loss on biodiversity intactness, thus advancing standard approaches for quantifying PA effectiveness. There were also significant differences in losses of NLC and BII between different types of PLCAs. However, contrary to the hypothesis, effectiveness did not depend on legal protection, as informal PLCAs were relatively more effective than some of the formally protected ones. NLC and BII losses were likely to occur at points within PLCAs that were closer to towns, further from roads, with low elevation, gentle slopes, within small and old PLCAs, and with low rainfall. This supports research on state-owned PAs, in which highly accessible areas were shown to be less effective due to higher human pressure. This study provides evidence that PLCAs are relatively effective, which is highly relevant given current discussions around their inclusion towards biodiversity targets. The study also highlights how different factors influence the effectiveness of PLCAs, which has important implications on where best to establish future PLCAs and how different management strategies and policies can be better placed to facilitate biodiversity conservation within PLCAs. The study contributes to the growing body of knowledge about PLCAs as a complementary biodiversity conservation strategy worth considering, which future studies can build upon.

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Abstrak

Wêreldwye pogings of biodiversiteit te beskerm kan nie eksklusief op beskermde gebiede (BG) wat deur die staat besit word staat maak nie. Privaat land bewaring gebiede (PLBGe) is een moontlike, komplimentêre bewaring strategie. Ten spyte van die immer-groeiende omvang en erkenning van dié gebiede wêreldwyd, is die mate tot wat PLBGe effektief is meestal steeds onbekend. Kennis met betrekking tot die invloed van kontekstuele sosiaal-ekologiese faktore op genoemde effektiewiteit is soortgelyk beperk. Danksy Suid-Afrika se lang geskiedenis met PLBG bewaring, sowel as sy diversiteit van PLBG instrumente, bied dié land 'n interessante gevallestudie wat hierdie kennisgaping kan aanspreek. Hierdie studie het dus die effektiwiteit van PLBGe regoor Suid-Afria, sowel as die faktore wat daardie effektiwiteit beïnvloed, gewantifiseer, Tot daardie doeleinde het ek verliesse in natuurlike gronddekking (NG), sowel as die biodiversiteit intaktheid indeks (BII), as proksies gebruik. Die NG was gebaseer op die nasionale landbedekking kaart van 1990 en 2013 en die BII verteenwoordig die persentasie van 'n meerderheid taksa wat in 'n area kan voortbestaan onder verkillende landgebruike. Ek het punte binne PLBGe gepaar met kontroleer punte buite PLBGe om die voorspelling te toets dat, indien PLBG effetiewe bekserming bied, verliesse in NG and BII aansienlik laer sou wees binne hulle grense in vergelyking met onbeskermde kontroleer punte met soortgelyke omgewings. Ek het daarna NG en BII verliesse vergelyk tussen verskillende tipe PLBGe, met die verwagting dat PLBGe wat wettig beskerm word meer effektief sou wees as meer informele PLBGe. Ek was veral geïnteresseer in hoe verskillende kontekstuele faktore die effektiwitiet van PLBGe beinvloed, soos gemeet deur die vermoë van 'n PLBG om verliese van NG en BII te voorkom. Moontlike verklarende faktore het toeganklikheid (gemeet deur afstand-tot-by-naaste-pad en –dorp, sowel as elevasie en helling), reënval, ouderdom en gebied grote ingesluit. Daar was 'n aansienlike verskil in NG en BII verliesse tussen PLBGe en gepaarde onbeskermde punte. Terwyl PLBGe 3 % NG en 2 % BII verloor het tussen 1990 en 2013, het onbeskermde punte 6 % NG en 4 % BII verloor. My bevindings dui op die relatiewe effektiewiteit van PLBGe, bied insig oor die implikasies van NG verlies vir biodiversiteit intaktheid, en bevorder die tegniese kwantifisering van BG effektiewiteit. Daar was ook aansienlike verskille tussen NG- en BII verliese tussen verskillende kategorieë PLBGe. Teen verwagtinge was effektiwiteit egter *nie* afhanklik van wettige beskerming nie, aangesien informele PLBGe relatief meer effektief as sekere kategorieë formele gebiede was. NG en BII verliesse is meer gereëld gevind op punte wat nader aan dorpe en verder van paaie was, punte met laer elevasie, sagter hellings, en laer reënval, en punte binne ouer en kleiner beskermde gebiede. Hierdie resultate is ondersteunend van navorsing op staat-beheerde BG, wat wys dat meer toeganklike areas minder effektief is as gevolg van hoër mensdruk. Hierdie studie verskaf bewys dat PLBGe relatief effektief is, 'n bevinding wat uiters relevant is vir kontemporêre besprekings rondom hulle insluiting in formele biodiversiteit teikens. Hierdie studie beklemtoon ook hoe verskeie faktore die effektiwiteit van PLBGe beïnvloed, wat belangrike

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implikasies het vir die optimale vestiging van toekomstige BGs, sowel as die mees gepaste besturing strategieë en beleide vir biodiversiteit bewaring in verskillende PLBGe. Die studie lewer bydrae tot die toenemede kennis van PLBGe as ‘n komlimentêre bewarings strategie.

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Dedication

This thesis is dedicated to Josiah and Shylet Shumba

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Acronyms

BII	Biodiversity Intactness Index
CBD	Convention on Biological Diversity
CIESIN	Center for International Earth Science Information Network
DEA	Department of Environmental Affairs
DEM	Digital Elevation Model
GIS	Geographical Information System
GLM	Generalized Linear Model
GLMM	Generalized Linear Mixed Model
IUCN	International Union for Conservation of Nature
IUCN-WCPA	IUCN-World Commission on Protected Areas
NGO	Non-Governmental Organization
NLC	Natural Land Cover
OECM	Other Effective area-based Conservation Measures
PA	Protected Area
PACAD	Protected Areas and Conservation Areas Database
PLCA	Private Land Conservation Area
PPA	Privately Protected Area
QGIS	Quantum Geographic Information System
SACAD	South African Conservation Areas Database
SAPAD	South African Protected Areas Database
SANParks	South African National Parks
SES	Social Ecological System
SRTM	Shuttle Radar Topography Mission
WDPA	World Database of Protected Areas

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Preface

This thesis is presented in five chapters. Chapter one is the introduction and literature review for the study. Chapter 2 covers the general methods used. Thesis data chapters (chapter 3 and 4) were written as standalone ready for submission journal articles following the journal of Biological Conservation guidelines. There is therefore some repetition among the chapters.

Chapter 1	**General Introduction** This chapter provides the background and context within which this study is framed. The section is aimed, to provide sufficient information to readers with no prior knowledge of private land conservation areas (PLCAs) in South Africa. It therefore includes background information about what is known about PLCAs, their importance, history, limitations and the knowledge gaps which this study aims to address.
Chapter 2	**General Methods** The chapter provides an overview of the methods used in the thesis. Methods specific to each data chapter are however repeated within the respective chapters.
Chapter 3	**Effectiveness of private land conservation areas in conserving natural land cover and biodiversity intactness** In this chapter the effect of protection offered by PLCAs was quantified by comparing losses in natural land cover and biodiversity intactness between points within PLCAs and unprotected control areas with similar environmental variables.
Chapter 4	**Factors influencing the effectiveness of private land conservation areas in conserving natural land cover and biodiversity intactness** Here the objective was to understand how different environmental and social-ecological factors influence the retention of natural cover and biodiversity intactness, and to establish the best models for explaining variations in natural cover and biodiversity intactness losses within PLCAs.
Chapter 5	**General Discussion** This section consolidates results from the two data chapters and the introduction to come up with a synthesized understanding of the contribution of PLCAs. The section provides key conclusions about PLCAs, limitations of this study, offers recommendation for biodiversity conservation in the context of PLCAs and how future studies can be done to further understand PLCAs.

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Chapter 1: General Introduction

Biodiversity losses due to human activities have been widely documented and the establishment of protected areas (PAs) has been a key conservation strategy (Watson et al. 2014, Maron et al. 2018). Most PAs are however predominantly state-owned, and it is apparent that despite their importance they cannot achieve biodiversity conservation targets alone (Brooks et al. 2004, Hayes and Ostrom 2005, Venter et al. 2017, Maron et al. 2018). Shortcommings in the effectiveness of PAs attributed to poor funding particulaly in developing countries, biases in their location towards marginal, high elevation areas, unfavourable for agriculture, which are not usually biologically diverse (Rodrigues et al. 2004, Joppa and Pfaff 2009, Venter et al. 2017), and increased external pressures from surrounding communities (Jones et al. 2018) have been realised.

Consequently, there have been increased calls for conservation to look beyond state-owned PAs and explore alternative strategies such as private land conservation areas (PLCAs) (Borrini-Feyerabend et al. 2013, Stolton et al. 2014, Dudley et al. 2018). A PLCA is a privately owned piece of land managed for biodiversity conservation, protected with or without formal government recognition (Carter et al. 2008, Stolton et al. 2014, Clements et al. 2016a). Private entities involved can include individual(s), communities, corporations, non-governmental organisations (NGOs), universities and / or religious groups (Stolton et al. 2014). The importance of PLCAs in biodiversity conservation through increasing the total area available for conservation, increasing connectivity, and representation of threatened habitats has increasingly been recognised over the past decades (Fitzsimons and Wescott 2008, Gallo et al. 2009, Stolton et al. 2014, Maciejewski and Cumming 2015). However, the effectiveness of their contribution remains poorly understood (Langholz and Lassoie 2001, Carter et al. 2008). Reasons such as their small size (Langholz and Lassoie 2001, Baldwin and Fouch 2018), profit oriented management systems (Clements et al. 2016a), and uncertain land tenure systems (Langholz and Krug 2004) are widely thought to compromise their long-term importance in biodiversity conservation, hence their exclusion in most national or regional conservation plans and strategies (Stolton et al. 2014).

There is thus an urgent need to quantify the effectiveness of PLCAs in protecting biodiversity given the increase in their extent and recognition as a potential complementary strategy to state-owned PAs. South Africa is one country appropriate to address this knowledge gap. The country is biologically diverse and has a long history of conservation on private lands, with a significant amount of land under private ownership (Wright et al. 2018, De Vos et al. 2019), hence it is appropriate to understand the effectiveness, opportunities and threats associated with PLCAs as a conservation strategy.

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This introduction gives a background about what is known about PLCAs, why they can be relevant in biodiversity conservation today, their history, the social-ecological factors that shape their long-term effectiveness, and the different ways their effectiveness can be quantified. The section is then concluded with a motivation of why South Africa is a good place to understand PLCA effectiveness, and a statement of the problem as well as study objectives.

1.1 The need for PLCAs: Limitations with state-owned protected areas

In recent years, scientists and policy makers have increasingly recognised that largely state-owned PAs are inadequate to exclusively achieve global biodiversity conservation targets (Brooks et al. 2004, Hayes and Ostrom 2005, Venter et al. 2017, Maron et al. 2018). A number of reasons have been identified as shortcomings, hence the need for alternative conservation strategies (Jones et al. 2018, Maron et al. 2018). There is a notable bias in where most state-owned PAs are located, i.e. in areas with high elevation, low rainfall, high temperatures, high occurrences of livestock diseases, far from roads, and cities, which are often not the most threatened environments (Rodrigues et al. 2004, Joppa and Pfaff 2009, Venter et al. 2017). It can thus be argued that these PAs were designed with little or no consideration of ecological importance but rather in areas where they will not compete with humans i.e. in areas less attractive for agriculture (Joppa and Pfaff 2009, Venter et al. 2017). Although the establishment of PAs now involves systematic planning incorporating biological significance, threats, and connectivity, biases in where state-owned PAs are located still exist (Myers et al. 2000, Sarkar et al. 2006, Joppa and Pfaff 2009, Venter et al. 2017). Furthermore, particularly in developing countries state-owned PAs are typically poorly funded and many suffer from over-exploitation by humans that live near their boundaries (Bruner et al. 2001, Jones et al. 2018).

Also, the establishment of state-owned PAs has traditionally been problematic from a social justice perspective leading to negative perceptions from surrounding communities (Naughton-Treves et al. 2005). Up until the 1990s, most state-owned PAs were established with protectionist, state-centred approaches that had few benefits to society, and hence lacked support from respective communities even up to today (Naughton-Treves et al. 2005). When societal interests and livelihoods are threatened by a conservation model, its chances of succeeding are often compromised (Kideghesho et al. 2007). In such circumstances surrounding communities end up having negative attitudes towards PAs which consequently affects their effectiveness in conserving biodiversity (Naughton-Treves et al. 2005). In some cases, the establishment of state-owned PAs has involved forced displacement of communities from their traditional lands and exclusion from accessing different ecosystem services (McIvor 1994). This has led to conflicts with society, which not only affects how PAs are viewed, but also excludes the role of indigenous knowledge in biodiversity conservation

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(West et al. 2006, Palomo et al. 2014). In other cases, conflicts between park officials and communities have led to complete degazettement of PAs (Newmark et al. 1993). Because state-owned PAs were historically established mostly on areas with poor agricultural potential (Joppa and Pfaff 2009, Venter et al. 2017), they are often associated with poor surrounding communities, which have to rely on biodiversity for income, food, fuel wood and building materials (Struhsaker et al. 2005, Brockington et al. 2006, Ferraro et al. 2011, Bowker et al. 2017). Consequently, there is a link between poverty and conservation, even though conservation can play an important role in poverty alleviation and human well-being (Fisher and Christopher 2007, Naidoo et al. 2019). Furthermore, reduced natural resource exploitation and infrastructure development around PAs is argued to intensify poverty among surrounding communities (Brockington et al. 2006, Ferraro et al. 2011). Nevertheless, the perceptions and relationships between surrounding communities and PAs can influence biodiversity conservation effectiveness. Although this has been mainly been shown among state-owned PAs, negative community perceptions can also affect PLCAs, and eventually such areas can become degraded or surrounded by degraded environments which affects general biodiversity conservation and provision of ecosystem services (McNeely 1994, Bowker et al. 2017). Positive perceptions towards state-owned PAs and community endorsement can however be improved by having inclusive conservation models that incorporate and benefit surrounding communities or are otherwise run by the communities themselves (Magome and Murombedzi 2003).

In the past decades, PLCAs have emerged as an important alternative *in situ* conservation strategy (Borrini-Feyerabend et al. 2013, Stolton et al. 2014). The Convention on Biological Diversity (CBD), Aichi target 11, of at least 17% of terrestrial land protected by 2020 (CBD 2010), even when achieved will leave 83% of the land unprotected. Consequently, PLCAs, and communal areas have a big role to play in covering the deficit, especially given that most ecological and evolutionary processes occur at scales larger than what state-owned PAs can cover (Maron et al. 2018, Donald et al. 2019). PLCAs increase the total area available for biodiversity conservation without increasing conservation costs on respective states (Stolton et al. 2014, Mitchell et al. 2018b). Consideration of PLCAs, is thus essential given that state land is limited, and further expansion of their PA estate may be inhibited by socio-economic and political factors (Mitchell et al. 2018b). The ‘Half-Earth’ call by Wilson, in which he argues that to effectively conserve biodiversity there is need to dedicate at least half of the earth to conservation (Wilson 2016, Dudley et al. 2018), despite some criticism (Büscher et al. 2017), can also only be achieved when state-owned PAs are considered together with PLCAs. PLCAs have also been shown to protect some threatened species and disappearing habitats underrepresented within state-owned PAs, thus serving as custodians in the absence of government institutions (Langholz and Krug 2004, Gallo et al. 2009, Clements et al. 2019).

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Given the biases in where state-owned PAs were established (Rodrigues et al. 2004, Joppa and Pfaff 2009, Venter et al. 2017), considering PLCAs might thus be essential for the representation of habitats underrepresented within state-owned PAs. PLCAs also improve connectivity between fragmented PAs thus serving as corridors for species migration (Fitzsimons and Wescott 2008, Stolton et al. 2014, Maciejewski and Cumming 2015).

The increase in the profile of PLCAs, and increase in literature documenting their importance shows that their recognition is gaining momentum (Stolton et al. 2014, Jonas and MacKinnon 2016, Bingham et al. 2017, Mitchell et al. 2018b). PLCAs have been shown to be innovative and flexible in options for funding because of their economically driven objectives and the ability of landowners to have collaborative relations with other managers and stakeholders (Langholz and Krug 2004). This is critical in developing countries where funding for conservation activities is limited (Bruner et al. 2001, Jones et al. 2018). PLCAs can thus pursue objectives benefitting the economy, society and biodiversity, for example through hunting and ecotourism activities (Langholz and Lassoie 2001, Pfaff and Robalino 2012), provision of non-timber forest products to surrounding communities and employment (Bond et al. 2004, Jones et al. 2005). Furthermore, PLCAs can more easily attract funding for conservation work from the general public, donors and NGOs than state-owned PAs, which is essential especially in countries with high corruption and poor accountability (Laurance et al. 2006). PLCAs thus appear as a strategy worth considering in modern-day conservation in which neoliberalism is the norm, whereby both economic and conservation need to be considered concurrently (Pfaff and Robalino 2012). However, despite the potential and recognition of PLCAs, their contribution and effectiveness in biodiversity conservation remains largely unknown (Bingham et al. 2017, Selinske et al. 2019). There is therefore an urgent need to quantify the effectiveness of PLCAs and to understand whether they can be relied upon as a long term biodiversity conservation strategy as well as to quantity how different factors influence their effectiveness (Langholz and Lassoie 2001, Carter et al. 2008, Holmes 2013, Selinske et al. 2019).

There have also been increased cases in which state-owned PAs are degazetted, downsized or downgraded (De Vos et al. 2019, Kroner et al. 2019). This is when changes in policy or law result in a PA being made smaller or completely losing its protection status (Kroner et al. 2019). When that happens their effectiveness in biodiversity conservation is subsequently compromised, hence the need for other alternative strategies. Although PLCAs also experience cases of degazettement, downsizing, and downgrading (De Vos et al. 2019) their existence provides an essential back-up when state-owned are degazetted for mining, urban development or other political or economic reasons (Kroner et al. 2019).

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1.2 History of private land conservation areas

PLCAs in their modern form have existed in various arrangements since the 18th century (Alderman 1994, Stolton et al. 2014). They were formed when individuals, organisations or communities set aside land for hunting, managing species, regulation of exploitation, protecting fodder for livestock as well as for cultural and religious reasons (McNeely 1994, Child 2004, Chape et al. 2005, Stolton et al. 2014). For example, in 1824, Veracruz, Mexico, a botanist secured a large piece of land which he managed both as a coffee plantation and as a private reserve (Ochoa-Ochoa et al. 2009). In Germany, one of the pioneer PLCAs was established in the 1880s to protect the scenic beauty of a mountain range, which was threatened by mining activities (Stolton et al. 2014). The United Kingdom also has a notable history of conservation on private lands. Organisations such as the National Trust and the Royal Society for the Protection of Birds are prominent pioneer organisations to establish PLCAs across the country (Langholz and Krug 2004, Stolton et al. 2014), with the former known to have attained its first reserve in Cambridgeshire in 1899 (Alderman 1994). In the 20th century the number of PLCAs substantially expanded in the United States of America, Australia and a number of developing countries, which was attributed to the growth of ecotourism and hunting (Langholz and Lassoie 2001, Adams 2004, Stolton et al. 2014).

However, conservationists and policy makers have not paid much attention to PLCAs, and their growth and extent has been typically under-reported when compared to state-owned PAs (Langholz and Krug 2004, Bingham et al. 2017). The first notable recognition of PLCAs was in 1962, when delegates at World Congress on National Parks acknowledged that a number of nature reserves were privately owned (Adams 1962, Langholz 2010). Their proliferation was however, relatively unnoticed, until the 1990s, when Alderman published research on private lands used for nature tourism in Africa and Latin America (Alderman 1994). By the 1990s several countries had laws and policies to support conservation on private lands (Langholz 2010, De Vos et al. 2019). Another landmark motion in the role of PLCAs was the 5th World Parks Congress in Durban, South Africa, 2003. The congress saw 154 participating nations approving the Private Protected Area Action Plan (IUCN 2005), which officially recognised and endorsed the role of PLCAs in biodiversity conservation. To date a number of action plans for incorporating PLCAs into national protected area systems, building their capacity in biodiversity conservation, and providing guidelines for best practices have been produced, indicating increased interests in private areas as a biodiversity conservation strategy (Stolton et al. 2014, Jonas and MacKinnon 2016, Mitchell et al. 2018b). The extent and number of PLCAs today continues to grow, particularly in Europe, Australia, Latin America, Canada, and southern Africa (Langholz and Lassoie 2001, Stolton et al. 2014). The increase in ecotourism opportunities, and societal interest in biodiversity conservation,

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together with failure of governments to effectively deal with biodiversity conservation issues are some of the major motivations behind the increase of PLCAs worldwide today (Langholz and Lassoie 2001, Stolton et al. 2014, Selinske et al. 2017, De Vos et al. 2019).

However, widespread scepticism regarding the conservation potential and sustainability of PLCAs has been a major drawback in efforts to recognise and incorporate PLCAs in national and regional conservation action plans (Stolton et al. 2014, Bingham et al. 2017, Mitchell et al. 2018b). The lack of a universally agreed definition of a PLCAs is one reason why their inclusion has been problematic (Bingham et al. 2017, Donald et al. 2019). To date there are up to 50 definitions with terms such as private conservancy, private park, private game reserve, private nature reserve and other effective area-based conservation measure (OECM), being widely used to refer to PLCAs, despite different contextual meanings (Carter et al. 2008, Capano et al. 2019, Donald et al. 2019). In an effort to standardize definitions, the International Union for Conservation of Nature (IUCN) recommended a privately protected area (PPA) to be defined as a protected area under private governance, which can be by individual(s), NGOs, universities or religious groups (Cumming and Daniels 2014, Stolton et al. 2014). According to the IUCN definition of a "privately protected area", an area first has to qualify as a PA (a clearly defined area dedicated to the conservation of biodiversity and ecosystem services, through legal or other effective means) to be included under the definition (Stolton et al. 2014). In that view, not all conservation efforts on private lands can or should be considered protected, while some require minor management changes to be considered protected (Stolton et al. 2014, Mitchell et al. 2018b). Consequently, the definition fails to accommodate areas dedicated to biodiversity conservation without formal or legally recognised protection status. The term private land conservation area (PLCA) is therefore used in this thesis to refer to private land managed for biodiversity conservation, with diverse levels of legal protection ranging from minimum protection to strongly protected, thus incorporating both formal and informal conservation efforts (Carter et al. 2008, Cousins et al. 2008, Pasquini et al. 2010, Von Hase et al. 2010, Clements 2016). This definition is also in line with the recent definition of an OECM, produced by the CBD in November 2018, at the 14th conference of the parties, in which parties officially recognised and defined OECM as non-protected areas, managed in ways ensuring long-term biodiversity conservation, and associated ecosystem services (Dudley et al. 2018, Mitchell et al. 2018a, WCPA and IUCN 2018). In this study all private lands dedicated for biodiversity conservation in South Africa were considered thus including official PPA and OECM. Although the inclusion of a broader spectrum of PLCAs especially the informally protected ones brings uncertainty regarding their sustainability, such areas remain important in conservation efforts (Cadman et al. 2010, Selinske et al. 2017).

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Another reason why PLCAs have been relatively less recognised in biodiversity conservation is because of poor reporting of their locations and extent especially the informal ones (Rissman et al. 2017). Although anecdotal data shows that the number of PLCAs continues to increase, with their global extent believed to have nearly doubled over the past two decades, their absolute number remains unknown (Langholz and Lassoie 2001, Rissman et al. 2017, Fouch et al. 2019). The current number of formally recorded PLCAs according to the World Database of Protected Areas (WDPA) is approximately 14 296 areas, existing over 25 countries covering a total area of 161, 634 km^2 (Stolton et al. 2014, Bingham et al. 2017, Mitchell et al. 2018b), which is however not an exhaustive list (Rissman et al. 2017). For reasons such as poor reporting, keeping information about private areas 'private', political reasons, and lack of mapping capacity there is no complete spatial data about PLCAs i.e. where they are located, their respective sizes, dates of establishment and their governance (Rissman et al. 2017, Clements et al. 2018). The availability of such information is crucial in conservation. Without current or historical information, it is difficult to account for and evaluate the contribution of PLCAs as well as incorporate them into regional and national conservation plans (Langholz and Lassoie 2001, Rissman et al. 2017, Clements et al. 2018). Nevertheless, PLCAs continue to proliferate, with communities, organisations, and individuals getting involved in their establishment (Bertzky et al. 2012), hence the urgent need to understand their contributions to biodiversity conservation.

1.3 Private land conservation areas as social-ecological systems

Biodiversity conservation is no longer strictly the protection of rare and endangered species but is becoming more human-centred, in which sustainable use is the objective (Dudley 2008, Cumming et al. 2015). Consequently, PLCAs cannot be viewed as ecological islands (Janzen 1983), but rather as social-ecological systems (SESs), characterised by human and biophysical components, interacting across multiple scales (Berkes and Folke 1998, Palomo et al. 2014, Cumming et al. 2015). A SES can be defined as a nested multilevel system, in which social and ecological components interact in an interdependent manner with the society depending on and modifying the ecosystem (Berkes and Folke 1998, Gunderson and Holling 2002, Biggs et al. 2015). These SESs produce a set of ecosystem services essential for humans such as clean water, food, air, pollination, climate regulation and protection from climatic hazards such as floods and storms, recreation and other cultural services as well as contribute to local and national economies through hunting and ecotourism (Biggs et al. 2012, Cumming et al. 2015). However, because an ecosystem can provide a bundle of products and services, management choices, economic factors, and other social-ecological factors influence the services a SES can provide (De Fries et al. 2004, Biggs et al. 2012). It is therefore essential to understand PLCAs as complex SES whose effectiveness is dependent on multiple

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factors (Cumming et al. 2015, Clements et al. 2016a). Land use and management choices among PLCAs are thus associated with trade-offs for other services knowingly or unknowingly (Cumming and Allen 2017). For example, a decision to grow crops within a PLCA will provide more food and financial options, but will reduce the capacity of the system to conserve biodiversity, regulate water quality, prevent soil erosion and possibly expose it to invasive species (Reyers et al. 2009, Biggs et al. 2012).

The use of fire is another example of how SESs in Savanna and Grassland biomes are influenced by management decisions and environmental factors. While controlled periodic burning of parts of the property leads to a diverse and resilient system, no burning or very frequent burning has opposite results (Kruger and Bigalke 1984, Carpenter et al. 2001). Periodic use of fire stimulates regrowth of succulent foliage for wildlife, while no burning causes the accumulation of moribund material, which not only hinders grass regrowth but increases the fuel load thus setting the scene for larger uncontrolled fires which have adverse effects on biodiversity (Belsky 1992, Carpenter et al. 2001, Brockway et al. 2002). Other management choices such as the use of fences, animal translocations, manipulation of animal densities through supplementary feeding and creation of waterholes also influence the effectiveness of PLCAs in positive and negative ways (Child et al. 2013, Cumming and Allen 2017). In the worst case, undesirable results such as soil erosion, vegetation cover loss, bush encroachment and occurrence of invasive species are the indicators of ineffectiveness in biodiversity conservation and provision of ecosystem services (Carpenter et al. 2001, Biggs et al. 2012, Cumming et al. 2015, Cumming and Allen 2017). It is therefore useful to consider the complexity of the factors that influence the effectiveness of SESs at patch, area, landscape, national and regional scales and to use a social-ecological approach in understanding and managing PLCAs (Ban et al. 2013, Palomo et al. 2014).

A number of studies have focused on state-owned PAs as SESs and how the different factors influence their effectiveness based on different proxies (Andam et al. 2008, Ament and Cumming 2016, Bowker et al. 2017). It is therefore timely to understand how different factors affect PLCAs and how they can be managed. This is particularly important since there has been a lot of criticism about the contribution of PLCAs, with size, motives, management strategies and unstable tenure systems being cited as some of the main reasons behind their omission in regional conservation action strategies (Langholz and Lassoie 2001, Clements et al. 2016a, Baldwin and Fouch 2018). By being typically small, i.e. < 10, 000 ha, PLCAs have been criticised for not being appropriate to support mega fauna requiring large areas (Langholz and Lassoie 2001, Baldwin and Fouch 2018). Small areas are also susceptible to edge effects due to large boundary-to-area ratios, making them highly susceptible to biodiversity loss, especially when surrounded by human communities (Langholz and Lassoie 2001, Joppa et al. 2008, Pfeifer et al. 2012). Also, by being small, PLCAs are potentially

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impacted by lack of spatial diversity (spatial heterogeneity), which consequently affects the diversity of the species they can protect (Allen et al. 2016). As a result, bigger PAs are preferred in conservation planning because they are likely to encompass diverse habitat patches that may make them more resilient to biodiversity loss (Allen et al. 2016, Lacher et al. 2019). Compared to smaller areas, large PAs can also easily attract funding for conservation work such as for law enforcement from government and NGOs, thus making them more likely to effectively conserve biodiversity than smaller ones (Blackman et al. 2015).

Profit generation is an important priority in many PLCAs particularly in southern Africa (Clements et al. 2016a). Consequently, such small parcels of land are often intensively managed and overstocked with animals ideal for the tourism and hunting markets at the expense of the vegetation and diversity of smaller vertebrate species (Cumming et al. 1997, Maciejewski and Kerley 2014, Clements 2016, Clements and Cumming 2017). Currently, ecotourism is regarded the world's fastest growing industry (Langholz and Krug 2004, El-Haggar and Samaha 2019) and in southern Africa it has been shown to generate revenue equivalent to that generated from farming, forestry, and fisheries combined (Scholes and Biggs 2004). This has led to doubts about the long-term effectiveness of PLCAs in protecting biodiversity particularly if profit generation is the major motive.

Accessibility is also another factor that has been shown to have a significant influence on the effectiveness of state-owned PAs and likely to have a similar effect on PLCAs (Bowker et al. 2017). Areas closer to human settlements, closer to road networks, with gentle slopes and low elevation are generally more accessible and hence vulnerable for exploitation and are also closer to markets for nature based products hence highly threatened (Pfeifer et al. 2012, Bowker et al. 2017).

The effectiveness of PLCAs can also be influenced by political factors especially associated with land tenure (Langholz and Krug 2004). Most PLCAs are informal and lack substantial legal framework to support their activities and existence (Langholz and Krug 2004). By existing under different levels of protection, ownership and permanence, PLCAs have uncertainties in their commitment to biodiversity conservation, with some having short-term arrangements of 5-10 years (Cadman et al. 2010). In such circumstances, perpetual biodiversity conservation cannot be guaranteed given the dynamic political and economic factors especially given that such agreements might at least in South African context coincide with elections, which might result in scepticism from a landowner's perspective in case there is change in the ruling party. Consequently PLCAs with long-term agreements might be essential for perpetual biodiversity conservation rather than the informal PLCAs which are conserved with minimum or no land use restrictions or regulations (Pasquini 2007, Cadman et al. 2010).

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1.4 Quantifying the effectiveness of conservation areas

With the understanding that PAs and PLCAs are SESs that affect and are affected by a number of factors comes the need to quantify how such factors influence their ability to protect biodiversity and provide the relevant ecosystem services. Despite the establishment of state-owned PAs and PLCAs, and huge investments in their management, biodiversity loss within PAs and PLCAs continues to be a problem (Salafsky et al. 2002, Chape et al. 2005, Jones et al. 2018). It therefore seems effective ways to prevent biodiversity loss are yet to be mastered. The existence of PAs whether state-owned, private, legally recognized or informal can only remain relevant if they can effectively conserve biodiversity and provide ecosystem services within their boundaries and or beyond (Salafsky et al. 2002, Hockings 2003). There is therefore need to objectively quantify the effectiveness of conservation strategies and to understand what makes them work, how, and why, as well as their threats and weakness so that lessons can be learnt to enhance or maintain their effectiveness through adaptive management strategies and policies (Balmford et al. 2001, Hockings 2003). There is also need for conservationists, managers and policy makers, to know how their efforts and investments contribute to biodiversity conservation (Salafsky et al. 2002, Parrish et al. 2003, Cook et al. 2014). However, despite its importance, reliable information about the effectiveness of PLCAs and state-owned PAs is limited (Honey-Roses et al. 2011).

Effectiveness is a challenging entity to measure (Parrish et al. 2003). In this context, effectiveness refers to how well a strategy (e.g., PLCAs) achieves its goals and objectives, which is primarily the conservation of biodiversity (Hockings et al. 2006, Stoll-Kleemann 2010). This can involve either quantifying threats or ecological integrity (Ervin 2003, Parrish et al. 2003). Although there might not be a direct relationship between threat level and biodiversity, a proxy for threats can give an indication about effectiveness (Parrish et al. 2003). For example, reduced poaching is a good indication of anti-poaching activities, but alone it is not a sufficient indicator of effectiveness, since systems are seldom threatened by one factor (Parrish et al. 2003). Alternatively, ecological integrity can be considered by using aspects such as species richness, genetic diversity, and vegetation characteristics (Karr and Dudley 1981, Parrish et al. 2003).

Because of limited funding and methodological constraints, it is challenging to systematically carry out direct and extensive evaluations of effectiveness across PLCAs and state-owned PAs (Nelson and Chomitz 2011, Fouch et al. 2019). There have thus been increased efforts to quantify and model biodiversity remotely (Nagendra 2001, Gillespie et al. 2008). Remote sensing technology and Geographical Information Systems (GIS) offer an opportunity to assess the contribution of PLCAs and state-owned PAs to biodiversity conservation at local, landscape, regional, and global spatial scales (Nagendra 2001). The

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technology provides a consistent and systematic view of the earth's surface which allows comparisons and changes to be assessed in a relatively inexpensive manner compared to direct field monitoring (Nagendra 2001, Gillespie et al. 2008). Assessments based on remote sensed data have thus become a common way of quantifying effectiveness among PAs (Bruner et al. 2001, Liu et al. 2001, Clark et al. 2008, Gillespie et al. 2008, Pfeifer et al. 2012, Heino et al. 2015, Ament and Cumming 2016, Bowker et al. 2017, Andam et al. 2018). Data such as forest cover loss (Pfeifer et al. 2012, Bowker et al. 2017, Donald et al. 2019), occurrence of forest fires (Nelson and Chomitz 2011), NLC change (Bruner et al. 2001, Ament and Cumming 2016), habitat change and fragmentation (Liu et al. 2001) have been used to quantify the effectiveness of many systems of state-owned PAs around the world. By comparing changes and rates of change of such proxies before and after protection as well as by comparing areas with and without protection the effect of a conservation strategy can be established as well as help display possible trajectories of change (Naughton-Treves et al. 2005, Beresford et al. 2013). With most studies having focused on state-owned PAs, it is essential to use such proxies and methodologies to understand the contribution of PLCAs.

Changes in land cover and use, especially from natural to less natural conditions affect biodiversity through influencing species composition and species movements (Vitousek et al. 1997, Fischer and Lindenmayer 2007). The direct relationship of NLC with biodiversity, nutrient cycling, soil structure and climate also makes NLC an important proxy for quantifying the effectiveness of a conservation strategy (Nagendra 2001). NLC loss involves cases in which the earth surface is transformed from its natural pristine state for infrastructure development, crop and livestock production as well as other land uses. This consequently affects biodiversity through promoting invasive species, habitat loss and fragmentation (Hamilton et al. 2013). Changes in land cover/use on a PLCA can be viewed as a continuum, with natural undisturbed areas with indigenous vegetation on one end, while at the other end, land is completely converted to agriculture or urban use (Foley et al. 2005, Hamilton et al. 2013). Areas can be at different stages along the continuum and the time to pass through the different stages varies, depending on management and different socio-economic factors (Foley et al. 2005). Vegetation cover proxies such as the amount of forest cover, and Normalized Difference Vegetation Index (NDVI) have thus been used to represent the proportion of natural and transformed areas within a landscape (Pfeifer et al. 2012, Bowker et al. 2017). NLC among conservation areas can thus be used to understand effectiveness and resilience of a strategy to changes in social, economic, ecological, and political factors (Brooks et al. 2009).

A lot of detail is however lost in attempts to classify remotely sensed data into a binary variable (natural or non-natural) in the use of NLC, which is a limitation. This is especially important given that different land cover types impact biodiversity differently. Although a farm

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and a mine are both non-natural cover types, a farm will likely support more populations of insects and birds than a mine (Scholes and Biggs 2005, Child et al. 2009). It is therefore essential to have more direct ways of quantifying biodiversity, or rather, ways that appreciate how different land cover / uses affect biodiversity. The Biodiversity Intactness Index (BII) is one such multiplicative proxy that has been developed to provide a quick assessment of biodiversity (Scholes and Biggs 2005). The index uses land use scenarios and population impacts of different land uses under different biomes (Forest, Fynbos, Savanna, Grassland, Thicket, Nama Karoo and Succulent Karoo) on different taxa to come up with a measure of intactness (Scholes and Biggs 2005). The index is thus a measure of the proportion of major taxonomic groups (plants, mammals, birds, reptiles, and amphibians) that can exist in an area depending on the land use (protected, moderately used, degraded, cultivated, plantation and urban), relative to a pristine undisturbed population (Scholes and Biggs 2005, Biggs et al. 2006). Despite its ability to provide insight into the consequences of NLC change for biodiversity, the index has not yet been used in assessments of PA effectiveness.

The use of proxies remain essential in quantifying the contribution of different conservation strategies to biodiversity conservation (Bowker et al. 2017, Fouch et al. 2019). Among the studies focussing on state-owned PAs, most were shown to be effective in protecting NLC, forest loss and biological diversity, relative to unprotected areas (Pfeifer et al. 2012, Bowker et al. 2017). There are however some methodological designs which have been shown to overestimate effectiveness (Andam et al. 2008, Pfeifer et al. 2012, Bowker et al. 2017). PAs and PLCAs are not randomly distributed, but were established with biological, economic, social and political reasons in mind (Andam et al. 2008, Venter et al. 2017). Most state-owned PAs have been shown to be disproportionately established on landscapes unfavourable for human use i.e. less accessible, low rainfall and low agricultural potential (Joppa and Pfaff 2009, Venter et al. 2017). There are therefore differences between PAs and areas immediately surrounding them, making direct comparisons inappropriate. Consequently, methods that do not account for contextual differences by comparing PAs with areas immediately surrounding them have been shown to overestimate effectiveness (Oliveira et al. 2007, Andam et al. 2008, Pfeifer et al. 2012). Factors such as distance to roads, distance to town, climate, vegetation, soils, elevation and slope can influence NLC loss, deforestation or biodiversity loss (Andam et al. 2008). Without protection, some areas will logically experience low levels of NLC or biodiversity loss by virtue of their location and inaccessibility, cushioning them from exploitation rather than the ‘protection’ itself (Andam et al. 2008, Bowker et al. 2017). Reliable conclusions about the effectiveness can thus be achieved by comparing PAs with unprotected areas exposed to similar environmental variables (Ament and Cumming 2016, Bowker et al. 2017). This can be achieved through a matching analysis which pairs

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protected and non-protected areas based on similarities in different contextual factors, resulting in improved assessments of effectiveness (Joppa and Pfaff 2010, Ho et al. 2011).

A number of studies have used the matching approach to quantify the effectiveness of state-owned PAs in reducing deforestation and NLC loss (Andam et al. 2008, Nagendra 2008, Gaveau et al. 2009, Ament and Cumming 2016, Bowker et al. 2017). No study has however been undertaken to quantify the effectiveness of PLCAs in Africa, with one similar study having been conducted in the USA, in which PLCAs were shown to be ineffective in comparison to unprotected areas and state-owned PAs in protecting NLC (Fouch et al. 2019). The main objective of this study was thus to use a matching approach to understand the effect of protection offered by PLCAs across South Africa as well as to understand how different social-ecological and environmental variables influenced their effectiveness using NLC and BII as proxies. NLC loss was based on respective 1990 and 2013 national land cover maps (DEA 2015a, b), while BII represented a measure of the proportion of major taxa that can persist in an area given different land use scenarios (Scholes and Biggs 2005).The prediction was that if PLCAs offer significant protection (i.e. are effective), losses in NLC and BII will be significantly lower within their boundaries in comparison to unprotected areas exposed to similar conditions. With the environmental and social-ecological factors, the aim was to determine the best model for explaining losses in NLC and BII, and establish how the different factors influenced losses in NLC and BII. The use of the two proxies together gives better understanding into the contribution of PLCAs and also the consequences of NLC loss on biodiversity intactness, which is unique since previous studies quantifying effectiveness have involved the use of single proxies.

1.5 Private land conservation areas in southern Africa

In southern Africa, conservation on private lands has mainly been documented in Namibia, South Africa, and Zimbabwe, mostly because these countries have significant amounts of land under private ownership, and have policies and legal framework that enable PLCAs to thrive (Bond et al. 2004). Around the 1960s legislation emerged that promoted the decentralisation of wildlife ownership and use which was previously monopolised and controlled by respective states (Bond et al. 2004, Child et al. 2012). This allowed landowners to partake in commercial wildlife farming which was a rapidly expanding, lucrative, market driven industry both regionally and globally, despite little support from the respective states (Bond et al. 2004, Child et al. 2012). Individuals and NGOs thus started purchasing pieces of land and managing them as conservancies and nature reserves across southern Africa, with some converting their cattle ranches to wildlife ranches (Carter et al. 2008). This also coincided with the fact that in the 1960s many conservationists argued that climatic conditions in southern Africa were better suited for wildlife than livestock production, and advocated for exclusive wildlife ranching or

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mixed wildlife and livestock farming within most southern African countries (IUCN 1963, Child et al. 2012).

In South Africa, PLCAs are well integrated into the existing national conservation strategies and form a significant part of the national conservation estate (Bond et al. 2004, Cadman et al. 2010, Cumming and Daniels 2014, Stolton et al. 2014, Bingham et al. 2017). This has been partly due to policies and legislation that allow the declaration of PAs on private and communal land (Magome and Murombedzi 2003, Wright et al. 2018, De Vos et al. 2019). The country thus has a long history of conservation on private lands (De Vos et al. 2019). With 79% of the country's total area privately owned, and its protected area expansion model emphasizing on private areas (DEA 2016c), the country offers a good case study to understand the effectiveness of PLCAs.

Currently, policies and programs behind the establishment of PLCAs in South Africa include the National Environmental Management Protected Areas Act (Act 57 of 2003), the National Protected Area Expansion Strategy, and the Biodiversity Stewardship Program, (Cadman et al. 2010, DEA 2016b, Wright et al. 2018). The aim of these strategies is to improve biodiversity conservation by encouraging private landowners within important biodiversity areas to commit to conservation through formal agreements with conservation authorities (Chapin III et al. 2010, Barendse et al. 2016, Wright et al. 2018). The above mentioned strategies however direct the establishment of mostly formally recognised PLCAs, while for informal ones, tourism and hunting might be major forces with the National Biodiversity Economy Strategy being the major driver (DEA 2016a).

South African PLCAs can be classified into four main categories namely, contractual national parks (contractual parks), private nature reserves (nature reserves), biodiversity agreements and voluntary conservation areas (DEA 2013). Accordingly, these PLCAs have differences in how they are established, their objectives, level of protection, duration, incentives, and management restrictions (Cadman et al. 2010, Cumming and Daniels 2014, Mitchell et al. 2018b). Contractual parks and nature reserves are formally protected through the Protected Areas Act. Biodiversity agreement areas receive moderate protection, through contractual laws between the landowner and the provincial conservation authority (Cadman et al. 2010, Wright et al. 2018). Informal PLCAs (conservancies and conservation areas) on the other hand are dedicated to conservation, but without legally binding contractual laws or agreements guiding their management (DEA 2013, 2016c). They thus rely on protection from their respective owners (DEA 2013). Table 2.2 (Chapter 2), shows further detailed descriptions of the different types of PLCAs considered in this study, while Figure 1.1 shows the general spatial representation of all PLCAs and PAs in South Africa i.e. both state and privately owned. The diversity in types of PLCAs also makes South Africa a good case study to not only investigate the contribution of PLCAs, but also to understand how differences in legal support

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and incentives influence the effectiveness of PLCAs. Among state-owned PAs, effectiveness was shown to vary with IUCN protection category with PAs with stricter protection being more effective than those with less strict protection (Nagendra 2008, Bowker et al. 2017). In this study the hypothesis that that formally protected PLCAs (i.e. with more legal support and strict land use restrictions) would be more effective than the voluntarily protected informal ones was thus tested.

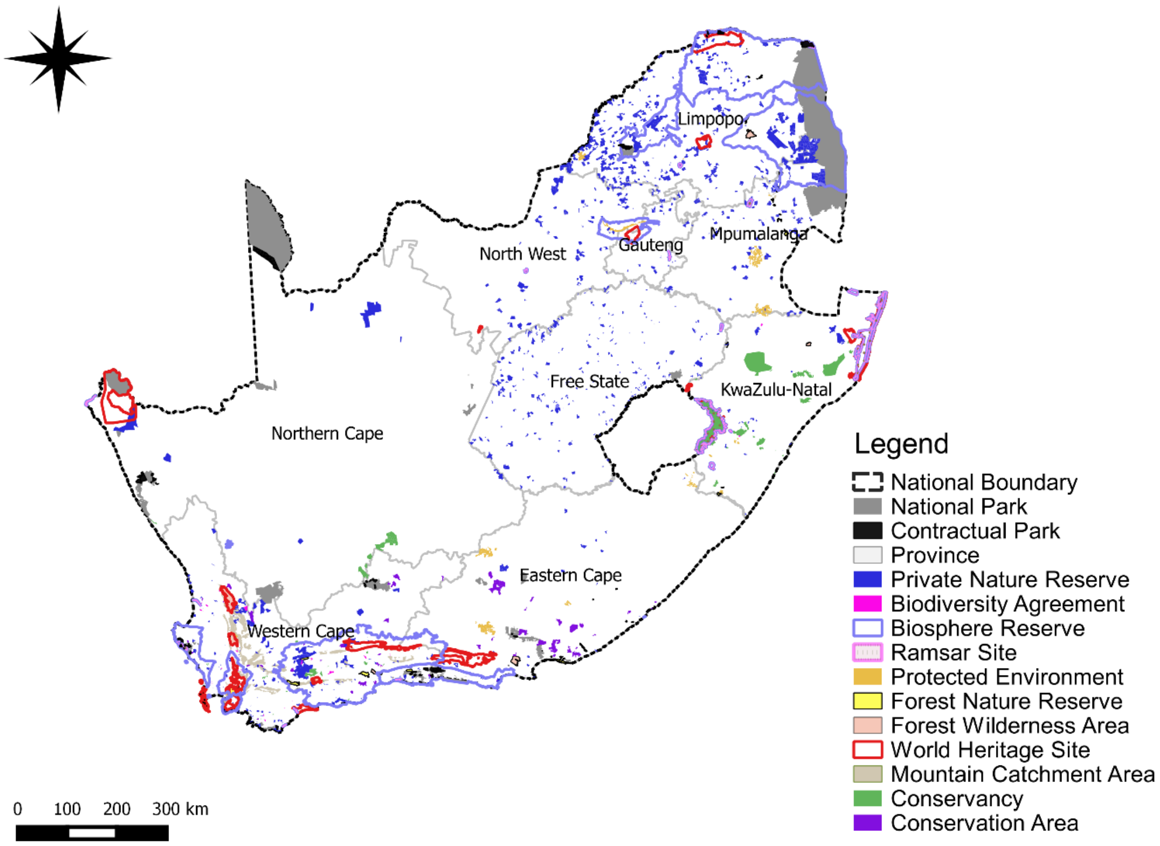

Figure 1.1. Map showing the different types of terrestrial private land conservation areas and state-owned protected areas in South Africa.

1.6 Problem statement

It has become apparent that state-owned PAs are unlikely to be sufficient to effectively protect biodiversity on their own. PLCAs are one option that might complement the existing PAs in achieving global biodiversity conservation targets. The increased recognition of the contribution of PLCAs to biodiversity conservation and literature documenting their importance, and their proliferation globally shows that they might be a conservation strategy worth considering. However, their effectiveness in actually conservation biodiversity remains unclear. Given the different motives and management systems under which PLCAs exist, it is not clear whether they can be relied upon as an effective and sustainable biodiversity conservation strategy. This study was thus motivated by the need to quantify the effectiveness of PLCAs and how different social-ecological factors influenced their effectiveness.

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1.7 Research objectives and specific questions

1) The first objective (addressed in Chapter 3) was to understand the effectiveness of South African PLCAs in preventing losses in NLC and BII between 1990 – 2013.
   a) Are there significant differences in NLC and BII losses between PLCAs and unprotected areas with similar environmental conditions?
   b) Are there significant differences in NLC and BII losses between PLCAs with different levels of protection (contractual parks, nature reserves, biodiversity agreements, conservancies and conservation areas)?
   c) Are there significant differences in NLC and BII losses across different South African biomes?
2) The second objective (addressed in Chapter 4) was to determine how different environmental, and social-economic factors influenced the effectiveness of PLCAs.
   a) What are the best models to explain variation in NLC and BII losses among PLCAs?
   b) What influence does accessibility (distance to town, distance to major road, elevation and slope) have on NLC and BII losses?
   c) What effect does size and age of PLCAs have on NLC and BII losses?
   d) What effect does rainfall have on NLC and BII losses?

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Chapter 2: General Methods

2.1. Study areas

The study focused on PLCAs in South Africa, distributed across the seven main biomes namely; Forest, Fynbos, Grassland, Nama-Karoo, Savanna, Succulent Karoo, and Thicket, with the highest number of study sites occurring in the Savanna biome (Figure 2.1). Biomes represent major differences in climatic conditions and vegetation characteristics (Mucina and Rutherford 2006, Rutherford et al. 2006). Table 2.1 below includes further details about the biomes in South Africa (Mucina and Rutherford 2006, Rutherford et al. 2006, Esler and Archer 2018) while Figure 2.1 shows their spatial distribution in relation to study sites.

Table 2.1. Description of South African biomes within which private land conservation areas considered in this study were located; AAR = Average Annual Rainfall, AAT = Average Annual Temperature, % Coverage = proportional coverage of each biome in relation to the national total area, and Study sites = total area in each biome covered by study sites (Mucina and Rutherford 2006, Rutherford et al. 2006, Esler and Archer 2018).

Biome	**Description**	**AAR (mm)**	**AAT (ºC)**	**% Coverage**	**Study sites (km^{2})**
Forest	Occurs typically as patches within other biomes. The vegetation is mainly adapted to cool temperate conditions, mainly shrubs, trees and climbers.	943	17	0.3	4876.1
Fynbos	Endemic to South Africa. Found within the Cape region, characterised by winter rainfall, with evergreen and fire prone shrub vegetation.	483	15.7	6.6	5737.5
Grassland	Covers the high central plateau of South Africa, mainly KwaZulu-Natal and the Eastern Cape provinces. Rainfall is received during the summer, and the vegetation is predominantly grasses.	661	14.7	27.9	7501.3
Nama Karoo	Covers the central plateau and the western region of South Africa, with summer rainfall. Vegetation is mainly shrubs, annual forbs and grasses.	208	16.3	19.5	3765.8
Savanna	Covers the majority of South Africa, characterised by summer rainfall. Typical about its mixture of trees and grasses.	495	18.7	32.5	19263.2
Succulent Karoo	Found in the Cape Region, with winter rainfall and occasional summer rainfall. Vegetation is mainly shrubs and seasonal geophytes.	168	16.8	6.5	2006.6
Thicket	Found in the South eastern parts of the country, characterised by dense vegetation, with rainfall mainly in spring and autumn.	431	17.2	2.2	409.3

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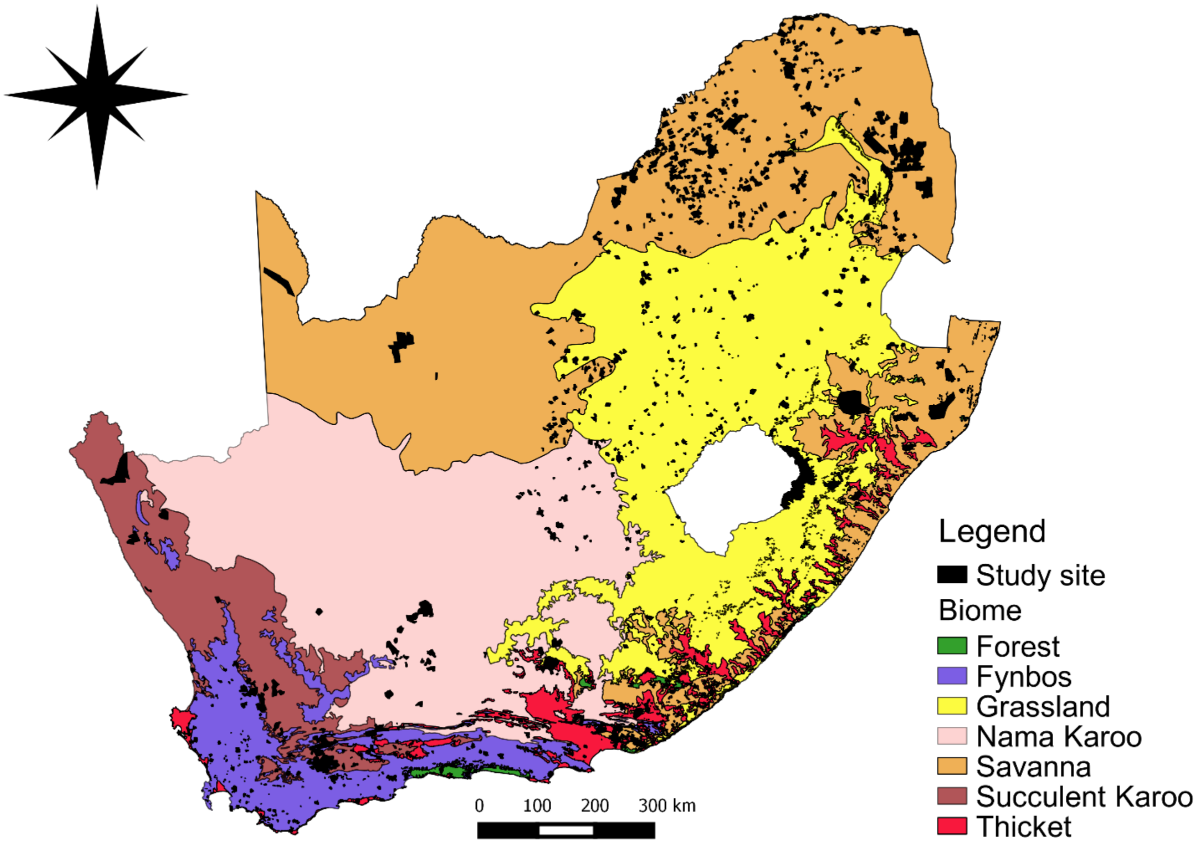

Figure 2.1. Map showing the spatial distribution of South African biomes and their relation to private land conservation study areas.

Study sites satisfied the definition of a PLCA as a piece of land privately owned and managed for biodiversity conservation with or without formal government or legal recognition (Pasquini et al. 2010, Clements et al. 2016a). These study sites included contractual parks and nature reserves which are formally protected through the National Environmental Management Protected Areas Act (Act 57 of 2003), biodiversity agreements which are protected by contractual laws and conservancies and conservation areas which are informally protected. Study sites represented different levels of protection, longevity, landowner commitment to conservation, land use restrictions and incentives (Table 2.2, Figure 2.3) which suited the objectives of this study.

Contractual parks and nature reserves are characterised by strict land use restrictions that prohibit extractive activities such as mining, require management plans and are monitored through annual management audits (Cadman et al. 2010, DEA 2016b). Their permanence and perpetuity ranges from 30 to 99 years with tittle deeds and property laws guaranteeing long-term protection despite ownership changes (Stolton et al. 2014, Mitchell et al. 2018b). With biodiversity agreements, provincial conservation bodies enter an agreement with landowners in important biodiversity areas, to get them committed to biodiversity conservation, through contractual laws, with some land use restrictions. Informally protected PLCAs include

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conservation areas and conservancies. These do not have land use restrictions and can have multiple land uses such as crop and livestock farming, besides ecotourism and hunting (Cadman et al. 2010, DEA 2013). They are thus protected on a voluntary basis, receiving protection from their respective landowners without any legally binding laws. Table 2.2 provides further descriptions of the considered PLCA categories and the different ways under which they are established and or managed. These data were available mainly through the South African Protected Areas Database (SAPAD) and the South African Conservation Areas database (SACAD) (DEA 2013, 2016c) which are updated on quarterly basis. However, these databases mainly represent the formally recognised PLCAs (De Vos et al. 2019); for the informal ones the data are not exhaustive, and many remain undocumented (Rissman et al. 2017). Additional data on informal conservation areas were thus sourced through provincial conservation bodies (Mpumalanga Tourism and Parks Agency, Cape Nature, Ezemvelo Kwa Zulu Natal, Wildlife, Limpopo Department of Economic Development & Tourism), South African National Biodiversity Institute (SANBI), manual digitising from google maps and previous studies (Clements 2016, De Vos et al. 2019).

In total, 5,121 PLCA study properties were considered, which included 127 contractual parks, 4741 nature reserves, 82 biodiversity agreements, 98 conservancies, and 73 voluntary conservation areas. The Figure 2.2 below shows the different types of PLCAs used as study sites for this study. It is important to note that the majority of study sites were nature reserves which contributed 71% (31, 092.3 km^2) of the area within study sites, followed by conservancies which a covered 7, 624 km^2 (17%), conservation areas which covered 3, 256.9 (7.4%), contractual parks 1, 816.7 km^2 (4.1%), and lastly biodiversity agreements which contributed 224.7 km^2 (0.5%) of the study area. These properties were classified on a protection gradient as shown on Figure 2.3, based on available legal support and landowner commitment to conservation, to enable comparison of effectiveness across diverse types of PLCAs. Of interest was also how effectiveness could be influenced by age and size of the PLCAs. In that regard when dates of establishment (gazzettment) were recorded age could be determined, with the oldest PLCA being 92 years, while size ranged between 0.0002 km^2 and 2, 348.9 km^2.

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Table 2.2. Description of PLCAs considered in this study and the different legal tools and policies under which they are established and managed, ranked by associated degree of protection (Cadman et al. 2010, DEA 2013, 2016c).

Category	Description	Guiding framework	Incentives	Data source(s)
Contractual park	Established through joint agreements between South African National Parks (SANParks) and the private landowner or community. They are typically small portions of private land within a national park. They are also formed through the state buying land adjacent to a national park and then giving it to a community to manage as contractual park (Reid 2001, Magome and Murombedzi 2003). They are formally declared and protected through the Protected Areas Act, with duration of 30-99 years.	National Environmental Management Protected Areas Act (Act 57 of 2003).	Exclusion from municipality property rates. Reduced tax payments through the Biodiversity tax rules. Advanced technical support with habitat and species management. Better recognition and marketing opportunities. .	South African Protected Areas Database https://egis.environment.gov.za
Nature reserve	Formally declared through section 23 of the Protected Areas Act, with strict restrictions on exploitative land uses such as mining with longevity ranging from 30 to 99 years.			
Biodiversity Agreement	Formed when the provincial conservation body lobbies to get private landowners within highly biodiverse areas or threatened habitats committed to conservation through contract agreements. Longevity can be between 5-10 years with possibility of extension. These areas are not considered part of South Africa's formal PA estate, but contracts include a management plan, which gives the areas a medium level of protection (DEA 2013, Wright et al. 2018). They can also be upgraded to the same protection level as nature reserves through more legally binding long-term contracts	Biodiversity Stewardship program, Biodiversity Act, Contractual laws. The South African National Biodiversity Strategy and Action Plan.	Assistance with management plans and technical support in managing species and habitats from the provincial authority. Assistance with management of fire and invasive species.	South African National Biodiversity Institute http://bgis.sanbi.org
Conservancy	Informal areas dedicated to conservation on a voluntary basis, with other land uses such as cattle ranching or crop farming being typical. They are typically composed of many properties in which adjacent landowners cut down internal fences and manage their properties as one big continuous landscape, with a shared biodiversity conservation vision (Kreuter et al. 2010, DEA 2013). They have no fixed length of years to be committed.	National Biodiversity economy strategy	Basic extension support with habitat and species management.	South African Conservation Areas Database https://egis.environment.gov.za/Clements 2016)
Conservation area	Dedicated to conservation without any law influencing their management, with protection only coming from landowners. They are characterised by multiple land uses that include ecotourism, hunting and livestock and crop farming, and do not have specific number of years to be committed.			

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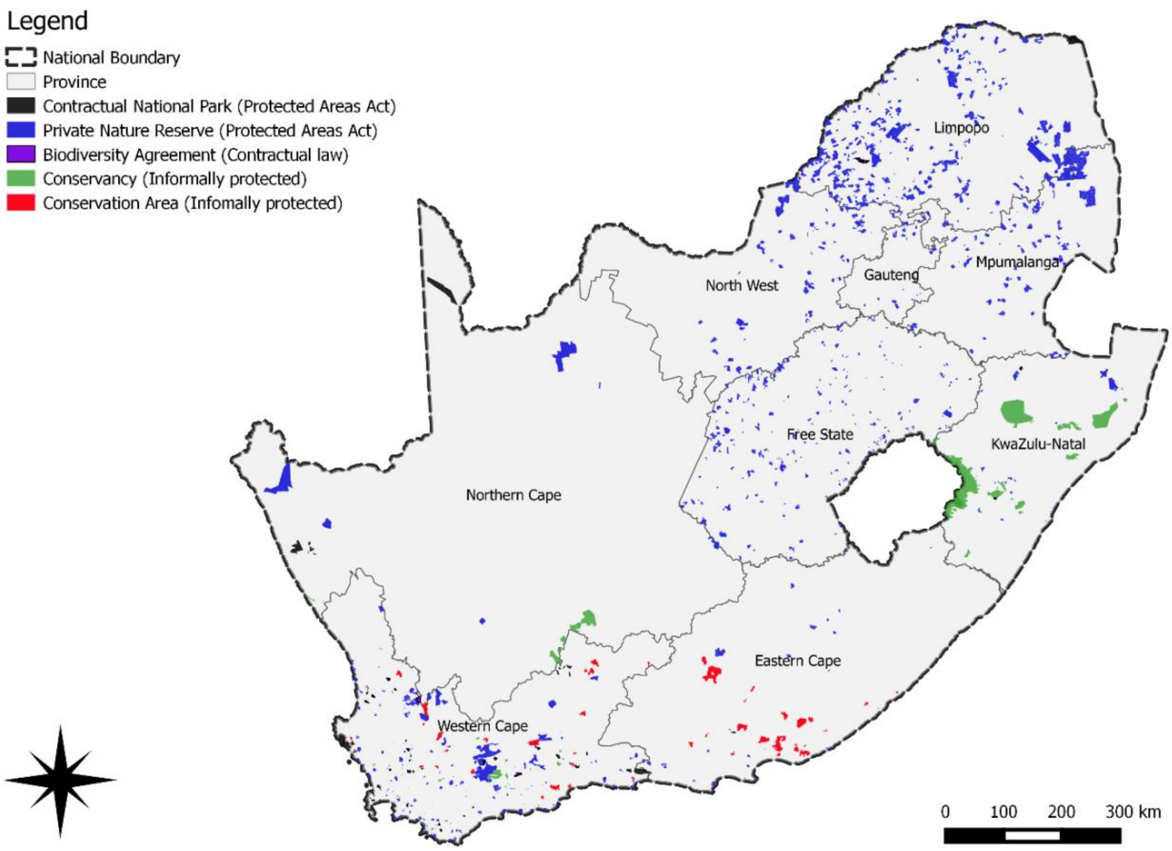

Figure 2.2. Private land conservation areas (PLCAs) assessed in this study, classified into the different categories under which they are managed and protected.

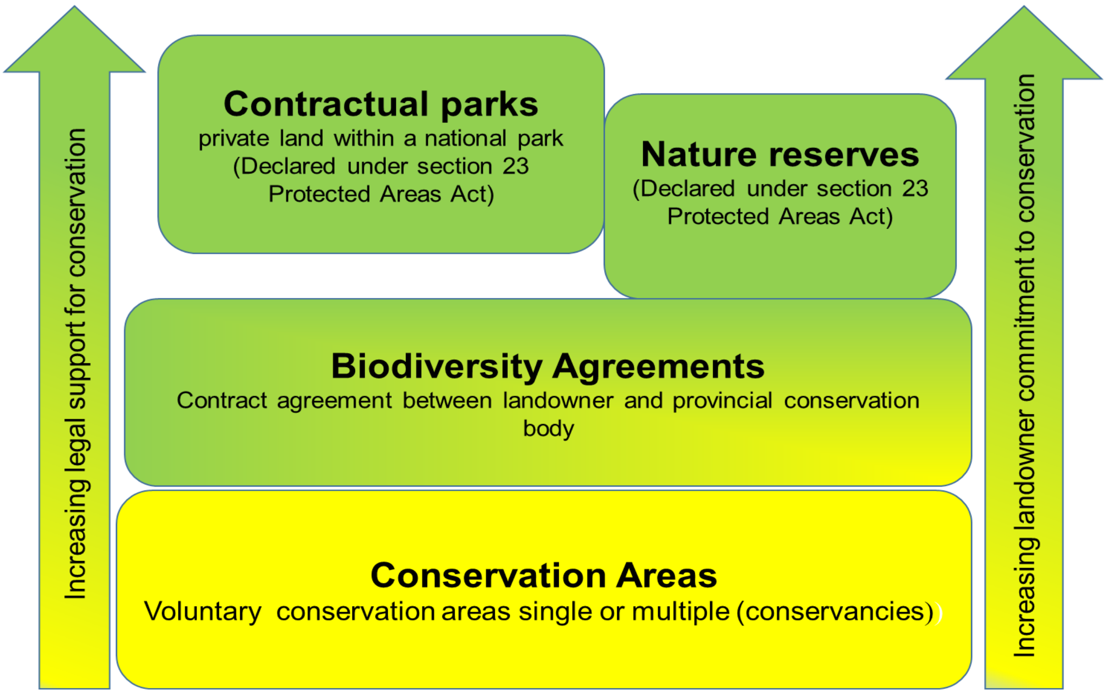

Figure 2.3. The different categories under which private land conservation areas fall, differentiated by differences in levels of protection and landowner commitment to conservation (Cadman et al. 2010, DEA 2013, Wright et al. 2018).

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2.2. Land cover data

South African national land cover maps for 2013 and 1990 courtesy of the DEA were used to assess changes in natural land cover (NLC) among PLCAs in South Africa. See (DEA 2015a, b), for detailed procedures of how the data sets were created.

The 2013 land cover map has a total of 35 land classes (Table 2.3) with 30 x 30 m resolution. Landsat 8 satellite images from April 2013 to March 2014, aided by cloud free Landsat 5 imagery accessed from United States Geological Survey data archives, were used to create the dataset (http://glovis.usgs.gov/). Semi-automated modelling procedures were then used to classify multi-seasonal images with each 30 m^2 pixel representing the dominant land cover class, based on spectral reflectance, shape, size and texture. Specific ground control points were then used to determine the accuracy of the map, and a mapping accuracy of 82.53% was established, with Kappa index of 80.87 and a mean land cover accuracy of 88.36% (DEA 2015a, b).

Using the same methodology, the 1990 national land cover map for South Africa was created based on historic remote sensed data (DEA 2015a, b). Landsat 4 and 5 imagery acquired between April 1989 and October 1991, with 30 x 30 m resolution were thus subjected to the same classification methodology as with the 2013 product, resulting in a map with 35 land cover classes. Because of the lack of historic reference points for 1990, the accuracy of the map could not be assessed, however the DEA reports the accuracy is analogous to the 2013 map since the same methodology was used (DEA 2015a).

Of interest in this study were losses in NLC across PLCAs and matched unprotected areas and how different factors influenced the losses. The 35 land classes across two national cover maps were thus reclassified to represent whether a pixel was natural or non-natural based on reclassification rules shown on Table 2.3, using the QGIS, grass plugin re-class function (QGIS Development Team 2015). Categories with human modification such as cultivated lands, settlements, roads and mines were thus reclassified as non-natural, while vegetated categories were classified as natural (Table 2.3). This ensured comparability of the products for investigation for NLC loss between 1990 and 2013, excluding finer changes within classes such as from woodland to bushland. Water bodies were not classified as either natural or not, because of the uncertainty in the form in which they could exist (i.e. no clear differentiation between man-made or natural water bodies) and also given their dynamic nature (i.e., changing in size based on rainfall).

Limitations of both the 2013 and 1990 national land cover products in this study are acknowledged particularly to do with resolution and accuracy. Both maps fail to distinguish self-seeded bush encroached areas and areas covered by invasive species (DEA 2015a, b). Consequently, despite such classes being forms of land degradation, the map includes them

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within the bushland and thicket categories i.e. natural in this study (DEA 2015b). Because of such limitations, there are no attempts to make fine-scale assessments of land cover change but rather the focus is on understanding how performing a nation-wide assessment at the scale of 'natural' verses 'non-natural' land cover can help quantify the effectiveness of PLCAs. Such coarse-scale maps have been used to perform similar assessments on state-owned PAs in South Africa and elsewhere (Pfeifer et al. 2012, Ament and Cumming 2016, Bowker et al. 2017). In this case these maps provide justifiable, nation-wide datasets to investigate the contribution of PLCAs, mainly because they were produced using a similar methodology, with the same land cover classes making them directly comparable. The maps also have a reasonable temporal resolution for enquiries about changes and to how different social, political economic and environmental factors may influence NLC.

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Table 2.3. Descriptions of the 35 land cover/ use types in the South African national land cover data for 1990 and 2013 and how they were reclassified (1; natural and 0; non-natural) to investigate losses in natural land cover (NLC) and biodiversity intactness index (BII) in this study, see DEA, 2015b for detailed descriptions of the land cover types and Scholes and Biggs, 2005 for description BII classes.

Class	Name	Description	NLC	BII class
1	Indigenous forest	Natural indigenous forest areas dominated by tall trees of > 5m and > 75% canopy cover.	1	Moderate
2	Thicket and dense bush	Natural tree or bush dominated areas with trees between 2-5 meters and canopy cover between 60-75 %. Can include bush encroached areas.	1	Moderate
3	Woodland/ open bush	Dominated by natural trees, with heights between 2-5m, and canopy densities ranging between 40-75 %, which will be in association with other sparser areas with 15-20 % canopy cover. Can included transitional wooded grasslands and bush encroached areas.	1	Moderate
4	Low shrub land	Dominated by natural shrubs ≤ 2m in height. Associated with bare ground, fire scars, and low woody shrubs.	1	Moderate
5	Forest plantation	Planted forest plantations dominated by timber trees at different growth stages, include also clear-cut stands, smaller woodlots and windbreaks	0	Plantation
6	Cultivated annuals	Cultivated areas characterised by rain-fed annual crops for commercial markets.	0	Cultivated
7	Cultivated irrigated	Cultivated areas with irrigation for commercial production of annual crops	0	Cultivated
8	Cultivated orchards	Permanent, cultivated areas with rain-fed and irrigated commercial production of orchard crops (citrus, tea, coffee, grapes pineapples among others)	0	Cultivated
9	Cultivated vines	Permanent rain-fed and irrigated commercial grape farms	0	Cultivated
10	Cultivated subsistence	Cultivated areas for local market or home production of annual rain-fed crops.	0	Cultivated
11	Settlements	Built up areas including formal and informal structures	0	Urban
12	Wetlands	Vegetated areas in association with permanent surface water	1	Excluded
13	Grassland	Natural areas dominated by grass, with trees having < 20 % canopy.	1	Moderate
14	Fynbos forest	Forest areas (class 1) within the Fynbos biome	1	Moderate
15	Fynbos thicket	Thicket areas (class 2) in the Fynbos biome	1	Moderate
16	Fynbos open bush	Woodland or open bush (class 3) in the Fynbos biome	1	Moderate
17	Fynbos low shrub	Low shrub land (class 4) in the Fynbos biome	1	Moderate
18	Fynbos grassland	Grassland (class 13), in Fynbos	1	Moderate
19	Fynbos bare ground	Bare ground within Fynbos	1	Moderate
20	Nama karoo forest	Forest areas (class 1) within Nama Karoo biome	1	Moderate
21	Nama karoo thicket	Thicket areas (class 2) within Nama Karoo biome	1	Moderate
22	Nama Karoo open bush	Woodland or open bush (class 3) within Nama Karoo biome	1	Moderate
23	Nama Karoo low shrub land	Low shrub land (class 4) within Nama Karoo biome	1	Moderate
24	Nama Karoo grassland	Grassland (class 13), in Nama Karoo	1	Moderate
25	Nama Karoo bare ground	Bare ground within Nama Karoo	1	Moderate
26	Succulent Karoo forest	Forest areas (class 1) within Succulent Karoo biome	1	Moderate
27	Succulent Karoo thicket	Thicket areas (class 2) within Succulent Karoo biome	1	Moderate
28	Succulent Karoo open bush	Woodland or open bush (class 3) within Succulent Karoo	1	Moderate
29	Succulent Karoo low shrub	Low shrub land (class 4) within Succulent Karoo biome	1	Moderate
30	Succulent Karoo grassland	Grassland (class 13), in Succulent Karoo	1	Moderate
31	Succulent Karoo bare ground	Bare ground within Nama Karoo	1	Moderate
32	Mines	Mining activities current and abandoned (extraction pits, quarries, open pits)	0	Urban
33	Water	All natural and man-made surface water sources	Null	Excluded
34	Bare ground non vegetated	Bare non -vegetated areas as a result of natural or human activities. Includes roads, fire scars, dry river beds, eroded areas	0	Degraded
35	Degraded man induced	Sparsely vegetated areas as a consequence of human activities.	0	Degraded

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2.3 Biodiversity intactness data

The biodiversity intactness index (BII) developed by Scholes and Biggs (2005), was used as the second proxy to assess the effectiveness of PLCAs and how different factors influenced effectiveness. The index is a measure of the average proportion of different species and functional groups within the major taxa (plants, mammals, birds, reptiles, and amphibians) that can persist in an area given the land use (moderately used, degraded, cultivated, plantation or urban) and the biome in which they occur (Fynbos, Forest, Savanna, Grassland, Thicket, Nama Karoo and Succulent Karoo). In this study, pixel based BII estimates for 1990 and 2013 were calculated using the formula

$$BII = (\sum_i \sum_j (R_{ij}\, I_{ijk}) / \sum_i \sum_j R_{ij}) \qquad \text{(Eq. 1)}$$

where R_{ij} is the species richness of taxon *i* in biome *j* and I_{ijk} is the population-level impact, i.e. the proportion of a population of taxon *i* that can persist under land use activity *k* in biome *j,* relative to an undisturbed pristine population in a similar ecosystem/biome.

Scholes and Biggs (2005) estimated population impacts I_{ijk} through a structured expert interview process. At least three specialists working on each taxon were interviewed, rating how a land use would affect the abundance of their study species in different biomes, on a scale from 100% (no impact, in pristine protected environments) to 0% (no probability of a species existing under a land use or cover type) (Scholes and Biggs 2005). A matrix of values was thus generated representing the average proportion of the original group of taxa that can exist under each land use type relative to an undisturbed pristine population in a PA. These values were then validated using empirical data (Scholes and Biggs 2005).

The index further incorporates species richness (R) per broad taxon, per ecosystem, originally compiled for the eight taxonomic groups across the seven South African biomes (Le Roux 2002). It is therefore assumed that each species occurs throughout the extent of a particular ecosystem in which it was recorded, and that there were no changes in richness per ecosystem type since 2002.

To apply the index to the national land cover maps used in this study, the 35 land cover classes in the 1990 and 2013 maps were reclassified to represent five categories (moderately used, degraded, cultivated, urban and plantation) related to the ones originally used in Scholes and Biggs (2005) as shown on Table 2.3, excluding wetlands by virtue of there not being significantly large wetlands in South Africa (Scholes and Biggs 2005). Using national land cover maps, together with known population impacts, species richness, and biomes, pixel-based BII values for 1990 and 2013 were thus calculated (Equation 1; Table 2.4) for random points within and outside PLCAs. This gave a second proxy to quantify the contribution of

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PLCAs. The index is thus a multiplicative proxy that takes into account how different land uses influence biodiversity; detail missed when a proxy such as natural cover is used.

Like any other proxy, the index has inherent limitations. By requiring a range of data for its calculation errors in input data can easily be multiplicatively amplified. The accuracy of the index is thus dependent on accurate land use / cover maps which are seldom available. The index has also been criticized for inflating intactness in some cases through underestimating losses (Martin et al. 2019). Because the index depends on the proportion of major taxa that can remain in an area relative to an undisturbed pristine population (population impact), it is compromised by the rarity of such pristine unaltered systems, which the index does not account for (Martin et al. 2019, Newbold et al. 2019). Its failure to represent the same hotspots indicated by other metrics such as the biomass intactness has also been a cause of concern (Martin et al. 2019). There are thus concerns that its simplicity might compromise accuracy. However, the index is not intended to give high-resolution data or highlight specific species losses but to highlight local and global trends in biodiversity necessary for decision and policy making (Scholes and Biggs 2005, Biggs et al. 2006, Newbold et al. 2015, Newbold et al. 2016). The index thus remains one of the few available cost effective proxies for quantifying the state of biodiversity (Mace 2005). In this study, its use is not only complementary to the NLC analysis but adds an extra layer of information on how changes in land cover influences biodiversity. The use of BII together with NLC thus enables a better understanding of the effectiveness of PLCAs, in South Africa and its drivers.

Table 2.4. Weighted proportion of major taxa that can exist under the different land cover types and biomes, used to determine the contribution of private land conservation areas to biodiversity intactness between 1990 to 2013 (Scholes and Biggs 2005).

	Land use / cover type				
Biome	Moderate use	Degraded	Cultivated	Urban	Plantation
Forest	0.88	0.49	0.21	0.08	0.26
Fynbos	0.94	0.52	0.21	0.09	0.25
Grassland	0.94	0.49	0.27	0.13	0.27
Savanna	0.95	0.59	0.25	0.13	0.27
Thicket	0.95	0.58	0.26	0.13	0.27
Nama Karoo	0.91	0.55	0.30	0.15	N/A
Succulent Karoo	0.90	0.55	0.25	0.11	N/A

2.4. Explanatory variables

A total of nine variables, predicted to influence the effectiveness of PLCAs in preventing losses in NLC and BII were considered. Table 2.5 below shows their descriptions and sources. Three of these variables are associated with the different PLCA properties i.e. PLCA type (contractual park, nature reserve, biodiversity agreement, conservancy and conservation area), their age since gazzettment, and their total area (size). These were available as

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metadata courtesy of the SAPAD and SACAD databases, and from previous studies (DEA 2013, Clements 2016, DEA 2016c, De Vos et al. 2019).

Factors related to PLCA location namely; distance to town, distance to road, elevation, slope, biome and rainfall (Table 2.5) were also considered. Distance to closest town, distance to road, elevation and slope represented accessibility, with the general prediction being that highly accessible areas will be less effective by being more susceptible to human exploitation (Bowker et al. 2017). Rainfall was also considered because of its association with agriculture i.e. the higher the rainfall the more attractive an area is for cultivation.

Data for town locations were obtained as vector points from the Global Rural – Urban Mapping project which represented towns with population sizes equal to or greater than 50, 000 by the year 2000 (CIESIN 2017b), also see (CIESIN 2017a) for detailed information about how the data were compiled. A proximity to raster layer representing distances from the South African towns was then created using QGIS grass, r.grow.dist command with the resultant raster having 30 x 30 m resolution. Data on South African roads were obtained from the worldwide road layer in polygon format (FAO 2015). This represented major roads that existed since 1980. Because the objective was to determine how distance to the road influenced losses in NLC and BII, the data were likewise converted to raster format using the r.grow.dist command to represent the distance from road on a 30 x 30 m resolution. Data on elevation and slope were obtained from the Digital Elevation Model (DEM), Shuttle Radar Topography Mission (SRTM) at 1 x 1 km resolution (USGS 2004). Precipitation data for South Africa were obtained from worldclim-global climate data (http://www.worldclim.org/current). This dataset was compiled from multiple weather stations over the period from 1950 to 2000, with a 1 x 1 km resolution. A total of 12 rasters, each representing the total rainfall recorded per pixel per month (January – December) were thus obtained. These rasters were then added together using the QGIS raster calculator to obtain one raster file representing the total annual rainfall received per pixel over the years. For more details about the worldclim-global climate dataset see (Hijmans et al. 2005).

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Table 2.5. Variables considered to influence changes in natural land cover and biodiversity intactness among private land conservation areas.

Variable	Type	Description	Source
PLCA type	categorical	Different types of PLCAs (Contract parks and nature reserves, biodiversity agreements, conservancies, and conservation areas)	DEA, SAPAD and SACAD databases
size	continuous (km^2)	The size of the PLCA	DEA, SAPAD and SACAD databases
age	continuous (years)	Number of years since PLCA establishment	DEA, SAPAD and SACAD databases
biome	categorical	Type of biome within which each PLCA occurs (Figure 2.1)	DEA
distance to town	continuous (meters)	Distance to the nearest human settlement	CIESN (2017b)
distance to road	continuous (meters)	Distance to major road	Worldwide roads (FAO 2015)
elevation	continuous (meters)	Height of point above sea level	(USGS 2004)
slope	continuous (degrees)	Steepness of a point	(USGS 2004)
rainfall	continuous (mm)	Total rainfall received	(Hijmans et al. 2005)

2.4. Sampling

A total of 500 random points 100 m apart (to limit spatial autocorrelation) in each PLCA, or the maximum possible within small PLCAs were generated. This resulted in total of 1,123,098 random points inside PLCAs. Outside PLCAs, a total of 1,000,000 random control points were generated for comparisons. These control points were generated 15 km away from any PLCA or PA (Figure 1.1) to counter for spill over effects (Ament and Cumming 2016), and where also at least 100m apart. Covariates and metadata associated with each point, were then extracted using the point sampling tool in QGIS 2.8 (QGIS Development Team 2015). These included, the name of the PLCA, its protection category (contractual park, nature reserve, biodiversity agreement, conservancy or conservation area), size, age (for points inside PLCAs), land cover class in 1990 and 2013, NLC classification (1; natural or 0; non-natural) in 1990 and 2013, BII in 1990 and 2013, biome, annual precipitation, distance to town, distance to roads, elevation and slope. Table 2.5 shows how the different covariates considered were sourced, processed and used in this study.

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Chapter 3: Effectiveness of private land conservation areas in conserving natural land cover and biodiversity intactness

Abstract

Private land conservation areas (PLCAs) are increasingly looked to for meeting the deficit left by state-owned protected areas (PAs) in reaching global conservation targets. However, despite the increasing extent and recognition of PLCAs as an alternative conservation strategy, little research has been done to quantify their effectiveness; a critical consideration if they are to be counted towards international biodiversity conservation targets. The long history of PLCAs in South Africa provides an interesting case study to address this knowledge gap. In this study, the effectiveness of South African PLCAs in biodiversity conservation was quantified by comparing losses in natural land cover (NLC) within PLCAs to that of unprotected control points with similar environmental characteristics. Furthermore, consequences of NLC loss on biodiversity intactness were assessed, thus advancing standard approaches for quantifying effectiveness. Between 1990 and 2013, PLCAs lost significantly less NLC (3%) and biodiversity intactness (2%) than matched unprotected areas (6% and 4%, respectively), indicating their effectiveness in protecting NLC and biodiversity intactness. Of the NLC lost within and outside PLCAs, most was converted to cultivated land. Farms can support more species than other land uses (e.g. mines), a likely explanation for why loses in biodiversity intactness were less than losses in NLC. However, contrary to the predicted pattern, effectiveness did not increase with level of protection; informal PLCAs with no legal protection had better NLC and biodiversity intactness retention, with most losses recorded among PLCAs with moderate protection. This study provides the first national-scale evidence that PLCAs can be an effective conservation mechanism, which is highly relevant given current discussions around their inclusion towards biodiversity conservation targets. **Keywords:** biodiversity intactness index, biodiversity conservation, effectiveness, land cover loss, matching methods, privately protected area, South Africa.

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3.1. Introduction

The establishment of protected areas (PAs) which are mostly state-owned remains a dominant strategy for biodiversity conservation (Watson et al. 2014). However, there is growing evidence that in addition to failing to meet international area-based targets, many of these PAs perform poorly in terms of: (1) representing biodiversity due to their bias towards high elevation and unproductive areas, and (2) protecting biodiversity due to insufficient funding, management and governance problems (Joppa and Pfaff 2009, Venter et al. 2017, Maron et al. 2018). Consequently, there is growing interest in private land conservation areas (PLCAs), as an alternative conservation strategy (Stolton et al. 2014). PLCAs are pieces of land predominantly managed for biodiversity conservation, protected with or without formal government recognition, and owned or otherwise secured by individuals, communities, corporations, NGOs, universities or religious groups (Pasquini et al. 2010, Stolton et al. 2014, Clements 2016). There has been increased recognition of the role of PLCAs in complementing state-owned PAs in biodiversity representation and increasing landscape connectivity (Langholz and Lassoie 2001, Gallo et al. 2009), as well as their contribution to national and local economies through hunting and ecotourism as well as provision of ecosystem services (Biggs et al. 2012, Stolton et al. 2014, Mitchell et al. 2018b). However, comprehensive data on countries' PLCA estates is generally lacking (Bingham et al. 2017, Rissman et al. 2017), making it difficult to quantify their contribution in conserving biodiversity (Langholz and Krug 2004, Carter et al. 2008, Bingham et al. 2017). Consequently, despite the long history of conservation on private lands, little is known about their long-term effectiveness as a conservation strategy (Stolton et al. 2014).

Concerns regarding the effectiveness of PLCAs have mainly focused on their small size which makes them susceptible to edge effects (Langholz and Lassoie 2001, Allen et al. 2016). In addition, diverse motives behind ownership and management of PLCAs (Clements et al. 2016a, Selinske et al. 2019) may influence their effectiveness. For example, profit-oriented management systems, influenced by expectations of hunters and tourists, may not always align with conservation needs (Langholz and Lassoie 2001, Cousins et al. 2008, Clements et al. 2016b). Consequently, high densities of charismatic megafauna and the introduction of extralimital species have been documented among PLCAs in southern Africa, which comes at the expense of vegetation and overall species diversity (Cumming et al. 1997, Maciejewski and Kerley 2014, Clements et al. 2016b). These uncertainties regarding PLCA effectiveness are a major reason for their exclusion in most national conservation planning and reporting towards international targets (Stolton et al. 2014).

A conservation strategy can be regarded a success or failure through objective measurement of effectiveness (Salafsky et al. 2002). Here effectiveness represents how well

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a strategy (e.g. PLCAs) achieves biodiversity conservation objectives (Hockings et al. 2006, Stoll-Kleemann 2010). There is therefore a need to understand what conservation strategies work, how, and why (Pullin and Knight 2001). However because of methodological constraints, approaches to quantify effectiveness have often involved GIS and remote sensing approaches (Nagendra 2001, Gillespie et al. 2008, Nelson and Chomitz 2009), with proxies such as deforestation (Pfeifer et al. 2012, Bowker et al. 2017), forest fires (Nelson and Chomitz 2011), NLC loss (Ament and Cumming 2016), and habitat change and fragmentation (Liu et al. 2001) being common. By comparing protected with non-protected areas, the contribution of a strategy as well as predictions of possible future trajectories of change can be established (Naughton-Treves et al. 2005, Brooks et al. 2009, Beresford et al. 2013). Changes in NLC directly translate to changes in biodiversity and provisioning of ecosystem services, thus making it an important proxy for understanding the effectiveness of a conservation strategy (Vitousek et al. 1997, Nagendra 2001, Fischer and Lindenmayer 2007). Detail is however lost by classifying remotely sensed data into a binary variable (e.g., natural or non-natural), given that different land cover types impact biodiversity differently (Scholes and Biggs 2005). For example, although farms and mines might both be non-natural, a farm will support more species than a mine. There is therefore a need to have a complementary proxy which consider the magnitude of effect of different land uses in quantifying effectiveness, especially given that remote sensed data are often at course scales with certain levels of mapping accuracy.

The biodiversity intactness index (BII) has been proposed as one measure for assessing the impact of different land uses on a broad range of taxa (Scholes and Biggs 2005)) The index represents the proportion of major taxa that can persist in an area given different land use / cover scenarios relative to an undisturbed population (Scholes and Biggs 2005). The index can thus serve as an important proxy for assessing effectiveness, in the absence of detailed field data which often cannot be compared across scales and regions (Mace 2005). Accordingly, the index has been proposed as one of the best available methods for capturing the role of biodiversity in supporting earth system functioning under the planetary boundaries framework (Steffen et al. 2015).

In this study the effectiveness of PLCAs across South Africa was assessed using losses of NLC and BII between 1990 and 2013. The use of these two proxies in this study not only allow better understanding of effectiveness, but also help in understanding how losses in NLC relate to losses in BII, thus advancing standard approaches for quantifying effectiveness, which has not been common among previous studies (Pfeifer et al. 2012, Bowker et al. 2017). In South Africa, PLCAs are characterised by diverse levels of protection, land use restrictions and incentives (Cadman et al. 2010, Mitchell et al. 2018b). They include contractual parks and nature reserves, legally protected through Protected Areas Act (Act 57 of 2003) (DEA 2016c), biodiversity stewardship agreements protected through contractual laws and conservation

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areas which are informally protected (Cadman et al. 2010). The diversity in which PLCAs exist and their long history in South Africa makes the country an interesting case study to understand the role, challenges, opportunities, and effectiveness of PLCAs (Gallo et al. 2009, Mitchell et al. 2018b). The main prediction here was that if PLCAs are an effective biodiversity conservation strategy i.e. if they offer significant protection, losses in NLC and BII will be significantly lower within their boundaries when compared to unprotected control areas with similar environmental variables. In addition to assessing their overall effectiveness, the hypothesis that effectiveness is dependent on legal support and landowner commitment to conservation was tested, in which legally protected PLCAs were predicted to be more effective than informally protected ones. Also of interest was understanding which land uses the points which lost NLC were transformed to, as well as how rates of NLC and BII loss compared across the different biomes (i.e. do some biomes experience greater losses in NLC and BII, given their differences in climatic and vegetation associations?).

3.2. Methods

3.2.1. Study areas

PLCAs across South Africa were assessed (see Chapter 2, Figure 2.2). These included contractual parks, nature reserves, biodiversity agreements, conservancies and conservation areas. Contractual parks and nature reserves are formally protected through the Protected Areas Act (Act 57 of 2003) and constituted the formally protected PLCAs for this study. Biodiversity agreements are moderately protected through contractual agreements between landowners and the provincial conservation authorities (Cadman et al. 2010). Conservancies and conservation areas constituted informally PLCAs for the study. Table 2.2 (Chapter 2) shows detailed descriptions of PLCAs considered in this study and the different legal tools and policies under which they are managed.

In total, 5,121 PLCA properties were considered, which included, 127 contractual parks, 4,741 nature reserves, 82 biodiversity agreements, 98 conservancies, and 73 voluntary conservation areas (Chapter 2, Figure 2.2).

3.2.2. Sampling

A total of 1,123,098 random points within PLCAs and 1,000,000 random control points outside PLCAs were generated for comparisons as described in Chapter 2. NLC and BII (Chapter 2), for 1990 and 2013 were then obtained for sample points together with data on other covariates (biome, rainfall, distance to town, distance to roads, elevation and slope), and for points inside PLCAs the name of the PLCA and its protection category. Table 2.5 (Chapter 2) shows how

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the different covariates considered were sourced, processed and used for this study. All spatial analyses were performed in QGIS 2.8 (QGIS Development Team 2015).

3.2.3. Data analysis

To positively ascertain the effect of protection offered by PLCAs by controlling for other factors that might influence losses in NLC and BII, sample points inside PLCA were paired with environmentally similar random unprotected control points outside. This was essential to have a good understanding of NLC when no protection is available. A nearest neighbour matching approach was thus used to match points inside PLCAs with control points outside based on similarities in distance to road, distance to town, rainfall, elevation and slope. These factors have been shown to influence losses of NLC on state-owned PAs, and failure to account for them can overestimate effectiveness (Andam et al. 2008, Ament and Cumming 2016, Bowker et al. 2017). Without protection, some areas will experience minimum losses in NLC or BII due to contextual factors, which are controlled for by the matching process resulting in a 1:1 ratio of PLCAs to unprotected points, with similar environmental variables, with which the genuine effect of protection can be established (Andam et al. 2008, Bowker et al. 2017).

A 0.5 standard deviation calliper was thus used as the maximum allowable difference between point pairs, which paired respective PLCA and non-protected points with replacement using the MatchIt package in R (Ho et al. 2011, Ament and Cumming 2016, R Core Team 2018).

Because the interest was how losses in NLC and BII compared between PLCAs and unprotected areas, only points which had NLC in the initial year (1990) were considered. Although an increase in NLC or BII is a good measure of effectiveness, it was appropriate to only focus on losses given the time period required for a pixel to regain NLC and biodiversity intactness and how such gains might be influenced by bush encroachment and invasive species (Seymour et al. 2010, Bowker et al. 2017).

The effect of protection (1 - PLCAs; 0 - non-protected) on NLC loss, a binary response (1 - NLC lost; 0 - NLC retained), and BII loss (%), was then determined by fitting generalized linear models (GLMs). These GLMs are a family of non-parametric models which consider nonlinear response variables, in this case a Binomial model with logit link for NLC loss and a Gaussian model with identity link for BII loss. To understand what points that lost NLC were transformed to, points that lost their initial cover were isolated, both within and outside PLCAs, so as to determine the proportional representation of the land uses they were transformed to by 2013.

To test the hypothesis that losses in NLC and BII would be less likely among PLCAs with more legal support for conservation (Figure 2.3), GLMs with NLC and BII loss as response variables were fitted with protection category as the explanatory variable. In case significance

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was recorded, Tukey post-hoc test with Bonferroni correction was used to determine how the specific categories differed. Similarly, to test whether losses in NLC and BII varied across biomes (Figure 2.1), both within and outside PLCAs, GLMs were fitted with NLC and BII loss as the response variables while biome was the explanatory variable, with a Tukey post hoc test, with Bonferroni adjustment being used to determine pairwise comparisons. All analysis were done using the lme4 package in R statistical software at a significance level of 0.05 (R Core Team 2018).

3.3. Results

To quantify PLCA effectiveness, a total of 937,682 points inside PLCAs (89.9% of points, which had NLC in the initial year) were successfully matched with unprotected control points. Of these sample points inside PLCAs that had NLC in 1990, 3.06% (SE 0.017%) had lost their NLC by 2013, thus retaining approximately 97% of their original NLC. In contrast, matched unprotected control points lost approximately double that amount (6.08%; SE 0.025%). A similar pattern was also observed with BII. Between 1990 and 2013 PLCAs lost 1.97% (SE 0.012%) BII, while unprotected control points lost 3.87% (SE 0.016%). Protection offered by PLCAs thus significantly reduced losses in NLC ($p < 0.005$, odds ratio, 0.72) and BII ($p < 0.005$, odds ratio 0.02).

In both PLCAs and matched unprotected areas, a large proportion of the points that lost NLC (66.4% and 65.9% respectively) were converted to some form of cultivated land (Figure 3.1). Among PLCAs, these cultivated areas (Table 2.3) comprised annual crops (38.5%), cultivated areas with irrigation (12.9%), plantation forest (8.7%), subsistence cultivation (3%), commercial orchards (2.4%), and commercial vines (0.9%). Among unprotected areas, settlements and degraded areas were important outcomes of transformation (11.3% and 15% respectively), along with annual crops (30.6%), subsistence cultivation (19.7%), and plantation forests (10.6 %), while irrigated areas (3.2%), orchards (1.3%), and vines (0.4%) were less frequent outcomes (Figure. 3.1).

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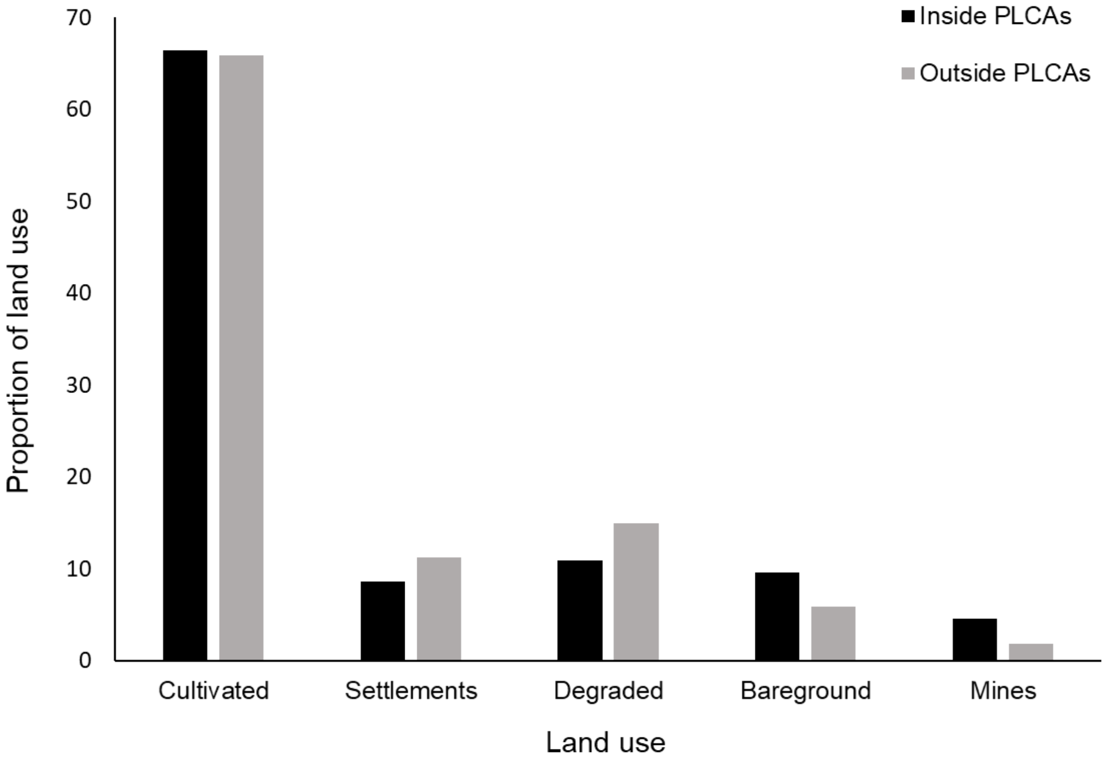

Figure 3.1. Proportional distribution of land uses to which points that lost natural land cover were transformed, inside private land conservation areas (PLCAs) and outside (non-protected areas) from 1990 to 2013.

There were significant differences in NLC loss among the different PLCA categories ($F_{(5, 1875363)} = 2002.5$, $p < 0.05$). All pair wise comparisons were significant ($p < 0.05$) except between contractual parks and conservation areas, and contractual parks and conservancies (Appendix 1). The greatest losses of NLC were recorded in PLCAs under biodiversity agreements, while contractual parks and conservation areas had the least losses (Figure 3.2). Nevertheless, all PLCA categories lost signficantly less NLC than matched unprotected control points (Figure 3.2).

Similarly, there were significant differences in BII loss across the different PLCA categories ($F_{(5, 1875363)} = 1859.9$, $p < 0.05$) with significant pair-wise comparisons except between conservation areas and contractual parks, biodiversity agreements and nature reserves and between biodiversity agreements and conservancies (Appendix 2). Most losses in BII were recorded among nature reserves and biodiversity agreements with conservation areas and contractual parks having the least losses (Figure 3.2). Likewise, none of the PLCA categories lost BII to the same extent as unprotected control areas (Figure 3.2).

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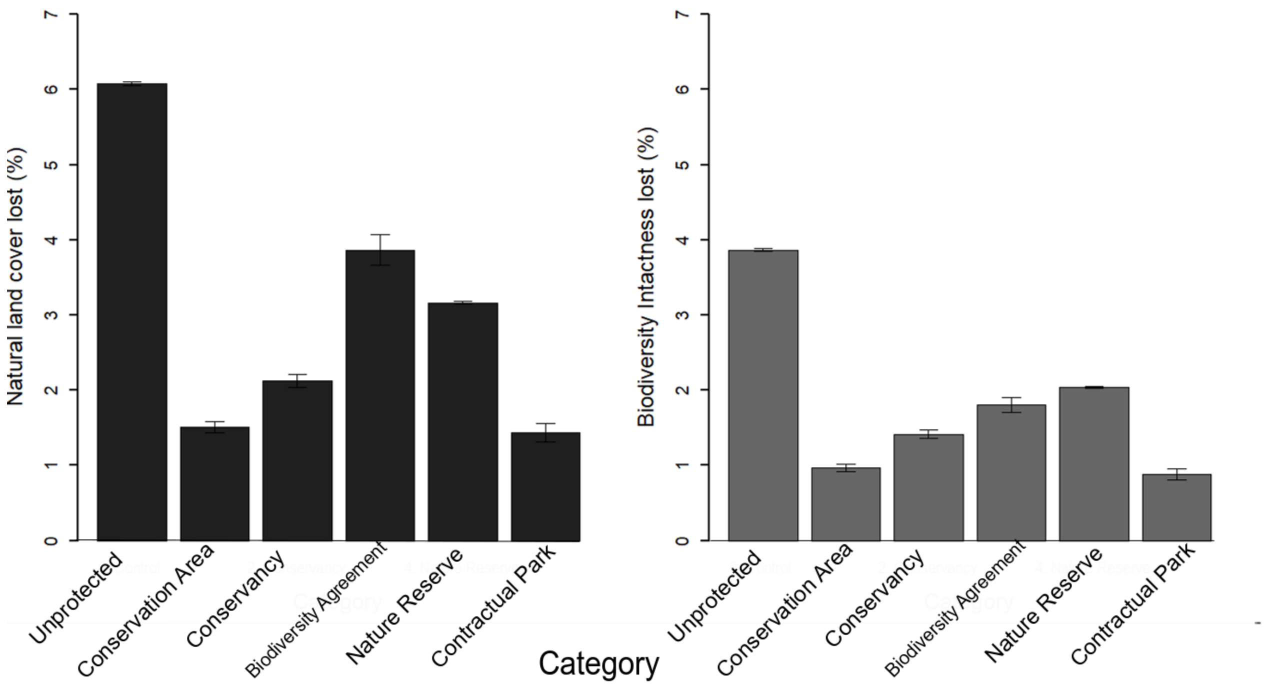

Figure 3.2. The percentage of points that lost natural land cover and biodiversity intactness and their respective standard errors of the proportion of points, across different categories of private land conservation areas (PLCAs) and unprotected control areas, arranged in order of increasing legal support and landowner commitment to conservation.

There was also significant influence of biome on NLC loss ($F_{(6,\ 1875357)} = 2430.1$, $p < 0.05$) and BII loss ($F_{(6,\ 1875363)} = 2703.4$, $p < 0.05$) across PLCAs and unprotected control points. The most NLC and BII losses were recorded within the grassland biome, while the Succulent and Nama Karoo experienced the least losses (Figure 3.3). Appendix 3 and 4 show the pair-wise comparisons from the Tukey's post hoc analysis between the different biome pairs.

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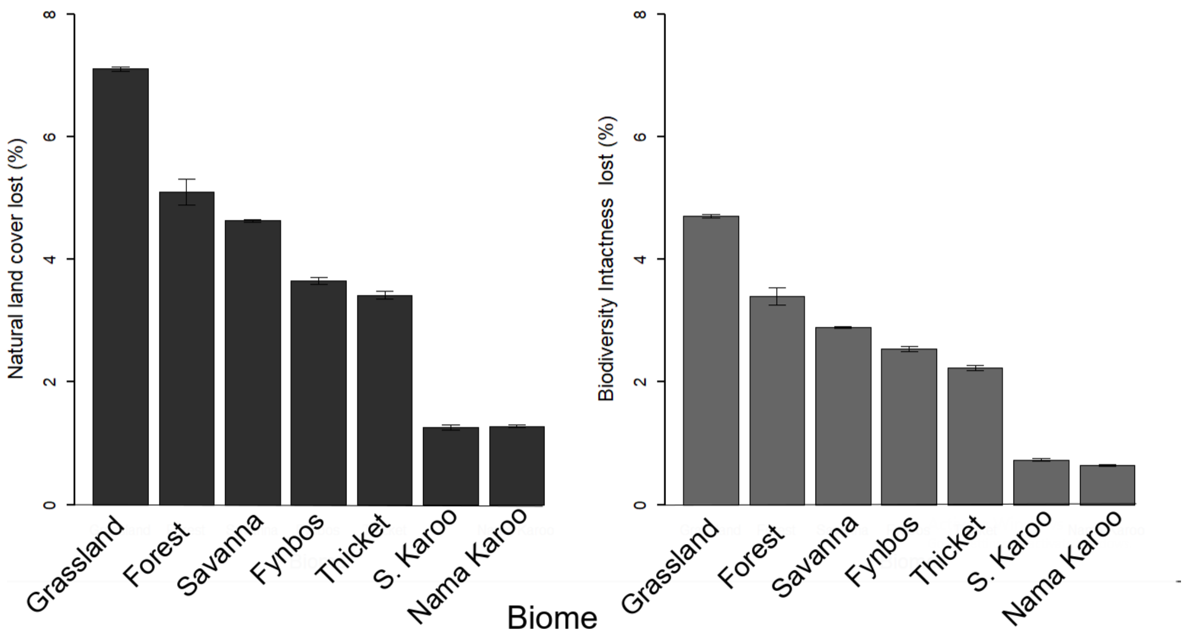

Figure 3.3. The proportion of points that lost natural land cover and biodiversity intactness and the respective standard errors of the proportion of points, across different South African biomes using data from within and outside PLCAs between 1990 to 2013.

3.4. Discussion

This study provides the first national-scale evidence that PLCAs are an effective mechanism for reducing losses of NLC and BII, with losses recorded among unprotected areas being almost double those recorded within PLCAs. PLCAs lost 3% NLC between 1990-2013 which is also comparable to NLC loss recorded across South African state-owned PAs, which was estimated at 2% over the period 2000-2009 (Ament and Cumming 2016). PLCAs thus offer a comparable level of protection of NLC to state-owned PAs. With the expanding number of PLCAs around the world and the increasing reliance on them as a conservation tool, evidence of their effectiveness in South Africa is reassuring.

Of the 3% NLC that was lost within PLCAs, a large proportion was converted for cultivation and settlements (Figure 3.1). This is likely due to multiple land uses being allowed within informal PLCAs (Cadman et al. 2010, Child et al. 2013). Most informal PLCAs do not receive incentives or reliable funding for conservation activities (Wright et al. 2018), and a variety of land use strategies (ecotourism, hunting, farming) are essential for funding conservation objectives and ensuring economic viability (Child et al. 2013). However, these diverse land uses may negatively impact their effectiveness in ensuring long-term biodiversity conservation. For example, although growing crops might provide more food and revenue options, it reduces biodiversity intactness (Table 2.4), reduces capacity of the system to regulate water quality and increases risk for invasive species (Reyers et al. 2009, Biggs et al. 2012). The notable increase in ecotourism in southern Africa and the world at large since the 1980s (Langholz and Krug 2004, Scholes and Biggs 2004, El-Haggar and Samaha 2019), and

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the need to keep PLCAs competitively attractive for tourist markets may also explain the conversion of NLC to settlements (lodges and picnic sites). Such changes in land cover/ use highlight the complexities of conservation in landscapes where financial viability is both a driver of wildlife-based land-uses as well as a potential constraint on their effectiveness in conserving biodiversity (Child et al. 2013, Clements and Cumming 2017). Nevertheless, results from this study indicate the importance of PLCAs in reducing NLC loss and maintaining biodiversity when compared to areas without any form of protection, which is an important finding considering how PLCAs have been criticised for their small size and profit-oriented management principles (Langholz and Lassoie 2001, Clements et al. 2016a).

This study recognizes that the consequences of NLC loss for biodiversity loss depend on both the biome (the greatest BII losses occurred in Grassland, Figure 3.3) and what the land transforms to (greatest BII losses in settlements and mines; Chapter 2 Table 2.4). The finding that 3% NLC loss corresponded to 2% BII loss within PLCAs can thus be explained by cultivation being the land use that predominantly replaced NLC on PLCAs, since cultivation retains greater biodiversity intactness than NLC transformed to, for example, urban or plantation land uses (Scholes and Biggs 2005). The study thus provides important insight into the consequences of NLC loss on biodiversity that are typically overlooked in land cover change studies, which assume all NLC loss is equal. Consequently, this study provides a significant advancement to the common approaches for assessing PA effectiveness which have often been based on a sole proxy such NLC loss.

Among state-owned PAs, effectiveness has been shown to vary with IUCN protection category, with strictly protected PAs being more effective (Nagendra 2008, Bowker et al. 2017). However, in this study the effectiveness of PLCAs did not increase with level of protection (Figure 3.2). Notably, informally protected conservation areas and conservancies lost significantly less NLC and BII than some formally protected PLCAs (Figure 3.2, Appendix 1, Appendix 2), with the highest losses being recorded within biodiversity agreements which receive moderate protection (Chapter 2, Table 2.2, Figure 2.3). This may be because of the multiple land use allowed in such areas as well as due to land use prior to be designated biodiversity agreements (Cadman et al. 2010). With the Biodiversity Stewardship Program under which biodiversity agreements are established having started in the early 2000s (Wright et al. 2018) and this study covering the period of 1990 to 2013, it is likely that some areas were proclaimed biodiversity agreements after NLC and BII had already been lost. These biodiversity agreements are established through agreements between a provincial conservation authority and a landowner within biological diverse landscapes (Wright et al. 2018). They are set up in an incentive based system in which the landowner receives help in management advice as well help in managing fires and invasive species (Chapter 2, Table 2.2) from the provincial conservation body in exchange for the landowner's commitment to

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managing the area at natural or semi-natural conditions (Cadman et al. 2010, Wright et al. 2018). Consequently, when the provincial conservation authority is incapacitated in terms of skilled personnel and funds to provide extension services, biodiversity conservation within biodiversity agreement areas can be compromised (Wright et al. 2018). This might explain why biodiversity agreements had the biggest losses in NLC and BII, compared with informal conservation areas and conservancies whose effectiveness is not largely dependent on extension services but rather on landowner commitment and motivation for conservation. In other words, a landowner might see the value of protecting NLC and biodiversity for ecotourism or hunting opportunities and hence be more likely to invest in protection than someone whose motivation to conserve is based on incentives. Nevertheless, follow up future studies which are site specific are essential to properly understand future performance of biodiversity agreements as well the other PLCAs. This study however demonstrates the importance of both formal and informal PLCAs in NLC and BII retention. In India, informal PLCAs were also shown to be equally important as the formal ones especially in cases when the formal ones are surrounded by communities and cultivated areas (Bhagwat et al. 2005).

Finally, differences across PLCA categories could also be due to proximity to state-owned national parks, which can be associated with both negative and positive spill over effects (Andam et al. 2008, Ament and Cumming 2016). Negative spill over involves cases when adjacency to a national park encourages activities leading to loss of NLC such as creation of more tourist accommodation and markets for wood curios around PLCAs close to state-owned PAs (Ament and Cumming 2016). Alternatively, positive spill overs occur when proximity to national park motivates PLCA landowners to maintain NLC and biodiversity in their properties, thus boosting their own chances of attracting tourists (Andam et al. 2008, Ament and Cumming 2016). This might thus explain why contractual parks, which are situated within or adjacent to national parks (Chapter 1, Figure 1.1), had lower NLC and BII losses than nature reserves (Figure 3.2) despite both being formally protected through the protected areas act (Chapter 2, Table 2.2).

In this study most NLC and BII losses were recorded within the grassland biome. The biome has been shown to support most human populations, with high levels of utilisation for agricultural and commercial rangeland production and hence highly threatened both regionally and globally (Carbutt et al. 2011, Egoh et al. 2011, Valkó et al. 2016). In South Africa previous studies have shown the grassland biome to have the highest number of threatened ecosystems, with cultivation, overgrazing, urbanization, and mining being major threats, while Nama and Succulent Karoo were the least threatened (Reyers et al. 2005, Esler and Archer 2018), which was also portrayed in this study (Figure 3.3).

While the methods applied in this study have been commonly used to understand broad-scale trends in NLC among PAs (Ament and Cumming 2016, Bowker et al. 2017),

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inherent limitations of the data used are acknowledged, notably the resolution of land cover products and their inability to distinguish bush encroachment and invasive species from natural habitat (DEA 2015b). These forms of land degradation reduce overall species diversity and ecosystem functioning (Pejchar and Mooney 2009, Goodenough 2010). Consequently, products that do not use fine-scale resolution, hyperspectral remote sensing fail to account for such detail, which is a limitation when drawing conclusions (He et al. 2011, Olsson et al. 2011).

Nevertheless, the study demonstrates that PLCAs, whilst not a panacea for achieving conservation goals, are an effective conservation mechanism. Conservationists, policy makers and the general public should thus recognize the importance of PLCAs in biodiversity conservation, and advocate for better policies and best management practices that can improve their effectiveness and incorporate them in national, regional and global conservation action plans. Future studies should however ensure that complete data about where PLCAs are located, when they were established, what they are managed for and by who are documented and publicly available to ensure easy evaluation and accountability of their contribution to biodiversity conservation. More can also be done in terms of using high resolution data, to understand site specific factors that can improve or compromise the effectiveness of PLCAs in biodiversity conservation as well as how social, ecological and political factors influence their effectiveness.

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Chapter 4: Factors influencing the effectiveness of private land conservation areas in maintaining natural land cover and conserving biodiversity

Abstract

Biodiversity loss is a major global issue, and it is apparent that state-owned protected areas (PAs) alone cannot achieve global biodiversity targets. Private land conservation areas (PLCAs) are one complementary strategy that increases the total area available for biodiversity, improves connectivity and representation of diverse habitats. However, despite their recognition, little has been done to quantify the effectiveness of PLCAs, or the factors that influence their ability to effectively conserve biodiversity. This study addresses this gap by quantifying how different factors influenced the effectiveness of PLCAs in South Africa. The ability of PLCAs to retain natural land cover (NLC) and the biodiversity intactness Index (BII) were used as measures of effectiveness, with accessibility (distance to town, distance to road, elevation, slope), rainfall, PLCA age and size being considered as drivers. The main objective was to determine the best respective models for explaining variations in losses in NLC and BII, as well as establish how the different factors influenced such variation (magnitude and direction of effect). The main prediction was that highly accessible areas would have low NLC and BII retention due to human pressure and models with accessibility variables would be the best to explain losses in NLC and BII. The best model to explain NLC loss among PLCAs included distance to town, distance to road, elevation, slope and rainfall, while the best model for explaining BII loss included distance to town, slope, rainfall and age as explanatory variables. Higher probabilities of NLC and BII retention were found among areas farther away from towns, close to roads, with high elevation, steep slopes, high rainfall, and in bigger and younger PLCAs. These results demonstrate that the effectiveness of PLCAs is influenced by multiple factors, which should be considered in the establishment of future PLCAs, in designing policies for improving the capacity of PLCAs to conserve biodiversity, and in future research on PLCA effectiveness.

Keywords: biodiversity intactness index, effectiveness, privately protected area, South Africa.

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4.1 Introduction

Biodiversity loss is a major problem worldwide, and the establishment of protected areas (PAs), most of which are state-owned, has been a key intervention strategy (Maron et al. 2018). Poor funding, particularly in developing countries (Bruner et al. 2001), and biases in the location of PAs towards marginal, high elevation areas where they do not compete with other land uses such as agriculture (Rodrigues et al. 2004, Joppa and Pfaff 2009, Venter et al. 2017) mean that state-owned PAs alone can not achieve global biodiversity conservation targets (Jones et al. 2018, Maron et al. 2018). There has thus been increased focus on private land conservation areas (PLCAs) as a complementary conservation strategy (Stolton et al. 2014). PLCAs are privately owned pieces of land predominantly managed for biodiversity conservation, protected with or without formal government recognition (Stolton et al. 2014, Clements 2016).

PLCAs complement state-owned PAs through increasing the total area available for biodiversity (Stolton et al. 2014) and representation of some highly threatened ecosystems (Gallo et al. 2009).They also play an important role in local and national economies through ecotourism and hunting, as well as provide essential ecosystem services such as clean water and air (Biggs et al. 2012, Maciejewski and Kerley 2014, Cumming et al. 2015). However, despite the long history of PLCAs (De Vos et al. 2019) and increased acknowledgement of their role in biodiversity conservation and human well-being (Fitzsimons and Wescott 2008), little is known about their effectiveness in biodiversity conservation (Bingham et al. 2017) and how different socio-economic, political and ecological factors influence their effectiveness in achieving biodiversity conservation objectives (Selinske et al. 2019). There is therefore an urgent need to understand what makes PLCAs effective or ineffective in relation to different threats to establish appropriate measures to ensure their long-term contribution to biodiversity conservation (Salafsky et al. 2002, Hockings 2003).

Over the years views of biodiversity conservation have shifted from strictly focusing on the protection of rare and endangered species for future generations, to a more human-centred approach that focuses on sustainable use of biological resources (Dudley 2008, Cumming et al. 2015). Consequently, PLCAs should not be viewed as ecological islands (Janzen 1983, Palomo et al. 2014), but rather as complex, social-ecological systems (SESs) in which humans and biophysical components interact across multiple scales (Palomo et al. 2014). The effectiveness of PLCAs is thus greatly influenced by a number of social, economic, ecological, and political factors. It is thus important to understand how these factors influence the ability of PLCAs (as SESs) to retain their composition, structure and function (Biggs et al. 2015, Cumming 2016, Folke et al. 2016). Most studies have focused on state-owned PAs, and their effectiveness has been shown to be associated with factors such as accessibility, size,

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age, funding, law enforcement, and community commitment (Nagendra 2008, Pfeifer et al. 2012, Bowker et al. 2017). With accessibility it has been shown that highly accessible areas i.e. closer to settlements and roads and with low elevation and gentle slopes are relatively more susceptible to NLC loss through deforestation and by also being more attractive for infrastructure development, and agricultural transformation (Pfeifer et al. 2012, Bowker et al. 2017). Also political factors associated with land tenure (Langholz and Krug 2004) and conflicts with surrounding communities have been shown to have a negative influence on the effectiveness of state-owned PAs biodiversity (Naughton-Treves et al. 2005). Given that like PAs, PLCAs are also SESs, the above mentioned factors are likely to have comparable influences on the effectiveness of PLCAs, hence the need to quantify their effect.

One major criticism of PLCAs as a conservation strategy has been that they are typically small in size (i.e. <10, 000 ha), hence susceptible to edge effects (Langholz and Lassoie 2001). When such small areas are intensively managed for profit generation, for example to promote charismatic megafauna attractive for tourists, their effectiveness is compromised, with consequences such as vegetation loss, soil erosion, bush encroachment and occurrence of invasive species (Carpenter et al. 2001, Biggs et al. 2012, Cumming et al. 2015, Cumming and Allen 2017). Because SESs provide a bundle of products and services, choices made by managers, stakeholders, communities and various aspects of their ecological, economic and political context have an effect on services they can provide (Cumming et al. 2015). For example, growing crops in a conservation area will provide food, but may affect the ability of the system to cope with soil erosion and increase vulnerability to invasion (Reyers et al. 2009, Biggs et al. 2012).

Due to limited funding and methodological constraints, systematically measuring effectiveness and its drivers across PLCAs is problematic (Nelson and Chomitz 2011). Attempts to quantify effectiveness and its drivers among state-owned PAs have often involved cost- and time effective remote sensing based proxies such as NLC change (Pfeifer et al. 2012, Ament and Cumming 2016, Bowker et al. 2017). In this study two proxies namely NLC and the biodiversity intactness index (BII) (Scholes and Biggs 2005) were used to determine how environmental and social-ecological factors influence the effectiveness of PLCAs. Changes in land cover are directly linked to climate, biodiversity, ecosystem services, and habitat loss and fragmentation, making it an important parameter to measure (Vitousek et al. 1997, Fischer and Lindenmayer 2007).The BII complements this by representing the proportion of major taxa that can persist under different land use scenarios (Scholes and Biggs 2004). Using species richness and known impacts of land uses on species in different ecosystems, the BII provides another proxy for quantifying the influence of different factors on the effectiveness of PLCAs in conserving different species (Scholes and Biggs 2005).

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A total of seven variables which could influence the effectiveness of PLCAs were considered. These included accessibility of the area (distance to major town, distance to road, elevation, and slope), size, age, and rainfall. The main objective was to establish which factors could best explain variations in NLC and BII losses within PLCAs in South Africa, their relative influence, relationship and magnitude of effect. With accessibility, the prediction was that easily accessible areas will have higher chances of losing NLC and BII than inaccessible ones by virtue of being more exposed to humans (Bowker et al. 2017). Size was also considered, with the hypothesis being that small PLCAs will likely have higher losses of NLC and BII due to edge effects and the risks of overstocking large mammals (Langholz and Lassoie 2001). Among previous studies done in state-owned PAs, younger PAs were found to be more effective than older ones, with reasons such as community endorsement and better management regimes being suggested (Blackman et al. 2015, Bowker et al. 2017). Likewise, in this study younger PLCAs were predicted to be more effective. Rainfall was also included because it can influence effectiveness by virtue of high rainfall areas being more attractive for agriculture, leading to higher chances of losing NLC and BII. Understanding how such factors influence losses or retention of NLC and BII, can add to the growing literature about PLCAs, and shed light on how different environmental and social-ecological factors influence their effectiveness. This has important implications for the establishment of effective policies and management strategies to improve the effectiveness of PLCAs as a long-term conservation strategy.

4.2 Methods

4.2.1 Study area

A total of 5,121 properties satisfying the definition of a PLCA as a privately owned piece of land managed for biodiversity conservation, protected with or without formal government recognition (Stolton et al. 2014, Clements 2016), distributed across seven biomes were considered (Figure 2.1). PLCAs considered included 127 contractual parks, 4,741 nature reserves, 82 biodiversity agreements, 98 conservancies, and 73 voluntary conservation areas (Chapter 2, Figure 2.2).

Contractual parks and nature reserves are formally gazetted and protected through the Protected Areas Act (Act 57 of 2003). Biodiversity agreement areas are established through agreements between a landowner and a provincial conservation authority hence are moderately protected (Wright et al. 2018), while cconservancies and conservation areas are informally protected. The Table 2.2 shows detailed descriptions of the different types of PLCAs considered in this study.

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4.2.2 Sampling

Two proxies, NLC loss and BII loss, were used as response variables to quantify how different explanatory variables influenced the effectiveness of PLCAs. NLC was derived from the South African national land cover maps for 2013 and 1990 data (DEA 2015a, b) while the BII was calculated using algorithms originally developed by (Scholes and Biggs 2005) as described in Chapter 2. NLC and BII were thus obtained for a total of 1,123,098 random points inside PLCAs. Explanatory variables (distance to town, distance to road, elevation, slope, size, age and rainfall) associated with each point, were then extracted using the QGIS 2.8 point sampling tool (QGIS Development Team 2015).

4.2.3 Data analysis

The objective of the analysis was to assess how different explanatory variables influenced losses in NLC (a binomial variable; 1 - NLC lost, 0 - NLC retained) and BII (a continuous variable, %), between 1990 and 2013. Points which had NLC in the initial year (1990), which included 92.8% of the original points generated, were considered. Of the 1,042,234 points that had NLC in 1990, a full set of variables could be obtained for a total for 789,959 points (76%). Dropped points were mainly from informal PLCAs whose dates of establishment were unknown hence a full set of explanatory variables was obtained for mostly contractual parks and nature reserves.

Generalized linear mixed models (GLMM) with PLCA identity as a random effect (to account for site-specific factors such as management styles that might influence NLC or BII losses) and different explanatory variables as fixed effects were used. These GLMMs were performed using the lme4 package in R software (R Core Team 2018). All explanatory variables were standardized to ensure that they had a similar scale, with a mean of zero and standard deviation of one. All continuous explanatory variables were tested for collinearity, and correlations below 0.70 were deemed acceptable (Spearman 1987). None of the explanatory variables were however significantly correlated, hence all were retained for analysis.

Multiple plausible models with different combinations of explanatory variables were run to determine the best respective models to explain variations of losses in NLC and BII. Akaike Information Criterion corrected (AICc) was used to determine the best model, with the best model being the one with the lowest AICc and highest weight (Burnham and Anderson 2002). The amount of variance explained by fixed and random effects in the respective top models were then evaluated by calculating the marginal R^2 (variance explained by fixed effects only) and conditional R^2 (variance explained by both fixed and random effects), using the lme4 and MuMIn packages (Barton 2012, Nakagawa and Schielzeth 2013). The respective best models for NLC and BII losses were then assessed for how well they fitted the observed data, with the null hypothesis being that the models are correct. For the NLC loss logistic model the Hosmer-Lemeshow goodness of fit (GOF) test was used using the resource selection package in R (Hosmer et al. 1988, R Core Team 2018) while for the BII loss model the Chi-square GOF which compares the residual deviance to the χ^2 distribution was used based on the lme4 package (Dalgaard 2008, R Core Team 2018). Coefficients of the parameters in the best plausible models, based on delta AICc and AICc weight, were then obtained to determine their magnitude, direction of effect and significance in influencing respective variations in loss of NLC and biodiversity intactness.

4.3 Results

The best model for explaining NLC loss among PLCAs included the following explanatory variables: distance to town, distance to road, elevation, slope, and rainfall, with an AICc weight of 0.64 (Table 4.1), while the second ranked model had a delta AICc of 1.86 and AICc weight of 0.25, indicating biological plausibility. The Hosmer and Lemeshow GOF showed a good model fit for the data (χ^2 = -1957.8, d.f =8, p = 0.99). The marginal R^2 (variance explained by fixed effects) for the best model was 0.04, while the conditional R^2 (for both fixed and random factors) was 0.3. Both random and fixed effects thus explained 30% variance while fixed effects alone explained 4% variation in NLC loss. Table 4.1 shows the four best models to explain variations in NLC loss among PLCAs and the null intercept only model.

Table 4.1. Top four models for explaining variations in natural cover loss across South African private land conservation areas, based on AICc, with the number of model parameters (K), delta AICc (ΔAICc), Akaike weights (wi) and cumulative AICc weights (Cum wi) and the null intercept only model.

Model parameters	K	AICc	ΔAICc	wi	Cum wi
distance to town + distance to road + elevation + slope + rainfall	8	-581670	0	0.64	0.64
distance to town + distance to road + slope + rainfall +size	9	-581669	1.86	0.25	0.89
distance to town + slope + rainfall + age	7	-581667	3.59	0.11	1
distance to town + distance to road + elevation + slope + rainfall + age	10	-581642	28.79	0.01	1
intercept only	3	-578522	3148	0	1

Table 4.2 shows coefficients of parameters in the two best models with delta, that could explain NLC loss variations within PLCAs. Distance to town had a significant effect on NLC loss ($p < 0.001$). Increasing distance from town was associated with decreasing probability of NLC loss. Points further away from major towns were thus more likely to retain NLC than closer ones. In contrast, distance to major road, also a measure of accessibility, showed a

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positive relationship with NLC loss (points closer to roads were more likely to retain NLC than those further away), but this effect was not significant ($p > 0.05$). Elevation and slope also had significant influences on NLC loss. Within PLCAs, higher elevation points and those with steeper slopes had higher chances of retaining NLC, which also represented the effect of accessibility on NLC loss. Rainfall also significantly predicted NLC loss, with high rainfall associated with reduced probability of NLC loss. The size of each PLCA had a negative relationship with NLC loss, i.e. points within larger PLCAs had higher chances of retaining natural cover than those in smaller ones, however its effect was not significant ($p > 0.05$). All models with age as a variable performed badly (Table 4.1), with the best performing model with age as variable having a delta AICc of 3.59.

Table 4.2. Coefficient of parameters from the best mixed effects models developed to assess drivers of natural land cover loss in private land conservation areas in South Africa.

Variable	β	S.E.	t value	p-value
intercept	0.047	0.004	12.65	<0.001
distance to town	-0.04	0.001	-42.39	<0.001
distance to road	0.001	0.0004	1.77	0.08
elevation	-0.005	0.001	-3.78	<0.001
slope	-0.008	0.0003	-28.57	<0.001
rainfall	-0.02	0.0013	-14.43	<0.001
size	-0.002	0.005	-0.38	0.7

For BII, the best model for explaining variations included four explanatory variables: distance to town, slope, rainfall and age, and had an AICc weight of 0.91. Table 4.3 shows the other top ranked models that could explain variations in BII loss. The best model had marginal and conditional R^2 at 0.03 and 0.25 respectively, thus both random and fixed effects accounted for 25% of the observed variance while fixed effects alone explained 3% of the variation. According to the Chi-square GOF test, the best model did fit the data well ($\chi^2 = 7.876$ d.f =8, $p > 0.05$).

There were substantial similarities in how different explanatory variables influenced losses in BII and NLC, with the differences being magnitude of effect and significance (Table 4.4). In terms of direction and magnitude of effect, points closer to town, further away from road, with low elevation, gentle slopes, with low rainfall, in small and old PLCAs had higher chances of losing BII. However, among the above mentioned variables, the effects of distance to road, elevation and size were not significant (Table 4.4).

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Table 4.3. Top four models for explaining variations in biodiversity intactness loss across South African private land conservation areas based on AICc, with the number of model parameters (K), delta AICc (ΔAICc) Akaike weights (wi) and cumulative AICc weights (Cum wi) and the null intercept only model.

Model parameters	K	AICc	ΔAICc	w_i	Cum w_i
distance to town + slope + rainfall + age	7	6033150	0	0.91	0.91
distance to town + distance to road + elevation + slope + rainfall + size + age	10	6033155	4.55	0.09	1
distance to town + distance to road + elevation + slope + rainfall + size	9	6033194	43.98	0	1
distance to town + distance to road + elevation	6	6034522	1371.78	0	1
intercept only	3	6035795	2644.59	0	1

Table 4.4. Coefficient of parameters from the best mixed effects models developed to assess losses in biodiversity intactness across private land conservation areas in South Africa.

Variable	β	S.E.	t value	p-value
intercept	3.18	0.25	12.6	<0.001
distance to town	-2.01	0.06	-36.86	<0.001
distance to road	0.007	0.027	0.25	0.8
elevation	-0.08	0.09	-0.92	0.35
slope	-0.63	0.02	-33.07	<0.001
rainfall	-1.43	0.09	-15.77	<0.001
size	-0.28	0.36	-0.78	0.43
age	0.79	0.12	6.46	<0.001

4.4 Discussion

The effectiveness of PLCAs, as measured by the ability of sample points to retain natural land cover (NLC) and biodiversity intactness (BII), was influenced by a variety of environmental and social-ecological factors, though the amount of variability that could be explained by the factors measured was low. Variables considered to influence losses in NLC and BII were found to be quite similar, with the differences being in magnitude of effect and significance (Table 4.2, Table 4.4). Such a similarity is understandable given the relationship between NLC and the biodiversity it can support (Vitousek et al. 1997, Fischer and Lindenmayer 2007), and the fact that the BII was calculated based the national cover dataset used for NLC.

Accessibility, represented by distance to town, elevation and slope significantly influenced losses in both NLC and BII, and appeared in the best models to explain variations in NLC and BII loss in PLCAs. As predicted, increases in distance to town were associated with higher NLC and BII retention. This pattern has been shown among state-owned PAs, with areas nearer to towns being more susceptible to transformation due proximity to markets for

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wood and other nature based products (Pfeifer et al. 2012, Bowker et al. 2017). Similarly, higher and steeper areas were more likely to retain their NLC and BII, although elevation was not significant in the BII model. Exploiting areas with high elevation and steep slopes for infrastructure development is relatively expensive, hence such areas are at lower risk of transformation (Bowker et al. 2017). High and steep areas are also generally less attractive for agriculture, which is a major cause of NLC loss among PLCAs in the eastern United States (Lacher et al. 2019). In this study, a large proportion of NLC cover lost within both PLCAs and unprotected control was transformed to cultivated land (Chapter 3, Figure 3.1). It therefore makes sense for areas with low elevation, gentle slopes and close to towns to have higher NLC and BII losses because they are more favourable for agriculture. However, contrary to predictions, areas with high rainfall had better NLC and BII retention, perhaps due to the relationship between rainfall and primary productivity, and biological diversity (Nascimbene and Marini 2015). Areas closer to roads were also hypothesized to have greater NLC and BII losses by virtue of being easily accessible (Pfeifer et al. 2012, Bowker et al. 2017), but this was not supported. Although its effect was not significant in both respective models, losses in NLC and BII were likely to be recorded at points farther away from roads (Table 4.2, Table 4.4), perhaps due to better monitoring and law enforcement closer to roads or scepticism in establishing farms and infrastructure closer to the roads.

Also despite insignificance ($p > 0.05$) in both NLC and BII models, points within small PLCAs had a higher chance of losing NLC and BII than points in larger PLCAs (Table 4.2, Table 4.4). Such a relationship has also been shown among state-owned PAs, in which bigger PAs have been shown to do better than smaller ones (Bowker et al. 2017). The insignificance of size however suggests that there is considerable variability in this effect, perhaps explained by different management strategies (e.g., whether or not the PLCA had introduced megafauna; factors that could not be considered in this study due to incomplete data). In addition, in South Africa, many PLCAs are located adjacent or close to other PLCAs and national parks (Chapter 1, Figure 1.1). That is likely to counter for their small size and isolation, by buffering them from NLC or biodiversity loss through positive spill over effects and improved connectivity for species migration. This may be a possible reason why PLCAs size did not come up significant among other model parameters (Langholz and Lassoie 2001, Ament and Cumming 2016). Positive spill-overs occur when adjacency to a national park can encourage PLCAs managers to maintain NLC and biodiversity for ecotourism opportunities (Andam et al. 2008, Ament and Cumming 2016). Furthermore, being adjacent to a big, well managed PA can also reduce the risk of human encroachment and invasive alien species (Cantú-Salazar and Gaston 2010). Nevertheless, because smaller PLCAs had a higher chances of losing NLC and BII, than bigger ones (Table 4.2, Table 4.4), indicates the importance of bigger PLCAs in biodiversity conservation (Cantú-Salazar and Gaston 2010, Lacher et al. 2019). In Northern

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Piedmont, Eastern United States, PLCAs were shown to follow a clustering pattern which formed aggregations which greatly improved landscape connectivity and ecological resilience (Lacher et al. 2019). Consequently, establishing future PLCAs closer to existing PLCAs or other PAs can ensure long-term biodiversity conservation. Conservancies in South Africa, in which landowners of adjacent PLCAs voluntarily remove internal boundary fences and manage their properties as one landscape (Kreuter et al. 2010), are a good example of how continuous landscapes with different management systems and land tenure diversity can improve overall biodiversity conservation, economic viability and ecological resilience (Child et al. 2013, De Vos and Cumming 2019).

Age was an important variable in explaining variations in BII but performed badly in models for NLC loss (Table 4.1, Table 4.3). There were higher chances of BII loss among older PLCAs in comparisons to younger ones. In this study the oldest PLCAs was established in 1926, while the youngest was established in 2016. This inverse relationship with age has also been shown among state-owned PAs (i.e. older ones being less effective than younger ones), with reasons such as community endorsement, establishment with genuine conservation needs and better management being suggested as explanations (Blackman et al. 2015, Bowker et al. 2017). There is also fear that old PLCAs are not properly monitored for compliance with management restrictions, hence likely turning into 'paper PLCAs' characterised by high losses of NLC and BII (Bowker et al. 2017). It has also been shown that particularly in South Africa and Australia the majority of PLCAs landowners are mostly retired people over the age of 60 (Selinske et al. 2017). Consequently perpetual biodiversity conservation cannot be guaranteed when ownership of such PLCAs is transferred to new landowners who may not have the similar motivations for biodiversity conservation (Selinske et al. 2015, Selinske et al. 2017). Such dynamics in ownership might thus explain why old PLCAs gradually become ineffective. It is thus the responsibility of governments, provincial conservation authorities, conservationists to ensure that management plans, land use restriction and best practices for biodiversity conservation are ensured or enforced despite ownership changes among PLCAs (Selinske et al. 2017).

These results should however be treated with caution. Although a big sample size of PLCAs was considered, records of when most informal PLCAs were established were unknown and these areas were discarded from the final analysis. This analysis thus represents results from nature reserves and contractual parks, which are formally protected through the Protected Areas Act (DEA 2016c) and whose dates of establishment are formally recorded, which is not the case with the informal conservation areas, whose complete national and global extent is unknown (Rissman et al. 2017, Fouch et al. 2019). Although the sample of PLCAs analysed here is a good representative to understand the influence of environmental factors on PLCAs, complete data on when most informal PLCAs were established would have

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improved the insights that could be derived from the models. Consequently, the need for PLCAs to be properly documented, i.e. when they were established, their geographic position, what they are managed for, and by who is encouraged. This will not only help to assess their effectiveness and drivers of effectiveness, but ensure better conservation planning, and accountability as well as help in the bid for them to be seriously considered in biodiversity conservation (Stolton et al. 2014, Rissman et al. 2017, Fouch et al. 2019).

The effectiveness of PLCAs in maintaining NLC and biodiversity is dependent on multiple factors, interacting in complex ways at patch, property and regional levels (Cumming et al. 2015). Unlike previous studies where only NLC was used, the use of two proxies in this study allowed for a better understanding of the mechanisms by which different environmental variables influence effectiveness. Variables used were however influenced by data availability. There are therefore other site-specific factors associated with management, as well as other unmeasured social, ecological and economic contextual factors responsible for influencing the effectiveness of PLCAs (Leménager et al. 2014, Bowker et al. 2017). The contribution of such unmeasured variables is supported by the amount of variance explained by random effects in both the NLC and BII loss models (PLCAs was the random variable), as well the observed differences across the different types of PLCAs (Chapter 3, Figure 3.2), which differ in management, legal support for conservation, longevity, land use restrictions and incentives (Cadman et al. 2010, Cumming and Daniels 2014, Mitchell et al. 2018b).

Nevertheless, results from this study show the conditions under which retention of NLC and BII are higher within PLCAs. However, when it comes to using these results for planning on where to establish future PLCAs or state-owned PAs, there are inherent trade-offs: Do we establish PAs where they can do better (big and far away from humans), or establish them where threats are high and increase support for conservation through better policies and management principles? State-owned PA establishment has been shown to be biased to areas which are not necessary biologically diverse or threatened (Joppa and Pfaff 2009, Venter et al. 2017), while some PLCAs have been shown to represent highly diverse, threatened habitats underrepresented in state-owned PAs (Langholz and Lassoie 2001, Gallo et al. 2009). Consequently, for best biodiversity conservation results, most conservationists now support systematic planning of future of PLCAs and state-owned PAs to consider threat levels, biological diversity, and connectivity (Sarkar et al. 2006). In South Africa, the biodiversity stewardship program under which biodiversity agreements are established (Chapter 2, Table 2.2) represents such an approach, in which landowners within biologically diverse habitats are targeted (Cadman et al. 2010, Wright et al. 2018). Such systematic planning may, however, not be applicable to informal conservation areas where landowners have diverse motives for managing the land. For example a landowner would purchase land

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that best represents his/her interests, which may not always align with biodiversity conservation priorities (Mitchell et al. 2018b).

Large scale analyses such as this study reveal general patterns and drivers of effectiveness. Future studies can build on this work to include the effect of other factors not measured in this study, and increase the resolution and site specificity of data associated with different PLCAs to better understand how different factors influence their effectiveness (Fouch et al. 2019). Although there were differences in NLC and BII losses across different PLCAs (Chapter 3, Figure 3.2), there is need to investigate how different management systems, objectives, and motives influence effectiveness, through the use of proxies such as management intensity (Child et al. 2013), animal density, and measures of levels of hunting and ecotourism. There is a need to further understand how proximity or adjacency to national parks influence the effectiveness of PLCAs in NLC and BII retention. Both positive and negative spill over effects have been recorded around South African national parks (Ament and Cumming 2016), but there is need to understand how distance to national parks influences variations in NLC and BII loss from an island biogeography or metapopulation perspective. The theory of island biogeography states that species diversity in an area is dependent on size of the area and the distance from the source population or main land (MacArthur and Wilson 2001), while the metapopulation concept recognises that a population is made up of several populations in different habitats linked together naturally or artificially, through a source–sink relationship under which there is general overall population stability (Hanski et al. 1997, Howell et al. 2018). Accordingly, such concepts apply to PLCAs and how they relate to other PAs, either as source or sink habitats which can consequently explain variation biodiversity intactness amongst them.

In conclusion, this study offers an understanding of how different factors influence the effectiveness of PLCAs, which has important implications in where future PLCAs can be established for better biodiversity conservation results as well the conditions under which PLCAs are highly threatened, hence requiring more attention. The study also highlights the complexity of the factors influencing the effectiveness of PLCAs, which reinforces the need to recognise PLCAs as complex social-ecological systems that influence and are influenced by social, economic, political and ecological factors at different scales. Future studies can thus build upon this work and incorporate other social-ecological factors not considered in this study so that PLCAs can be better understood and better managed for sustainable biodiversity conservation.

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Chapter 5: General Discussion

5.1 Summary of findings

Calls for conservation to look beyond state-owned protected areas (PAs) and include private land conservation areas (PLCAs) and communal areas (Stolton et al. 2014, Maron et al. 2018, Donald et al. 2019), have been mainly driven by the realisation that state-owned PAs alone cannot achieve global biodiversity conservation targets (Maron et al. 2018). The motivation for this study was thus to quantify whether PLCAs are worth to be considered as an effective alternative biodiversity conservation strategy, given their proliferation and increased recognition in increasing the total area for conservation, and representation of some threatened habitats (Gallo et al. 2009, Stolton et al. 2014). The main objective of the study was thus to establish whether South African PLCAs offer significant protection in comparison to unprotected areas exposed to similar environmental conditions, using two proxies i.e. natural land cover (NLC) and the biodiversity intactness index (BII). The prediction was that if PLCAs are effective, losses in NLC and BII would be significantly less within PLCAs in comparison to unprotected areas. Of interest was also how effectiveness varied across different types of PLCAs given their diversity in legal frameworks guiding them, incentives and land use restrictions. The prediction was that formal, legally protected PLCAs would be more effective than the informal ones. The second phase was then to determine the best respective models for explaining variation in NLC and BII losses and understand how different social-ecological factors influenced the ability of PLCAs to retain NLC and BII. This was the first study to quantify the effectiveness of PLCAs and its drivers at a national scale in Africa, which was timely given the discussions towards including PLCAs in global biodiversity conservation targets.

The study provides the first evidence that a national system of PLCAs was an effective strategy for biodiversity conservation. Unprotected areas lost roughly double the amount of NLC and BII lost within PLCAs, indicating the significance of the protection offered by PLCAs (Chapter 3). Effectiveness did not however, depend on level of protection as predicted, with informal PLCAs having better NLC and BII retention than some formally protected PLCAs (Chapter 3, Figure 3.2). Notably, biodiversity agreements (moderate protection) and nature reserves (high protection) had the highest losses in NLC and BII, losing more than the conservation areas and conservancies which are informally protected (Appendix 1, Figure 3.2). Most NLC lost within PLCAs was converted to agriculture (Chapter 3, Figure 3.1), a trend also observed for informal conservation areas in the United States (Rissman and Merenlender 2008, Lacher et al. 2019), highlighting the complexity in trade-offs associated with balancing both economic and conservation objectives among PLCAs.

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The effectiveness of PLCAs in preventing losses in NLC and BII was mainly influenced by accessibility (distance to towns, elevation, and slope) which confirms known impacts of humans on biodiversity (Nagendra 2008, Pfeifer et al. 2012, Bowker et al. 2017) and demonstrates how easily accessible areas are more vulnerable to transformation for infrastructure development and for cultivation and deforestation (Chapter 3, Figure 3.1). The percentage of variability in NLC and BII losses that could be explained by considered variables was low thus demonstrating the complexity in the likely drivers of effectiveness in this diverse conservation mechanism.

5.2 Implications of the findings

Results from this study help shed light on the role of PLCAs in biodiversity conservation, as well as how policies and management strategies can be better placed to ensure the long-term contribution of PLCAs to biodiversity conservation. The understanding of how the different factors influence the effectiveness of PLCAs can also be used to inform decisions about where to establish future PLCAs and state-owned PAs.

The fact that most NLC lost within PLCAs was converted to agriculture brings uncertainty regarding the capacity of PLCAs to conserve biodiversity long term, given how agriculture can negatively affect biodiversity and affect the provisioning of ecosystem services (Reyers et al. 2009, Biggs et al. 2012). Although NLC lost was quite low (3%) the fact that most of it was converted to agriculture shows that this can a big threat to PLCAs and their role in biodiversity given the global human population increase and the need to increase agricultural food production in future. However, given that the majority of PLCAs in South Africa are self-funded, agriculture, and other land uses can provide funding for conservation work while ensuring financial viability for the landowner (Wright et al. 2018). For example, if ecotourism is not doing well one might turn to crop or livestock farming and vice versa. There is therefore fear that the effectiveness of PLCAs as a conservation strategy is conditional i.e. if they are reliant on ecotourism to maintain biodiversity and NLC, changes in political or economic factors negatively affecting ecotourism could cause landowners to turn to agriculture, which is not best for biodiversity conservation objectives. In the worst-case scenario, for example, a disaster or civil war, might affect tourist visits, which will then affect the relevance of PLCAs to biodiversity conservation if ecotourism is the major motivation to begin with. Nevertheless, it is essential to understand that one of the reason why PLCAs in general seem a worthwhile conservation strategy is that they are financially driven, which allows them to achieve both economic and ecological objectives (Norton 2000, Langholz and Lassoie 2001). Their relevance to both biodiversity conservation and to respective landowners managing them is thus dependent on the landowners realisation of their objectives and their financial viability, which is more of a ‘if it pays it stays’ scenario (Norton 2000, Pfaff and

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Robalino 2012). For example, despite genuine biodiversity conservation intentions, landowners have to at least cover running costs otherwise they will be making a loss and forced to turn to other land uses. Consequently, there are compromises and trade-offs to make between biodiversity conservation and economic viability when dealing with PLCAs, which policies and regulations should consider and ensure that landowner objectives are achieved without compromising biodiversity and vice versa (Norton 2000). Studies to better understand the motives, management objectives and expectations of landowners are thus encouraged to determine under which circumstances will biodiversity or economic viability be compromised.

On a positive note, despite agricultural transformation being the highest land use to which NLC was lost among PLCAs and unprotected areas (Chapter 3, Figure 3.1), its effects to biodiversity are relatively low compared to transformations to land uses such urban areas or mines (Chapter 2, Table 2.4). Agricultural lands have also been shown to be important for seed eating birds (Scholes and Biggs 2005, Child et al. 2009), which makes the effects of agricultural transformation within PLCAs tolerable. Nevertheless, if biodiversity protection is to be improved within PLCAs, policies advocating for minimum or no agriculture should be encouraged. One way is to incentivise NLC and biodiversity protection by rewarding good stewardship, for example through rewarding landowners that maintain their NLC or do less agriculture. The reduced emissions from deforestation and degradation (REDD) program is one successful exemplary program in which landowners and communities in developing countries are paid to maintain forests (Köhl et al. 2009, Pfaff and Robalino 2012).

The observed differences in NLC and BII losses across different types of PLCAs (Chapter 3, Figure 3.2) also highlight how differences in motives, incentives, legal frameworks, and restrictions influence effectiveness. Although the hypothesis of legally protected PLCAs being likely to be more effective was not supported, the results have important implications about how PLCAs are viewed at least in a South African context. Biodiversity agreements, established through the biodiversity stewardship program had the most losses in NLC and BII, despite being moderately protected (Cadman et al. 2010), losing more than the informally protected conservancies and conservation areas. In their establishment, biodiversity agreements involve contractual laws between landowners of properties containing highly diverse or threatened habitats, and the provincial conservation authority. In exchange for the landowner’s commitment to biodiversity conservation by keeping the area natural or semi-natural, the provincial conservation authority helps to draft a management plan, provide extension services, and helps with managing fires and invasive species (Cadman et al. 2010, Wright et al. 2018). Biodiversity agreements are thus an incentive based conservation strategy and their effectiveness likely depends on the provision of such incentives by the conservation authorities, which varies across the provinces (Pasquini et al. 2010, Wright et al. 2018). This is different to conservation areas and conservancies, which tend not to be motivated by

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extension services because they receive minimum assistance in this regard (Cadman et al. 2010). Instead, landowners and managers of conservation areas might be motivated by returns from ecotourism and hunting opportunities, hence might have better reason to maintain NLC and biodiversity under favourable economic and political conditions (Langholz and Lassoie 2001). This might thus explain why they did better in NLC and BII retention than biodiversity agreements.

Landowners of PLCAs under biodiversity agreements in South Africa have expressed some concerns with how the biodiversity stewardship programs are run (Pasquini et al. 2010). They have cited incompetence of the provincial conservation authorities to implement the program, their minimal participation in the design and implementation of the program, under appreciation of their efforts in biodiversity conservation, indifference of the conservation authorities to their needs as their major concerns (Pasquini et al. 2010). Lack of effective communication between the extension officers and the landowners is also another reason why landowners within biodiversity agreements have problems with the program, with most landowners expressing the need for more engagement and more site visits by extension officers (Pasquini et al. 2010, Selinske et al. 2015). Consequently, for biodiversity agreements and PLCAs in general to do better, there is need for sufficient funding to ensure effective provision of extension services and monitoring of landowners compliance to agreed management plans, and land use restrictions (Pasquini et al. 2010, Hanley et al. 2012, Selinske et al. 2017). There is also need for social learning, building trust, and social capital between conservation authorities and landowners if the effectiveness of PLCAs in general is to be enhanced (Pasquini et al. 2010, Selinske et al. 2015, Selinske et al. 2017). Transparency and governance approaches that foster for better cooperation and working together between PLCA landowners, governments, and provincial conservation authorities, policy makers, conservationists are also encouraged (Norton 2000, Pasquini et al. 2010, Hanley et al. 2012, Selinske et al. 2015).

Also, biodiversity agreements typically have durations of five to ten years (Cadman et al. 2010). Although they have option for extension beyond that period, there is no guaranteed long-term biodiversity conservation in such areas given how economic and political factors can change in favour of other land uses. The five to ten year contractual agreements between the landowner and the provincial conservation authorities also coincide with national elections which might also influence the landowner’s commitment to biodiversity conservation given the uncertainties associated with political regime changes. Such scepticism from a landowner’s perspective may explain why NLC and BII losses were more pronounced within biodiversity agreements. Long term contractual agreements guaranteeing perpetual commitment to biodiversity conservation by landowners are therefore encouraged.

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Nevertheless, these results confirm the need not to view PLCAs as islands but rather as social-ecological systems, whose overall performance in biodiversity is influenced by complex social, economic, political and ecological factors (Palomo et al. 2014, Clements et al. 2016a), in which trade-offs and compromises are inevitable (Norton 2000). On the ecological side, the study confirmed the influence of accessibility on effectiveness, which has relevance to how future PLCAs can be established to ensure biodiversity conservation as well as offer understanding on how more accessible PLCAs, under greater pressure, can be better protected. However given the known bias in state-owned PAs being in areas not necessarily biologically diverse or highly threatened (Joppa and Pfaff 2009, Venter et al. 2017), the dilemma is where should future PLCA be established. Should they be established far away from humans, where they are less at risk of NLC or biodiversity loss or they should be established where threats are high and be better protected despite the threat levels? Many conservationists support the latter, in which PLCAs and state-owned PAs are encouraged to be established in highly threatened habitats rather than in areas where species diversity would not be lost without protection (Myers et al. 2000, Joppa and Pfaff 2009). Consequently, the establishment of new PAs now includes systematic planning and the use of algorithms to predict best options for biodiversity conservation results, thus considering biological significance, threats level and connectivity (Sarkar et al. 2006). Such thinking at least in a South African context is being applied to the establishment of PLCAs under the biodiversity stewardship program in which the focus is on establishing them within biologically important habitats by encouraging landowners within such areas to sign up for the program (Cadman et al. 2010, Wright et al. 2018). However, for informal areas which are voluntarily protected through the independent decisions of respective landowners, such systematic planning may not be applicable since their emergence and commitment to conservation is depended on motivations, economic, political and other social factors (e.g., landowner lifestyle choices) (Cadman et al. 2010, DEA 2016c, Selinske et al. 2017). For example such properties may oscillate their land use between crop farming, livestock farming, wine farming, mixed wildlife and livestock, and ecotourism, depending on what is economically viable (Jones et al. 2005, Pasquini et al. 2010). Also considering that most private landowners will have clear motives and objectives when purchasing land, it is essential to understand that they will purchase land at a location that best represents their interests. If one’s priority is ecotourism, accessibility of the area in relation to towns, roads and airports would be a big factor to consider despite not being best for biodiversity conservation (Mitchell et al. 2018b). It is no coincidence that PLCAs that have been shown to generate most profits are closer to towns, roads and airports (Clements et al. 2016a). This is likely to compromise their effectiveness in preventing loss of NLC and protecting biodiversity since they will be required to ensure better accommodation, better roads, and higher stocking densities of charismatic species ideal for ecotourism at the

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expense of other species and vegetation (Cumming et al. 1997, Child et al. 2013, Clements et al. 2016a). Nevertheless, the results on factors that influence effectiveness indicate the kind of trade-offs associated with balancing threat level and biological significance.

Although its effect was not significant, losses in NLC and BII had a higher chance of occurring within small PLCAs, hence bigger ones were more effective (Table 4.1, Table 4.3), which is in line with other studies done on state-owned PAs (Bowker et al. 2017). One of the reasons why PLCAs have been neglected in conservation planning is the argument that they are too small to effectively conserve mega fauna requiring large areas, with bigger, spatially heterogeneous areas being preferred in that regard (Cantú-Salazar and Gaston 2010, Allen et al. 2016, Lacher et al. 2019). With that in mind, establishment of future PLCAs by governments, conservation authorities and landowners should aim for bigger and connected landscapes. One way of going about this is establishing PLCAs in clusters or near other already existing PAs to ensure large connected and ecological resilient landscapes (Langholz and Lassoie 2001, Lacher et al. 2019). This has been shown to be the reason why small parcels of PLCAs in the United States of America have managed to ensure connectivity and overall success in biodiversity conservation (Lacher et al. 2019). The theory of island biogeography also supports the establishment of bigger connected landscapes for conservation (MacArthur and Wilson 2001), which is appropriate given the relationship between size and BII and NLC losses among PLCAs. However, despite the importance of bigger PLCAs it is important to acknowledge that some small PLCAs are actually protecting the last remnants of critically endangered ecosystems (Gallo et al. 2009), and some studies have also indicated the importance of such small isolated patches in biodiversity conservation (Lindenmayer 2019, Wintle et al. 2019). Accordingly, even though big continuous landscapes are ideal, the contribution of small parcels to biodiversity conservation should not be overlooked (Tulloch et al. 2016).

5.3 Study limitations and recommendations for future studies

The limitations of this study should be considered when interpreting and applying its findings. The absolute number and location of informal conservation areas remains unknown not only in South Africa but globally (Rissman et al. 2017, Clements et al. 2018). This not only raises questions about their spatial coverage, but makes it difficult to draw conclusions about their contributions to biodiversity conservation. This study had a good sample size of PLCAs to make conclusions (5,121 properties) however the majority of sites used were nature reserves (Figure 2.2), courtesy of the quarterly updated SAPAD and PACA databases (DEA 2013, 2016c). This is mainly because formally protected PLCAs (nature reserves and contractual parks) follow a legal gazzettment process hence they are well documented (De Vos et al. 2019). This is not the case with informal PLCAs (Rissman et al. 2017, Clements et al. 2018).

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Consequently, because there is no formal database for informal PLCAs, data used in this study were based on availability, hence likely not to be an exhaustive list. Conclusions of this study might therefore be slightly biased. However, without crying foul of data shortages, researchers should supplement data by other means, rather than relying solely on publicly available data sources, which are often not sufficient (Fouch et al. 2019). In this study, such data were sourced from provincial conservation bodies, manual digitizing from google products and from previous studies (Clements 2016, De Vos et al. 2019). Better estimates about the contribution of PLCAs and models assessing how their effectiveness is influenced by different factors would have been more robust with a complete dataset though. Future studies should thus prioritise proper documentation and maintenance of data about PLCAs, where they are located, when they were established, the motives for establishment, what they are managed for and by who, which will not only improve estimates about their contribution, but will be essential in ensuring accountability and ensure that they are taken seriously as a biodiversity conservation strategy (Fouch et al. 2019). The lack of proper documentation about PLCAs is one of the main reasons why incorporating them into regional and national conservation action plans has been difficult (Stolton et al. 2014).

This study was based on proxies taken to represent the effectiveness of PLCAs in biodiversity conservation. Although such proxies have been used in multiple studies that have attempted to quantify the effectiveness of other PAs, with which governments have managed to make sound decisions and policies (Bruner et al. 2001, Liu et al. 2001, Scholes and Biggs 2005, Clark et al. 2008, Gillespie et al. 2008, Pfeifer et al. 2012, Heino et al. 2015, Ament and Cumming 2016, Newbold et al. 2016, Bowker et al. 2017, Andam et al. 2018, Fouch et al. 2019), they are not flawless. Their use is generally inspired by limited funding and methodological constraints to conduct direct measurements of biodiversity (Nelson and Chomitz 2011, Fouch et al. 2019). In this study their limitations are acknowledged.

With the national land cover maps, land cover classes based on spectral reflectance at 30m resolution, there are limitations about the amount of detail they can reveal even though a land use mapping accuracy of 88.4% was reported (DEA 2015b). There is therefore need for hyperspectral remote sensed data to get finer details about land classes and their dynamics over time, despite being costly and time consuming for nationwide studies (He et al. 2011, Olsson et al. 2011). Nevertheless, previous studies have made use of maps with 30 m^2, 90 m^2 and even 1 km^2 resolution to establish trends and contribution of strategies to biodiversity conservation (Biggs et al. 2006, Ament and Cumming 2016, Bowker et al. 2017, Fouch et al. 2019), which justifies the resolution of products used in this study. Of concern however is the failure of the maps to distinguish self-seeded bush encroached areas and invasive alien species from bushlands and thickets (Table 2.3) (DEA 2015b), which is a limitation given how bush encroachment and invasive alien species reduce overall species diversity (Pejchar and

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Mooney 2009, Goodenough 2010). Such differences and finer details associated with habitat quality and species differentiation can only be revealed when hyperspectral remote sensed data are used (He et al. 2011, Olsson et al. 2011), which future studies can focus on at smaller scales or at property level and get finer land cover changes. The Enhanced Freshwater and Terrestrial Observation Network (EFTEON) in South Africa is a good example in which different organisations work on smaller human-transformed landscapes to monitor and understand different aspects about biodiversity, ecosystem productivity, and ecosystem services (DST 2016).

The BII used also has limitations. For example, by requiring different data sets as inputs its accuracy is dependent on the accuracy of its inputs. The index has also received criticism for overestimating intactness in some instances (Martin et al. 2019). In its calculation the index makes reference to undisturbed pristine populations (Scholes and Biggs 2005) which are difficult to find nowadays, since some of the pristine reference populations have since experienced significant human modification, which is a limitation in its applicability (Martin et al. 2019, Newbold et al. 2019). Also, the failure of the BII to concur with other vegetation metrics such as the biomass intactness has also been regarded as a cause of concern (Martin et al. 2019). Nevertheless, the index offers a quick and cheap way of assessing the state of biodiversity, and reveal general trends without the need for extensive field work (Mace 2005). In this study the index proved essential in assessing the contribution of PLCAs to biodiversity conservation. Similar patterns in BII and NLC losses between PLCAs and unprotected areas as well as relationships with explanatory variables demonstrate its applicability and usefulness. The index has also been applied on global scale studies with implications in policy formulation (Scholes and Biggs 2005, Biggs et al. 2006, Newbold et al. 2015, Newbold et al. 2016).

Another limitation of the study was that variables considered to influence losses of NLC and BII and to match sample and control points were based on data availability. There are therefore a number of site specific variables responsible for variations in NLC and BII losses associated with property level management scenarios, ecological and economic factors not measured in this study (Leménager et al. 2014, Bowker et al. 2017). This study thus provides insight into some of the variables that can influence the effectiveness of PLCAs, while the effect of other factors and their complex interactions remains unanswered. The existence of such factors is evidenced by amount of variance explained by fixed and random effects among the respective best models for NLC and BII loss (Table 4.1, Table 4.3). Also the observed differences in losses in NLC and BII among different types of PLCAs (Figure 3.2) is confirmation that other unmeasured factors are responsible for the variations in NLC and BII. Future studies can thus build upon this work and include other variables so as get a full picture of how different factors influence the effectiveness of PLCAs. Some of the factors future

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studies can consider are management objectives or business models, management intensity represented by the amount of waterholes, translocations, reintroductions, and stocking density (Child et al. 2013) and understand how they influence the effectiveness of PLCAs in conserving biodiversity as well as how they are linked to land degradation, occurrence of invasive species and the provision of other ecosystem services. Such factors have a direct link to NLC and biodiversity and hence have the potential to influence the effectiveness of PLCAs (Child et al. 2013). What will also be interesting is to use metrics of connectivity and fragmentation and establish how they influence biological diversity and effectiveness within PLCAs. Such metrics can help better understand how the theory of island biogeography applies to PLCAs. One easier metric future studies can consider in that regard is how distance to state-owned PAs influences NLC and BII losses within PLCAs. Also with profit oriented management systems being one of the reasons why PLCAs are criticised, it is important for future studies to have proxies for ecotourism intensity such as number tourist visitation (De Vos et al. 2016) or profits made (Clements et al. 2016a) and how that relates to effectiveness in biodiversity conservation as well as to tourist perceptions, expectations and satisfaction. At a smaller scale, methods such as line transects and camera traps can be used to understand differences in the diversity and richness of mammals between PLCAs and unprotected areas as well as how such diversity varies across the different types of PLCAs and management systems. The densities of key indicator species such as insects can also be used by smaller scale future studies to quantify effectiveness of PLCAs relative to similar unprotected areas or across a management intensity gradient.

Conclusions in this study are also based on treating the national PLCA estate as homogeneous, which is not the case, despite fitting the scope of this study. In some regional studies some state-owned PAs were shown to experience greater than or equal levels of NLC loss than unprotected areas (Pfeifer et al. 2012, Bowker et al. 2017). Such parks are essentially ineffective and regarded as paper parks (Pfeifer et al. 2012, Bowker et al. 2017). By treating the PLCA estate as a single entity in this study, although different categories were recognised, the existence of 'paper PLCAs' is overlooked. Future studies should thus aim to reveal how individual PLCAs perform in comparisons to unprotected areas, so that effective and ineffective PLCAs can be identified. This will then open up avenues for investigating how they are managed, what they are managed for, where they are located, and how their effectiveness can be improved through better management techniques.

5.4 General conclusions

This study presents the first national-scale evidence that PLCAs offer relevant contributions to biodiversity conservation justifying the need to recognise and include them in broader conservation plans. Their effectiveness was influenced by multiple factors at different scales,

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associated with where they are located and how close they are to humans, which can be managed, incorporated into policies and considered in the establishment of future PLCAs.

Nevertheless, although PLCAs were shown to be effective in this study, they are not a panacea to all modern day biodiversity loss problems. Genuine limitations about their potential have been raised, which include their typical small sizes, profit oriented management systems, and uncertain tenure systems (Langholz and Lassoie 2001, Clements et al. 2016a, Clements and Cumming 2017). There is also empirical evidence that some PLCAs in the United States are relatively ineffective in comparison to unprotected areas raising questions about the generalizability of these findings to other parts of the world (Fouch et al. 2019). However, this study shows the potential of PLCAs in biodiversity conservation. PLCAs have been shown to have diverse land tenure systems, governance, management systems, and motives for their establishment, which is essential for biodiversity conservation (Fitzsimons and Wescott 2008, Child et al. 2013, De Vos and Cumming 2019). This is because of the link between diversity (ecological and institutional) and social-ecological resilience (Biggs et al. 2015, De Vos and Cumming 2019). The general agreement is that diverse systems are more resilient than homogenous ones, hence likely to protect functional groups despite changes in social, economic, ecological and political factors (Folke et al. 2016). Consequently, although differences in management objectives, incentives, restrictions and motives might bring about doubts about the contributions of PLCAs, it brings diversity and redundancy essential for protection of biodiversity in the face of changes in social, economic, political and ecological factors which will affect each PLCAs type differently (De Vos and Cumming 2019). Their contribution to biodiversity conservation is also further reinforced when PLCAs are considered together with state-owned PAs. This is because social, economic, and political factors affect state-owned PAs and PLCAs differently, which brings about complementarity essential for long-term biodiversity conservation (De Vos et al. 2019, Fouch et al. 2019). For example, PLCAs and state-owned PAs have differences in sources of funding, land tenure dynamics, social acceptability and management capacity among other factors which brings about synergies when the two run concurrently (Mitchell et al. 2018a). Also, PLCAs can easily attract a diverse set of stakeholders to invest in conservation through a bottom-up approach, that can easily access funds from individuals and organisations who are usually sceptical about giving such funds to respective governments to deal with conservation issues especially in countries with high corruption and poor accountability (Laurance et al. 2006). With a number of state-owned PAs having been shown to suffer from poor funding particularly in developing countries, it is ideal in such circumstances to have well established PLCA structures since they have been shown to have multiple sources of funding that can engage the general public in biodiversity conservation (Mitchell et al. 2018b). Nevertheless, PLCAs may not exclusively achieve biodiversity conservation targets alone but neither can state-owned PAs on their own,

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which thus brings the need for an inclusive landscape approach involving PLCAs, state-owned PAs and communal areas, if overall success in biodiversity conservation is to be achieved (Leménager et al. 2014, Mitchell et al. 2018b).

Efforts to incorporate PLCAs into national and regional action plans are therefore encouraged. Already the existence of literature providing guidelines and best practices for PLCAs, indicates increased interest in integrating them in conservation strategies (Stolton et al. 2014, Jonas and MacKinnon 2016, Mitchell et al. 2018b). However, problems still exist with what properties to consider and what not to consider as PLCAs, mainly because of the lack of a standard universally agreed definition of PLCAs (Stolton et al. 2014). This is especially due to the fact that IUCN does not recognise all conservation efforts as worth the consideration, particularly when there is no long-term legal support for protection or when biodiversity conservation is not perceived as the main objective (Stolton et al. 2014, Mitchell et al. 2018b). Although they now recognise some efforts as other effective area-based conservation measures (OECMs), results from this study and elsewhere (Bhagwat et al. 2005), highlight the importance of informal conservation efforts. Governments and conservation authorities should thus recognise the importance of PLCAs and especially the informal ones. In South Africa both informal and formally protected PLCAs, and state-owned PAs are counted towards the national conservation estate, despite the informal ones not being officially recognised (DEA 2013). The national protected areas expansion model through the biodiversity stewardship program also has biodiversity agreements as its main focus, with little focus on informal conservation areas and conservancies (DEA 2016c, Wright et al. 2018). However, based on these results there is need for the authorities to equally appreciate the role of informal PLCAs and invest in incentivising them as much as they do to the formal ones (Norton 2000, Selinske et al. 2015). Advocacy for the recognition of PLCAs, especially the informal ones, by states, NGOs, researchers, and the general public is thus encouraged. In the United States recognition through plaques and certificates was essential to get landowners motivated in having their properties officially recognised (Shafer 2004, Pasquini et al. 2010). Recognition through green certification, stewardship awards and media coverage, has also been shown to be essential in motivating PLCAs landowners in biodiversity conservation (Doremus 2003, Pasquini et al. 2010). Such recognition does not only encourage environmentally conscious landowners but allows them to realise they are part of big community striving to curb biodiversity loss.

Results here also show the need to recognise that like PAs, PLCAs are complex SESs which affect and are affected by social, economic, ecological, political and environmental factors at different scales (Janzen 1983, Cumming et al. 2015). The effectiveness, management and understanding of PLCAs thus has to shift from the traditional way of viewing them as independent isolated entities (Janzen 1983, Cumming et al. 2015). They exist under

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different management systems, are established with diverse motives and management objectives, land use restrictions and incentives which all influence their effectiveness together with other unmeasured variables. This study demonstrated how some of these factors influence PLCA effectiveness. However, the reality is that there are other factors, interacting at multiple scales not measured in this study that influence the effectiveness of PLCAs. Future studies can thus build upon this work and aim to deeply understand the complexity of the different factors influencing the effectiveness of PLCAs, which will have important implications on how they can be better managed to ensure long-term sustainable biodiversity conservation.

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Appendices

Appendix 1. Tukey post-hoc analysis showing results of multiple pairwise comparison of natural cover loss between different private land conservation areas categories and unprotected control points in South Africa.

Comparison	Difference	Lower	Upper	Bonferroni Adj p	Significance
Conservancy - Biodiversity Agreement	-0.017	-0.025	-0.010	<0.001	***
Conservation area - Biodiversity Agreement	-0.024	-0.031	-0.016	<0.001	***
Unprotected - Biodiversity Agreement	0.022	0.016	0.029	<0.001	***
Contractual Park- Biodiversity Agreement	-0.024	-0.033	-0.016	<0.001	***
Nature Reserve - Biodiversity Agreement	-0.007	-0.014	-0.001	0.019	*
Conservation area – Conservancy	-0.006	-0.011	-0.001	0.008	*
Unprotected – Conservancy	0.039	0.036	0.043	<0.001	***
Contractual Park- Conservancy	-0.007	-0.014	0.0001	0.07	ns
Nature Reserve – Conservancy	0.010	0.006	0.014	<0.001	***
Unprotected - Conservation area	0.046	0.042	0.049	<0.001	***
Contractual Park- Conservation area	-0.0007	-0.008	0.006	1.00	ns
Nature Reserve - Conservation area	0.016	0.013	0.02	<0.001	***
Contractual Park- Unprotected	-0.047	-0.05	-0.04	<0.001	***
Nature Reserve – Unprotected	-0.029	-0.03	-0.029	<0.001	***
Nature Reserve - Contractual Park	0.017	0.011	0.023	<0.001	***

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Appendix 2. Tukey post-hoc analysis showing results of multiple pairwise comparison of biodiversity intactness loss between different private land conservation area categories and unprotected control points in South Africa.

Comparison	Difference	Lower	Upper	Bonferroni Adj p	Significance
Conservancy - Biodiversity Agreement	-0.004	-0.009	0.0008	0.28	ns
Conservation area - Biodiversity Agreement	-0.008	-0.013	-0.004	<0.001	***
Unprotected - Biodiversity Agreement	0.021	0.016	0.025	<0.001	***
Contractual Park- Biodiversity Agreement	0.009	0.0035	0.015	<0.001	***
Nature Reserve - Biodiversity Agreement	-0.002	-0.006	0.002	1	ns
Conservation area – Conservancy	-0.004	-0.0078	-0.001	0.03	*
Unprotected – Conservancy	0.025	0.02	0.027	<0.001	***
Contractual Park- Conservancy	0.005	0.0007	0.01	0.02	*
Nature Reserve – Conservancy	-0.006	-0.009	-0.004	<0.001	***
Unprotected - Conservation area	0.03	0.026	0.031	<0.001	***
Contractual Park- Conservation area	0.0009	-0.004	0.006	1	ns
Nature Reserve - Conservation area	-0.01	-0.013	0.008	<0.001	***
Contractual Park- Unprotected	0.03	0.03	0.034	<0.001	***
Nature Reserve – Unprotected	0.02	0.018	0.02	<0.001	***
Nature Reserve - Contractual Park	0.01	0.007	0.02	<0.001	***

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Appendix 3. Tukey post-hoc analysis showing results of multiple pairwise comparison of natural land cover loss across different South African biomes using data inside and outside private land conservation areas.

Comparison	Difference	Lower	Upper	Bonferroni Adj p	Significance
Fynbos - Forest	-0.015	-0.02	-0.008	<0.001	***
Grassland - Forest	0.02	0.014	0.026	<0.001	***
Nama Karoo - Forest	-0.038	-0.044	-0.032	<0.001	***
Savanna - Forest	-0.005	-0.01	0.001	0.22	ns
Succulent Karoo - Forest	-0.38	-0.45	-0.03	<0.001	***
Thicket - Forest	-0.017	-0.023	-0.01	<0.001	***
Grassland - Fynbos	0.035	0.03	0.037	<0.001	***
Nama Karoo - Fynbos	-0.024	-0.03	-0.21	<0.001	***
Savanna - Fynbos	0.01	0.008	0.012	<0.001	***
Succulent Karoo - Fynbos	-0.024	-0.027	-0.021	<0.001	***
Thicket - Fynbos	-0.002	-0.005	0.0006	<0.001	***
Nama Karoo - Grassland	-0.058	-0.06	-0.057	<0.001	***
Savanna - Grassland	-0.025	-0.026	-0.024	<0.001	***
Succulent Karoo - Grassland	-0.06	-0.061	-0.06	<0.001	***
Thicket - Grassland	-0.037	-0.04	-0.035	<0.001	***
Savanna - Nama Karoo	0.034	0.03	0.035	<0.001	***
Succulent Karoo - Nama Karoo	-0.0001	-0.003	0.002	1	ns
Thicket - Nama Karoo	0.021	0.019	0.024	<0.001	***
Succulent Karoo - Savanna	-0.034	-0.036	-0.031	<0.001	***
Thicket - Savanna	-0.012	-0.014	-0.01	<0.001	***
Thicket - Succulent	0.022	0.018	0.025	<0.001	***

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Appendix 4. Tukey post-hoc analysis showing results of multiple pairwise comparison of biodiversity intactness loss across different South African biomes using data inside and outside private land conservation areas.

Comparison	Difference	Lower	Upper	Bonferroni Adj p	Significance
Fynbos - Forest	0.008	0.005	0.013	<0.001	***
Grassland - Forest	-0.013	-0.02	-0.01	<0.001	***
Nama Karoo - Forest	0.027	0.024	0.032	<0.001	***
Savanna - Forest	0.005	0.0013	0.009	0.002	
Succulent Karoo - Forest	0.027	0.023	0.03	<0.001	***
Thicket - Forest	0.012	0.008	0.016	<0.001	***
Grassland - Fynbos	-0.022	-0.023	-0.02	<0.001	***
Nama Karoo - Fynbos	0.02	0.017	0.02	<0.001	***
Savanna - Fynbos	-0.003	-0.005	-0.002	<0.001	***
Succulent Karoo - Fynbos	0.018	0.016	0.02	<0.001	***
Thicket - Fynbos	0.003	0.001	0.005	<0.001	***
Nama Karoo - Grassland	0.04	0.04	0.042	<0.001	***
Savanna - Grassland	0.02	0.017	0.02	<0.001	***
Succulent Karoo - Grassland	0.04	0.04	0.041	<0.001	***
Thicket - Grassland	0.025	0.023	0.026	<0.001	***
Savanna - Nama Karoo	-0.022	-0.023	-0.021	<0.001	***
Succulent Karoo - Nama Karoo	-0.001	-0.003	0.001	0.73	Ns
Thicket - Nama Karoo	-0.016	-0.017	-0.014	<0.001	***
Succulent Karoo - Savanna	0.022	0.02	0.023	<0.001	***
Thicket - Savanna	0.006	0.005	0.008	<0.001	***
Thicket - Succulent	-0.014	-0.017	-0.013	<0.001	***