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AN EVALUATION OF THE EFFECTIVENESS OF AREA-BASED CONSERVATION INTERVENTIONS IN AVOIDING BIODIVERSITY LOSS IN SOUTH AFRICA

L. VON STADEN

2023

AN EVALUATION OF THE EFFECTIVENESS OF AREA-BASED CONSERVATION INTERVENTIONS IN AVOIDING BIODIVERSITY LOSS IN SOUTH AFRICA

By

Lize von Staden

A thesis submitted
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Supervisor: Prof. Amanda Lombard
Co-supervisor: Dr Stephen Holness

DECLARATION

Name: Lize von Staden

Student number: 220019797

Qualification: Doctor of Philosophy

Title of project: An evaluation of the effectiveness of area-based conservation interventions in avoiding biodiversity loss in South Africa

In accordance with Rule G5.11.4, I hereby declare that the above-mentioned treatise/ dissertation/ thesis is my own work and that it has not previously been submitted for assessment to another University or for another qualification.

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This work is dedicated to my father, André Agenbag (1950·10·04 - 2019·02·13), my first role model and mentor who shared with me a deep appreciation for nature and South Africa's wild plants and encouraged me to pursue a research career in biodiversity conservation.

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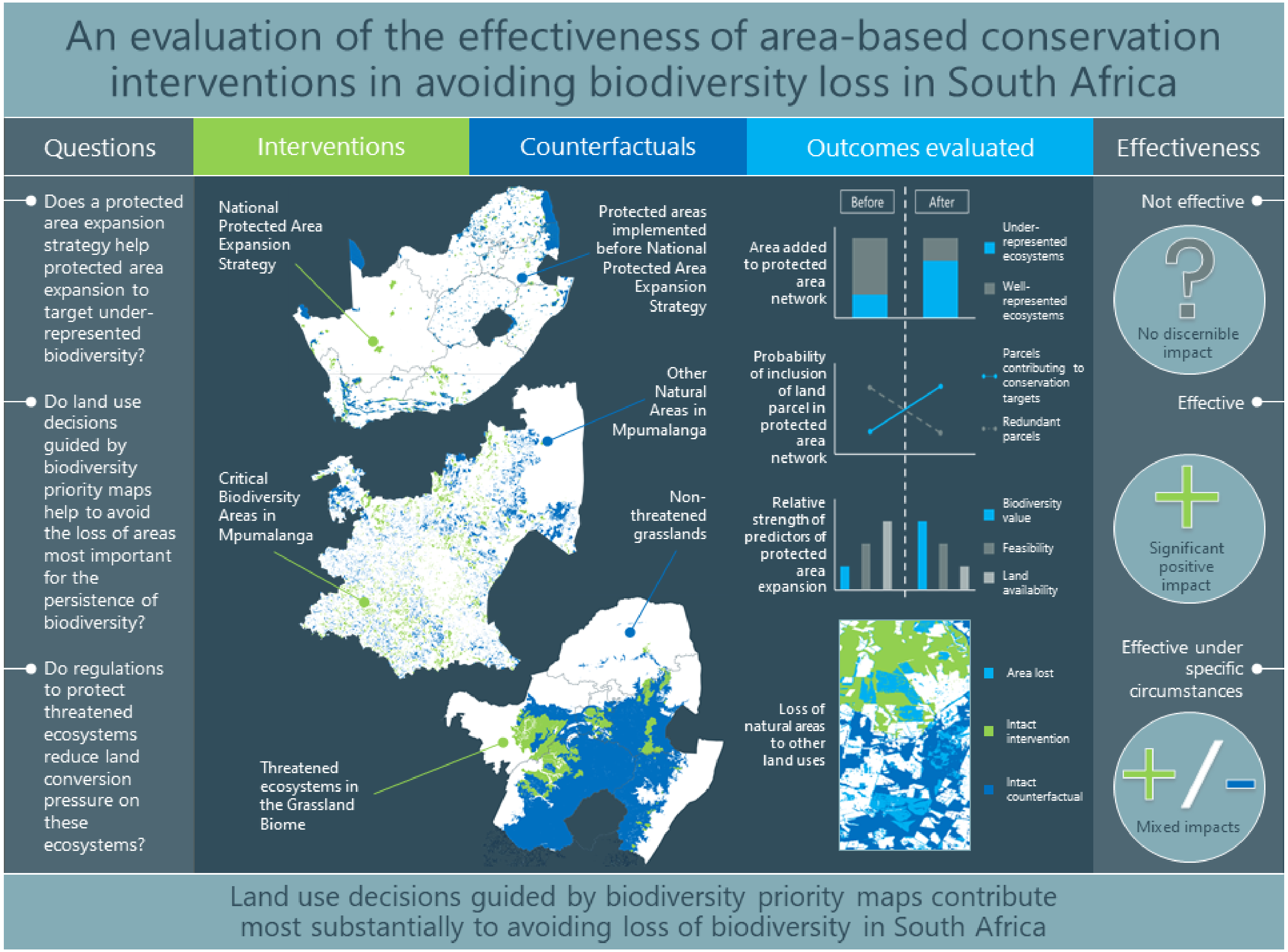
Graphical abstract



General abstract

Counterfactual impact evaluation studies form an important evidence base for the effectiveness of conservation projects, programs, and policies (collectively referred to as conservation interventions). In South Africa, counterfactual impact evaluation methods have rarely been applied to local conservation interventions, and therefore evidence for the effectiveness of key strategic national conservation approaches is lacking. This study evaluated three area-based interventions that together aim to avoid the loss of areas most important for the persistence of biodiversity in the terrestrial realm as evidence towards the effectiveness of South Africa's landscape approach to biodiversity conservation. The first intervention, South Africa's National Protected Area Expansion Strategy (NPAES), set ambitious targets to double the extent of South Africa's protected area network while ensuring that the expansion preferentially occurs in areas of under-represented biodiversity. The strategy was evaluated in terms of its effectiveness in guiding protected area expansion towards more equitable representation of South Africa's biodiversity through an assessment of changes in indicators of protected area expansion decision-making before and after the implementation of the strategy. The second intervention is the use of maps of biodiversity priorities to guide land use change decisions outside protected areas. Impact was evaluated as avoided loss of Critical Biodiversity Areas (CBAs), which need to remain in a natural condition to meet *in situ* conservation targets for species, ecosystems, and ecological processes. Avoided loss in CBAs was benchmarked against avoided loss in protected areas, to contextualize the effectiveness of land use planning as a conservation intervention. Lastly, the effectiveness of stricter land use regulations for threatened ecosystems to reduce land conversion pressure on these ecosystems was evaluated. Key findings were that protected areas are highly effective conservation interventions where they can be implemented, but their capacity for conservation impact is limited by severe constraints on strategic expansion. On the other hand, land use decisions that consider biodiversity priorities contribute substantially towards avoiding the loss of natural areas most important for the persistence of biodiversity across landscapes. They can be particularly effective in areas where it may not be feasible to implement protected areas. Protected areas and maps of biodiversity priorities therefore are important complementary interventions in a landscape approach to biodiversity conservation. Threatened ecosystems regulations did not have a significant impact on land conversion in threatened ecosystems, but there was large heterogeneity in effect estimates between ecosystems. This indicated that there are strong ecosystem-specific circumstances that determine whether the regulations have an impact or not. Meta-analysis indicated that the regulations successfully reduced land conversion pressure when ecosystems were mapped at a fine scale, they were included in multi-sector land use planning, and there were efforts to mainstream awareness of biodiversity priorities into economic sectors. Overall, this study found evidence that South Africa's landscape approach to biodiversity conservation is mostly working as intended in the regions that were studied, but that there is a need for further policy development, particularly if South Africa wants to achieve the goals of the Convention on Biological Diversity.

Keywords: environmental policy, counterfactual, National Protected Area Expansion Strategy, systematic conservation planning, Critical Biodiversity Areas, threatened ecosystems

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List of abbreviations

ATT	Average treatment effect in the treated
BA	Before-after study design
BACI	Before-after control-intervention study design
BMP	Biodiversity Management Plan
BRT	Boosted regression tree
CBA	Critical Biodiversity Area
CBD	Convention on Biological Diversity
CI	Control-intervention study design
CRSE	Cluster robust standard error
DEA	Department of Environmental Affairs*
DEAT	Department of Environmental Affairs and Tourism*
DFFE	Department of Forestry, Fisheries, and the Environment*
eCDF	Empirical cumulative density function
EIA	Environmental impact assessment
ESA	Ecological Support Area
GLM	Generalized linear model
IDP	Integrated Development Plan
ITS	Interrupted time series

IUCN	International Union for the Conservation of Nature
MBCP	Mpumalanga Biodiversity Conservation Plan
MBSP	Mpumalanga Biodiversity Sector Plan
NEMBA	National Environmental Management: Biodiversity Act
NPAES	National Protected Area Expansion Strategy
ONA	Other Natural Area
PA	Protected Area
PES	Payments for ecosystem services
RLE	Red List of Ecosystems
SANBI	South African National Biodiversity Institute
SAPAD	South Africa Protected Areas Database
SCP	Systematic conservation planning
SDF	Spatial Development Framework
SMD	Standardized mean difference
SMOTE	Synthetic minority over-sampling technique

* South Africa's environment ministry has had several name changes. It was DEAT from 1994-2009, DEA from 2009-2019, and is currently the DFFE.

1. Introduction

1.1. The evaluation of the effectiveness of conservation interventions

Biodiversity loss is a global concern as nature's capacity to maintain essential functions for life on earth is increasingly being eroded (Dasgupta 2021). Conservation actions are implemented with the belief that they will ensure the persistence of biodiversity (Pullin and Knight 2001, Pressey et al. 2017, Sutherland et al. 2004). Yet despite billions invested in conservation actions (Perry and Karousakis 2020) there are no signs of slowing or reversing biodiversity loss on the global scale (Díaz et al. 2019). One reason for the lack of success is insufficient resources allocated towards biodiversity conservation (Barbier 2022, Coad et al. 2019, Maxwell et al. 2020, McCarthy et al. 2012). Another is that many conservation actions are implemented without sufficient evidence that they are effective in avoiding biodiversity loss (Ferraro and Pattanayak 2006, Pullin and Knight 2001). This can result in scarce conservation resources being spent on actions that are ineffective, insufficient, or even counterproductive (Pressey et al. 2017).

Conservation actions are considered effective when they make a positive difference to the state or condition of elements of biodiversity such as ecosystems or species, through for example the reduction or elimination of threatening processes (Keene and Pullin 2011, Stem et al. 2005). In the evaluation of conservation projects, programs, and policies (hereafter referred to as conservation interventions) it is often assumed that monitoring the status of their target biodiversity elements is enough to detect effectiveness (Adams et al. 2019a, Mascia et al. 2014, Stem et al. 2005). However, upon closer scrutiny, these assumptions are often found to be invalid because they fail to causally link actions to outcomes (Ferraro 2009, Pressey et al. 2017). For example, protected areas are established primarily to prevent the loss of natural areas to other land uses and are judged as effective when natural areas within their boundaries remain intact (for example Pfeifer et al. 2012). Considering that protected areas are most often sited in areas where land conversion pressure is the lowest (Joppa and Pfaff 2009), should a protected area still be deemed an effective intervention if the area it protects would have remained intact even if it was not implemented?

The key element in detecting the effectiveness of conservation interventions is therefore to verify whether it makes a difference in a desired outcome (Pressey et al. 2017, Pressey et al. 2021). This can be determined by comparing the outcome between a scenario where the intervention is implemented to one where it is not implemented. The scenario representing the absence of the intervention is termed the counterfactual (Ferraro 2009). The difference in outcome is referred to as the intervention's impact (Baylis et al. 2016), and the size of the impact is called the effect size (Nakagawa and Cuthill 2007, Figure 1).

When impact is estimated, the most critical factor determining the accuracy of the estimation is a study design that systematically eliminates alternative or confounding explanations for the observed differences in outcome (Kimmel et al. 2021, Larsen et al. 2019). The study design that most effectively eliminates confounding is the randomized control trial, where an intervention is randomly assigned, and the outcome is compared with another randomly selected group of study units that did not receive the intervention. One of the proposed reasons for the lack of rigorous impact evaluation in conservation science is that practical and ethical considerations often preclude the implementation of such study designs (Baylis et al. 2016, Margoluis et al. 2009, Pynegar et al. 2021). For example, stakeholders may object to randomly withholding conservation interventions from species or ecosystems to create a control sample if it means that they could become increasingly imperilled as a consequence.

Conservation science is, however, not the only intervention-focused discipline facing this dilemma. Great advances have been made in impact evaluation in public policy, education, medical and social sciences through the adoption of quasi-experimental methods that construct counterfactuals from observational data (Imbens and Rubin 2015). Comparative studies have shown that when carefully designed, these quasi-experimental methods are able to approximate impact estimates derived from randomized control trials (Ferraro and Miranda 2014).

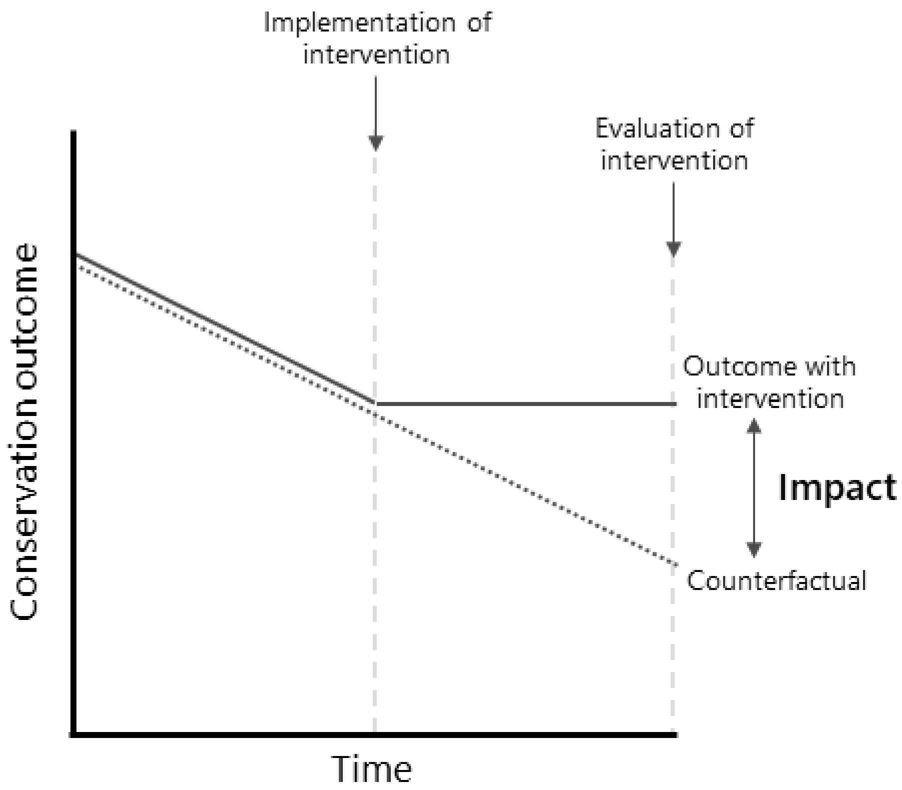

FIGURE 1 Terminology used in conservation impact evaluations. The intervention is implemented to alter a specific outcome, which is measured over time in a scenario where the intervention is implemented, as well as one where it is not implemented, known as the counterfactual. The impact of the conservation intervention is the difference in outcome between where it was implemented and the counterfactual.

Many argue that the lack of mainstream appreciation of counterfactual thinking in conservation science has impeded progress towards establishing credible evidence for the effectiveness of conservation interventions (Adams et al. 2019a, Ferraro 2009, Josefsson et al. 2020, Pressey et al. 2017). Counterfactual thinking encourages a more critical assessment of systemic biases in where and how interventions are applied, as well as a more thorough consideration of the various causal mechanisms that could potentially explain a particular outcome (Adams et al. 2019a). It supports the development of richer conceptual models of the complex social-ecological systems within which conservation interventions are implemented (Ferraro et al. 2019), which leads to study designs that more effectively eliminate alternative explanations for observed outcomes. Counterfactual studies with well-articulated conceptual models and theories of change do not just provide credible evidence of whether an intervention is effective or not, but they also help to explain why an intervention is effective or not.

For example, counterfactual impact evaluations of endangered species legislation in the USA and Australia found that in general, listing species on endangered species lists and the development of recovery plans does not lead to better population recovery outcomes than if they had not been listed (Bottrill et al. 2011, Ferraro et al. 2007). In the USA it was found that only if listing triggered substantial funding for recovery operations did species' populations recover significantly better than a counterfactual (Ferraro et al. 2007). Therefore, endangered species lists are not effective interventions to prevent extinctions. It is only when such lists serve as a catalyst for guiding spending on conservation actions that it is worth the effort of compiling them.

The global evidence base for the effectiveness of conservation interventions is growing, mainly around the biodiversity impacts of protected areas. Terrestrial protected areas lead to modest reductions in deforestation (Geldmann et al. 2013), but there is mixed evidence for effectiveness in maintaining and recovering bird and mammal populations (Geldmann et al. 2013, Terraube et al. 2020, Wauchope et al. 2022). In the marine environment there is evidence that protected areas have positive impacts on the populations of fish and invertebrate species (Sciberras et al. 2013). The evidence for the effectiveness of incentive-based interventions such as payments for ecosystem services (PES), Reducing Emissions from Deforestation and Forest

Degradation (REDD+) and provision of alternative livelihoods is limited (Börner et al. 2017, Pullin 2015, Roe et al. 2015, West et al. 2020). Some systematic reviews have also pointed out complete absences of evidence for the effectiveness of widely implemented policies and practices such as landscape approaches to conservation and development (Sayer et al. 2016) and systematic conservation planning (McIntosh et al. 2018).

In South Africa, counterfactual impact evaluation methods have rarely been applied to local conservation policy and practices. An evaluation of the Working for Water program, a government-funded initiative to clear invasive plant species, found that invasive species cover would have been 49% higher, had clearing not been implemented. It also found that the program could have been more effective if clearing efforts were primarily focused on large areas of intact natural vegetation, rather than production landscapes (McConnachie et al. 2015). Ament and Cumming (2016) evaluated avoided loss of natural areas in 19 national parks in South Africa between 2000 and 2009 and found mixed effectiveness between parks, with most parks contributing to avoiding loss of natural areas, but also that some parks had no impact compared to matched counterfactuals. Shumba et al. (2020) found no difference in persistence of natural areas and biodiversity intactness within formally declared private conservation areas compared with informal private conservation areas in South Africa and concluded that informal private conservation areas make a significant contribution to biodiversity conservation in South Africa.

With only these three studies on local interventions completed, large gaps remain in the evidence base for the effectiveness of South Africa's environmental policies and regulations, particularly those promulgated under the National Environmental Management: Biodiversity Act (NEMBA, Act 10 of 2004) This act is considered the cornerstone of South Africa's integrated and co-ordinated approach to biodiversity conservation (Cadman et al. 2010).

1.2. South Africa's approach to biodiversity conservation

South Africa is one of 17 countries rated as megadiverse, owing to very high levels of species richness and endemism (National Biodiversity Assessment Unit 2019). Together, these countries contain more than two thirds of the world's biodiversity (Mittermeier et al. 1997). South Africa is also one of only a few countries with three distinct terrestrial biodiversity hotspots, which are areas where high levels of species richness and endemism coincide with severe threats to biodiversity (Mittermeier et al. 2005).

South Africa is a party to the Convention on Biological Diversity (CBD), an international treaty whereby countries commit to safeguarding their biodiversity (CBD 2022). Given such high levels of diversity coupled with severe pressures on the richest regions, ensuring the persistence of a representative sample of all biodiversity elements is a significant challenge. When NEMBA was implemented, South Africa's biodiversity was considered still relatively intact (Biggs et al. 2006), and the most recent comprehensive assessment of the state of the country's biodiversity indicate that a minority of biodiversity elements require restoration or population recovery to meet their conservation targets, particularly in the terrestrial realm (Skowno et al. 2019). A pro-active conservation approach aimed at securing *in situ* persistence of biodiversity is therefore considered a more efficient conservation strategy (Drechsler et al. 2011, Walls 2018). However, in biodiverse regions, traditional approaches such as the development and implementation of action plans tailored to the individual conservation needs of specific elements of biodiversity are not practically feasible. South Africa has therefore opted to take an integrated landscape approach to biodiversity conservation (Arts et al. 2017), where the most important areas that are required for the *in situ* persistence of species, ecosystems and ecological processes are systematically identified, and policies and legislation are put in place to avoid the loss and degradation of these areas (Cadman et al. 2010).

The process of identifying the most important areas required for the persistence of biodiversity (hereafter biodiversity priority areas) is underpinned by comprehensive, systematic mapping and assessment of the status of species, ecosystems and ecological processes across terrestrial, marine and freshwater realms. It is driven by national assessments of the state of South Africa’s biodiversity, with three assessments completed to date (Driver et al. 2005, Driver et al. 2011, Skowno et al. 2019). A target is set for each individual biodiversity element, which is the minimum area (for ecosystems and ecological processes) or population (for species) that needs to be maintained to ensure the persistence of that feature (SANBI 2016). Systematic conservation planning (SCP) is then used to identify the areas where these targets can be met most efficiently.

SCP has several benefits over other methods that can be used to identify biodiversity priority areas (Watson et al. 2011): it allows for the development of spatially efficient priority maps while maintaining connectivity across natural and semi-natural areas to support larger-scale ecological processes. Where there are several options within the planning space to meet targets, it preferentially selects areas where potential conflict with the needs of other economic sectors such as mining, agriculture and infrastructure development is least severe (Naidoo et al. 2006). These features of the SCP process are extremely important in a developing country with a rapidly growing population where there is a need to balance the demand for land, resources, and economic growth with biodiversity conservation (Cadman et al. 2010).

SCP was first developed as a method to design efficient and ecologically representative protected area networks (Margules and Pressey 2000). After it was found that protected area networks designed using SCP were seldom implemented as planned, the concept was expanded to include stakeholder engagement in the development of maps prioritizing conservation actions, and the development of tools and strategic mechanisms to support the implementation of conservation plans (Pressey and Bottrill 2008). South African conservation scientists began experimenting with reserve selection algorithms in the 1990s (Botts et al. 2019), and since the 2000s South Africa has been at the forefront of the development of innovative, practical mechanisms to ensure the implementation of conservation plans developed to guide conservation actions across landscapes and seascapes (Balmford 2003, Botts et al. 2019, Pierce et al. 2005). One of the major achievements of these practical advances was the integration of maps of biodiversity priorities in land use decision-making processes legislated through NEMBA (Botts et al. 2019).

South Africa has a relatively limited protected area network, falling far short of protected area targets set by the CBD, and it does not yet represent South Africa's biodiversity equitably (DEAT and SANBI 2008). There are many opportunities for protected area expansion in remaining intact natural areas that could ensure better representation of South Africa's biodiversity, and South Africa is implementing a long-term strategy to expand the national protected area network to under-represented areas (DEAT and SANBI 2008). However, due to the intensifying pressures on natural areas (Skowno et al. 2019, Skowno et al. 2021) and the high cost of protected area implementation and management (DEA 2016, Cumming et al. 2017), it has been recognized that protected areas alone cannot be relied upon to prevent biodiversity loss (Cadman et al. 2010).

One of the central purposes of the National Environmental Management: Biodiversity Act (NEMBA) is to establish policies and processes to avoid the loss of biodiversity priority areas outside protected areas. Loss is regulated in two ways: pro-actively, through development planning and zoning, and reactively, through environmental authorization. NEMBA sets out guidelines and standards for the development of maps of biodiversity priorities and lists of threatened and protected biodiversity elements (species and ecosystems), stakeholder participation in the development process, and reviewing of outputs. Once formally gazetted, these outputs are required to be considered in spatial development planning and multi-sector land use zoning. Zoning decisions within biodiversity priority areas are required to be aligned with compatible land use categories for the persistence of biodiversity (Cadman et al. 2010, Driver et al. 2017).

A set of listed activities, typically actions that would impact intact natural areas or threatened and protected biodiversity elements, require environmental authorization before they may proceed (DEAT 2010). Environmental authorization is granted by independent competent authorities and is guided by Environmental Impact Assessments (EIAs). Environmental authorization decisions are required to consider gazetted land use guidelines that are compatible with the persistence of biodiversity in priority areas and are guided by the mitigation hierarchy (Phalan et al. 2018). Specifically, in instances where there are no alternative options for meeting the persistence targets of biodiversity elements if the activity would result in the decline of those elements, environmental authorization should not be granted (Brownlie et al. 2017).

1.3. Study aim and thesis outline

South Africa's integrated landscape approach to biodiversity conservation combines legislation and policies to encourage protected area expansion in areas with under-represented biodiversity as well as avoiding the loss of biodiversity priority areas that are not formally protected, but it is not yet known whether this strategy is effective at ensuring the persistence of South Africa's biodiversity. The aim of the study is therefore to assess the impact of a set of area-based policy interventions in South Africa that together aim to secure the persistence of natural areas needed to meet conservation targets for biodiversity elements. Impact is assessed for each intervention against a counterfactual representing no intervention. The study focuses on the terrestrial realm because mapping of priority areas and implementation of interventions to avoid loss has had a significant head start over other realms (Botts et al. 2019).

The consideration of biodiversity priorities in land use planning and decision-making is one of the core functions of NEMBA, but the act also regulates sustainable use of indigenous biological resources, management of invasive species, and the equitable sharing of the benefits of bioprospecting. The effectiveness of these regulations is also in need of study but will not be addressed in this study.

1.3.1. Constructing counterfactuals for South African area-based conservation interventions

Constructing counterfactuals for South African area-based conservation interventions is not easy and there are many limitations preventing the application of the most robust study designs. The legislation and policies evaluated in this study have been in place since the early 2000s, but implementation has often lagged and where it is the responsibility of provincial conservation authorities, it has occurred piecemeal (Botts et al. 2019, Botts et al. 2020). In addition, due to lags between decision-making, such as where to establish protected areas or the granting of environmental authorizations, and outcomes, such as protected area proclamation or land use change, the impact of interventions can also take some time to become apparent (Baylis et al. 2016). These challenges were addressed as far as possible by focusing evaluations where implementations have been in place for the longest time, and where data allows for lags between decision-making and outcomes to be considered.

The interventions are all evaluated retrospectively, which means that impact can be assessed only through quasi-experimental study designs constructed from observational data. In quasi-experimental designs there are various ways of selecting or defining a counterfactual, with different levels of robustness to hidden biases and assumptions (Adams et al. 2019a, Jones and Lewis 2015, Wauchope et al. 2021). Impact can be estimated either by comparing outcomes before and after an intervention is implemented (known as before-after designs), or by comparing simultaneous outcomes between study units that did and did not receive the intervention (known as control-intervention designs). Both these designs are susceptible to confounding and hidden biases but can be made more robust by introducing additional mechanisms to minimize the potential effects of confounding. Before-after (BA) comparisons using time series of repeated observations of an outcome before and after an intervention can more credibly link changes in outcome to an intervention if breaks or changes in direction of outcome trends can be shown to coincide with the timing of an intervention (Ewusie et al. 2020). Control-intervention (CI) designs can be strengthened by eliminating systemic differences between control and intervention units that are the result of the non-random application of interventions through statistical matching (Schleicher et al. 2020). The most robust study designs combine all these elements into panel or time series before-after control-intervention (BACI) designs (Jones and Lewis 2015, Wauchope et al. 2021). Data limitations, particularly very sparse and irregular covariate and outcome time series data prevented the application of BACI designs in this study, but as far as possible, BA and CI study designs were strengthened with measures to control for potential biases.

As is the case around the world, the most severe threat to terrestrial biodiversity in South Africa is the loss of natural areas to other land uses such as agriculture, mining, and infrastructure development (Skowno et al. 2019). The interventions related to land use planning and decision-making examined in this study are primarily designed to guide destructive land use changes away from areas required to meet conservation targets. Therefore, impact was assessed as avoided loss of natural areas. Avoided loss does not necessarily translate to effective outcomes for specific elements of biodiversity (Vincent 2016), but since no other impact evaluations have been conducted yet for the interventions evaluated here, it is important to first establish whether this baseline objective of avoided loss has been achieved. A challenge with counterfactual evaluations of avoided loss is obtaining suitable land cover data that accurately tracks conversion of natural areas to other land uses (Geldmann et al. 2019). The South African government commissioned a series of land cover datasets that are tailored to South African ecosystems and locally relevant land cover classes (DFFE 2021), but land cover data are available only for 1990, 2014, 2018 and 2020. This means that for most interventions, only a single data point (1990) is available pre-intervention, which is one of the main reasons why BACI designs could not be implemented.

1.3.2. Thesis outline

Each of the following three chapters evaluates a specific area-based intervention and is written as a standalone research article for publication in the peer-reviewed literature. One chapter is already published (Von Staden et al. 2022) – see Appendix I. Counterfactual impact evaluation methods are applied to each intervention, using the most robust study designs possible given the limitations of available data.

Chapter 2 evaluates the effectiveness of South Africa's National Protected Area Expansion Strategy (NPAES) in guiding protected area expansion towards more equitable representation of South Africa's biodiversity. It is a before-after study that assesses changes in indicators of protected area expansion decision-making as evidence for the effectiveness of the strategy. The study uses a conceptual model of protected area expansion to guide the use of statistical methods to control for confounding factors that may explain observed outcomes and assesses evidence from multiple indicators against a theory of change to strengthen conclusions about the effectiveness of the strategy in the absence of concurrent control observations.

Chapter 3 evaluates the effectiveness of the land use planning and decision-making process in avoiding the loss of biodiversity priority areas that are not formally protected in Mpumalanga Province. Mpumalanga was selected as a case study as it had one of the first biodiversity priority maps specifically developed to inform land use decision-making as required by NEMBA. It is a control-intervention study, with loss in biodiversity priority areas (intervention) compared to loss in natural areas not needed to meet biodiversity targets (control) over the same period. Differences in land conversion pressure between intervention and control observations are controlled through statistical matching. To contextualize the impact of this process, which is the first of its kind to be evaluated through counterfactual methods, it is benchmarked against avoided loss achieved by protected areas over the same time.

Chapter 4 evaluates the effectiveness of additional, stricter control measures over clearing of remaining natural areas in threatened ecosystems in reducing land conversion pressure on these ecosystems. It is also a control-intervention study and uses similar methods as Chapter 3. Avoided loss is estimated for each of the 34 threatened ecosystems in the Grassland Biome, South Africa's largest biome that has seen rapid intensification of land use change pressures over the last decade. Meta-analytic methods are used to gain an understanding of the drivers of differences in impact estimates among the ecosystems evaluated.

The thesis concludes with a synthesis chapter where the relative impact of the different interventions is compared, and their collective contribution towards evidence for the effectiveness of South Africa's integrated landscape-scale approach to biodiversity conservation is contextualized. The policy implications of the overall findings are discussed, and future directions for impact evaluations of South African environmental policies and legislation are outlined.

2. An evaluation of the effectiveness of a protected area expansion strategy in guiding protected area expansion towards conservation targets

Strategic protected area expansion has been proposed as a solution to the inequitable representation of species and ecosystems in existing protected area networks, but its effectiveness remains untested. South Africa implemented a National Protected Area Expansion Strategy (NPAES) in 2008, which now provides the opportunity to test the role that strategic decision-making played in guiding protected area expansion towards priority areas. This study evaluated three outcomes as evidence for strategic expansion – the share of annual protected area expansion that was aligned with the goals of the NPAES, the probability of protected area expansion in priority areas compared with non-priority areas, and changes in predictors of protected area expansion away from indicators of land availability towards indicators of biodiversity value. Protected area expansion post-implementation of the NPAES was generally aligned with the goals of the NPAES, but no evidence could be found that this result was due to changes in strategic decision-making around where to implement new protected areas. South Africa has a relatively small protected area network, which means that there are many opportunities to protect under-represented biodiversity, and therefore there is not yet a need for strong adjustments in protected area implementation practices. The study does however add evidence to the concern that in under-resourced conservation strategies, ambitious conservation targets can undermine the achievement of strategic biodiversity objectives. The pressure to expand protected areas rapidly to meet areal targets discourages strategic implementation in favour of opportunistic land acquisitions.

2.1. Introduction

Protected area expansion is a key conservation intervention to prevent species extinctions and the loss of ecosystems (Butchart et al. 2012, Geldmann et al. 2013). Global protected area expansion targets set by the Convention on Biological Diversity (CBD), an international treaty dedicated to the conservation and sustainable use of biodiversity, have been consistently met (UNEP-WCMC and IUCN 2021), but the achievement of these targets has not translated into a slowing or reversal of the global decline in biodiversity (Butchart et al. 2010, Pressey et al. 2015, Tittensor et al. 2014). This apparent conservation paradox is explained by the global protected area network not protecting species and ecosystems equitably (Barr et al. 2011, Brooks et al. 2004, Butchart et al. 2015, Hoekstra et al. 2005, Maxwell et al. 2020).

Strategic protected area expansion that targets gaps in the representation of species and ecosystems has been proposed as a solution to the existing shortcomings in the global protected area network (Kullberg et al. 2019, Venter et al. 2014, Visconti et al. 2015). These recommendations often assume that gaps in representation of species and ecosystems in protected area networks are due to a lack of awareness of where these gaps are, and that maps of priority areas for protected area expansion would guide protected area expansion towards better biodiversity outcomes. Analyses of the geographical characteristics of the global protected area network have however found significant and persistent spatial biases towards areas that are of low economic value (Joppa and Pfaff 2009, Pressey et al. 2015, Pressey et al. 2017). These studies suggest that availability of land for protected area expansion may be highly constrained by competing priorities such as economic growth.

Whether strategic protected area expansion can successfully overcome biodiversity protection shortfalls therefore depends on whether lack of awareness of biodiversity priorities or land availability are the most important drivers of the existing biases in protected area networks. Maps of biodiversity protection priorities are often compiled with the consideration of land value and landowner attitudes towards conservation (Carwardine et al. 2008, Guerrero et al. 2010, Knight et al. 2011), with the aim to avoid areas that are highly unlikely to be available for protected area expansion. It is therefore anticipated that better informed decision-making is all that is needed to overcome gaps in biodiversity protection. The reality is that conflicts between biodiversity priorities and other competing land uses cannot be avoided entirely (Liberati et al. 2019, Williams et al. 2003), and the socio-economic complexities of protected area expansion are often underestimated (Ban et al. 2013, Knight et al. 2008, Pressey et al. 2013).

The effectiveness of strategic guidance of protected area expansion as a solution to the biodiversity crisis has rarely been studied within real-world implementation of protected area expansion priority maps (McIntosh 2018). Carter et al. (2015) found that policies and practices within implementing agencies were the strongest predictors of where new protected areas were established following the development of a protected area expansion priority map for Wisconsin, USA. This resulted in areas with the highest practical feasibility for protected area establishment being added to the protected area network, regardless of their value for biodiversity and ecosystem services, land use pressures, or the socio-economic characteristics of the local population. Following the adoption of biodiversity target-based protected area expansion in Australia, Barr et al. (2016) found that recent protected area expansion was less biased towards areas of low economic value than historical trends. Progress in the protection of the most poorly represented bioregions was however slower than expected if decisions were guided solely by conservation targets, suggesting that substantial constraints on opportunities for protected area expansion remain. Global (Kuempel et al. 2016) and regional (Neugarten et al. 2020) analyses of recent protected area expansion found improvements in protection of under-represented ecoregions and Key Biodiversity Areas, but in both these studies, gains were less than what would be expected if strategic protected area expansion in known biodiversity priority areas was implemented. Similarly, Venter et al. (2018) found that if the more than 3 million km^2 added to the global protected area network between 2004 and 2014 had been strategically allocated, 30 times more threatened vertebrate species could have been protected.

A better understanding of the constraints on strategic protected area expansion is needed to gauge its effectiveness as a remedy to inequitable biodiversity protection. South Africa implemented a national protected area expansion strategy (NPAES) in 2008 (DEAT and SANBI 2008), following findings of strong biases in the representation of ecosystems in the national protected area network (Rouget et al. 2004). The aim of this study is to evaluate the effectiveness of this strategy by comparing the patterns and drivers of protected area expansion before and after its implementation, with a particular focus on what share of the changes post-implementation can be attributed to more strategic decision-making. The NPAES included a terrestrial and marine component, however only implementation within the terrestrial realm is evaluated here.

The NPAES produced a map of priority areas for protected area expansion but owing to the known unpredictability of land availability and the socio-economic complexities of protected area expansion, effectiveness cannot be evaluated against how much overlap there is between priority areas and protected area expansion post-2008. Instead, post-2008 protected area expansion is assessed in terms of how well it aligns with the goals of the NPAES, which is to increase the protection of under-represented ecosystems in the national protected area network (DEAT and SANBI 2008). An ecosystem is classified as under-represented when less than its conservation target is within the national protected area network. Once the conservation target is reached, the ecosystem is considered well-protected, and further protected area expansion in that ecosystem risks the creation of redundancy in the protected area network, at the expense of poorly protected biodiversity. At the time of development of the strategy, 349 out of 458 (76%) terrestrial ecosystems were under-represented.

Analyses were guided by a conceptual model of the process of protected area expansion (Figure 2A) and a theory of change (Figure 2B). The conceptual model was developed for this study based on factors known to influence the siting of protected areas, as documented in earlier studies (Barr et al. 2016, Carter et al. 2015, Guerrero et al. 2010, Joppa and Pfaff 2009, McDonald and Boucher 2011). These include variables related to land value and the socio-economic conditions within a region, which affect the availability of land for protected area expansion. Features that influence decisions on which available land parcels to protect include considerations of feasibility and cost of implementation, such as the intactness of natural areas, as well as the biodiversity value of the site. Measures of biodiversity value include both intrinsic value, such as species richness, as well as the contribution of the site towards meeting ecosystem conservation targets.

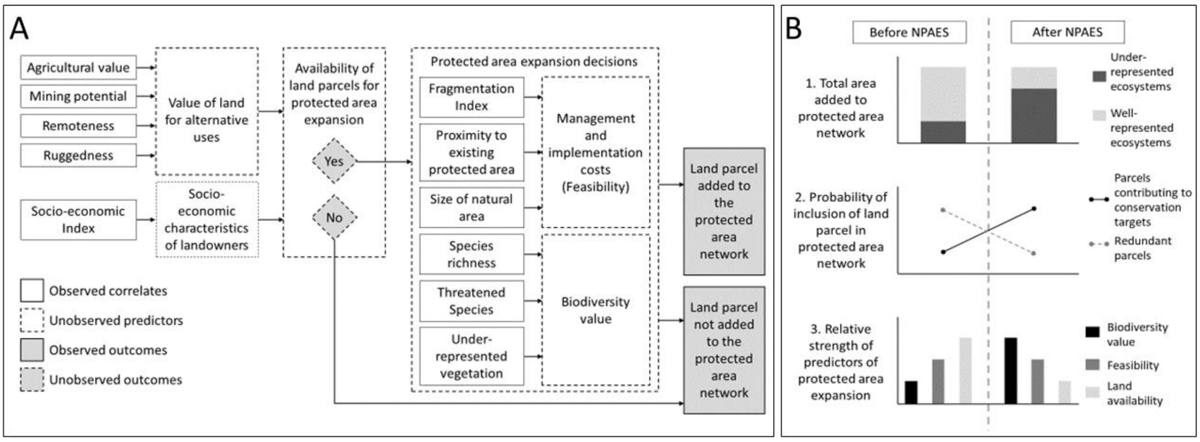

FIGURE 2 A. Conceptual model of protected area expansion. B. Theory of change for the effect of the National Protected Area Expansion Strategy (NPAES) on protected area expansion in South Africa.

The theory of change (Figure 2B) anticipates three outcomes if the NPAES is effective in guiding protected area expansion decisions towards more equitable representation of biodiversity. Firstly, if strategic protected area expansion is being implemented, it is expected that the share of area added to the protected area network that contributes to meeting ecosystem conservation targets should increase.

The second anticipated change is an increased probability that land parcels contributing to ecosystem conservation targets are added to the protected area network, relative to those that do not. The analysis is designed to isolate the effect of strategic decision-making in observed protected area expansion. It uses statistical matching methods to answer the question – if all other factors likely to influence the siting of new protected areas are equal, are land parcels that contribute to conservation targets more likely to be protected post-implementation of the strategy?

Lastly, the findings of the second analysis are further explored through changes in the relative strength of predictors of protected area expansion before and after the implementation of the NPAES. Following the findings of Barr et al. (2016), it is expected that if more strategic decision-making is all that is needed to overcome gaps in biodiversity protection, variables indicating biodiversity value would become the dominant predictors in models of recent protected area expansion, compared with historical patterns.

2.2. Methods

2.2.1. Protected areas

Protected Area data was obtained from the Department of Forestry, Fisheries, and the Environment (South Africa Protected Areas Database, SAPAD, version 2020Q4). This national database contains protected areas officially proclaimed in terms of the National Environmental Management: Protected Areas Act, Act 57 of 2003.

There are many overlapping features in this dataset, which represent the evolving designations of protected area types and management authorities over time. The analyses in this study were based on time series of protected area expansion, and therefore, the aim was to determine the earliest proclamation date for each land parcel within the protected area network. Therefore, later iterations of the same area were discarded when overlapping features were eliminated.

The database has the constraint that for protected areas that have expanded over time but remained the same type and under the same management authority, sections that were proclaimed at later dates are not differentiated. The dataset was therefore augmented with additional data from other sources, including relevant Government Gazettes, and national and provincial protected area management agencies, to obtain proclamation dates for individual land parcels within protected areas. Where proclamation dates could not be determined, or in instances where administrative problems meant that no official proclamation date existed, these protected areas were included in calculations of the extent of ecosystems that are protected but excluded from time series calculations of recent protected area expansion.

2.2.2. Time series analysis of changes in the share of annual protected area expansion contributing to conservation targets

Changes in the share of annual protected area expansion contributing to ecosystem conservation targets in response to the NPAES were analyzed using an interrupted time series (ITS) model. This method was selected because it allows for a causal link between the timing of an intervention and changes in an outcome to be inferred (Schober and Vetter 2021), and an effect size, which is the difference that the intervention made to the outcome, to be estimated (Wagner et al. 2002).

ITS models are constructed using repeated observations of an outcome variable at regular intervals over a period of time that includes a section before and after the implementation of the intervention. Linear regressions are fitted as separate segments on the time series before and after the intervention. Changes in the outcome in response to the intervention are inferred from changes in the intercept and slope between the two segments (Schober and Vetter 2021, Wagner et al. 2002). The effect size was estimated by comparing the difference in the predicted outcome of the fitted model to a counterfactual model where the intervention is assumed to have not taken place. To allow the effect size to be estimated as an absolute percentage point difference in outcome, the ITS was fitted as a linear probability model (Huang 2022, Uanhoro et al. 2021).

The SAPAD database contains protected areas proclaimed between 1903 and 2020, however, uncertainty around proclamation dates increases substantially with earlier proclamations. A time frame of 1980-2020 was therefore chosen for this analysis as a trade-off between sample size and uncertainty.

Protected area expansion does not necessarily occur only in intact natural areas (DEA 2016), but only natural areas contribute towards meeting ecosystem conservation targets. The outcome variable was therefore calculated as follows: South Africa's national vegetation map (version 2018, Mucina et al. 2018) was used as a representation of terrestrial ecosystems for the country. National land cover data sets (DFFE 2021) were used to eliminate areas that were no longer in natural condition from the vegetation data set. For each year in the time series, the proportion of the estimated historical extent of each ecosystem that was in natural condition and in protected areas was calculated and compared against its conservation target (Desmet and Cowling 2004). Ecosystems were then classified as under-represented when less than their conservation targets were protected, and well-protected when the target had been reached. The natural areas that were added to the protected area network by the following year were divided into the area that contributed to protection of under-represented ecosystems and the area that did not. The annual outcome variable was then calculated as the proportion of total protected area expansion in natural areas for that year that contributed towards increased protection for ecosystems that were under-represented at the time.

Time series data sets typically display autocorrelation, where observations closer in time are more similar, and non-stationarity, where there is a time-correlated change in the mean and variance (Bernal et al. 2017). Both these characteristics violate the assumption of independence of observations and therefore need to be accounted for in model specifications. The protected area expansion time series was tested for evidence of autocorrelation and non-stationarity, but neither was found.

A drawback of the ITS method is that it assumes an instantaneous response in the outcome following an intervention. This is however seldom the case in real-world scenarios, where there is often a lag between an intervention and apparent changes in the outcome. To solve this conundrum, Cruz et al. (2017) propose defining a change point in the outcome time series, through a maximum likelihood estimation. The break point between the regression segments is then defined not by the implementation date, but by the change point. Using this method, the maximum likelihood change point for the protected area expansion time series was estimated as 2009.

2.2.3. Probability of protected area expansion in areas contributing to ecosystem conservation targets

In this analysis, the probabilities of protected area expansion in areas that do, or do not contribute to ecosystem conservation targets, were compared. Differences in probabilities were compared over three decades between 1990 and 2020, which includes two decades before the implementation of the NPAES and one after. The selection of the time period was constrained by the availability of appropriate time series datasets for time-varying covariates. It includes a 10-15 year period preceding the widespread practical adoption of systematic conservation planning in land use decision-making in South Africa (Botts et al. 2019). The supporting tools enabling better protected area expansion prioritization and decision-making, such as ecosystem-specific conservation targets (Desmet and Cowling 2004), and maps of ecosystems that are under-represented in the protected area network became available from the mid-2000s (Reyers et al. 2007), while the NPAES was published in 2008 (DEAT and SANBI 2008), with an updated version released in 2016 (DEA 2016). It is therefore expected that there would be little to no discernment between protected area expansion in under-represented ecosystems compared to ecosystems that are well-protected in the 1990s, with possibly some evidence of more strategic protected area expansion in the 2000s. The strongest difference is anticipated to be evident in the 2010s, if there was general adoption of the goals of the NPAES among implementation agencies, and if strategic decision-making could overcome constraints such as land availability, feasibility, and cost of implementation.

The unit of analysis was land parcels known as farm portions. It was selected because it is the spatial unit at which land use decisions are made. At the start of each decade, all land parcels that were not within the protected area network were classified into one of two classes: land parcels that had at least one hectare of vegetation of an under-represented ecosystem were classified as contributing to conservation targets, while those that did not were classified as redundant. This threshold was chosen as a generous lower bound of what can be considered a contribution towards meeting the NPAES's objective of increasing protection of under-represented ecosystems. The outcome, which is whether or not a parcel was added to the protected area network, was assessed at the end of each decade.

To isolate the effect of strategic decision-making on the probability of protection, the contributions of other covariates that are likely to explain observed protected area expansion (Figure 2A) were controlled through statistical matching. This method has been used to estimate the causal impact of policies and interventions in observational studies in medical, social, political and conservation sciences where the implementation of randomized controlled trials is not possible (Austin 2011). Regression models using matched samples can approximate differences in outcomes obtained through randomized experiments more accurately than regression analyses on unmatched data (Ferraro and Miranda 2014).

Redundant parcels (the smaller of the two classes), were matched to parcels that contribute to conservation targets that were similar in terms of eleven variables predicting protected area expansion for each decade. The predictor variables included five variables related to land availability, three related to feasibility of implementation, and three indicating biodiversity value. The variables, their sources, and rationales for selection are summarized in Table 1. For variables mapped at a finer scale than land parcels, the average value was calculated for the land parcel, except for species richness variables, where the maximum value was used. For distance variables, the minimum distance between the target feature and the edge of the land parcel was used. Where variables varied over time, the nearest available data point in time to the start of the decade was used.

With matching, the aim is to balance covariates so that the only remaining difference between observations is the class that they belong to (Nguyen et al. 2017, Stuart 2010). A range of matching methods was tested using the R package MatchIt (Ho et al. 2011), and balance was assessed based on standardized mean difference (SMD), variance ratio and empirical cumulative density functions (eCDF), with SMDs <0.1, variance ratios close to 1 and eCDFs close to zero indicating good balance. Best balance was achieved through one-to-one nearest neighbour matching on Mahalanobis distance with replacement. To control for implementation agencies, which are typically provincial conservation departments, matching was restricted to parcels within the same province.

The difference in probability of protection for the two classes of land parcels was then estimated using the matched samples for each decade, by fitting binomial generalized linear models (GLMs) with the land parcel class as a binary predictor and whether the parcel was added to the protected network during the decade as a binary outcome.

The difference was quantified as the marginal odds ratio by using the logit link function (Greifer 2022). Therefore, the odds that a land parcel that contributes to meeting ecosystem conservation targets is added to the protected area network is expressed as a ratio of the odds that a similar redundant parcel is added to the protected area network during the same period. Odds ratios close to one indicate no or marginal differences, whereas values greater than one indicate that parcels contributing to conservation targets are more likely to be added to the protected area network. Values less than one indicate that parcels contributing to conservation targets are less likely to be added to the protected area network than redundant parcels.

Coefficient errors were estimated using cluster robust standard errors (CRSE) to account for sampling with replacement as well as pair membership of observations. It has been shown that CRSEs provide more accurate error estimates for these data structures typically encountered in matched samples than regular methods (Austin 2009).

2.2.4. Variables explaining protected area expansion

Following the matching analysis, the main drivers of protected area expansion in each decade between 1990 and 2020 were explored through boosted regression trees (BRTs), which is an ensemble classification method based on sequential iterations of decision trees, where each iteration successively fine-tunes the classification process by learning from the errors of the previous iteration. The relationship between the predictor variables and observed patterns of protected area expansion is likely to be complex, and possibly non-linear. In these contexts, BRT classification offer an advantage over multivariate regression techniques as it can model such complexities more accurately (Carvalho et al. 2018, De'ath and Fabricius 2000).

This analysis used the same datasets as for the matching analysis (Table 1), with the classification of land parcels according to their contribution to meeting ecosystem conservation targets included as a binary predictor variable. Models were fitted using the R package caret (Kuhn 2008), utilizing the extreme gradient boosting (xgboost) algorithm (Chen and Guestrin 2016).

To control for residual spatial autocorrelation, protected area expansion in each decade was modelled in two phases, following the method of Crase et al. (2012). In the first phase, the model was trained on the full dataset using repeated cross-validation, with 10 folds in each of 10 repeats. This allowed for the extraction of residuals of the fitted model for each of the observations, which was used to

TABLE 1 Variables that are likely to influence the siting of protected areas in South Africa used in matching of land parcels and boosted regression tree models. For time-varying covariates the data set nearest to the start of each decade was used. A limitation was that time series data sets such as land cover and national census data are very sparse, and therefore the same data sets had to be used for more than one decade. Time-invariant covariates are indicated by TIC.

				Nearest available data point for decade		
Variable group	**Variable**	**Definition and source**	**Rationale**	**1990s**	**2000s**	**2010s**
Land availability	Agricultural potential	The suitability of a land parcel for crop cultivation and rangelands. Agricultural potential is modelled in 15 classes, where 1 is lowest suitability and 15 is highest. Source: Agricultural Research Council – Institute for Soil, Climate and Water	Protected area expansion is more likely in areas of lowest suitability for agriculture (Joppa and Pfaff 2009).		TIC	
	Mining potential	Mining potential is quantified in four categories, based on criteria related to the size of known mineral deposits and their economic importance (Rouget et al. 2004). Source: National Spatial Biodiversity Assessment 2004 available at bgis.sanbi.org/nsba	Mining is of major importance to the South African economy. Environmental legislation forbids mining in protected areas, and therefore protected area proclamation in areas of high mining potential is unlikely		TIC	
	Remoteness	Distance in kilometres from nearest built-up area as mapped in national land cover data sets. Source: Department of Forestry, Fisheries, and the Environment, available at https://egis.environment.gov.za/sa_national_land_cover_datasets	Protected area expansion is more likely in remote areas (Joppa and Pfaff 2009).	1990	1990	2014
	Ruggedness	An index of terrain ruggedness was calculated from a digital elevation model following the method of Riley et al. (1999). Higher values indicate increasing ruggedness.	Protected area expansion is more likely in mountainous areas, where land development costs for other economic activities are likely to be higher (Joppa and Pfaff 2009).		TIC	

		Source: 90m resolution Shuttle Radar Topography Mission (SRTM) Digital Elevation Model.				
	Socio-economic index	An aggregated multidimensional poverty index by municipality. Higher values indicate a larger proportion of the population living in severe poverty. Source: Statistics South Africa (2014).	Wealthier areas are likely to have more available resources for conservation actions (McDonald and Boucher 2011).	2001	2001	2011
Feasibility of implementation	Area in natural condition	The area in km^2 of the land parcel that is in natural condition according to national land cover data sets. Source: As for remoteness.	Protected area expansion generally targets large, intact natural areas. Expansion in such areas is less costly to implement than in degraded or non-natural areas, which requires restoration and reintroduction of biodiversity features.	1990	1990	2014
	Fragmentation index	An index indicating the fragmentation of natural areas within a land parcel and its immediate neighbours. The index was calculated from land cover data using the Patch Cohesion Index method (Schumaker 1996). Lower index values indicate small, isolated remnants, with larger values indicating large, continuous natural areas.	Management of small, isolated remnants may be more costly due to edge effects. The long-term viability of biodiversity features present in small remnants may be more difficult to maintain.	1990	1990	2014
	Proximity to existing protected area	Distance in km from the edge of the land parcel to the edge of the nearest existing protected area. Source: protected area time series data compiled for this study.	Protected areas in South Africa are strongly clustered, and protected area expansion often proceeds through enlargement of existing protected areas. It is less costly to expand an existing protected area than to establish a new one.	1990	2000	2010
Biodiversity value	Species richness	A combination of species richness for eight taxonomic groups modelled at 1 km^2 resolution.	Areas of high species richness may have higher perceived biodiversity		This dataset represents an estimated historic species richness – recent declines in	

		Source: National Biodiversity Assessment 2018 (http://biodiversityadvisor.sanbi.org/planning-and-assessment/national-biodiversity-assessment-nba-2018/)	value than areas of lower species richness.	species richness owing to habitat loss and fragmentation was not possible to model.		
	Threatened animal richness	A combination of threatened species richness maps for seven animal groups modelled at 1 km^2 resolution. Source: as for species richness.	Non-strategic protected area expansion is more likely to be negatively correlated with threatened species richness, as threatened species tend to be concentrated in areas with higher land use pressures (Hoekstra et al. 2005).		2018	
	Threatened plant richness	Plant species richness modelled at 1 km^2 resolution. Source: as for species richness.				
	Contribution to meeting ecosystem conservation target	A binary value indicating whether a land parcel contains at least 1 ha of an ecosystem that is under-represented in the protected area network at the time. Source: protected area time series data compiled for this study.	This variable is most directly related to the goals of the NPAES and indicates changes in decision-making towards strategic protected area expansion. It is used as the classification variable in the matching analysis, and a predictor in models of protected area expansion.	1990	2000	2010

calculate the first-order neighbourhood autocovariate, which was then included as a covariate in the second phase of modelling. In the second phase, models were trained on a random subset of 70% of the observations, and predictive performance was tested on the remaining 30% of observations.

The binary outcome showed extreme class imbalance for all three datasets, with less than 1% of the observations belonging to the class indicating that the parcel was added to the protected area network (outcome = 1). Machine learning classifiers such as BRTs tend to perform poorly on such datasets, and therefore the minority class in each of the training samples in both modelling phases was amplified using the synthetic minority over-sampling technique (SMOTE, Chawla et al. 2002).

The xgboost hyper-parameters learning rate (eta), tree complexity (max_depth), number of iterations (nrounds), minimum loss reduction (gamma), minimum observation weights for terminal nodes (min_child_weight), subsample of observations per iteration (subsample) and subsample of predictors per iteration (colsample_bytree) were tuned for each model using the caret function adaptive resampling (Kuhn 2014). The xgboost algorithm has some stochastic components, such as random subsamples of observations and variables included with each individual tree fitting iteration, which means that not exactly the same model is fitted each time it is run, even if the training dataset and hyper-parameter settings are held constant (Elith et al. 2008). This results in some variability in the estimation of predictive performance as well as variable importance.

To quantify this variability, the final model for each decade was run 50 times on the training dataset and predictive performance was tested on the remaining data. Model performance was quantified using balanced accuracy and Cohen’s Kappa, which are both regarded as better metrics when the outcome class is severely imbalanced. Balanced accuracy is the arithmetic mean of sensitivity (proportion of positive observations correctly classified) and specificity (proportion of negative observations correctly classified), thereby giving equal weight to the accuracy of predictions for both classes. Cohen's Kappa assesses predictive performance against a random null model with the same class distribution as the test dataset (Warrens 2015).

For each run, variable importance was scored using the model-specific method in the vip R package (Greenwell et al. 2020). Variables were then ranked for each decade using their average scores across 50 runs. To gain a better understanding of how the value of individual variables relate to predicted outcomes, partial dependence profiles were constructed for each variable in each of the three models, using the R package pdp (Greenwell 2017). Partial dependence functions depict the effect of a predictor variable on the outcome after accounting for the average effects of all the other predictors (Elith et al. 2008).

2.3. Results

2.3.1. General trends in protected area expansion before and after the implementation of the NPAES

When the NPAES was developed, a relatively small proportion (6.5%) of South Africa's terrestrial area was formally protected (Figure 3). The strategy set ambitious targets to add 108 000 km^2 to the protected area estate, to increase the proportion of land protected to 12% by 2028 (DEAT and SANBI 2008). To meet this target, an average of 9000 km^2 need to be added to the protected area network per year. Between 1980 and 2008, an average of 1444 (±238) km^2 was added to the network annually. Achieving the goals of the NPAES therefore required implementation agencies to rapidly and significantly upscale protected area expansion efforts.

A review of the first phase of implementation (2008-2014) found that only 18% of the five-year target was achieved, with the majority (68%) of the expansion from conservation contracts with private and communal landowners, while a much smaller share was due to formal protection of state-owned land (19%) and land donations and purchases (13%, DEA 2016). There was an increase in the average area added to the protected area network per year (1755 ±155 km^2) between 2009 and 2020, but not enough to approach the long-term goals of the NPAES.

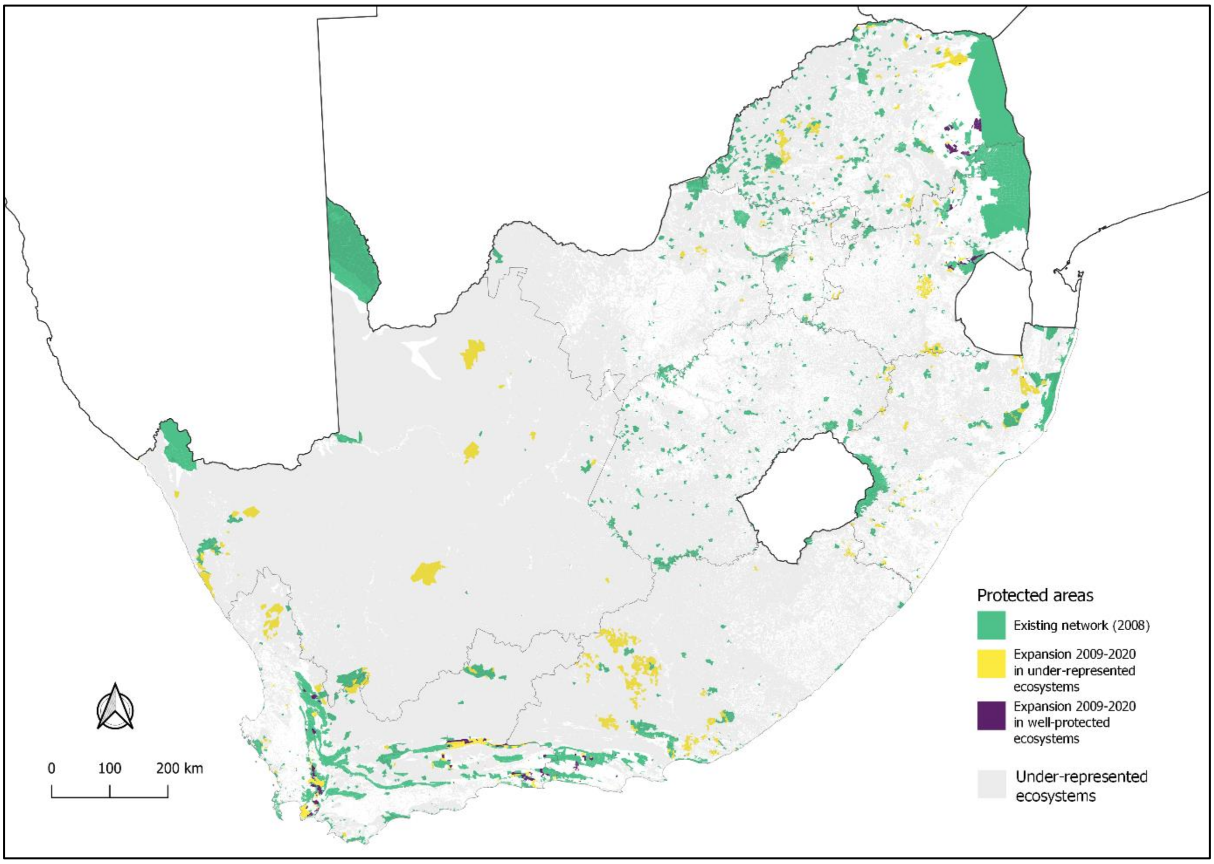

FIGURE 3 Protected area expansion in South Africa following the adoption of the NPAES showing expansion aligned with the goals of the NPAES compared with expansion in already well-protected ecosystems.

2.3.2. Time series analysis

Before 2008, an average of 84.6 (±3.1) % of annual protected area expansion contributed towards protection of under-represented ecosystems. Between 2009 and 2020 the average was 92.4 (±2.0) %. Protected area expansion that did not contribute to improved protection of under-represented ecosystems occurred mostly through enlargement of existing protected areas (Figure 3). The ITS analysis found no significant trends in the share of protected area expansion contributing to ecosystem targets either before or after the NPAES (Table 2, Figure 4). A relatively large level change of 15% was detected relative to a counterfactual of no intervention (Figure 4), but due to large inter-annual variability in the outcome, the effect was uncertain (95% CI -1.4 – 31.7%) and not statistically significant (p = 0.11, Table 2).

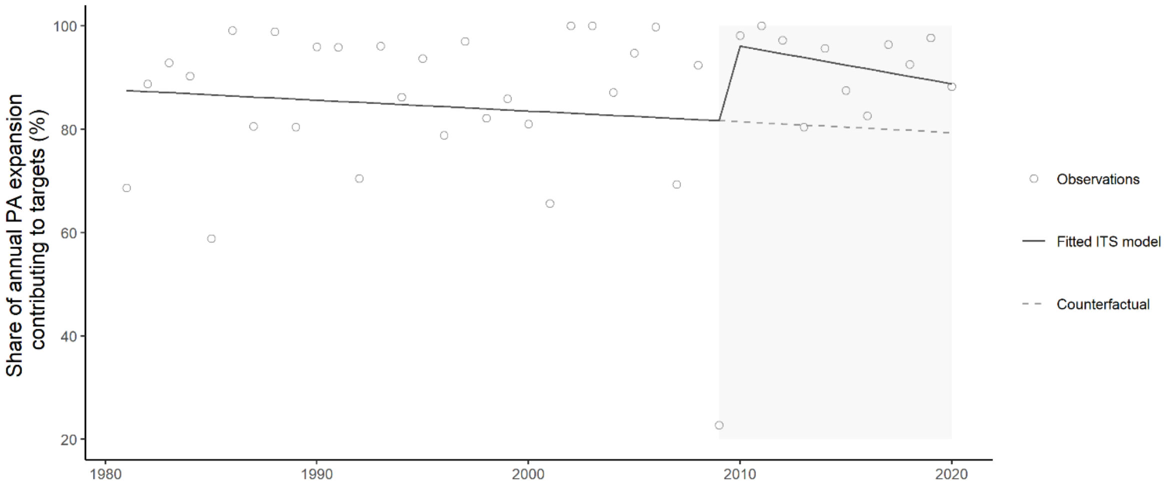

FIGURE 4 Results of the interrupted time series (ITS) model fitted to the annual share of protected area (PA) expansion that contributed to meeting conservation targets for under-represented ecosystems between 1980 and 2020. The shaded area indicates the time segment following the maximum likelihood estimated change point in the outcome in response to the National Protected Area Expansion Strategy.

2.3.3. Matching analysis

South Africa's well-protected ecosystems are clustered in the mountains of the Western Cape, the Mpumalanga and Limpopo Lowveld, northern KwaZulu-Natal and in the Drakensberg Mountains along the border with Lesotho (Rouget et al. 2004, DEA 2016, Skowno et al. 2019). The largest expanse of under-represented ecosystems is in the central arid region of the country (Figure 3). Therefore, the average characteristics of land parcels that would contribute to meeting ecosystem conservation targets if they were to be added to the protected area network are somewhat contrary to expectations from typical biases in protected area networks. Parcels contributing to targets are on average larger, and in larger continuous natural areas in remote areas further away from existing protected areas than redundant parcels. Parcels contributing to targets have lower than average species richness, while redundant parcels have higher than average species richness. In terms of agricultural value, mining potential and socio-economic status, there are no substantial differences (Supplementary Table S1, Appendix II).

The region of common support between the parcel classes was therefore limited, and where necessary, variable-specific calipers were used to improve post-matching balance to <0.1 SMD (see Supplementary Table S1, Appendix II). This resulted in large numbers of observations being discarded. However, for the question that this analysis attempted to answer, pair similarity was more important than generalization of results to any one of the parcel classes, and therefore this was not considered a concern. For example, a large proportion of observations trimmed from the redundant class included parcels that had no remaining intact natural areas. These parcels are much less likely to be considered for protected area expansion owing to their low biodiversity value. Observations that were discarded from the parcels that contribute to targets were much larger than average, more remote, and very far from existing protected areas because no similar redundant parcels exist.

TABLE 2 Results of an interrupted time series model fitted to the annual share of protected area expansion contributing to ecosystem conservation targets between 1980 and 2020.

	Coefficient	**95% confidence interval**	**Standard Error**	**p-value**
Trend in outcome before NPAES	-0.0021	-0.0088, 0.0046	0.0049	0.67
Trend in outcome after NPAES	-0.0052	-0.0266, 0.0162	0.0066	0.43
Effect of intervention on outcome	0.1514	-0.0140, 0.3168	0.0941	0.11

For the three matched samples, SMDs after matching were <0.1, variance ratios were between 0.90 and 1.17, and eCDFs were between 0 and 0.084 (Supplementary Table S1, Appendix II). The coefficients of the binomial GLMs fitted on the matched samples indicate that in the 1990s, land parcels that contribute to ecosystem conservation targets were significantly less likely to be added to the protected area network than similar redundant parcels (Table 3). During the 2000s, the odds ratio suggests that land parcels contributing to targets were three times more likely to be protected. However, the coefficient estimate is very uncertain (CRSE 0.98) and not statistically significant (p = 0.24, Table 3). The results for the 2010s indicate that parcels contributing to targets were less likely to be protected than similar redundant parcels, but the coefficient estimate is also somewhat uncertain (CRSE 0.78, p value 0.039, Table 3).

TABLE 3 Results of binomial generalized linear models testing the difference in odds ratio of protection for land parcels contributing to ecosystem conservation targets compared with matched redundant parcels. Matched samples were created separately for each decade.

Decade	Coefficient	Error	p-value	Odds ratio	95% confidence interval
1990s	-1.95	0.29	<0.0001	0.14	0.08, 0.24
2000s	1.14	0.98	0.242	3.14	2.48, 4.03
2010s	-1.62	0.78	0.039	0.20	0.15, 0.26

2.3.4. Predictors of protected area expansion

The boosted regressions modelled protected area expansion over each of the three decades with good accuracy, as indicated by balanced accuracy on independent test data ranging between 89-94% (Table 4). Cohen's Kappa values also indicated strong improvements in classification accuracy over null models (Table 4).

The results of the matching analyses are supported by the BRT models of protected area expansion. Rankings of variable importance in all three decades indicate that the contribution of a land parcel towards meeting ecosystem conservation targets plays no role in explaining patterns of protected area expansion either before or after the implementation of the NPAES (Figure 5). Partial dependence functions indicate that there is virtually no difference in probability of protection between the two types of parcels (Supplementary Figure S1, Appendix II).

During the 1990s and 2000s, protected area expansion was explained by a combination of land value and species richness (Figure 5). Protected area expansion was less likely in areas of high agricultural potential and low ruggedness (Supplementary Figure S1, Appendix II). Partial dependence profiles indicate that

TABLE 4 Hyper-parameter values used in final boosted regression tree models of protected area expansion in the 1990s, 2000s, and 2010s. Predictive performance was tested on independent data samples and averaged over 50 iterations of the final models.

Decade	Hyper-parameter values for final models							Predictive performance			
	Learning rate (eta)	Gamma	Terminal node minimum observation weights	Tree complexity	Number of iterations	Subsample % per iteration	Variable subsample % per iteration	Balanced accuracy	95% confidenc e interval	Cohen's Kappa	95% confidence interval
1990s	0.46	2.98	9	6	913	65	64	0.8966	0.8957, 0.8975	0.7677	0.7660, 0.7693
2000s	0.14	0.38	16	8	685	60	47	0.9388	0.9383, 0.9393	0.8306	0.8298, 0.8313
2010s	0.096	3.82	14	7	738	84	55	0.8919	0.8912, 0.8923	0.7396	0.7386, 0.7406

areas with very low species richness were avoided in protected area expansion, but expansion was also less likely in areas with the highest species richness (Supplementary Figure S1, Appendix II).

Among indicators of feasibility of implementation, proximity to existing protected areas was the strongest predictor, with areas adjacent to, or nearby, existing protected areas being preferred (Supplementary Figure S1, Appendix II).

There was a strong shift in predictors explaining protected area expansion during the 2010s, with variables related to land availability other than socio-economic circumstances becoming much less relevant (Figure 5). Species richness variables remained strong predictors of siting of new protected areas (Figure 5). During all three decades, protected area expansion in areas with no or very small areas in natural condition was strongly avoided (Supplementary Figure S1, Appendix II), but only in the 2010s did it become one of the most important predictors of protected area expansion. Fragmented areas were also avoided (Figure 5, Supplementary Figure S1, Appendix II).

2.4. Discussion

This study finds that protected area expansion post-2008 was generally aligned with the goals of the NPAES, with an average of 92% of annual protected area expansion within under-protected ecosystems. Studies of protected area expansion elsewhere in the world have however found that progress towards biodiversity representation targets in protected area networks is not necessarily the result of strategic protected area expansion, particularly when these networks are initially limited (Kuempel et al. 2016, Neugarten et al. 2020, Venter et al. 2018). It is therefore necessary to also consider additional indicators of strategic decision-making. Among the indicators considered in this study no evidence could be found that the way that protected area expansion is implemented in South Africa has changed substantially in response to the NPAES.

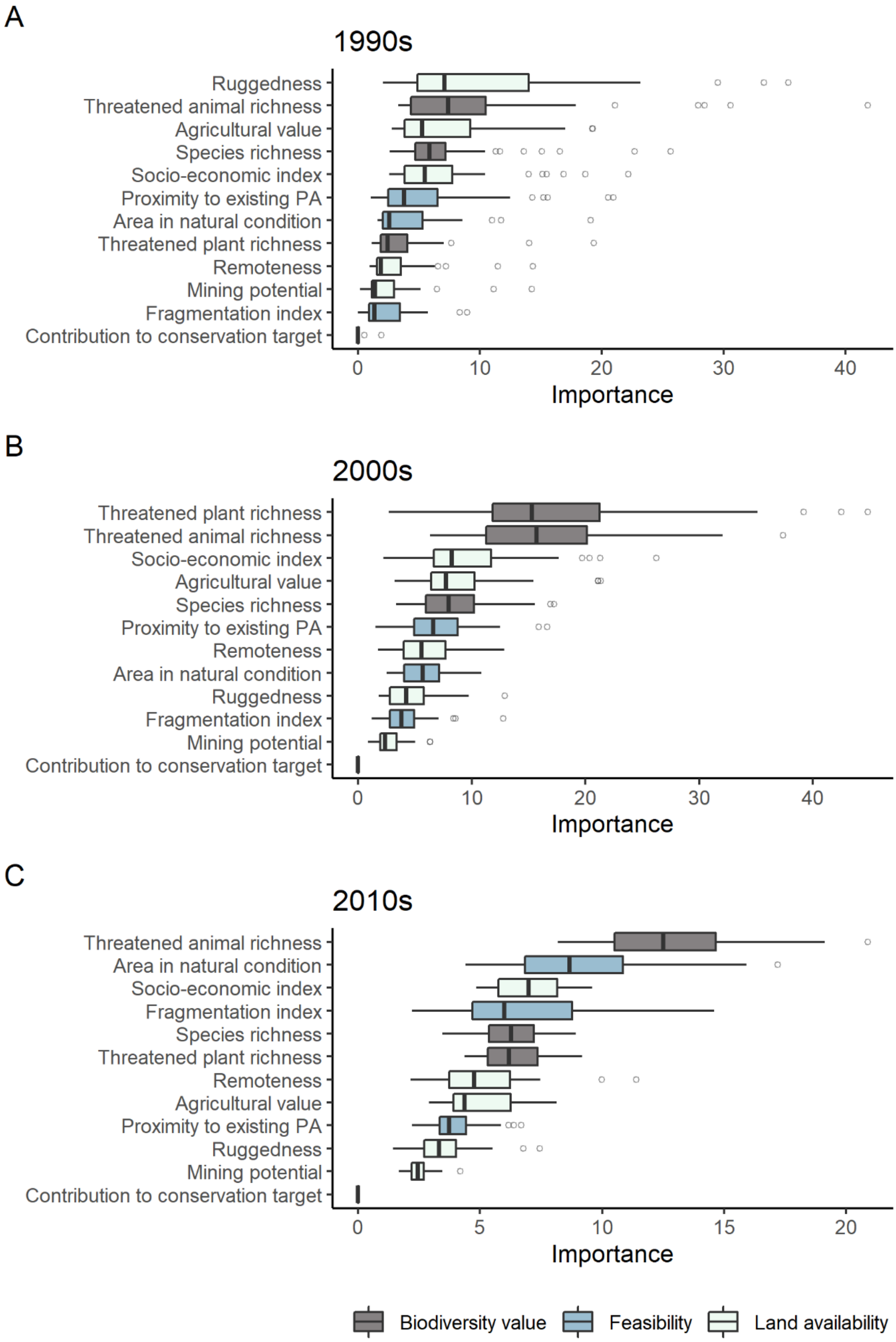

FIGURE 5 Importance of predictor variables in explaining protected area expansion during the A. 1990s, B. 2000s, and C. 2010s. Boxplots indicate variability in importance estimates over 50 runs of boosted regression tree models.

Between 2008 and 2014, eight out of twelve implementation agencies developed and adopted institutionally based protected area expansion strategies aligned with the 2008 NPAES (DEA 2016). It is therefore not suspected that the lack of evidence for strategic protected area expansion is due to typical planning-implementation gaps associated with maps of biodiversity priorities (Knight et al. 2008), but rather due to circumstances and constraints unforeseen by the theory of change.

Time series of the share of annual protected area expansion in under-represented ecosystems indicate that even before the development of the NPAES, the largest share of annual protected area expansion did generally occur within under-represented ecosystems, although the outcome was somewhat variable (Figure 4). This is likely due to South Africa's relatively limited protected area network, and most ecosystems being under-represented (Figure 3). In such circumstances, non-strategic protected area expansion can result in protection gains for under-represented biodiversity, because there are many options within the landscape where these gains can be made. This tendency is consistent with analyses of recent protected area expansion elsewhere in the world (Kuempel et al. 2016, Neugarten et al. 2020). Therefore, strategic decision-making may not yet be critical to ensuring equitable biodiversity protection, but may become much more important in future, when easily available options for protected area expansion in poorly represented ecosystems are exhausted.

A second possible reason for the lack of evidence that protected area expansion practices changed in response to the NPAES is that South Africa has traditionally prioritized areas of high biodiversity value for protected areas. South Africa is a biodiverse country, and wildlife-focused tourism is a major source of local and international revenue (Statistics South Africa 2021). Therefore, potential attractiveness of a proposed protected area to tourists is likely to be a consideration in implementation decision-making (Cousins et al. 2008, Di Minin et al. 2013, Maciejewski and Kerley 2014). This tendency likely explains the consistent high ranking of species richness variables in explaining protected area expansion before as well as after the implementation of the NPAES (Figure 5).

Protected area expansion that targets areas of high species richness, however, does not ensure complementarity in protected area networks (Veach et al. 2017). A recent analysis of species representation in protected areas against a population-based persistence target found that most terrestrial vertebrate species – those species most popular with tourists (Lindsey et al. 2007) - are already well-represented within South Africa's protected area network (Raimondo et al. 2019). Therefore, protected area expansion targeting species is not necessarily aligned with the goals of the NPAES, especially since areas of under-represented ecosystems are generally areas of lower species richness than ecosystems that are already well-protected (Supplementary Table S1, Appendix II).

The only post-NPAES outcome that was somewhat consistent with the theory of change is the decline in importance of variables related to land availability other than socio-economic circumstances in explaining protected area expansion in the 2010s (Figure 5). There is however an alternative plausible explanation for this trend: since the implementation of the NPAES protected area expansion in South Africa has shifted strongly from purchasing land to contracting privately and communally owned land for protection (DEA 2016, Cumming et al. 2017). This type of protected area expansion, known as biodiversity stewardship, is an incentive-driven process whereby landowners retain ownership of properties, but agree through contracts embedded within property deeds to manage natural areas on their properties as protected areas (Cumming et al. 2017). The first protected areas declared through biodiversity stewardship contracts occurred in 2008 (Cumming et al. 2017) and contributed to 68% of the terrestrial protected area expansion between 2008 and 2014 (DEA 2016).

This mechanism of protected area expansion significantly lowered the barriers to implementation within high economic value working landscapes, particularly areas of high agricultural value (Cumming et al. 2017), but it is reliant on landowners volunteering to participate. Incentives for participation include tax deductions against management expenses, reductions of property rates, and assistance from conservation agencies with practical conservation management of the contracted properties (Cumming et al. 2017). However, participation in biodiversity stewardship is not without cost to the landowner and there are limited scenarios where managing private land for conservation can be profitable (Clements et al. 2016). It is therefore not surprizing that while land value declined in importance, socio-economic circumstances remained a strong predictor of protected area expansion in the 2010s (Figure 5), with partial dependence profiles indicating that protected area expansion was more likely in wealthier areas (Supplementary Figure S1, Appendix II).

The analyses for this study were not designed to detect the effects of biodiversity stewardship and therefore it is only inferred from changing patterns in drivers of protected area expansion. It is possible that these patterns may be indicators of changes in decision-making but given that evidence from the other analyses is generally against strategic decision-making as an explanation, stewardship is considered the more plausible cause. These results point to the potential of incentive-based land protection to overcome the typical constraints of land availability on protected area expansion, but further studies are needed to confirm this. It also suggests that stewardship has its own constraints, namely the ability of landowners to afford participation. Stewardship however does not prevent strategic decision-making on where to implement protected area expansion as conservation agencies remain responsible for deciding which volunteered properties to accept into stewardship processes (Cumming et al. 2017).

The question therefore remains: if biodiversity stewardship unlocks more options for protected area expansion than previously available, why is there no evidence of strategic decision-making favouring protected area expansion in under-represented ecosystems? The answer is perhaps in the extreme pressure to rapidly expand the protected area network exerted by international agreements such as the CBD (to which South Africa is a party) as well as the area targets set within the NPAES. At the same time, protected area expansion remains severely under-resourced globally (Halpern et al. 2006, Waldron et al. 2013) as well as in South Africa (DEA 2016). Biodiversity stewardship substantially lowers the cost of acquiring land for conservation (Cumming et al. 2017), but is resource intensive in other ways, particularly staff capacity required for engagement with landowners, biodiversity assessments of proposed properties, and management assistance (DEA 2016).

Under such pressures and constraints, opportunistic protected area expansion is the only viable solution, and therefore the scenario envisaged in the matching analysis is probably not a practical reality. In other words, if rejecting opportunities for protected area expansion because they are not aligned with the biodiversity goals of the NPAES means that area goals will not be achieved, there may be strong incentives to relax biodiversity criteria to be more inclusive. This is not to argue that there may not be good reasons to continue protected area expansion in well-protected ecosystems, but underscores the concern that lack of resources and support for ambitious biodiversity targets can undermine efforts towards strategic achievement of outcomes that best support biodiversity persistence (Barnes et al. 2018, Lemieux et al. 2019, Pressey et al. 2021).

In retrospect, the framing of the theory of change for this study underestimated the complexities and constraints on strategic protected area expansion, which extend far beyond land availability and lack of awareness of biodiversity priorities. Additional factors that were uncovered include the role of tourism potential in feasibility considerations, which can bias protected area networks in ways not explained by the low-economic land value theory (Joppa and Pfaff 2009, McDonald and Boucher

2011), as well as the potential of incentive-driven protected area expansion to overcome land availability constraints independently of strategic decision-making.

South Africa was perhaps not the most ideal case study for testing the effectiveness of strategic protected area expansion, because there are theoretically many options within South Africa's terrestrial environment to improve protection for under-represented biodiversity, and therefore strategic protected area expansion should be relatively easy to achieve without the need for significant effort to adapt implementation practices. More analyses such as these in more constrained environments may yield better insights into the power of strategic decision-making for guiding equitable protected area expansion.

These less-than-ideal circumstances did, however, contribute to highlighting a much more serious unanticipated constraint on strategic protected area expansion, and that is the potential for extreme pressure to rapidly expand protected area networks to derail strategic, biodiversity priority focused efforts even when strategic protected area expansion should be easy to achieve. This study provides a real-world case study of how simplistic, politically driven conservation targets such as the CBD's protected area target, which is set to increase yet again (Waldron et al. 2020), can be a force against conservation impact (Lemieux et al. 2019, Pressey et al. 2021).

To conclude, South Africa is making progress on protection of under-represented ecosystems, but it is not possible to causally link this progress to better strategic decision-making on where to implement new protected areas. Conditions that are more likely to encourage strategic protected area expansion include better resourcing of expansion efforts, as well as a less intense focus on rapid growth of the protected area network.

3. An evaluation of the effectiveness of Critical Biodiversity Areas, identified through a systematic conservation planning process, to reduce biodiversity loss outside protected areas in South Africa

Systematic Conservation Planning (SCP) is a spatially explicit process used globally to prioritize conservation actions, but its effectiveness is difficult to quantify. In South Africa, terrestrial SCP processes are mainly used to identify important biodiversity areas outside of formal protected areas that are required to meet conservation targets. Environmental policy refers to these areas as Critical Biodiversity Areas (CBAs) and uses them to inform land use change decisions. Using Mpumalanga Province as a case study, avoided loss within CBAs is quantified using counterfactual matching methods. To contextualize the results, it is benchmarked against avoided loss achieved by protected areas during the same period. Significant reductions of 54-72% in land clearing were achieved in CBAs compared to other natural areas and were comparable to avoided loss achieved by protected areas in similar evaluations in other countries. Protected areas in Mpumalanga were found to be very effective (88% relative avoided loss) but are located in areas of lower land use change pressures than CBAs. Avoided loss was quantified as 1058 ha for Irreplaceable CBAs, 5285 ha for Optimal CBAs and 20 586 ha for protected areas. The consideration of biodiversity priorities in land use change decisions outside protected areas was found to be an effective complementary strategy to protected areas to avoid loss of biodiversity in areas typically not available for protected area expansion.

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3.1. Introduction

Degradation and destruction of natural ecosystems is considered the leading cause of terrestrial biodiversity loss world-wide (Newbold et al. 2015). Establishment of protected areas is a commonly used conservation mechanism to avert ecosystem degradation and destruction. Studies have shown, however, that protected areas generally make small contributions to avoiding biodiversity loss (Geldmann et al. 2013), because they are established mostly in areas unsuited to other land uses (Joppa and Pfaff 2009), while the most threatened biodiversity tends to be concentrated in areas of high human pressures (Hoekstra et al. 2005, Venter et al.

2016). Complementary conservation strategies are therefore needed to safeguard biodiversity in areas that are not available for protected area expansion, or where protected areas are not practical to implement (Dudley et al. 2018). Examples of alternative area-based conservation interventions include territories conserved by indigenous peoples and local communities (Carranza et al. 2014), payments for ecosystem services (PES, Costedoat et al. 2015), and land use zoning (Bruggeman et al. 2018).

South Africa uses maps of areas most important for the persistence of biodiversity to guide the consideration of biodiversity in land use change decisions as a complementary area-based strategy to avoid extinction of species and collapse of ecosystems that are not yet well-represented within the protected area network (Cadman et al. 2010). Biodiversity priority maps are developed using systematic conservation planning (SCP), a spatially explicit approach for prioritizing conservation actions to secure the persistence of biodiversity features *in situ* (Margules and Pressey 2000).

The SCP process identifies spatial biodiversity priority areas by setting quantitative conservation targets for species, ecosystem types, and ecological processes in combination with other design criteria such as connectivity, avoiding implementation conflicts and minimizing implementation costs. Planning domains, which are typically provinces in South Africa, are divided into discrete spatial units known as planning units. Prioritization algorithms assign each planning unit a value indicating its importance for meeting the conservation targets and objectives set for the planning domain. Planning units that are not within existing protected areas but are considered necessary to meet conservation targets either because they are irreplaceable, or because they contribute to the design criteria mentioned above, are classified as Critical Biodiversity Areas (CBAs, SANBI 2017).

South African environmental regulations require that maps of biodiversity priorities are accompanied by guidelines detailing appropriate land uses compatible with each category of biodiversity priority included in the map. When maps of biodiversity priorities are formally gazetted as bioregional plans, the associated land use guidelines must be considered when land use changes are authorized (DEAT 2009), and land use decisions that are contrary to the guidelines may be legally challenged.

CBAs are areas that are mostly in a natural or near-natural state at the time of the conservation assessment, and associated management guidelines typically require that they be maintained in a natural or near natural state (SANBI 2016). In other words, land use changes that would result in the destruction or degradation of these areas should preferably not be authorized.

What is not yet known, is how effective this system of SCP-based land use decision-making is in avoiding loss of biodiversity. Although SCP processes have been used to identify priority areas for conservation actions since the 1980s (Adams et al. 2019b, Watson et al. 2011), a recent exhaustive review of the conservation literature found a lack of published evidence for conservation outcomes of SCP processes (McIntosh et al. 2018). Moreover, there are not yet any established methods for evaluating the conservation outcomes of SCP processes (Bottrill and Pressey 2012, McIntosh et al. 2018).

The evaluation of the effectiveness of a conservation intervention requires the quantification of impact on a specific outcome attributed to the intervention, which is the difference between the observed outcome under the intervention and the outcome in the absence of the intervention (Adams et al. 2019a, Ferraro and Pattanayak 2006). Such differences are most reliably estimated through randomized controlled trials, which is often not practical or ethical in conservation interventions (Pynegar et al. 2021).

An alternative approach is to use quasi-experimental methods to construct counterfactual estimates of conservation outcomes in the absence of an action (Ferraro and Hanauer 2014). This method has been used to estimate the effectiveness of spatially explicit conservation interventions such as protected areas in avoiding biodiversity loss, for example Andam et al. (2008), Honey-Rosés et al. (2011) and Joppa and Pfaff (2010).

According to Bottrill and Pressey (2012), appropriate and informative evaluations of SCP need to be framed around the ultimate conservation goals of planning. In the South African conservation context, one of the most important goals of conservation planning is to avoid the loss of natural areas that are needed to meet conservation targets for species, ecosystems, and ecological processes (Cadman et al. 2010). The aim of this study is therefore to quantify the avoided loss of biodiversity priority areas (CBAs) as a result of the implementation of CBA-associated land use guidelines, by adopting the spatially explicit counterfactual methods developed to evaluate the effectiveness of protected areas.

The first question this study aims to answer, is whether land use decisions guided by biodiversity priority maps have led to a reduction in loss of CBAs. Mpumalanga Province was chosen as a case study, as it had one of the first conservation plans specifically developed to inform land use decision-making as required by environmental legislation (National Environmental Management: Biodiversity Act 10 of 2004). CBAs in Mpumalanga were matched to areas under similar land use pressure that are not needed to meet conservation targets, with land use changes in these non-prioritized areas representing a counterfactual where land use decisions do not need to consider biodiversity priority guidelines. This analysis provides an estimate of the effect size of the intervention, but to gauge its effectiveness, it needs to be contextualized through comparisons to alternative conservation interventions that are designed to achieve the same outcome (Rasolofoson et al. 2015, Robalino et al. 2015, Sims et al. 2017).

The primary goal of protected areas is the same as land use guidelines associated with CBAs: to avoid the loss of natural areas to other land uses. In protected areas destructive land use changes are legally prohibited (Government of South Africa 2004), and therefore protected areas provide a benchmark of the most secure form of land protection. Therefore, the effect size of protected areas in avoiding loss of natural areas is estimated using the same matching methods and over the same time period (2010-2020) as the implementation of SCP-based land use decision making in Mpumalanga Province and is used to contextualize the effectiveness of CBAs.

3.2. Methods

3.2.1. Study area and time frame of analysis

Mpumalanga is a land-locked province in eastern South Africa with an area of 76 520 km^2 (Figure 6). The province's first SCP-based map of biodiversity priorities, known as the Mpumalanga Biodiversity Conservation Plan (MBCP) was developed in 2006, and associated land use guidelines were published for implementation in 2007 (Ferrar and Lötter 2007). An updated map and land use guidelines, known as the Mpumalanga Biodiversity Sector Plan (MBSP) was published in 2014 (Lötter et al. 2014), based on a reassessment of spatial biodiversity priorities following improvements in available biodiversity data, changes in environmental regulations, and incorporating the effects of protected area expansion and land use changes since 2006.

The MBCP assessment was based on land cover data from 2000, which means that there is a considerable lag between the implementation of the plan and the baseline data. Methods of land cover classification improved significantly between 2000 and 2010, when a second land cover dataset was developed for the province, which means that it is difficult to distinguish real land cover changes between 2000 and 2010 from those that are due to methodological differences. Due to these complexities, it is not possible to evaluate the impact of conservation planning outcomes on land use changes in Mpumalanga Province before 2010, and the analysis is therefore restricted to the period 2010-2020.

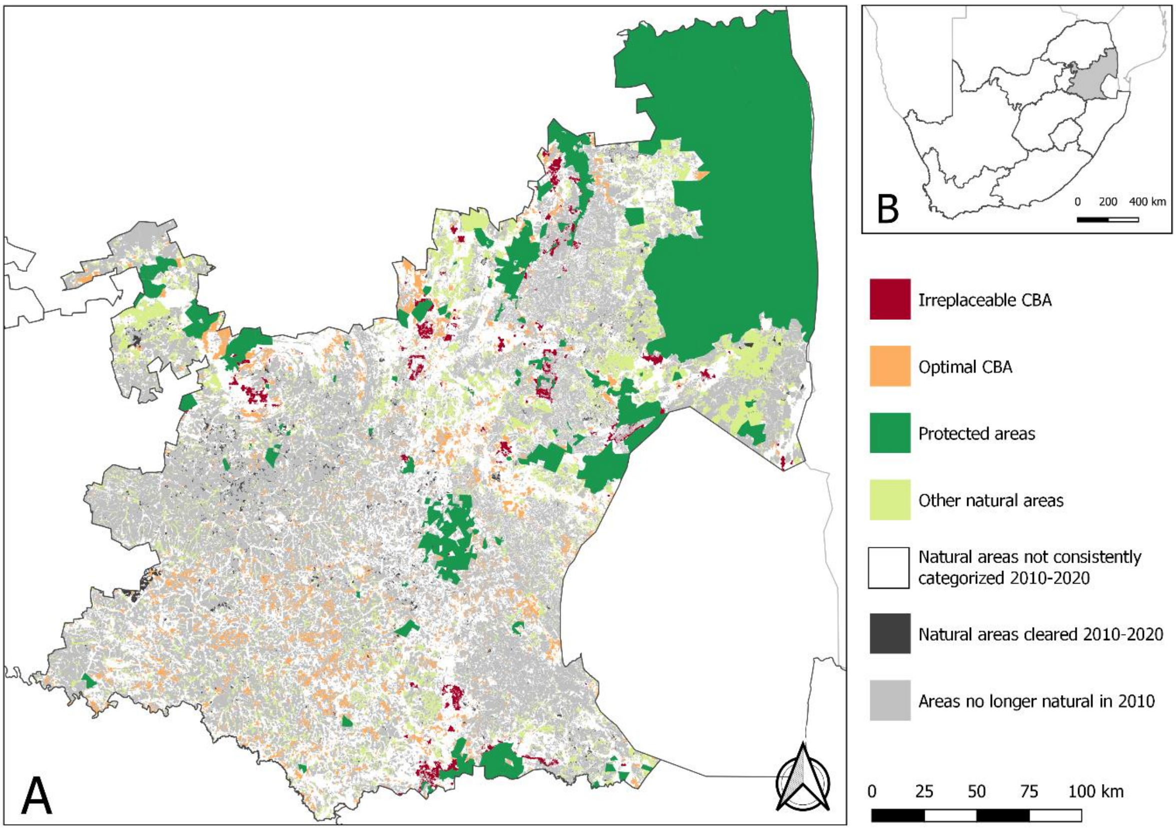

FIGURE 6 A. Map of consistently categorized biodiversity priority categories in the two conservation plans developed for Mpumalanga Province, with an indication of natural areas that were cleared between 2010 and 2020. B. The location of Mpumalanga Province in South Africa.

One of the challenges with evaluation of the conservation outcomes of SCP processes is the often-protracted implementation of recommended conservation actions (Bottrill and Pressey 2012), and therefore a longer time interval between the conservation assessment and evaluation is more likely to provide evidence of impact. The fact that conservation planning was already being applied in land use change decisions in Mpumalanga since 2007 means that there is no expected time lag in implementation that could undermine the causal link between planning and avoided loss of biodiversity priority areas.

3.2.2. Unit of observation, treatment, control and outcome variable

The study area was divided into 100m x 100m (1 ha) grid cells, with the centroid of each grid cell representing a unique sampling point, or unit of observation.

Sampling points within Critical Biodiversity Areas (CBAs) were evaluated as treatment observations. South African maps of biodiversity priorities recognise two types of CBAs. Irreplaceable CBAs represent ecosystems or areas of habitat for species for which the remaining intact areas are near or below the conservation target for the biodiversity feature. If such areas were to be lost, it means that the persistence of the biodiversity elements present would be compromised (SANBI 2016). Loss of these areas would therefore contribute to a loss of biodiversity.

The second type of CBAs are Optimal CBAs. These are areas where other options remain within the landscape for meeting the conservation targets for the biodiversity elements present, but these areas represent the most efficient locations for meeting their targets. Loss of these areas would therefore not necessarily contribute to a loss of biodiversity if alternative areas with similar biodiversity features could be secured for conservation through for example biodiversity offsets. Since land use guidelines are designed around the potential biodiversity impacts of the loss of the different types of CBAs, the two types of CBAs were analysed separately.

Areas that are in natural, near natural or semi-natural condition that are not required to meet conservation targets for ecosystem types, species or ecological processes are classified as other natural areas (ONAs) within biodiversity priority maps. These areas have no additional restrictions in terms of land use change authorisations, and therefore control observations were derived from sampling points within these areas.

Improved biodiversity data, as well as protected area expansion and loss of natural areas between the 2006 assessment and 2014 assessment, resulted in priority areas not being exactly aligned between the two plans. To ensure consistency in the evaluation of treatment and control samples, sampling was restricted to areas that were consistently classified as Irreplaceable CBA, Optimal CBA and ONAs in both the MBCP and the MBSP (Figure 6).

South African maps of biodiversity priorities include another category that needs to remain in a natural or near natural condition, known as Ecological Support Areas (ESAs, SANBI 2016). These are areas that support the ecological functioning of CBAs and protected areas or are important for ecosystem services. The MBCP did not include priority categories that are conceptually equivalent to ESAs, and therefore avoided loss in ESAs could not be evaluated for this study.

For the comparative analysis, treatment observations were drawn from protected areas included within the 2014 MBSP. Control observations were selected from ONAs outside protected area buffers (see section 3.2.7 on spillover effects for clarification).

The latest available land cover classification for the province is based on satellite imagery from 2020 (MTPA 2020) and was used to estimate loss of natural areas to other land uses. Outcome was measured as a binary value indicating whether a particular unit of observation was lost to other land uses (outcome = 1) or remained in a natural condition (outcome = 0) between 2010 and 2020.

3.2.3. Covariates

Within observational studies, it is necessary to control for confounding variables that are likely to affect selection of observation units into treatment, or the observed outcome (Austin 2011). In evaluations of avoided loss of natural areas, variables explaining development pressure are important covariates to consider, because development pressure is not evenly distributed across the landscape. Covariates reported in the literature as known predictors of development pressure that were included in this study were remoteness, distance from roads, terrain ruggedness and agricultural potential. In addition, covariates related to the local regulatory and economic context, such as mining pressure and development restrictions within protected area buffer zones were also included.

Bioregions were included as a covariate predicting selection into treatment. Bioregions are composite spatial terrestrial units of similar biotic and physical features at the regional scale, representing an intermediate vegetation classification between vegetation type and biome (Mucina and Rutherford 2006). Specific land use types are more prevalent within some bioregions than others, for example plantation forestry in Mesic Highveld Grasslands, and different bioregions are unequally represented within the protected area network. Sampling points within more highly transformed bioregions that are also poorly protected are therefore more likely to be classified as CBAs.

Bioregion was preferred over the finer scale vegetation type, as conservation targets are set on vegetation types. Therefore, in some highly threatened vegetation types, all remaining natural areas may be required to meet their conservation targets, which means that it would not be possible to match control units to observations within these vegetation types. Variables that perfectly predict selection into treatment are known to reduce precision of treatment effect estimates and should therefore be avoided (Bergstra et al. 2019). A summary of all covariates considered in this study, including data structure and sources is provided in Table 5.

3.2.4. Matching

Matching methods are used to balance covariate differences between treatment and control samples in observational studies (Rubin 1973), thereby controlling for bias due to observable confounders in the estimation of the treatment effect. Random samples of 50 000 points were taken from each of the treatment groups (two types of CBAs and protected areas). For protected areas, samples were restricted to observations that were in a natural condition according to 2010 land cover. Matches for each of the treatment groups were drawn from the full set of control observations in ONAs (772 491).

TABLE 5 Covariates of development pressure used in matching analyses for CBAs and protected areas. * indicates categorical covariates where exact matching was required.

Covariate	Definition and source	Rationale
Agricultural potential*	The Land Capability classification system defines the agricultural potential of an area. Land Capability is modelled in 15 classes indicating the suitability of land for rangelands and crop cultivation, with 1 indicating lowest suitability and 15 indicating highest suitability. Source: Agricultural Research Council – Institute for Soil, Climate and Water.	Areas with higher agricultural suitability are likely to be under higher land conversion pressure than areas with low or poor suitability. Protected areas are generally located in areas of lower agricultural suitability.
Mining potential*	Mining potential is quantified in four categories, based on criteria related to the size of known mineral deposits and their economic importance (Rouget et al. 2004). Source: National Spatial Biodiversity Assessment 2004 available at bgis.sanbi.org/nsba	Mining is of major importance to the South African economy, and Mpumalanga Province is rich in mineral deposits. Mining pressure is expected to be higher in areas with high mining potential.
Remoteness	Distance in kilometres of sampling point from nearest built-up area as mapped in 2010 land cover dataset. Source: Mpumalanga Tourism and Parks Agency	Urban expansion is most likely in areas immediately adjacent to existing settlements, therefore sampling points nearer to built-up areas are likely to be under higher development pressure than more remote areas. Protected areas are more likely to be in more remote areas.
Ruggedness	An Index of Terrain Ruggedness was calculated from a digital elevation model following the method of Riley et al. (1999). Higher values indicate increasing ruggedness. Source: 90m resolution Shuttle Radar Topography Mission (SRTM) Digital Elevation Model	Development pressure is likely to be highest in areas with lowest ruggedness, as development costs may be higher in more rugged terrain.
Distance from road	Distance in kilometres of sampling point from nearest road. Source: NGI Topo Data June 2020 http://www.cdngiportal.co.za/cdngiportal/	Development is more likely in accessible areas, therefore sampling points closer to roads are likely to be under higher development pressure than those further away from roads.
Bioregion*	A categorical value indicating within which one of six bioregions present within Mpumalanga Province the sampling point is located. Source: Vegetation map of South Africa, Lesotho and Swaziland, 2018 version. http://bgis.sanbi.org/vegmap	Specific land use types are more prevalent within some bioregions than others. Planning units within highly transformed bioregions are more likely to be designated as CBAs.

Protected area buffer	A binary value indicating whether the sampling point falls within a designated protected area buffer zone or not. Source: 2014 Mpumalanga Biodiversity Sector Plan http://bgis.sanbi.org/MBSP	Stricter development regulations apply within protected area buffer zones than elsewhere, regardless of whether the area falls within a CBA or not.

Matching was performed using the R package MatchIt (Ho et al. 2011), using the method of one-to-one nearest neighbour matching with replacement for the set of covariates listed in Table 5. Exact matching was required on categorical variables (see Table 5), while Mahalanobis distance was used for continuous covariates. No other common support restrictions or calipers were used.

Covariate balance of the matched samples were assessed according to standardized mean differences (SMDs), variance ratios and empirical cumulative density functions (eCDFs). Good matches are indicated by SMDs < 0.1, variance ratios close to 1 and maximum eCDFs close to 0.

Where treatment observations had to be discarded due to lack of common support, particularly due to requirements for exact matching on categorical covariates, the trimmed samples were examined for differences in sample means and variances from the original sample. This is to ensure that the treatment effects estimated for the trimmed samples remained generalizable to the treatment category it was derived from.

3.2.5. Estimation of treatment effect using post-matching regressions

Effect size was estimated as average treatment effect in the treated (ATT) by fitting binomial generalized linear models on the matched datasets with treatment category as a binary predictor and loss to other land uses as a binary outcome. The identity link function was used to quantify the ATT as the absolute difference between the outcome in CBAs/protected areas (treatment) and ONAs (control), therefore a negative coefficient indicates avoided loss. As all SMDs for the matched samples were below 0.1, no additional covariate adjustment was included in the binomial regressions (Nguyen et al. 2017).

Uncertainty was estimated using cluster robust standard errors (CRSE), which controls for paired observations as well as repeat sampling of some control observations (Austin 2009).

After estimating the effect sizes, the avoided loss in hectares for each treatment category was calculated by using the % loss in the matched control samples to calculate its equivalent in hectares from the total area of each treatment category that was in a natural condition in 2010. Avoided loss is then the difference in between the hectares derived from the control sample and the observed total loss in the treatment category between 2010 and 2020.

Relative effects were calculated following the method of Carranza et al. (2014), according to the formula C-T/C. Where C is the observed loss of natural areas in a sample of control observations matched to a treatment sample, and T is the loss in the treatment sample. Relative effects therefore allow for the comparison of effect sizes when they have been estimated against different background or baseline rates of land conversion.

3.2.6. Sensitivity analysis

Matching on observable covariates is expected to approximate the effect size estimates that would be derived from randomized controlled trials, but this relies on the assumption that there are no unobserved confounders affecting the probability of selection into treatment. It is not possible to directly assess this potential bias, but Rosenbaum (2002) developed a nonparametric method to quantify the magnitude of hidden bias that would be needed to change inferences about the treatment effect. A constant, Γ (gamma), representing the odds ratio of two observations with the same values for observable covariates but different probabilities of treatment assignment, is manipulated to determine how large it must be before inferences about the significance of the estimated treatment effect would be invalidated. The larger the value of Γ at the point where significance is overturned, the more robust the treatment effect estimate is to hidden bias. Sensitivity bounds on p-values for treatment effects were estimated for Γ values between 1 and 8 using the R-package rbounds (Keele 2010).

3.2.7. Testing for spillover effects

It has been demonstrated that spatial conservation interventions often do not reduce environmental pressures, but merely displace them to nearby areas where the interventions are not present (Ewers and Rodrigues 2008, Heilmayr et al. 2020), a phenomenon termed spillover effects (Schleicher et al. 2019). Displacement of destructive land use changes from areas of high conservation priority to areas of low priority is a desired outcome of the land use change authorization system, however areas of displacement may not represent a realistic counterfactual of land use change in the absence of the policy intervention. Therefore, potential spillover effects were assessed by comparing loss of ONAs that were within 1 km of CBAs in both the MBSP and MBCP to ONAs more than 1 km from CBAs in both plans. The same sampling and matching methods were used as for the CBA-ONA comparisons, and binomial generalized linear models were again used to quantify the difference in development pressure.

Development within protected area buffer zones is more strongly regulated than in natural areas beyond the buffers, to support the ecological integrity of protected areas (DEAT 2010). Therefore, spillover effects are not anticipated in relation to protected areas, but sampling of control observations was restricted to areas outside protected area buffers, to provide a realistic counterfactual of avoided loss.

3.3. Results

3.3.1. Land cover changes in Mpumalanga Province between 2010 and 2020

Land cover changes between 2010 and 2020 in Mpumalanga Province indicate that 1.5% of all areas that were in natural condition in 2010 were lost to other land uses by 2020. Loss was predominantly to mining (38.4% of all loss), agriculture (21.9%) and expanding human settlements (16.1%, Figure 7A).

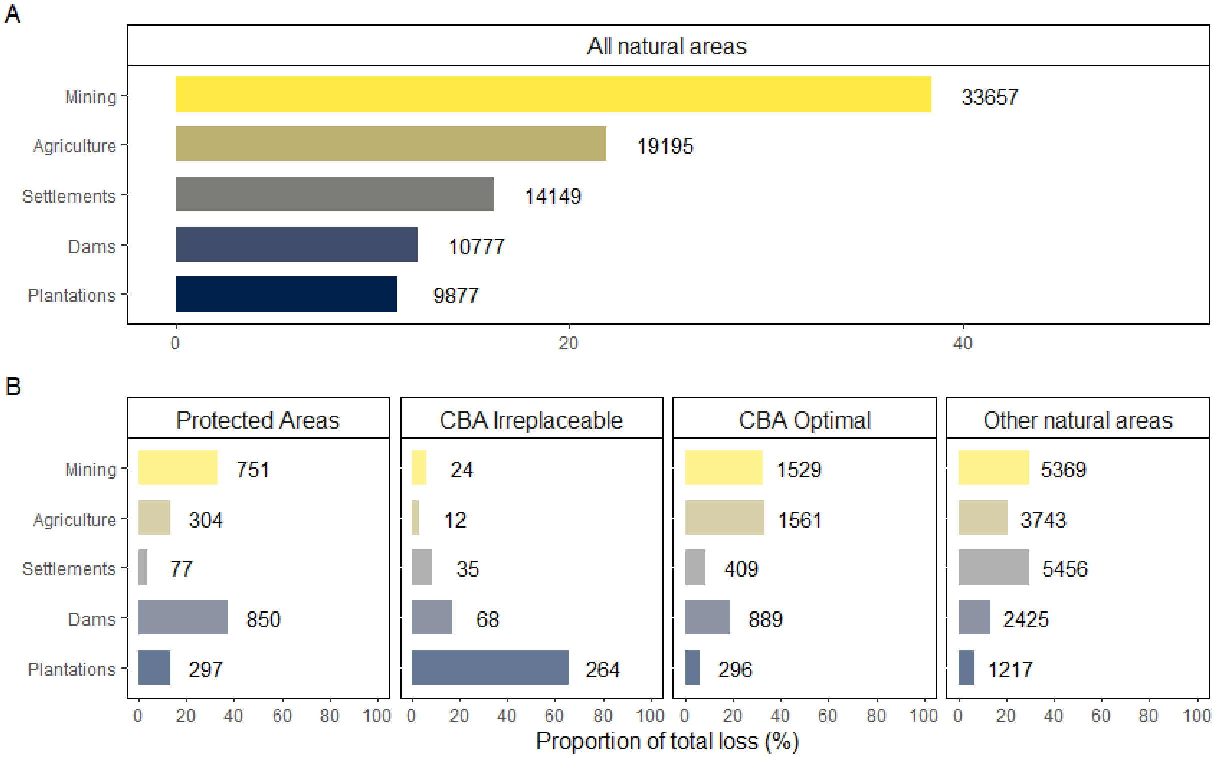

FIGURE 7 A. Major drivers of land use changes across all natural areas in Mpumalanga Province between 2010 and 2020. B. Major drivers of land use changes within different biodiversity priority categories over the same period. Labels indicate the number of hectares lost in each category. Hectares for biodiversity priority categories do not add up to the value for all natural areas, as not all natural areas were consistently categorized between 2010 and 2020.

Within SCP-defined biodiversity priority categories, loss was the lowest in protected areas (0.1%), and highest in ONAs (2.4%). Loss was 0.4% in Irreplaceable CBAs and 1.2% in Optimal CBAs. Loss of natural areas within Irreplaceable CBAs was mainly to plantations, while in Optimal CBAs it was to agriculture, mining, and dams (Figure 7B). Within protected areas, loss was mainly to dams, mining, and agriculture, while in ONAs, it was to settlements, mining, and agriculture (Figure 7B).

3.3.2. Matched samples

Before matching, CBA observations were in more remote and rugged areas further away from roads, and more likely to be in areas of lower agricultural potential, but with higher mining potential than ONAs (Table 6). After matching, all SMDs for the

TABLE 6 Covariate balance before and after matching for three treatments and potential spillover effects evaluated for Mpumalanga Province. Good matches are indicated by standardized mean differences (SMD) < 0.1, variance ratios close to 1 and maximum empirical cumulative density functions (eCDF) close to 0. It was not possible to calculate variance ratios for binary variables.

		Before matching			After matching				
Treatment	**Covariate**	**Means treated**	**Means control**	**SMD**	**Means treated**	**Means control**	**SMD**	**Variance ratio**	**eCDF**
Irreplaceable CBAs (N = 49 662)	Agricultural potential	6.12	7.50	0.70	6.13	6.13	0	1	0
	Mining potential	1.07	0.55	0.43	1.06	1.06	0	1	0
	Remoteness	1.99	1.44	0.47	1.99	1.93	0.05	1.07	0.04
	Ruggedness	53.54	24.97	0.68	53.34	49.98	0.08	1.23	0.03
	Distance from road	1.32	0.84	0.45	1.32	1.27	0.05	1.10	0.03
	Bioregion	3.03	2.45	0.68	3.02	3.02	0	1	0
	Protected area buffer	0.48	0.24	0.48	0.47	0.47	0	-	0
Optimal CBAs (N = 49 832)	Agricultural potential	7.22	7.50	0.17	7.23	7.23	0	1	0
	Mining potential	1.11	0.55	0.55	1.11	1.11	0	1	0
	Remoteness	1.56	1.44	0.11	1.56	1.54	0.02	1.07	0.01
	Ruggedness	23.98	24.97	0.04	23.93	23.62	0.01	1.10	0.01
	Distance from road	0.95	0.84	0.12	0.95	0.94	0.02	1.10	0.01
	Bioregion	2.82	2.45	0.56	2.83	2.83	0	1	0
	Protected area buffer	0.16	0.24	0.22	0.16	0.16	0	-	0
Protected areas (N = 45 737)	Agricultural potential	7.20	7.49	0.20	7.26	7.26	0	1	0
	Mining potential	0.23	0.64	0.56	0.24	0.24	0	1	0
	Ruggedness	19.85	24.50	0.16	20.45	20.53	0.003	1.13	0.04
	Distance from road	1.46	0.84	0.44	1.50	1.45	0.04	1.25	0.02
	Bioregion	2.31	2.58	0.35	2.15	2.15	0	1	0
Spillovers – ONAs <1 km from CBAs (N = 49 951)	Agricultural potential	7.23	7.60	0.22	7.23	7.23	0	1	0
	Mining potential	0.88	0.35	0.53	0.88	0.88	0	1	0
	Remoteness	1.51	1.46	0.04	1.51	1.49	0.01	1.08	0.01
	Ruggedness	26.82	24.14	0.10	26.81	26.68	0.005	1.05	0.01
	Distance from road	0.88	0.88	0.004	0.87	0.87	0.01	1.09	0.01
	Bioregion	2.73	2.27	0.62	2.73	2.73	0	1	0
	Protected area buffer	0.13	0.31	0.55	0.13	0.13	0	-	0

covariates were below 0.08, all variance ratios were below 1.24, and all eCDF maximum differences were below 0.05 (Table 6).

The requirement of exact matching on categorical variables resulted in 338 treatment observations being excluded from the Irreplaceable CBA sample and 168 from the Optimal CBA sample for lack of suitable matches among the control observations. Comparisons of SMDs, variance ratios and eCDFs indicated no substantial differences in covariate values between the original randomized treatment samples and the remaining samples post-matching.

Matching was difficult for protected areas because the region of common support with ONAs was limited. The main reason for lack of common support was that remoteness ranged from 0.1-26.4 km for protected areas, while for ONAs it was limited to 0-10.2 km. The protected area observations that fell outside of the region of common support were concentrated within the Kruger National Park, South Africa’s largest and one of its oldest protected areas. While remoteness is generally considered to be a driver of siting of protected areas (Joppa and Pfaff 2009), in the case of the Kruger National Park it is a consequence of its size and age. The relatively high agricultural potential and low ruggedness of sampled observations inside the park suggest that the area would have been under high development pressure, had the land not been protected. In fact, 45% of unprotected land outside the park that is similar in terms of bioregion, ruggedness and agricultural potential is no longer in a natural condition. For these reasons, remoteness was excluded from the covariates used in matching the protected area sample.

3.3.3. Treatment effects

Effect size estimates reveal that CBAs and protected areas had statistically significant impacts in reducing land clearing compared to counterfactuals observed within ONAs (Table 7). The effects, as inferred from the coefficients of the fitted logistic regressions, were a 1.1 percentage point difference in loss in Irreplaceable CBAs

(CRSE 0.0027, p<0.0001), 1.4 percentage points in Optimal CBAs (CRSE 0.0020, p<0.0001) and 1.09 in protected areas (CRSE 0.0009, p<0.0001). The absolute effect sizes appear to be small, but is due to the relatively low rate of land conversion in Mpumalanga Province between 2010 and 2020.

Effect size estimates were robust to hidden bias, with critical Γ values ranging between 2.1 for Optimal CBAs and 7 for protected areas (Table 7). These values indicate that very strong predictors of treatment categories are needed to overturn the significant estimates of avoided loss.

TABLE 7 Effect size estimates for three spatial conservation interventions aimed at reducing loss of natural areas in Mpumalanga Province between 2010 and 2020. Percentages in brackets under observed and avoided loss indicate % of extent in 2010.

Treatment	Coefficient	Error	p-value	Critical Γ at p=0.05	Observed loss (ha)	Avoided loss (ha)	Loss in matched ONAs (%)
Irreplaceable CBAs	-0.0111	0.0027	<0.0001	3.2	403 (0.42%)	1058 (1.11%)	1.53
Optimal CBAs	-0.0139	0.0020	<0.0001	2.1	4684 (1.20%)	5285 (1.35%)	2.56
Protected areas	-0.0109	0.0009	<0.0001	7.0	2279 (0.12%)	20 586 (1.11%)	1.24

The estimated avoided loss in Irreplaceable CBAs is 1058 hectares, which translates to a 72% relative reduction in the number of hectares that would have been lost in these highly valuable natural areas, had biodiversity priorities not been considered in land use change authorisations. Avoided loss in Optimal CBAs was 5285 hectares (54% relative reduction) and for protected areas it was 20 586 hectares (88% relative reduction).

3.3.4. Spillover test

No evidence was found for spillover effects within the immediate vicinity of CBAs. The difference in development pressure between ONAs within 1 km of CBAs and more than 1 km from CBAs was significant (0.79 percentage points, CRSE 0.0013,

$p<0.001$), but rates of land clearing were lower within the immediate vicinity of CBAs (2%) than further away (2.8%). These results were more sensitive to hidden bias than the treatment effects (critical Γ 1.4), suggesting a higher likelihood that differences in rates of land clearing could be explained by unobserved confounders rather than proximity to CBAs.

3.4. Discussion

Systematic Conservation Planning is used globally to prioritize areas for conservation actions, but its effectiveness has been difficult to quantify. It has been argued that counterfactual evaluations of SCP are not feasible because the need to compare conservation outcomes between areas with and without maps of biodiversity priorities introduces too many social, political, and biological complexities that are difficult to control for (Bottrill and Pressey 2012, Margoluis et al. 2009). While this study is not able to evaluate the effectiveness of SCP as a conservation mechanism, it demonstrates that the way that South Africa implements SCP-based maps of biodiversity priorities in land use decisions is an effective strategy to reduce biodiversity loss outside protected areas.

SCP-guided land use decisions led to significant reductions in loss of both Irreplaceable and Optimal CBAs. Consideration of biodiversity priorities in land use decisions is confirmed to be an effective conservation intervention, when contextualized among other spatially explicit interventions aiming to avoid biodiversity loss that were evaluated using similar counterfactual methods (Figure 8).

Protected areas in Mpumalanga Province were also found to be extremely effective in avoiding biodiversity loss, when compared to CBAs as well as other spatially explicit conservation interventions (Figure 8). These results are not unexpected, as most destructive land clearing is legally prohibited in protected areas, while the legal requirement for CBAs is only that land use guidelines must be considered when land use changes are authorized. Land clearing in protected areas was not completely avoided during the evaluation period, mainly due to the illegal expansion of mining into an adjacent protected area (Figure 7B), but overall impacted a very small proportion (0.12%) of the protected area estate (Table 7).

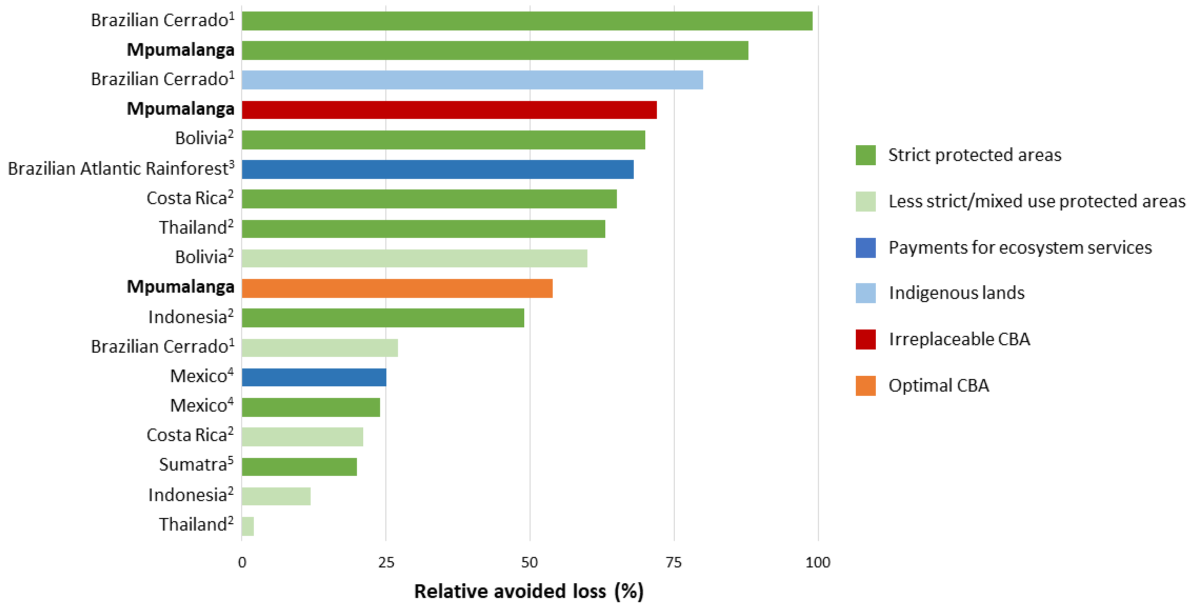

FIGURE 8 Ranked effect size of CBAs and protected areas in Mpumalanga Province among similar counterfactual evaluations of spatial conservation interventions reported in the literature. Data sources: [1]Carranza et al. 2014, [2]Ferraro et al. 2013, [3]Ruggiero et al. 2019, [4]Sims et al. 2017, [5]Gaveau et al. 2012.

An important finding was that when effectiveness was estimated in terms of absolute percentage point differences (Table 7), both Irreplaceable and Optimal CBAs performed better than protected areas. This is because CBAs are situated in areas of relatively higher land use pressure than protected areas, as inferred from observed loss in matched control ONA samples (Table 7). Larger absolute estimates of avoided loss are consistently correlated with conservation interventions located in areas of higher land use pressures in counterfactual (Carranza et al. 2014, Nolte et al. 2013, Pfaff et al. 2014) and simulated (Sacre et al. 2020) studies.

Critiques of the effectiveness of protected areas as conservation interventions often centre around the fact that they tend to be placed in areas of low land use pressure, and therefore these sites are likely to have remained in a natural condition, even if the protected areas were not in place (Ferraro and Pattanayak 2006, Joppa and Pfaff 2011, Pressey et al. 2021). The results for protected areas in Mpumalanga are consistent with these concerns. The results for CBAs however indicate that land use change decisions can achieve significant reductions in loss of biodiversity priority areas despite higher land use pressures and less severe legal restrictions on land clearing than protected areas.

The comparison of effectiveness between protected areas and CBAs suggest that protected areas are still the most effective option where resources are available. Protected area expansion should therefore not be deprioritized in favour of reliance on land use decisions as a means to avoid biodiversity loss, but CBAs do provide an effective complementary mechanism to secure the persistence of biodiversity in areas where land may not be available for protected area expansion, or protected areas may not be feasible to implement.

The correlation between higher estimates of avoided loss and higher land use pressure has led to recommendations that decisions on where to implement conservation interventions should be guided by measures of threat (e.g. Sacre et al. 2020). While such an approach may ensure that conservation action is implemented where it is most urgently needed, there are also risks and potential unintended consequences when maximizing avoided loss becomes the primary aim of a conservation intervention (Barnes et al. 2018, Vincent 2016).

Consistently prioritizing conservation over other land uses in areas of highest threat, which are also likely areas of highest economic growth potential, may give highest returns in terms of hectares spared, but is also most likely to provoke conflicts and backlash against conservation policies (Vincent 2016). Observed examples are pre-emptive clearing of threatened ecosystems and critical habitat for threatened species earmarked for legislative embargos on land clearing (Lueck and Michael 2003, Simmons et al. 2018).

A strength of SCP is that it avoids conflict with competing land uses as far as possible through the inclusion of cost surfaces in the assessment phase, as well as stakeholder engagement during the development of the plans (Naidoo et al. 2006, Lötter et al. 2014). The implementation of SCP-based land use decision making does not attempt to forestall economic development, because development is not legally prohibited within CBAs. The distinction between Irreplaceable and Optimal CBAs allows for a flexible and responsive approach to transparently and defensibly weigh up the relative costs and benefits of conservation against economic gains on a case-by-case basis.

The reason South Africa's land use change authorization system is an effective mechanism to avoid biodiversity loss outside formal protected areas is that SCP makes land use decisions in favour of biodiversity persistence defensible. Greater absolute as well as relative avoided loss in Irreplaceable CBAs compared to Optimal CBAs demonstrate that the importance of Irreplaceable CBAs for the persistence of biodiversity is understood and considered within land use change authorizations in Mpumalanga Province.

Further evidence for the effectiveness of land use decision-making guiding destructive land use changes away from biodiversity priorities can also be found in the relative contributions of different types of land use change sectors to loss within CBAs compared to other natural areas. Mining is the leading cause of clearing of natural areas in Mpumalanga (Figure 7A). Even though CBAs are in areas of higher mining pressure than ONAs (Table 6), mining was one of the lowest contributors to loss of Irreplaceable CBAs, while Optimal CBAs had proportionally similar losses to mining to ONAs (Figure 7B).

It has been proposed that international biodiversity targets should be framed around avoided loss (Pressey et al. 2021). There is however a risk that targets aimed at maximizing avoided loss will fail to achieve biodiversity benefits in the same way that area-based protected area targets have failed, because it assumes that all hectares spared translates to equally valuable outcomes for biodiversity persistence (Vincent 2016). There are not yet enough studies quantifying the impact of avoided loss of natural areas on specific elements of biodiversity, but one recent example illustrates that it does not necessarily translate to positive impacts on wildlife populations (Terraube et al. 2020).

SCP provides a powerful, yet simple measure of the importance of a particular area for the persistence of a range of biodiversity features, called irreplaceability. Irreplaceability is a measure of the likelihood of a site being required to meet a particular conservation objective (Margules and Pressey 2000). South Africa's system of considering biodiversity persistence in land use change authorizations provides an example of how irreplaceability can be used to focus conservation actions and policies where biodiversity benefits such as avoided extinctions of species or avoided collapse of ecosystems are more likely to be tangible.

3.4.1. Conclusion

SCP-guided land use decision-making is an effective complementary conservation intervention to protected areas because it focuses conservation action on areas that matter the most for persistence of biodiversity, while avoiding potential conflicts with economic development as far as possible. Protected areas are more effective in terms of relative reductions in land clearing, and therefore they need to established where feasible, but the consideration of maps of biodiversity priorities outside protected areas can achieve near comparable levels of avoided loss in areas that are generally under higher development pressure, and where protected areas are less likely to be established.

South Africa's conservation community has invested over two decades into developing systematic conservation plans, and maps of biodiversity priorities with associated land use guidelines are now being implemented in land use decisions in all nine provinces (Botts et al. 2019). This is the first quantitative study that validates this strategy and provides evidence that legal support for the implementation of SCP can result in real-world benefits to biodiversity.

4. A counterfactual estimate of the contribution of an environmental policy to avoided loss of threatened ecosystems of the Grassland Biome, South Africa

South Africa implemented regulations to reduce land conversion pressure on threatened ecosystems in 2011. This study uses counterfactual methods to estimate the impact of these regulations on 34 threatened ecosystems in the Grassland Biome between 2014 and 2020. A small positive overall effect of 0.2% avoided loss was found, but the estimate is not statistically significant due to large between-ecosystem heterogeneity in avoided loss. The effect of the regulations within individual ecosystems ranged from 21.5% avoided loss to 8.6% higher loss than in matched counterfactuals. Following robustness tests on effect estimates, positive impacts of avoided loss were found in six ecosystems, while six ecosystems had negative impacts. For the rest of the ecosystems it is not certain whether the regulations had any discernible impact. Meta-analysis methods were used to evaluate potential moderators of effectiveness of the regulations. The regulations were most effective where ecosystems were mapped at a fine scale, maps were integrated into multi-sector land use planning, and efforts were made to mainstream awareness of biodiversity priorities into economic sectors. Negative impacts were found where land conversion pressure was the most severe and there was a sole reliance on land use decisions guided by fine scale biodiversity priority maps to avoid loss. In these ecosystems land use decisions led to internal displacement of land conversion pressure which increased loss relative to counterfactuals. The study concludes with recommendations to strengthen threatened ecosystem conservation in South Africa including more investment in incentive-based land conservation, better integration of threatened ecosystems into land use planning, and the development of integrated management plans for threatened ecosystems.

4.1. Introduction

The Red List of Ecosystems (IUCN-CEM 2022) documents ecosystems' risk of collapse and has the potential to guide conservation policy and action in a similar way to the Red List of Species (Rodríguez et al. 2011, Keith et al. 2015). The Red List of Species is considered a key conservation tool (Rodrigues et al. 2006) that helps to identify sites for conservation actions (Venter et al. 2014), informs environmental impact assessments (Meynell 2005), and contributes to monitoring and assessment of the state of biodiversity (Akçakaya et al. 2006). There is evidence that actions guided by the Red List of Species have prevented the extinction of many vertebrate species (Hayward 2011, Hoffman et al. 2010, Young et al. 2014).

While the Red List of Species has been influencing conservation actions for over 50 years (Betts et al. 2020), internationally standardized methods for assessing ecosystems' risk of collapse are a comparatively recent development (Rodríguez et al. 2015, Bland et al. 2017). Hence ecosystem-focused conservation policies have not yet been widely adopted (Alaniz et al. 2019). A recent review of the impacts of the Red List of Ecosystems (RLE) on conservation policy and practice found that the European Union and seven other countries have implemented legislation and other regulatory policies to protect threatened ecosystems, but there is not yet evidence that these regulatory instruments are effective in avoiding ecosystem collapse (Bland et al. 2019).

South Africa independently developed criteria for assessing the threat status of ecosystems in the early 2000s in response to environmental legislation (National Environmental Management: Biodiversity Act 10 of 2004), which implemented special land use change regulations for ecosystems assessed as Critically Endangered or Endangered (Botts et al. 2020, Skowno and Monyeki 2021). A list of ecosystems that are threatened and in need of protection was published in the government gazette in December 2011 (Government of South Africa 2011), before the IUCN RLE criteria were formally established. A process is underway to update the national list of threatened ecosystems to be aligned with the outcome of a reassessment following IUCN RLE criteria (Skowno and Monyeki 2021). Therefore, although South Africa's national Red List of Ecosystems is not yet aligned with international standards, there is an opportunity to assess the impact of a threatened ecosystem-focused conservation policy after a decade of implementation.

South African ecosystem threat status assessments are primarily driven by historical and ongoing conversion of natural areas to other land uses such as agriculture, mining, and infrastructure development (Skowno and Monyeki 2021). To restrain these ongoing pressures on remaining natural areas, land use change in South Africa is regulated through spatial planning and environmental authorization. Local authorities, which are typically municipalities, are required to consider gazetted biodiversity priority areas in the development of Integrated Development Plans (IDPs) and Spatial Development Frameworks (SDFs), which are used to guide multi-sector land use zoning. Section 54 of the Biodiversity Act specifically requires listed threatened ecosystems to be considered in zoning decisions, and that land use guidelines in threatened ecosystems should be framed around avoiding loss and degradation of these areas.

Environmental authorization is required whenever land use change would result in the clearing of more than 300 m^2 of natural vegetation (DEAT 2010). Environmental authorizations are guided by independent Environmental Impact Assessments (EIAs), but the costs of assessments are carried by the applicants. Within Critically Endangered and Endangered ecosystems, no natural vegetation may be cleared without environmental authorization, and therefore in remaining natural areas of these ecosystems, essentially any activity that would result in the destruction of natural vegetation requires an EIA (Government of South Africa 2011). Environmental authorizations in threatened ecosystems also typically require more stringent and costly mitigation, for example through securing biodiversity offsets that are 30 times larger than the area cleared for Critically Endangered ecosystems, and 20 times for Endangered ecosystems (Brownlie et al. 2017).

The purpose of these regulations is to guide destructive land use changes away from remaining natural areas of threatened ecosystems both pre-emptively through planning and zoning, as well as reactively, by making environmental authorization more difficult and costly to obtain. It is however not yet known whether these regulations are effective deterrents to further loss of threatened ecosystems. The objective of this study is therefore to use counterfactual methods to estimate avoided loss of threatened ecosystems as a result of the implementation of these regulations. Counterfactual methods estimate conservation outcomes in the absence of an intervention, by using carefully constructed study designs to account for potential confounding caused by the non-random implementation of interventions (Ferraro 2009). The impact of a conservation intervention can then be estimated as the difference between the observed outcome in areas that received the intervention and a constructed counterfactual scenario where the intervention was not implemented (Baylis et al 2016).

This study is focused on the Grassland Biome, South Africa's most extensive biome, occurring in all nine provinces (Mucina and Rutherford 2006). Two of South Africa's most densely populated provinces, Gauteng and KwaZulu-Natal are situated in the Grasslands Biome, and it supports the cultivation of 60% of South Africa's commercial crops. More than 90% of commercial timber plantations are concentrated in mist-belt grasslands along South Africa's eastern escarpment (SANBI 2013). More than 40% of the biome is already irreversibly modified (Skowno et al. 2021) and for these reasons the biome has the second highest number of threatened ecosystems after the Fynbos Biome (Skowno and Monyeki 2021). The Grasslands Biome was the focus of a major project to mainstream awareness of biodiversity priorities into various economic sectors and encourage alignment of sector activities with land use guidelines for biodiversity priority areas (Ginsburg et al. 2013). It is therefore anticipated that an evaluation of the impact of threatened ecosystem regulations in the Grassland Biome is less likely to be obscured by a protracted implementation phase.

A recent study on land use change in South Africa found that land conversion pressures on remaining intact grasslands have intensified since 2014, but that these pressures are not evenly distributed across the landscape (Skowno et al. 2021). Studies on the impacts of protected areas in avoiding forest clearing in other parts of the world have found avoided loss can vary substantially by individual protected areas, and that these differences can be explained by differences in land conversion pressure as well as governance effectiveness (Nolte et al. 2013, Oldekop et al. 2015, Shah and Baylis 2015, Shah et al. 2021). Therefore, avoided loss is estimated separately for each Critically Endangered and Endangered ecosystem in the Grassland Biome. Meta-analysis of the outcomes for each ecosystem is then used to estimate the overall effect of threatened ecosystem regulations and explore the role of land conversion pressure and governance in the effectiveness of these regulations.

4.2. Methods

4.2.1. Threatened ecosystems

Maps of 34 Critically Endangered and Endangered ecosystems in the Grassland Biome that were gazetted in 2011 were obtained from the South African National Biodiversity Institute's Biodiversity GIS repository (bgis.sanbi.org). These maps were based on the 2006 version of South Africa's National Vegetation Map (Mucina and Rutherford 2006) as well as important areas highlighted in provincial conservation plans.

South Africa's threatened ecosystem criteria at the time allowed for sub-regions of ecosystems that are important for meeting explicit biodiversity targets while also being under high threat to be included in the national list of ecosystems that are threatened and in need of protection. These areas often include mosaics of grassland and forests. Because forests are protected by specific legislation not applicable to other biomes in South Africa (National Forests Act 84 of 1998), the extent of grasslands as mapped in 2006 was used to select only the grassland portions of these areas for evaluation.

4.2.2. Land cover

The Department of Forestry, Fisheries, and the Environment (DFFE) co-ordinates the development of land cover data sets for South Africa that are specifically aimed at tracking loss of natural areas to other land uses (DFFE 2021). Consistent methods of classification and a defined set of land use classes enable comparisons over time, however, there has been some refinements of approaches in more recent data sets. To account for these methodological differences, the data sets were processed following the methods of Skowno et al. (2021) into three classes: natural areas, non-natural areas, and secondary natural areas, which are areas that were in a non-natural condition in earlier datasets but have since had some recovery of native vegetation.

The data sets for 1990, 2014 and 2020 were used in this study, with 1990 used as a pre-intervention reference point, 2014 as a baseline for assessing the impact of the threatened ecosystem regulations, and 2020 for the outcome. There is no data set for 2011, the year that the regulations were implemented, but 2014 is considered a reasonable baseline as it allows for lags in the adoption of threatened ecosystem maps into revised municipal SDFs and land use change authorizations, that may have been granted before 2011 but not yet implemented by then, to be excluded from the analysis.

The drivers of loss of natural areas in grassland ecosystems over the implementation period between 2014 and 2020 was quantified using a simplified six class system based on Skowno et al. (2021).

4.2.3. Unit of observation, sampling approach, and outcome variable

The study area consisted of grasslands within the Free State, Gauteng, KwaZulu-Natal, Limpopo, Mpumalanga and North West provinces as mapped in the 2006 version of the National Vegetation Map (Figure 9). The study area was restricted to these provinces because the remaining provinces did not have any threatened grassland ecosystems. The study area was divided into 100m x 100m (1 ha) grid cells, with the centroid of each grid cell representing a sampling point, or unit of observation. Because the goal of the threatened ecosystems regulations is to avoid the loss of remaining natural areas in threatened ecosystems, sampling points that fell within areas that were in a natural condition in 2014 were used in the analysis, with non-natural and secondary natural areas excluded.

For each threatened ecosystem 5000 observations were randomly sampled from large ecosystems. Some ecosystems with little remaining intact natural areas, or ecosystems with very limited extent did not have 5000 available observation points, and in these ecosystems all available observations were used. Counterfactual observations were sampled from Vulnerable and Least Concern grassland ecosystems within the study area (hereafter collectively referred to as non-threatened grasslands). Although Vulnerable ecosystems are technically also considered threatened, the threatened ecosystem land use regulations do not apply to them, and they are therefore grouped with the control observations.

Land use change outcomes were derived from the 2020 land cover dataset as a binary variable, with observation points that were in areas lost to other land uses by 2020 coded as outcome = 1, and those in areas that remained in a natural condition coded as outcome = 0.

4.2.4. Covariates influencing land use changes

A counterfactual for the effect of threatened ecosystem regulations is an estimate of loss of natural areas in the absence of these regulations. The regulations did not apply to the ecosystems included under non-threatened grasslands, and therefore loss of natural areas in these ecosystems represent a potential counterfactual scenario. However, threatened ecosystems typically occur in areas where land conversion pressure is the highest (Rodríguez et al. 2007). A comparison of predictors of land conversion pressure between threatened and non-threatened grasslands indicated that threatened ecosystems are generally in areas of much higher agricultural potential and higher mining potential. Threatened ecosystems are in areas that are less rugged, less remote, and closer to roads and more densely populated. They are typically more fragmented than non-threatened ecosystems (Table 8).

Therefore, to construct a valid counterfactual, these differences in land conversion pressure need to be considered to obtain an unbiased estimate of the impact of land use regulations. Matching was used to select observations from non-threatened grasslands that are likely to be under similar land conversion pressure as each threatened ecosystem based on variables listed in Table 8. These included eight variables related to land conversion pressure, and two factors that are likely to influence land use change decisions independently of threatened ecosystems regulations. These factors are the provincial decision-making authorities and maps of biodiversity priority areas. Each province has its own land use decision-making authority, known as competent authorities. To account for differences in decision-making practices between different authorities, intervention and control observations were matched only if they occurred in the same province.

Maps of biodiversity priorities that include important areas not only for the persistence of ecosystems, but also species and ecological processes, are also required to be reported in EIAs and considered in land use change authorizations. These maps classify natural areas into a standard set of four categories according to their importance for biodiversity persistence and each category has specific compatible land use guidelines that must be considered in land use change decisions (Driver et al. 2017, Table 8). It was therefore necessary to also match intervention and control observations on biodiversity priority category. This means that this study evaluates the impact of threatened ecosystem regulations separately from the effect of biodiversity priority maps (the latter were evaluated in Chapter 3).

There are many variables that explain land conversion pressure within grasslands in general (Table 8), however, not all of these were locally relevant within specific threatened ecosystems. Therefore, for each threatened ecosystem, a subset of these

TABLE 8 Covariates of land conversion pressure and land use change authorizations used in matching for threatened grassland ecosystems, indicating differences in mean values between threatened and non-threatened ecosystems for continuous and ordinal variables. Categorical variables where exact matching was required are indicated by *. For these variables, differences in means could not be calculated.

Variable group	**Variable**	**Definition, data source and rationale**	**Mean value in threatened ecosystems**	**Mean value in non-threatened ecosystems**
Land conversion pressure	Agricultural potential	The Land Capability classification system indicates the agricultural potential of an area in 15 classes, with 1 indicating lowest suitability for rangelands and crop cultivation and 15 indicating highest suitability. Source: Agricultural Research Council – Institute for Soil, Climate and Water. Rationale: Land conversion pressure is likely to be higher in areas with higher agricultural potential (Curtis et al. 2018, Shah et al. 2021).	7.2	6.6
	Mining potential	Mining potential was quantified in four categories for South Africa, based on criteria related to size of known mineral deposits and their economic importance (Rouget et al. 2004). Source: National Spatial Biodiversity Assessment 2004 http://bgis.sanbi.org/nsba. Rationale: Mining is of major importance to the South African economy. Pressure for mining expansion is likely to be higher in areas with high mining potential.	0.7	0.4
	Ruggedness	An Index of Terrain Ruggedness was calculated from a digital elevation model following the method of Riley et al. (1999). Higher values indicate increasing ruggedness. Source: 90m resolution Shuttle Radar Topography Mission (SRTM) Digital Elevation Model. Rationale: Land development pressure is likely to be highest in areas with lowest ruggedness, as development costs may be higher in more rugged terrain (Shah et al. 2021).	17.9	22.1
	Remoteness	Distance in kilometres from non-natural areas as mapped in the 2014 national land cover data set. Source: Department of Forestry, Fisheries and the Environment https://egis.environment.gov.za/sa_national_land_cover_datasets.	0.4	0.6

		Rationale: Land conversion is known to be highly spatially autocorrelated (Overmars et al. 2003): loss of natural areas is therefore most likely within the immediate vicinity of existing other land uses such as crop fields, plantations, mines, or built-up areas.		
	Distance from road	Distance in kilometres to the nearest major road. Source: NGI Topo Data http://www.cdngiportal.co.za/cdngiportal/ Rationale: Land development is more likely in accessible areas, therefore sampling points closer to roads are likely to be under higher development pressure than those further away from roads (Bowker et al. 2017).	0.8	0.9
	Local intactness	The % of pixels within a 250 m radius around the observation point that is in a natural condition in 2014. Source: As for remoteness. Rationale: Threatened ecosystems are often more fragmented than non-threatened ecosystems (Rodríguez et al. 2007). Highly fragmented natural landscapes are an indication of historical and potentially ongoing high land conversion pressure.	83	90
	Population density	The number of residents per km^2. Source: Statistics South Africa 2011 National Census data. Rationale: Land conversion pressure is likely to be higher in more densely populated areas.	80	30
	Dominant local land use*	The dominant land use class within a 2 km radius around the observation point in 2014. Source: As for remoteness and local intactness. The data set was reclassified into a simplified six class system based on Skowno et al. (2021). Rationale: The main purpose of this variable is to account for potential unmeasured patterns and drivers of land use change associated with specific existing land use types and therefore exact matching was required on this variable.	Not applicable (categorical variable)	
Land use change authorizations	Decision-making authority*	The province within which the observation point is located. Rationale: Each province has its own decision-making authority responsible for granting environmental authorizations. Differences in decision-making practices between these authorities could explain differences in outcomes for ecosystems.	Not applicable (categorical variable)	
	Biodiversity priority category	Biodiversity priority categories as mapped in provincial systematic conservation plans. Categories were Critical Biodiversity Areas (CBAs), Ecological Support Areas (ESAs), other natural areas (ONAs) and protected areas (PAs).	Not applicable (categorical variable)	

	Sources: Free State Biodiversity Plan (2015): Department of Economic, Small Business Development, Tourism and Environmental Affairs (DESTEA) Gauteng Conservation Plan 3.3 (2011): Gauteng Department of Agriculture and Rural Development (GDARD) KwaZulu-Natal Systematic Conservation Plan (2012): Ezemvelo KZN Wildlife Limpopo Conservation Plan (version 2, 2013): Limpopo Department of Economic Development, Environment & Tourism (LEDET) Mpumalanga Biodiversity Sector Plan (2014): Mpumalanga Tourism & Parks Agency (MTPA) North West Biodiversity Sector Plan (2015): Department of Rural, Environment and Agricultural Development (READ) Rationale: Decision-making authorities are required to consider biodiversity priority categories and their associated land use guidelines in granting environmental authorizations

variables was selected for matching (see Supplementary Materials in Appendix III for a description of the method used to select variables). Exact matching was done on three categorical variables, dominant local land use, province, and biodiversity priority category, and these three variables were included in the matching variable selection for all threatened ecosystems. Robustness tests for the impact of variable selection on effect estimates were performed following the recommendations of Desbureaux (2021). The Supplementary Materials in Appendix III provide additional details.

4.2.5. Matching

Statistical matching eliminates systemic differences in confounding covariates between intervention and control samples that may bias estimates of the impact of conservation interventions when they are not randomly implemented (Schleicher et al. 2020). Intervention observations in threatened ecosystems were matched to similar control observations in non-threatened grasslands using the R package MatchIt (Ho et al. 2011).

When matching for spatial conservation interventions, it is necessary to consider potential displacement effects of such interventions, where impacts that are avoided within intervention zones are shifted to immediate surrounding areas (Ewers and Rodrigues 2008, Heilmayr et al. 2020). Where these displacement or spillover effects are present, study designs need to control for potential biases in impact estimates due to these effects (Schleicher et al. 2020). There was no evidence that restrictions on land cover change displaced land conversion pressure to the immediate vicinity of threatened ecosystems, and therefore no spatial restrictions were placed on the sampling of matched controls from non-threatened ecosystems (see Supplementary Materials in Appendix III for details).

Covariate balance post-matching was assessed on standardized mean difference (SMD), variance ratios and empirical cumulative density functions (eCDF). Different matching methods were tested until SMDs for all covariates were <0.1. When SMD is <0.1, effect sizes can be estimated on the matched samples without further need to include covariates (Nguyen et al. 2017). Nearest neighbour matching on Mahalanobis distance provided better balance than propensity score matching, and therefore Mahalanobis was used as the distance metric.

Matching intervention observations to multiple control observations can reduce bias and increase robustness of effect size estimates (Linden and Samuels 2013). Since ecosystem samples were relatively small, and the outcome of loss of natural areas very sparse within the data set (4.5% of observations lost by 2020), each observation within a threatened ecosystem was matched to three observations within non-threatened ecosystems. For most threatened ecosystems, the region of common support with non-threatened ecosystems was large enough that no observations needed to be discarded for lack of good matches, however, threatened ecosystem observations were generally clustered towards the more extreme end of land conversion pressure within non-threatened ecosystems, and therefore although the pool of available observations for non-threatened ecosystems was eight times larger than the threatened ecosystem samples, there were relatively few observations within the same variable combination space. Therefore, matching with replacement had to be used to obtain well-balanced matches. If three to one nearest neighbour matching with replacement did not result in SMDs <0.1 for all covariates, variable-specific calipers were applied to remaining variables with SMDs >0.1. Population density was the most difficult variable to balance for most ecosystems. It is highly variable, ranging from 1 resident per km^2 to 40 730 per km^2. Using a strict caliper on population density often resulted in many treatment observations being discarded, and therefore the criteria for a good match on population density was relaxed to SMD <0.25. This compromise still resulted in relatively good balances, with the difference in means between threatened ecosystems and their matched samples being less than 20 residents per km^2. Post-matching balance assessments and calipers used for each ecosystem are summarized in Supplementary Table S4 (Appendix III).

4.2.6. Estimation of treatment effect using post-matching regressions

The impact of threatened ecosystems regulations was estimated as the absolute percentage point difference in loss of natural areas within threatened ecosystems and matched observations in non-threatened ecosystems between 2014 and 2020. Linear probability models (Uanhoro et al. 2021, Huang 2022) were fitted to matched samples with intervention or control designation as a binary predictor and loss to other land uses by 2020 as a binary outcome. Weights were included within model specifications to account for multiple controls matched to each intervention observation.

Coefficient errors were estimated using cluster robust standard errors (CRSE) to account for sampling with replacement as well as pair membership of observations. It has been shown that CRSEs provide more accurate error estimates for these data structures typically encountered in matched samples than regular methods (Austin 2009).

4.2.7. Meta-analysis of differences in outcomes among threatened ecosystems

Evidence for the effectiveness of threatened ecosystem regulations was summarized using meta-analysis methods. These methods are typically employed to assess evidence across a range of independently published studies, but also offer some advantages over a single overall point estimate of effectiveness when there is substantial variation in outcomes across study units, which in this analysis are individual threatened ecosystems. Primarily, variance is partitioned into two types: the first type is uncertainty around the effect size estimates for individual studies (or ecosystems in this case) and is used to inversely weight the contributions of each effect estimate in the calculation of the overall effect. Thus, more uncertain estimates contribute less to the overall estimate than more precise estimates (Hardy and

Thompson 1998). The second type, between study (or between ecosystem) variability in effect size is quantified separately, with large heterogeneity (>50% of overall variance) providing evidence that external factors may be driving differences in outcomes between ecosystems (Harrer et al. 2021, Song et al. 2001).

Between ecosystem heterogeneity in point estimates of avoided loss was quantified using the Sidik-Jonkman estimator (Sidik and Jonkman 2005) for large τ^2 (effect size variance). A random-effects model, which considers both within and between ecosystem variability in effect size was used to estimate the overall or pooled effect of threatened ecosystem regulations, with Knapp-Hartung adjustments for small samples (Knapp and Hartung 2003) used to calculate the confidence intervals around the pooled effect. Meta-analyses were performed using the R packages meta (Balduzzi et al. 2019), metafor (Viechtbauer 2010) and dmetar (Harrer et al. 2021), following the methods described in Harrer et al. (2021).

Hierarchical partitioning (Chevan and Sutherland 1991) was used to explore possible moderators of avoided loss between threatened ecosystems. Hierarchical partitioning quantifies the independent contribution of each predictor variable towards explaining the outcome variable in a multivariate regression model. The difference in goodness-of-fit of a model containing the variable compared to a model without the variable is calculated across all possible variable combinations, and then averaged to arrive at an importance score for each variable (Mac Nally 2002). This method is recommended when it is not certain which variables among a range of possible predictors are the most influential in explaining a dependent variable (Mac Nally 2000). Hierarchical partitioning was performed using the R package hier.part (Mac Nally and Walsh 2004).

Previous studies of variation in the impact of spatial conservation interventions found that uneven land conversion pressure and governance explained differences in outcomes, and therefore a selection of ecosystem-level variables relating to these factors were tested as possible modifiers of the effectiveness of ecosystem regulations (Table 9). Differences in governance among implementation authorities were quantified by assessing uptake of threatened ecosystems into municipal and metro SDFs and IDPs, as well as which provinces each ecosystem was located in. The potential effect of mainstreaming efforts by the Grasslands Programme was also considered. Lastly, avoided loss was assessed as a percentage difference at a standard spatial scale, while threatened ecosystems varied substantially in size (18 km^2- 22 741 km^2). Ecosystem extent was therefore included as a possible confounder because the scale at which loss was measured is likely to bias effect estimates more in smaller ecosystems than larger ones (Avelino et al. 2016).

TABLE 9 Potential moderators of avoided loss in grassland threatened ecosystems.

Variable group	Variable	Definition and data source	Rationale
Land conversion pressure	Pre-intervention land conversion pressure	Percentage of natural areas in ecosystem that were lost to other land uses between 1990 and 2014. Source: Department of Forestry, Fisheries, and the Environment https://egis.environment.gov.za/sa_national_land_cover_datasets	Higher overall land conversion pressure is likely to be correlated with percentage avoided loss estimates (Nolte et al. 2013).
	Formal protection	Percentage of ecosystem protected by 2011. Source: national protected area time series data sets compiled for Chapter 2.	Formal protection substantially reduces land conversion pressure as most destructive land use changes are legally prohibited within protected areas.
Governance	Land use change decision-making authority	Percentage of ecosystem falling within each of the provinces containing threatened grassland ecosystems.	Differences in decision-making practices between these authorities could explain differences in outcomes for ecosystems.
	IDP and SDF coverage	Percentage of ecosystem falling within local municipalities and metros that included threatened ecosystems within their IDPs or SDFs within five years of gazetting of threatened ecosystems. Source: National Treasury Municipal Finance Management Act	Consideration of threatened ecosystems in spatial planning is likely to result in greater avoided loss.

		Documentation http://mfma.treasury.gov.za/	
	Ecosystem threat status	Binary variable indicating whether ecosystem threat status is Critically Endangered (1) or Endangered (0). Source: Government of South Africa (2011).	Remaining natural areas of Critically Endangered ecosystems are already less than what is required by their conservation targets. Therefore, all further loss within Critically Endangered ecosystems needs to be avoided. This variable tests whether the ecosystem's threat status affects land use decisions.
	Grasslands Programme mainstreaming	Percentage of ecosystem falling within the direct spatial footprint of the Grasslands Programme mainstreaming efforts. Source: Grasslands Programme.	Greater awareness of biodiversity priorities within economic sectors is likely to lead to better alignment of sector activities with management guidelines for threatened ecosystems.
Potential confounder	Ecosystem size	Ecosystem size in km^2 according to gazetted maps. Source: https://bgis.sanbi.org/SpatialDataset/Detail/501	The spatial scale at which ecosystem loss between 2014 and 2020 was measured is likely to have a larger impact on effect size estimates in smaller ecosystems than larger ones (Avelino et al. 2016).

All predictor variables were standardized to z-scores and tested for collinearity. No substantial collinearity was present other than between ecosystem size and percentage of ecosystem in the Free State (r = 0.91). The Free State Province contains the majority of the largest threatened grassland ecosystem, Vaal-Vet Sandy Grassland (VSG in Figure 9). Variable importance was assessed using a linear model with the point estimate of effect size as the outcome variable, as well as an ordinal logistic regression on impact categories adjusted for sensitivity to sampling and matching model specifications. See Supplementary Table S8 in Appendix III for definitions of the impact categories.

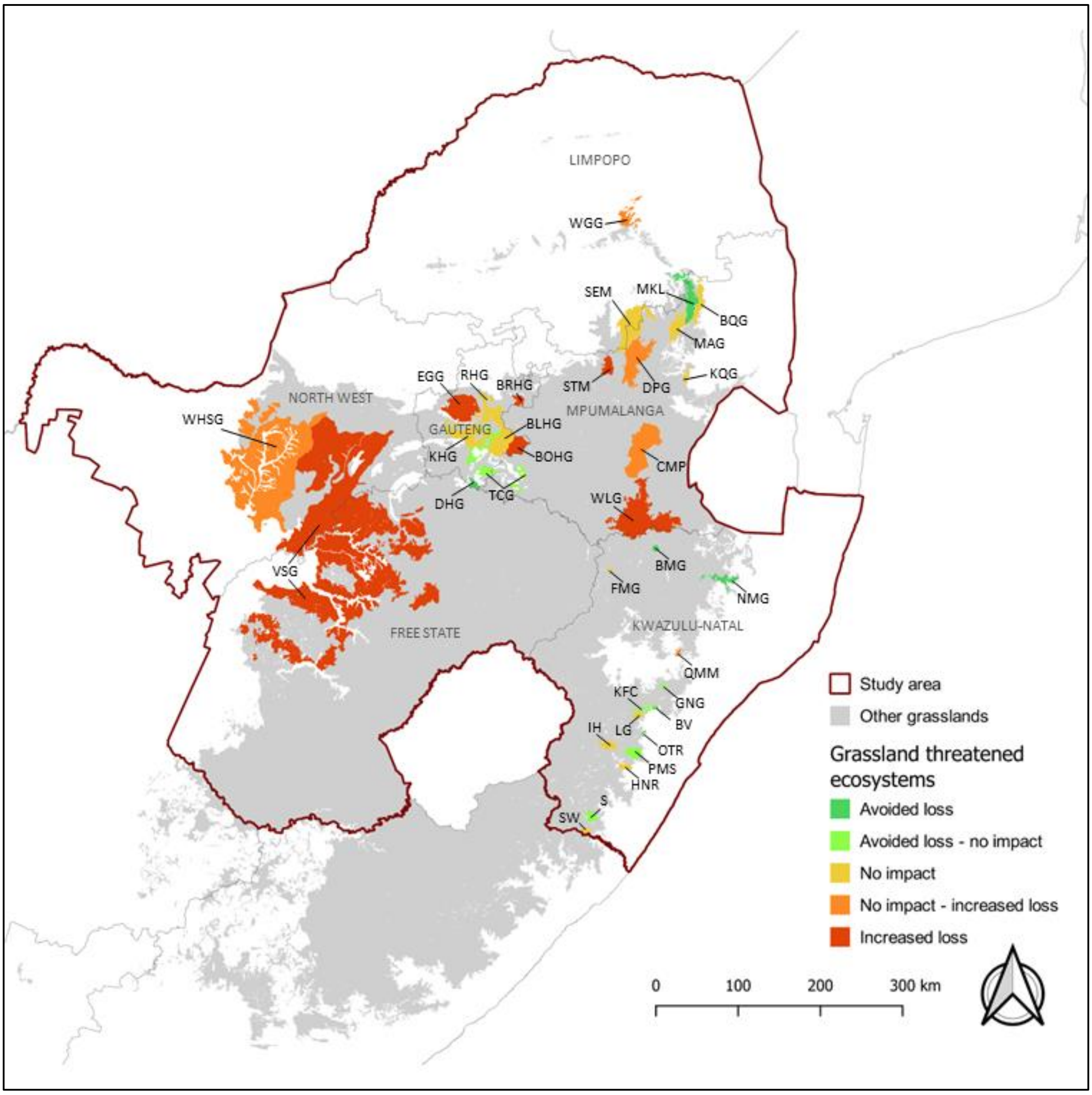

FIGURE 9 Map of the study area indicating the location of threatened ecosystems within the Grassland Biome. Ecosystem categories indicate the impact of threatened ecosystem regulations on grassland threatened ecosystems, adjusted for uncertainty due to sampling and matching model specifications. For a key to ecosystem abbreviations, see Figure 10.

4.3. Results

4.3.1. Effectiveness of threatened ecosystem regulations

Between 2014 and 2020, approximately 6.2% of remaining natural areas of threatened grassland ecosystems were lost to other land uses. In comparison, only 4.3% of non-threatened grasslands were lost over the same period. There was considerable variation in observed loss within individual threatened ecosystems, with 0.5% in Mauchesburg Alpine Grasslands, up to 26.3% in Egoli Granite Grassland (Figure 10).

Point estimates of the effect of threatened ecosystem regulations on land conversion in individual ecosystems ranged from 21.5% avoided loss in Oakland and Townhill Ridge, to 8.6% more loss than in the matched counterfactual for Egoli Granite Grassland (Figure 10). When sensitivity of effect estimates to sampling and matching model selection is considered, a positive impact of ecosystem regulations is confirmed in six ecosystems. Four of these are in KwaZulu-Natal, and one each in Mpumalanga and Gauteng (Figure 9).

In six ecosystems, observed loss between 2014 and 2020 was significantly higher than in their matched counterfactuals, and robustness tests indicated that this is likely to be a true effect. Increases were mostly small (1-2%), except in Bronkhorstspruit Highveld Grassland and Egoli Granite Grassland, which both had more than 8% higher loss (Figure 10). Both these ecosystems are in Gauteng, South Africa’s most densely populated and economically most productive province, where land conversion pressure is the most severe (Figure 9).

For the rest of the ecosystems, it is not certain whether threatened ecosystem regulations had any discernible impact on land conversion. The impact of regulations was not better within Critically Endangered ecosystems, where it is most important that any further loss should be avoided, than in Endangered ecosystems. Instead, loss was most often avoided in smaller ecosystems, regardless of their status, while more extensive ecosystems had worse outcomes (Figure 11).

The overall effect of the regulations, estimated by pooling the effect estimates of individual ecosystems, indicate avoided loss of 0.2%, but it is not statistically significant (Figure 10). Meta-analysis revealed very high between-ecosystem heterogeneity in effect estimates (I^2 = 91%), indicating that the effectiveness of

Ecosystem	Sample	Coefficient	Error	p-value	Observed loss (%)	Counterfactual (%)	Effect size (%)	95% CI	Weight
Oakland and Townhill Ridge (OTR)	143	-0.2145	0.0353	***	8.33	31.04	-21.45	[-28.37; -14.53]	2.1%
Blinkwater Valley (BV)	312	-0.1378	0.0323	***	5.45	19.23	-13.78	[-20.11; -7.45]	2.2%
Deneysville Highveld Grassland (DHG)	2770	-0.0414	0.0143	**	5.99	10.13	-4.14	[-6.94; -1.34]	3.0%
Tsakane Clay Grassland (TCG)	5000	-0.0329	0.0485	ns	7.17	10.49	-3.29	[-12.80; 6.22]	1.5%
Loskop Grasslands (LG)	1578	-0.0314	0.0682	ns	13.69	16.83	-3.14	[-16.51; 10.23]	1.0%
Pietermaritzburg South (PMS)	5000	-0.0295	0.0242	ns	4.42	7.33	-2.95	[-7.69; 1.79]	2.6%
Karkloof Forest Collective (KFC)	2314	-0.023	0.0143	ns	1.90	4.21	-2.30	[-5.10; 0.50]	3.0%
Bivane Montane Grassland (BMG)	4723	-0.02	0.0024	***	1.21	3.21	-2.00	[-2.47; -1.53]	3.3%
Greytown North Grasslands (GNG)	874	-0.0118	0.0198	ns	21.28	22.46	-1.18	[-5.06; 2.70]	2.8%
Ngome Mistbelt Grassland and Forest (NMG)	5000	-0.0117	0.0040	**	4.02	5.17	-1.17	[-1.95; -0.39]	3.3%
Malmani Karstlands (MKL)	5000	-0.0095	0.0017	***	0.72	1.67	-0.95	[-1.28; -0.62]	3.3%
Highover Nature Reserve and Roselands Farm Surrounds (HNR)	708	-0.0066	0.0077	ns	3.11	3.77	-0.66	[-2.17; 0.85]	3.2%
Sihleza (S)	5000	-0.006	0.0094	ns	3.86	4.42	-0.60	[-2.44; 1.24]	3.2%
Fort Metcalf Grasslands (FMG)	2866	-0.0048	0.0025	ns	0.66	1.14	-0.48	[-0.97; 0.01]	3.3%
Rietvleiriver Highveld Grassland (RHG)	5000	-0.0044	0.0379	ns	12.35	12.82	-0.44	[-7.87; 6.99]	1.9%
Southern Weza State Forest (SW)	4009	-0.0033	0.0035	ns	1.37	1.70	-0.33	[-1.02; 0.36]	3.3%
Mauchesburg Alpine Grasslands (MAG)	5000	-0.0017	0.0015	ns	0.54	0.71	-0.17	[-0.46; 0.12]	3.3%
Sekhukhune Mountainlands (SEM)	5000	-0.0016	0.0023	ns	1.89	2.06	-0.16	[-0.61; 0.29]	3.3%
Blyde Quartzite Grasslands (BQG)	5000	-0.0003	0.0019	ns	1.04	1.07	-0.03	[-0.40; 0.34]	3.3%
Impendle Highlands (IH)	5000	0.0051	0.0101	ns	2.92	2.41	0.51	[-1.47; 2.49]	3.2%
Qudeni Mountain Mistbelt Forest and Grassland (QMM)	2567	0.0055	0.0023	*	1.17	0.62	0.55	[0.10; 1.00]	3.3%
Kaapsehoop Quartzite Grasslands (KQG)	2811	0.0058	0.0035	ns	2.06	1.48	0.58	[-0.11; 1.27]	3.3%
Wakkerstroom-Luneburg Grasslands (WLG)	5000	0.0095	0.0030	**	3.89	2.93	0.95	[0.36; 1.54]	3.3%
Vaal-Vet Sandy Grassland (VSG)	4996	0.0117	0.0036	**	5.95	4.82	1.17	[0.46; 1.88]	3.3%
Klipriver Highveld Grassland (KHG)	5000	0.0144	0.0097	ns	21.55	20.06	1.44	[-0.46; 3.34]	3.2%
Boesmanspruit Highveld Grassland (BOHG)	5000	0.0155	0.0052	**	7.24	5.63	1.55	[0.53; 2.57]	3.3%
Blesbokspruit Highveld Grassland (BLHG)	5000	0.0164	0.0215	ns	11.52	9.86	1.64	[-2.57; 5.85]	2.7%
Dullstroom Plateau Grasslands (DPG)	5000	0.0195	0.0083	*	4.35	2.43	1.95	[0.32; 3.58]	3.2%
Stoffberg Mountainlands (STM)	5000	0.0233	0.0041	***	8.62	6.27	2.33	[1.53; 3.13]	3.3%
Woodbush Granite Grassland (WGG)	4939	0.0271	0.0039	***	3.10	0.40	2.71	[1.95; 3.47]	3.3%
Chrissiesmeer Panveld (CMP)	5000	0.0323	0.0307	ns	9.06	5.83	3.23	[-2.79; 9.25]	2.3%
Western Highveld Sandy Grassland (WHSG)	4992	0.0366	0.0107	***	7.68	3.97	3.66	[1.56; 5.76]	3.2%
Bronkhorstspruit Highveld Grassland (BRHG)	4184	0.0804	0.0122	***	14.29	6.25	8.04	[5.65; 10.43]	3.1%
Egoli Granite Grassland (EGG)	4999	0.0864	0.0222	***	26.31	17.43	8.64	[4.29; 12.99]	2.7%
Pooled effect size from random effects model							**-0.19**	**[-1.80; 1.42]**	**100.0%**
Prediction interval								**[-9.46; 9.08]**	

Heterogeneity: $I^2 = 91\%$, $p < 0.01$

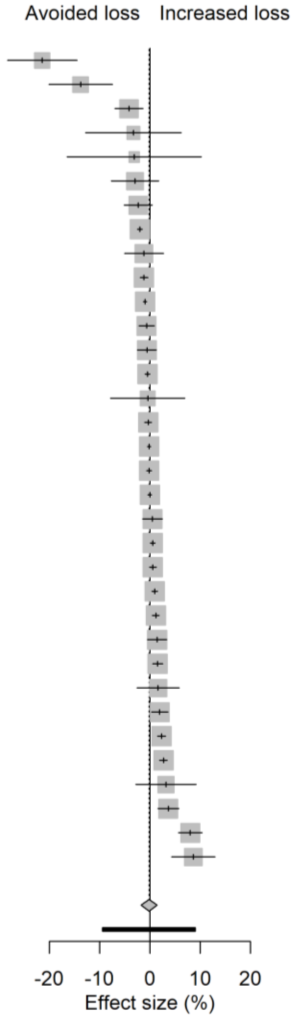

FIGURE 10 Forest plot of the range of point effect estimates for the impact of threatened ecosystem regulations on grassland ecosystems. The size of grey squares around point estimates indicates the weight given to the ecosystem in the calculation of the pooled effect size. Sample indicates the number of observations for the ecosystem that was matched to similar observations in non-threatened grasslands. The location of each ecosystem within the Grassland Biome is indicated by its abbreviation in Figure 9. P-value codes are *** <0.0001, ** <0.001, * <0.05, ns >0.05.

threatened ecosystem regulations is strongly dependent on the individual circumstances of each ecosystem. These results suggest that it is important to investigate the conditions under which the regulations are effective, and when they fail.

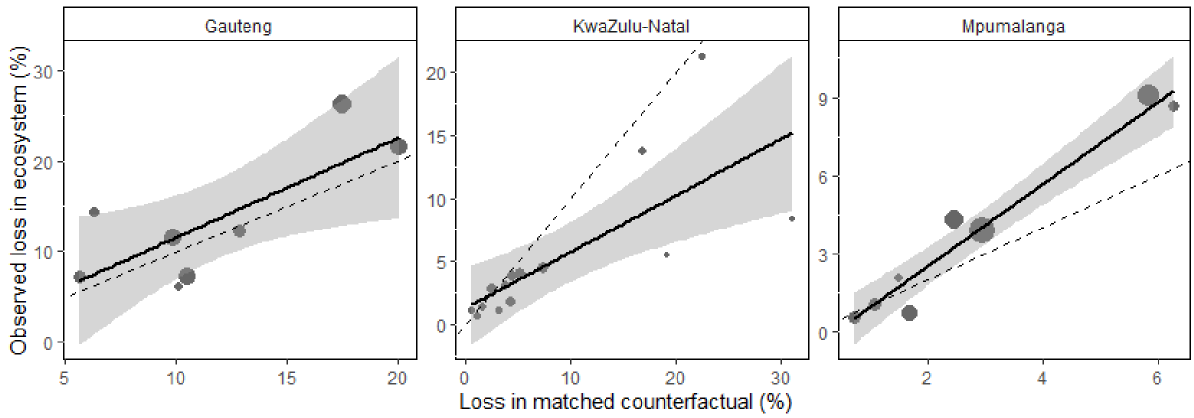

FIGURE 11 Trends in effectiveness of threatened ecosystem regulations by province and ecosystem extent. Size of dots indicates the relative size of the ecosystem in km^2. Dashed diagonal lines indicate the threshold of no impact, with dots below the line representing ecosystems where loss was avoided (observed loss < counterfactual), while dots above the line indicate negative impacts (observed loss > counterfactual). Dots furthest away from the dashed line indicate the largest impacts. The shaded area around the trend line indicates 95% confidence intervals. Other provinces had too few data points to fit trend lines.

4.3.2. Variables explaining differences in threatened ecosystems outcomes

Meta-analysis of the variance in effect estimates between threatened ecosystems clearly indicated that independent, ecosystem-specific circumstances are affecting the effectiveness of threatened ecosystem regulations. Hierarchical partitioning of the independent contributions of potential modifiers of the effectiveness of regulations however did not provide a clear indication of what these circumstances may be.

The linear model suggested that land conversion pressure is the strongest predictor of avoided loss, with ecosystems that had a higher percentage loss before the implementation of the intervention showing stronger positive impacts (Figure 12). The ordinal model on the other hand suggested that ecosystem extent is the strongest predictor of impact, with smaller ecosystems showing more positive impacts than larger ecosystems (Figure 11, Figure 12). Both models ranked provincial distributions of ecosystems highly, indicating that decision-making authorities are playing a role in the effectiveness of the regulations. The linear model ranked KwaZulu-Natal's competent authority as a strong predictor of positive effects, as most ecosystems within KwaZulu-Natal had positive impacts (Figure 11, Figure 12). It also indicated that there is a strong effect within Gauteng, but in this case, an ecosystem located in Gauteng was associated with mostly negative impacts (Figure 11, Figure 12). The ordinal model quantified the contributions of provincial decision-making authorities differently, ranking provinces with ecosystems clustered within one or two impact categories higher than provinces where ecosystems fall within a range of impact categories (Figure 9, Figure 12).

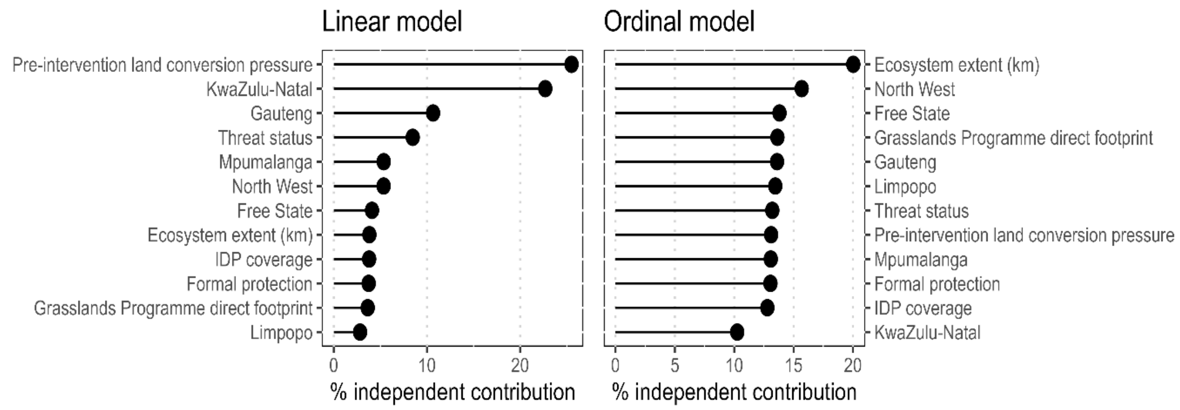

FIGURE 12 The independent contributions of moderator variables towards explaining differences effectiveness of threatened ecosystems regulations in different ecosystems, estimated using hierarchical partitioning. Variables on y axes are ranked from most to least influential.

The ordinal model also suggested that the Grasslands Programme contributed to positive outcomes in threatened ecosystems. The Grasslands Programme had a direct footprint within nine of the 11 ecosystems that had positive impacts (Avoided loss and Avoided loss – no impact). In the 11 ecosystems that had negative impacts (No impact – increased loss and Increased loss), the Grasslands Programme had a direct footprint in only three of these, and the spatial impact of the footprint was relatively smaller than in ecosystems where impacts were better. The analysis of potential modifiers of the impact of regulations in different ecosystems was limited by a small sample size. Further study of the effect of regulations in a broader range of ecosystems is needed to gain clearer insights.

4.3.3. Drivers of land use changes in the Grassland Biome 2014-2020

Between 2014 and 2020, the major causes of loss of natural areas in the Grassland Biome were agricultural expansion (50% of total loss), expansion of settlements and infrastructure (17% of total loss), and expansion of timber plantations (15% of total loss). These were also the main drivers of loss within threatened ecosystems (Figure 13). In ecosystems that had relatively better impact estimates (Avoided loss and Avoided loss – no impact) observed loss was mainly to plantations and settlements (Figure 13). On the other hand, in ecosystems where the regulations had the least impact (No impact – increased loss and Increased loss) most land conversion was due to agricultural expansion (Figure 13), except for two ecosystems, where loss was predominantly as a result of the expansion of settlements.

Land use regulations may fail to control loss of natural areas when the drivers of loss are bypassing spatial planning and environmental authorizations, such as expansion of informal settlements. Therefore, to gain a better understanding of why the regulations failed in some ecosystems, loss to agriculture and settlements were also assessed by subclasses within these main drivers of loss (Supplementary Figure S2, Appendix III). In ecosystems where the regulations failed to reduce land conversion pressure, loss to informal settlement expansion was a relatively larger proportion of the total loss, but it was not the main cause (Supplementary Figure S2, Appendix III). Instead, most of the loss to settlements and infrastructure development was shared between formal urban expansion and expansion of rural settlements in all impact classes. While expansion of rural settlements may also be less well regulated, there was no indication that it is more strongly associated with ecosystems where the regulations failed to have an impact (Supplementary Figure S2, Appendix III).

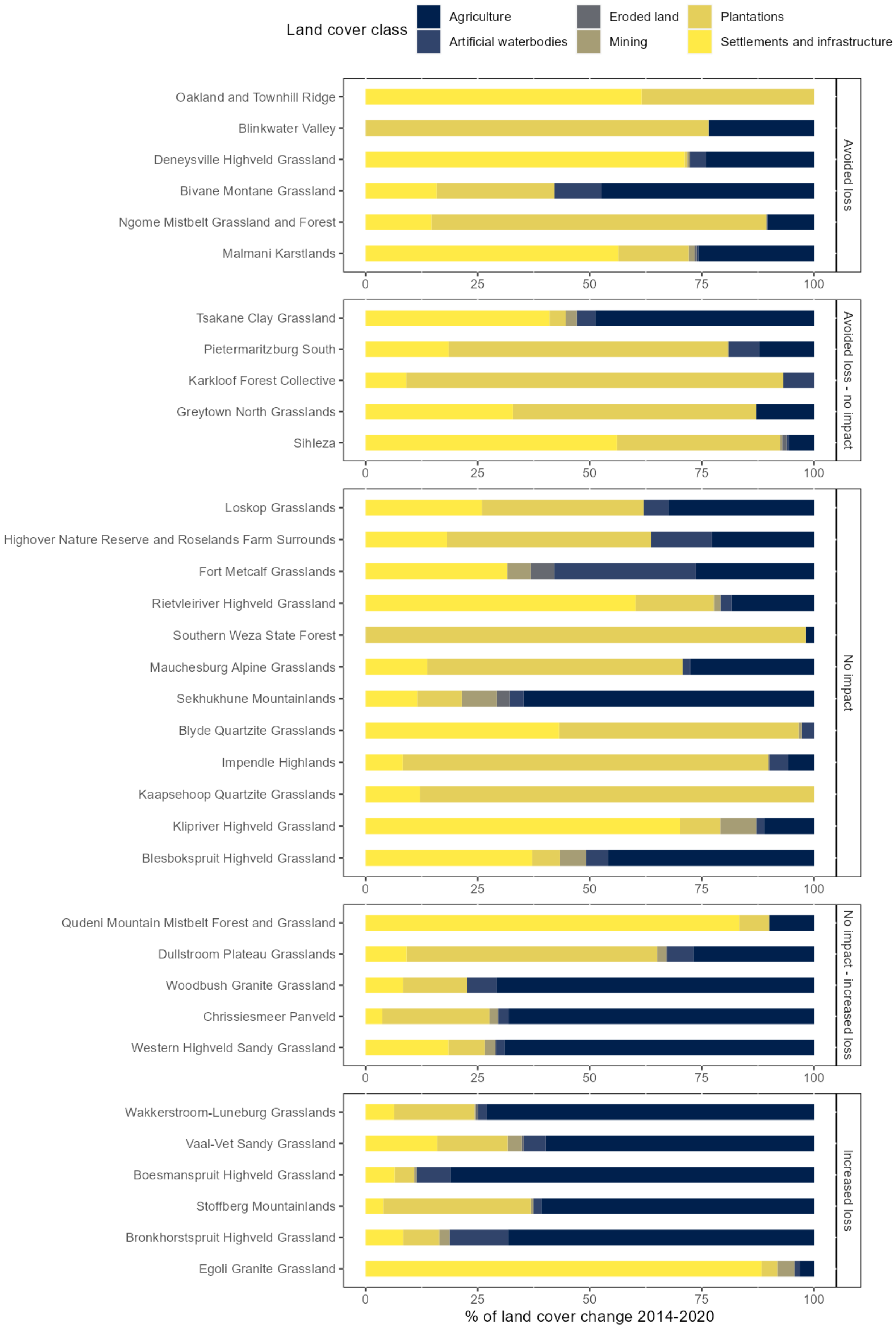

FIGURE 13 The main causes of loss of natural areas within threatened grassland ecosystems between 2014 and 2020. Ecosystems are grouped by the effectiveness of threatened ecosystem regulations within each ecosystem during this period.

Similarly with agriculture, less well-regulated expansion of subsistence agriculture was not associated with land cover change in ecosystems where the regulations failed, and it appears that the loss of natural areas in these ecosystems is mainly driven by expansion of commercial agriculture (Supplementary Figure S2, Appendix III).

4.4. Discussion

The overall estimate of the contribution of environmental regulations to avoided loss of threatened ecosystems in the Grassland Biome indicates that it had no significant impact on land conversion in these ecosystems. However, assessing the impact of the regulations at this level only risks overlooking local circumstances where the regulations were effective, and does not provide insights into the conditions that led to the regulations failing to have a measurable effect.

Most counterfactual studies of the impact of spatial conservation interventions have found similar variable effects in individual spatial units (Nolte et al. 2013, Shah and Baylis 2015, Shah et al. 2021, West et al. 2020). Nolte et al. (2013) for example found that the effectiveness of protected areas in avoiding deforestation in the Brazilian Amazon depends on land conversion pressure as well how strictly protection is enforced. At low land conversion pressure, less strict mixed-use protection has similar impacts to strictly enforced protected areas, but at higher land conversion pressure less strict protection has substantially lower effectiveness than strict protection. Threatened ecosystems occur in areas of relatively higher land conversion pressure, but regulations do not limit land use change as strictly as in protected areas. Following the findings of Nolte et al. (2013), this may partially explain the apparent overall ineffectiveness of the regulations, but the trend is not consistent and depends on the location of the ecosystem. In Gauteng and Mpumalanga, the regulations were somewhat effective in areas with lower land conversion pressure but failed in areas of high land conversion pressure (Figure 11). In KwaZulu-Natal however, the regulations had a stronger impact in ecosystems where land conversion pressure is the highest (Figure 11).

Several factors set KwaZulu-Natal apart from other provinces where threatened grassland ecosystems occur. The province had the most extensive uptake of threatened ecosystems in municipal IDPs and SDFs of all six provinces, with only two out of 14 ecosystems in the province not considered in these multi-sector spatial planning processes. Gauteng also had relatively good representation of threatened ecosystems in IDPs and SDFs, but evidence for compliance in other provinces was poor. The Grasslands Programme also implemented wide-ranging mainstreaming efforts within KwaZulu-Natal, focusing mainly on the plantation forestry sector. Only three threatened ecosystems within the province had no direct mainstreaming impacts. Other than Chrissiesmeer Panveld in Mpumalanga, which was the focus of efforts to implement a large landowner cooperative protected environment, the Grasslands Programme had a much smaller spatial footprint in grassland threatened ecosystems outside KwaZulu-Natal. Lastly, the provincial conservation authority opted to define all their listed threatened ecosystems at a finer scale than the national vegetation map, which meant that the implementation of the regulations was focused on areas where they were most needed. Analyses of moderators of the effectiveness of ecosystem regulations were not able to discern the influence of some of these factors (Figure 12). This is most likely because it is the combination of factors that explain the effectiveness of the regulations in KwaZulu-Natal, rather than their independent contributions.

It is possible that the spatial scale at which land cover change is measured may bias effect estimates in small ecosystems, because one hectare of avoided loss is a much larger proportion of an ecosystem of only a few hectares in extent, compared with an ecosystem that is several thousand square kilometres in size (Avelino et al. 2016). The two ecosystems with the largest absolute point effect estimates of avoided loss (Oakland and Townhill Ridge and Blinkwater Valley) are also the two smallest ecosystems (Figure 10), and these are therefore potential outliers due to spatial scale (Figure 11). It does not appear, however, that large negative effect estimates were due to spatial scale, because the largest effects were found in some of the largest ecosystems (Figure 9, Figure 11).

It is not uncommon for counterfactual studies to uncover effects that are the opposite of the intention of conservation interventions (Nolte et al. 2013, Wauchope et al. 2022, West et al. 2020). Sometimes these effects are due to unforeseen threats that disproportionately affect areas that received the intervention. For example, West et al. (2020) found that large forest fires inside protected areas led to negative deforestation impacts relative to counterfactuals. In other studies, aspects of the implementation of interventions failed to effectively mitigate threats. For example, Wauchope et al. (2022) found that waterbird populations declined in protected areas relative to counterfactual unprotected populations when protected areas were not specifically managed to support the persistence of suitable habitats for waterbirds.

Threatened ecosystems regulations were not successful in guiding destructive land use changes away from threatened ecosystems in instances where ecosystems are extensive and land conversion pressure is high. The reason for this is the way in which biodiversity priorities are considered in environmental authorizations. Threatened ecosystem regulations are primarily structured to discourage development applications through spatial zoning and by making authorizations more expensive, but when applications are submitted despite these hurdles, threatened ecosystems are then considered alongside maps of biodiversity priorities in environmental authorization decisions. Land use guidelines associated with biodiversity priority maps require to Critical Biodiversity Areas (CBAs) and Ecological Support Areas (ESAs) to remain in a natural or near-natural condition because these areas are needed to meet minimum persistence targets for biodiversity features and ecological processes. Biodiversity priority areas are typically mapped at finer scales than ecosystems.

It was shown in Chapter 3 that land use decisions guided by biodiversity priority maps are effective in avoiding loss of CBAs in Mpumalanga Province and therefore loss of natural areas within threatened ecosystems was assessed by biodiversity priority category (Figure 14A). Most of the remaining intact areas in threatened ecosystems are classified as Critical Biodiversity Areas (CBAs), and except for some ecosystems in Gauteng, loss in CBAs in threatened ecosystems was extremely low, even in ecosystems where the overall effect estimate of the regulations was negative (Figure 14A). Instead, most loss occurred within Other Natural Areas (ONAs) within threatened ecosystems (Figure 14A). Areas classified as ONAs in biodiversity priority maps are not needed to meet biodiversity targets, and land use guidelines associated with ONAs permit destructive land use changes (Driver et al. 2017).

There is therefore evidence that where land conversion pressure is high and threatened ecosystems cover extensive areas of the landscape, biodiversity priority maps are prioritized over ecosystem threat status in land use decisions. The effect of this pattern of decision-making is that land conversion pressure on threatened ecosystems is not reduced, but rather mostly displaced internally within the ecosystems. The preference for biodiversity priority maps elevates land conversion pressure on ONAs in threatened ecosystems relative to ONAs in non-threatened grasslands, and where this effect is strong, it explains the excess loss observed in some threatened ecosystems compared with their counterfactuals (Figure 14B). In Stoffberg Mountainlands, Vaal-Vet Sandy Grassland and Wakkerstroom-Luneburg Grasslands, loss in protected areas, CBAs, and ESAs were similar in the ecosystem and their matched counterfactuals, but loss in ONAs was much higher (Figure 14B). When this effect emerged in the results, another spillover test using the same methods as for buffer zones around protected areas was done, but this time comparing loss in ONAs in threatened ecosystems matched to ONAs that are not in threatened ecosystems. Loss in ONAs inside threatened ecosystems was significantly higher than ONAs that are not in threatened ecosystems (Supplementary Table S3).

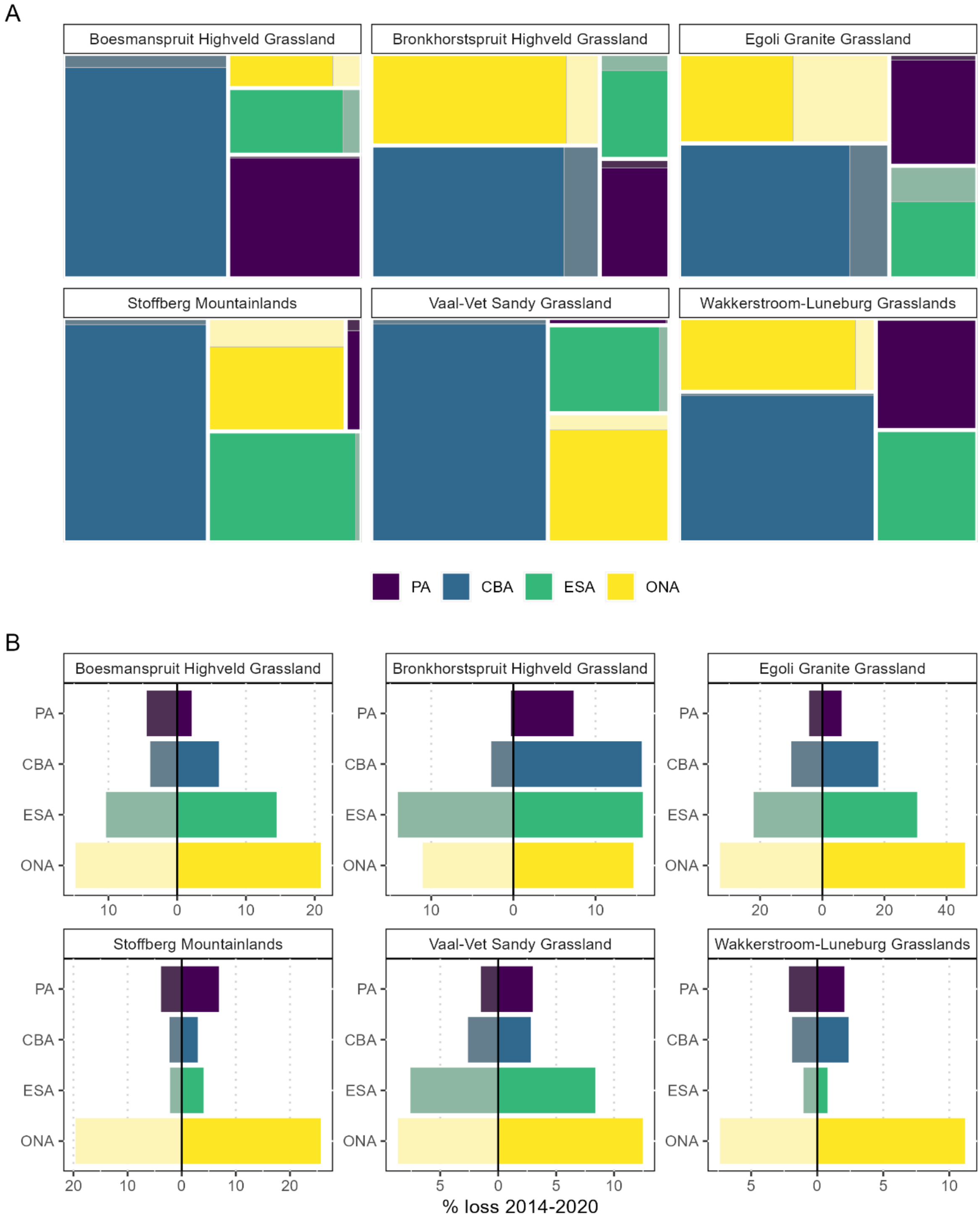

FIGURE 14 Evidence for internal displacement of land conversion pressure in six ecosystems where loss of natural areas between 2014 and 2020 was greater than in matched grassland ecosystems that were not subjected to threatened ecosystem regulations. A. Area of rectangles indicate the relative proportion of the remaining intact areas of the ecosystem in 2014 assigned to each biodiversity priority category. Lighter sections of each rectangle indicate the proportion of the biodiversity priority category lost to other land uses by 2020. B. The percentage loss in each biodiversity priority category in each ecosystem (darker colours to the right of the vertical line at 0) compared to the proportion lost within the matched sample (lighter colours to the left of the vertical line at 0).

Internal displacement of land conversion pressure explains excess loss in threatened ecosystems in Mpumalanga, the Free State and North West, but in Gauteng, loss was higher in all biodiversity priority categories than in matched counterfactuals (Boesmanspruit Highveld Grassland, Bronkhorstspruit Highveld Grassland and Egoli Granite Grassland in Figure 14B). In Gauteng, threatened ecosystems cover large sections of the province (Figure 9), and loss in matched counterfactuals indicates much higher land conversion pressure than in Mpumalanga (Figure 11). From the trends presented in Figure 14, it is clear that land use planning and decision-making processes are not able to reduce pressure on natural areas, it can only displace it. The challenge within Gauteng is that there are limited areas where developments can be displaced to (Figure 9). Therefore, within Gauteng, land use planning and decision-making is likely to be able to avoid the loss of only the highest biodiversity priority areas, such as irreplaceable sites.

4.4.1. Policy recommendations and conclusions

Alaniz et al. (2019) recommend that ecosystem-focused conservation policies should be directed at reducing risks to threatened ecosystems. South Africa’s threatened ecosystem regulations achieved this objective where ecosystems are mapped at a fine scale, the maps are integrated into multi-sector land use planning, and effort has been made to mainstream awareness of biodiversity priorities into economic sectors.

Where these factors are not in place, threatened ecosystems depend on environmental authorizations alone to restrain land conversion pressure. However, because environmental authorizations give precedence to biodiversity priority maps, which document areas most important for the persistence of biodiversity at finer scales than ecosystems, avoided losses due to land use decisions does not necessarily scale up to ecosystem-level impacts. The consideration of biodiversity priorities in land use decisions is working as intended, but the consequences are that these conservation impacts do not contribute towards positive trends in biodiversity indicators tracking the threat status, extent, and ecological integrity of ecosystems over time (Rowland et al. 2020).

There are therefore ways in which threatened ecosystem conservation could be strengthened in South Africa. There is, for example, much potential for stronger engagement with the commercial agricultural sector through mainstreaming programmes, because agriculture was the main driver of loss of natural areas where the threatened ecosystem regulations failed to have an impact. Evans (2016) recommends incentive-based policies rather than increased constraints on land use in situations where there are strong conflicts between economic sectors and conservation needs. South Africa has already had some success with incentive-based management of private and communally owned land for conservation (Cumming et al. 2017). Directing more resources towards incentive-based land conservation is likely to substantially benefit threatened ecosystems.

There is also much need for better integration of threatened ecosystems into multi-sector land use planning and zoning. The Spatial Planning and Land Use Management Act (no. 16 of 2013) requires all municipalities to develop and implement integrated land use schemes for areas under their jurisdiction. Section 24(2)(b) of the act requires that when land use schemes are being developed, applicable environmental management instruments and legislation must be considered. This encourages a proactive approach towards using biodiversity information to inform open space zonation, but municipalities often lack the capacity to appropriately interpret and use biodiversity priority maps. The South African National Biodiversity Institute recently developed guidelines to help municipalities integrate biodiversity priorities into their land use planning (SANBI 2021), but specific efforts to build capacity and better environmental awareness in municipal authorities may also be beneficial.

Alaniz et al. (2019) also recommend that policies are tailored to address the reasons why ecosystems are threatened. Thus far South Africa has implemented a single set of regulations for protecting all threatened ecosystems, but there are regulatory mechanisms available for crafting more specialized interventions. The National Environmental Management: Biodiversity Act allows for the development of Biodiversity Management Plans (BMPs) for ecosystems (DEA 2012). BMPs are developed in consultation with stakeholders such as landowners and conservation authorities and outline actions required for the integrated conservation management of an entire ecosystem. When plans are gazetted, responsibility for their implementation is assigned to specific entities. To date, however, very few BMPs have been developed for ecosystems. BMPs have the potential to better plan for and integrate conservation outcomes at the ecosystem level and may therefore help to overcome the displacement effects of independent land use decisions.

South Africa's threatened ecosystem regulations provide a case study of how land use planning and decision-making can be used to reduce land conversion pressure on threatened ecosystems, but it is not necessarily the most effective approach for ecosystem conservation. More counterfactual studies of other types of interventions are needed to gain better insights into effective actions for threatened ecosystems.

5. Synthesis: Evidence for the effectiveness of an integrated landscape approach to biodiversity conservation

To honour its commitments to the Convention on Biological Diversity (CBD) South Africa has implemented an integrated landscape-scale approach to biodiversity conservation. The aim of this approach is to secure the *in situ* persistence of a representative sample of all biodiversity elements (Cadman et al. 2010). Conservation strategies such as South Africa's landscape approach to biodiversity conservation often lack evidence for their effectiveness because they involve a range of actions that are implemented over long time frames within complex socio-ecological systems (Salafsky et al. 2021, Sayer et al. 2017). A recommended practical approach for the evaluation of conservation strategies is to isolate causal pathways for key mechanisms that are expected to lead to intermediate outcomes and test whether they are functioning as anticipated (Salafsky et al. 2022). Such individual tests of impact can contribute towards a collective evidence base for the effectiveness of a strategy. Where mechanisms or interventions are found to not work as expected it can contribute towards revising and adjusting the strategy to improve effectiveness (Salafsky et al. 2021).

An intermediate outcome for South Africa's landscape approach to conservation is to avoid the loss of natural areas that need to remain intact to meet the conservation targets for biodiversity elements such as species, ecosystems, and ecological processes. This study evaluated the impacts of three area-based interventions that together aim to avoid the loss of such areas as evidence towards the effectiveness of South Africa's strategy for biodiversity conservation.

The first intervention, protected areas and protected area expansion, was evaluated in Chapters 2 and 3. Protected areas in Mpumalanga Province were found to be very effective in preventing the loss of natural areas. An estimated 20 586 ha of natural areas in Mpumalanga would have been lost between 2010 and 2020, had the province's protected areas not been in place. After scaling this impact to a relative effect size, which allows for comparisons of effectiveness between studies, protected areas in Mpumalanga were found to rank highly among other area-based conservation interventions around the world. However, the effectiveness of protected areas in Mpumalanga cannot necessarily be generalized to the whole country. If relative effectiveness is calculated using the same formula as used in Chapter 3 (section 3.2.5), Ament and Cumming (2016) found a 70% relative reduction in the loss of natural areas in 18 national parks around the country, while Shumba et al. (2020) found a lower relative effectiveness of 50% reduction in loss of natural areas for private conservation areas.

Protected areas, particularly formal protected areas, are highly effective conservation interventions in South Africa because destructive land use changes are legally prohibited inside such protected areas (Government of South Africa 2004), and the protection of natural areas inside these areas is relatively well-enforced. The limitation of protected areas is that they tend to be implemented most often in areas of lower land conversion pressure (Joppa and Pfaff 2011, Pressey et al. 2015). Land conversion pressure is inferred from land conversion in matched counterfactual areas and for instance, in Mpumalanga, loss in natural areas matched to protected areas was 1.2% between 2010 and 2020, while the overall conversion of natural areas in the province over the same period was 1.5%. Ament and Cumming (2016) found no significant impacts for national parks located in South Africa's central arid interior and along the West Coast, which are areas of low land conversion pressure (Skowno et al. 2021). It must be noted however that while these parks do not contribute towards avoiding the conversion of natural areas, they do support the persistence of animal species that are otherwise absent from similar landscapes outside the parks.

Another limitation of the protected area network is that it is limited in extent and does not represent South Africa's biodiversity equitably (Driver et al. 2005). South

Africa's National Protected Area Expansion Strategy (NPAES) set ambitious targets to double the extent of South Africa's terrestrial protected area network over a 20-year period while ensuring that the expansion preferentially occurs in areas of under-represented biodiversity (DEAT and SANBI 2008). In Chapter 2 it was found that although protected area expansion has accelerated since the implementation of the strategy, it has not been enough to meet the strategy's area targets. A review of the strategy in 2016 found that insufficient resources and capacity committed towards protected area expansion are limiting the successful implementation of the NPAES (DEA 2016). These constraints mean that it is not possible to make strategic decisions regarding where to implement new protected areas, as all available land that can be rapidly secured for protection needs to be added to the protected area network to meet area targets. Fortunately, more than 90% of areas recently added to the protected area network contributed towards improved protection of under-represented biodiversity. The main finding of Chapter 2 was that this outcome is not due to strategic protected area expansion, but rather because most biodiversity is still under-represented in the protected area network, and therefore even non-strategic protected area expansion can result in biodiversity representation gains.

The second intervention that was evaluated in this study is the consideration of biodiversity priorities in land use decisions outside protected areas. The biodiversity priority maps that guide land use decisions are based on comprehensive, systematic mapping and assessment of species, ecosystems, and ecological processes, and maps have been developed for all nine provinces (Botts et al. 2019). These maps therefore have the potential to have a conservation impact throughout the landscape (Cadman et al. 2010).

The critical factor that ensures that South Africa's biodiversity priority maps are implemented is that the National Environmental Management: Biodiversity Act (NEMBA) requires the map to be accompanied by explicit land use guidelines for each biodiversity priority category included in the map and that these guidelines must be considered in land use decisions (DEAT 2009). The legislation however does not forbid land use decisions that are not aligned with land use guidelines, although such decisions may be legally challenged when maps have been formally gazetted as bioregional plans. The effectiveness of this component of South Africa's landscape approach to biodiversity conservation is therefore dependent on land use decisions being mostly aligned with land use guidelines, but this outcome is not necessarily assured.

Another risk is that land use changes are implemented without environmental authorization. Similar legislation in Australia, which requires land use changes that could impact threatened species or ecological communities to be reviewed for approval was found to have failed because more than 90% of clearing of natural areas proceeded without review (Ward et al. 2019). South Africa's environmental authorization system is not set up in a way that land use changes can be easily verified against environmental authorizations, and therefore effectiveness can only be inferred from overall trends in land use change.

Despite these risks and uncertainties, significant avoided loss of Critical Biodiversity Areas (CBAs) was found in Mpumalanga Province between 2010 and 2020. The largest relative effect of 72% reduction in land clearing was found in irreplaceable CBAs, which must remain intact, because their loss would mean that in situ persistence targets for biodiversity elements cannot be achieved. This result in particular was taken as evidence that decision-making authorities are considering biodiversity impacts in land use decisions, and are making decisions that are aligned with the goals of biodiversity planning, which is to avoid the loss of areas most important for the persistence of biodiversity. However, the findings in Mpumalanga cannot necessarily be generalized to the rest of the country without further study. The analysis of factors that influence the effectiveness of threatened ecosystem regulations in Chapter 4 found that provincial decision-making authorities play an important role in conservation outcomes. Therefore, the impact of biodiversity guidelines on land use decisions in other provinces may vary by how seriously decision-making authorities choose to follow these guidelines.

Chapter 4 also uncovered additional indirect evidence that biodiversity priority maps are playing an important role in shaping land use change patterns in other provinces. Even in threatened ecosystems with high land conversion pressure, loss in CBAs within the ecosystems was generally minimal. This occurred despite most of the remaining intact areas of these ecosystems being classified as CBAs. Loss was instead concentrated in Other Natural Areas (ONAs), which are areas that are not needed to meet persistence targets for biodiversity.

The third intervention that was evaluated in this study is regulations to protect threatened ecosystems. The regulations are designed to deter land use change applications in Critically Endangered and Endangered ecosystems by pro-actively embedding threatened ecosystems in multi-sector land use planning as well as requiring land use authorizations for any activity that would result in the clearing of natural areas. These regulations are acting in conjunction with those requiring biodiversity priority maps to be considered in land use decisions, and a central question of this study was whether these regulations have an additional impact to avoided loss due to biodiversity-aligned land use decisions.

It was found in Chapter 4 that in general, the regulations do not have a significant impact on land conversion in threatened ecosystems, but there was large heterogeneity in effect estimates between ecosystems. This indicated that there are strong ecosystem-specific circumstances that determine whether the regulations have an impact or not. It was however difficult to determine what these circumstances are, since meta-regressions delivered conflicting indications of which moderating factors were most important in explaining differences in effects between ecosystems. It was therefore inferred that it is possibly combinations of factors that determine whether the regulations are effective. This study evaluated impacts in threatened ecosystems across six provinces, and there was some evidence that the decision-making practices of the provincial authority make a difference. There was also limited evidence that mainstreaming efforts and consideration of threatened ecosystems in municipal land use planning strengthen the impact of threatened ecosystem regulations.

A significant finding of Chapter 4 was that avoided loss of biodiversity priority areas, which are mapped at a finer scale than ecosystems, does not necessarily scale up to avoided loss at the ecosystem level. This occurs primarily when land use decisions cause internal displacement of land use pressure within threatened ecosystems, rather than displacing it to non-threatened ecosystems.

5.1. Key findings

1. Protected areas are highly effective conservation interventions where they can be implemented, but their capacity for conservation impact is limited by severe constraints on strategic expansion.

2. There is evidence that land use decisions that consider biodiversity priorities contribute substantially towards avoiding the loss of natural areas most important for the persistence of biodiversity across landscapes. They can be particularly effective in areas where it may not be feasible to implement protected areas. The effectiveness of this process is however strongly reliant on compliance with the requirement for environmental authorization as well as decision-making authorities following land use guidelines in their decisions.

3. Threatened ecosystems regulations successfully reduce land conversion pressure when ecosystems are mapped at a fine scale, they are included in multi-sector land use planning, and there are efforts to mainstream awareness of biodiversity priorities into economic sectors. The regulations have no impact where ecosystems are extensive and land use decisions are the only mechanism for controlling loss, because land use decisions do not reduce land conversion

pressure but can help to displace it. When the displacement is largely within threatened ecosystems, it can lead to negative impacts relative to counterfactuals.

5.2. Evidence for the effectiveness of South Africa's landscape approach to biodiversity conservation

This study therefore concludes that, among the interventions evaluated, land use decisions guided by biodiversity priority maps make the most substantial contribution towards the effectiveness of South Africa's landscape approach to biodiversity conservation. The NPAES had no discernible impact on the way protected area expansion is implemented in South Africa, and therefore protected areas remain constrained in their capacity for conservation impact. Threatened ecosystem regulations also had no significant overall impact, although there were a few instances where the implementation of mainstreaming and multi-sector land use planning supported positive outcomes for threatened ecosystems.

5.3. Implications of the findings of this study for national and international biodiversity targets and conservation practice

The South African National Biodiversity Institute is responsible for regular national assessments of the state of the country's biodiversity (Driver et al. 2005, Driver et al. 2011, Skowno et al. 2019). These assessments form the basis of South Africa's country reports to the CBD. The headline indicators of national biodiversity assessments track changes in the Red List status and formal protection of species and ecosystems over time.

5.3.1. Implications of the findings of this study for Red List-based indicators

To prove that South Africa is meeting its obligations to conserve biodiversity, negative trends in indicators based on the Red List status of species and ecosystems need to at least be stabilized and ideally reversed (Mace et al. 2018). Stabilizing or reversing negative trends in the Red List status of species and ecosystems threatened by land conversion of natural areas requires that this threat is reduced to the extent that they cease to decline (Bubb et al. 2009).

The conservation interventions evaluated in this study are restrictive interventions, in other words, they restrict where threatening processes can occur (Evans 2016, Lambin et al. 2014). Such interventions, if they are effective, reduce threats where they are implemented, but they achieve this by displacing threats to other areas where restrictions are not in place. Such spillover or leakage effects have been well-documented (Dou et al. 2018, Ewers and Rodrigues 2008, Heilmayr et al. 2020, Leijten et al. 2021, Pfaff and Robalino 2017), and have led to most conservation efforts being described as "managed decline" (Mace et al. 2018).

Systematic conservation planning is a displacement intervention by design. One of its most acclaimed benefits is that it identifies the most efficient places within a planning region where conservation actions must be implemented to achieve a conservation objective, thus creating a spatial configuration where conservation can coexist with other economic sectors (McIntosh et al. 2017). In this sense, the use of SCP in South Africa to guide protected area expansion and land use decisions is working as intended, because the evidence presented in this study indicates that loss is being avoided in priority areas. The conservation bar is however set very low because the conservation objective in South Africa is to preserve only enough to avoid extinctions of species and collapse of ecosystems and ecological processes. The focus on minimum biodiversity targets is therefore facilitating a managed decline, which means that South Africa's Red List-based biodiversity indicators will not show stabilizing or increasing trends. Evidence of loss in threatened ecosystems in Gauteng (Chapter 4) suggests that it may become increasingly difficult to maintain even the minimum persistence targets for biodiversity features when options within the landscape for the displacement of threatening processes are progressively diminished.

5.3.2. Implications of the findings of this study for protected area-based indicators

Many concerns have been raised that ambitious global area targets for protection are directing protected area expansion towards areas where protection is easiest to implement, rather than where it is likely to have the best conservation impact (Barnes et al. 2018, Geldmann et al. 2021, Lemieux et al. 2019, Pressey et al. 2017, Pressey et al. 2021). Despite good intentions encapsulated in the NPAES, protected area expansion in South Africa is likely to follow this same trajectory, because the focus on rapid expansion is not allowing sufficient consideration of strategic allocation of limited resources towards protected area expansion that would have the best conservation impact.

Indicators tracking the representation of biodiversity features in protected area networks are unable to distinguish strategic protected area expansion from protected area expansion driven by expediency (Kuempel et al. 2016, Neugarten et al. 2020, Pressey et al. 2015). Representation of biodiversity features in protected areas is also not necessarily correlated with reductions in threats (Costelloe et al. 2016, Terraube et al. 2020, Wauchope et al. 2022), more so when protected areas are implemented for expediency rather than conservation impact (Pressey et al. 2015, Pressey et al. 2017). Therefore, although these indicators may show positive trends over time, they are not necessarily tracking conservation progress.

5.3.3. Implications of the findings of this study for international conservation policy and practice

This study makes a number of novel contributions towards a practical understanding of the strengths and weaknesses of South Africa's approaches to biodiversity conservation at a landscape scale, which has the potential to inform international conservation policy and practice. Strategic protected area expansion has often been proposed as the solution to the failure of the global protected area network to reduce biodiversity loss. Many maps of priority areas for protected area expansion have been produced (for example Kullberg et al. 2019, Rodrigues et al. 2004, Venter et al. 2014, Visconti et al. 2015). However, the real-world implementation of such plans has been rarely studied (McIntosh 2018), and therefore the effectiveness of strategic protected area expansion as a solution to the biodiversity crisis has remained largely theoretical. This study contributes a real-world case study of an attempt to implement strategic protected area expansion targeting under-represented biodiversity. It demonstrates that strategic protected area expansion is constrained by many factors that are often not fully appreciated by the proposers of this solution. This study also provides real-world support for the concerns that ambitious global area-based protected area expansion targets are likely to discourage strategic, biodiversity-focused protected area expansion that could result in more impactful protected area networks (Barnes et al. 2018, Geldmann et al. 2021, Lemieux et al. 2019, Pressey et al. 2021).

The effectiveness of protected areas has been relatively well-studied (Geldmann et al. 2013), but other than payments for ecosystem services (PES), the effectiveness of complementary area-based conservation interventions remains poorly known. This study contributed the first counterfactual evaluation of the effectiveness of the consideration of biodiversity priorities in land use decisions for avoiding biodiversity loss outside protected areas. This type of intervention has the potential to have a much wider impact across landscapes than protected areas, particularly in areas that are typically not available for protected area expansion. The consideration of biodiversity priorities in land use change decisions has the potential to make a significant contribution to the persistence of biodiversity if implemented more widely around the world.

Lastly, this study presented the first counterfactual evaluation of regulations designed to avoid the loss of threatened ecosystems. The effectiveness of legislative measures for the protection and recovery of threatened species has been studied (Bottrill et al. 2011, Ferraro et al. 2007) but a recent policy review pointed out the lack of evidence for effective measures to prevent ecosystem collapse (Bland et al. 2019).

Bland and colleagues also found that few countries have thus far implemented policies and practices to protect threatened ecosystems. The meta-analysis of conditions under which South Africa's threatened ecosystems regulations succeed and fail can serve as important guidance for the development of regulations and legislation elsewhere in the world. For example, there has been debates around the scale at which ecosystems should be mapped for assessment, with South Africa's approach judged too fine scale (KBA SAC 2022). The results of this study indicates that ecosystem regulations are more likely to be effective in reducing loss in ecosystems mapped at a very fine scale. Another important consideration in policy and regulation development is the implementation of mechanisms to avoid displacement of land conversion pressure, particularly within threatened ecosystems.

5.4. Policy recommendations

In a developing country such as South Africa, there is a need to balance economic growth with biodiversity conservation. South Africa's strategy for biodiversity conservation is designed to not unnecessarily constrain economic development by identifying only the most important areas within the landscape necessary for the persistence of species, ecosystems, and ecological processes, and to focus conservation actions in these areas. This study presented evidence that this strategy is mostly working as intended in the regions that were studied.

However, assessing the conservation impacts of South Africa's strategy for biodiversity conservation against the goals of the CBD gives cause for reflection. Is there more that can be done than just avoiding the extinction of species and collapse of ecosystems? Are there ways that pressures on biodiversity can be reduced, rather than just displaced?

A recent review of the economics of biodiversity concluded that unless we fundamentally change how biodiversity is valued in economic systems, ecological sustainability cannot be achieved (Dasgupta 2021). South Africa's biodiversity is a national asset (Statistics South Africa 2020, Turpie et al. 2017) that provides many benefits (SANBI 2019). Yet perceptions prevail that environmental regulations impose unreasonable costs on economic development (Retief and Chabalala 2009, Roos et al. 2020) and that the benefits of biodiversity conservation are not shared equitably (Musavengane and Leonard 2019, Poudyal et al. 2018).

South Africa's biodiversity conservation strategy could be enhanced by policies that support the generation of social and economic value from biodiversity conservation. NEMBA has implemented regulations to ensure the equitable sharing of economic benefits from bioprospecting, but there is a much wider scope for exploring causal pathways between biodiversity intactness and positive social and economic impacts (Guerry et al. 2015). A better understanding of such causal pathways can support the development of policies that more accurately value biodiversity (Barbier 2022) and encourage a reframing of environmental regulations and policies as mechanisms to enhance, rather than constrain economic and social well-being (Dasgupta 2021, Maze et al. 2016).

There is evidence that incentive-based conservation policies such as PES, provision of alternative livelihoods, and supply-chain initiatives can achieve net reductions in pressures on biodiversity (Lambin et al. 2018, Roe et al. 2015), and therefore this represents an important avenue for policy development. The development of such policies needs to be supported by a better understanding of the economic benefits of intact biodiversity. Such understanding can guide more accurate cost-benefit analyses of incentive-based conservation policies (Barbier 2022, Guidice and Börner 2021) and help to provide evidence that such policies offer benefits that outweigh costs (Ferraro and Simpson 2002). Studies have been done in this area to show that the benefits of alien invasive plant clearing outweigh costs (Holmes et al. 2020) and can serve as a guide for further investigation.

There also needs to be more consideration of cross-sectoral policies, such as policies that encourage agricultural intensification rather than expansion, as mechanisms to reduce pressures on biodiversity (Baylis et al. 2021). The development of such policies needs to be guided by more sophisticated indicators of the drivers of biodiversity loss (Driscoll et al. 2018).

5.5. Directions for further study

This study has contributed some evidence towards the effectiveness of South Africa's landscape approach to biodiversity conservation but a conclusion that the strategy is effective is premature. A limitation of this study is that impact evaluations other than the NPAES were focused on specific sub-regions of the country. There is therefore a further need for the evaluation of the impact of protected areas, land use decisions, and threatened ecosystem regulations in other parts of the country, particularly to inform a more precise understanding of the role of governance in biodiversity outcomes.

The success of South Africa's implementation of biodiversity priority maps has often been attributed to efforts to mainstream awareness of biodiversity priorities and capacity for the interpretation of planning products into multiple sectors (Cadman et al. 2010, Ginsburg et al. 2013, Manuel et al. 2016). This study uncovered limited evidence that mainstreaming is making a difference to biodiversity outcomes, and there remains a need to assess more rigorously the impact of various mainstreaming processes through counterfactual methods.

Another limitation of this study is that it was not possible to apply the most robust study designs such as randomized control trials and before-after control-intervention (BACI) designs. In Chapter 4 correlations were found between pre-intervention ecosystem decline and the impact of threatened ecosystem regulations. BACI designs are better able to account for the influence of pre-intervention trends in outcomes on effect estimates (Jones and Lewis 2015, Wauchope et al. 2021), and could have added more robustness to the uncertain effect estimates for individual threatened ecosystems. Wauchope et al. (2022) found substantial differences in estimates of the impact of protected areas on waterbird populations when using simpler before-after (BA) or control-intervention (CI) designs compared with BACI designs. In the BA design of Chapter 2 it was difficult to distinguish the impact of the NPAES from other concurrent changes in protected area implementation practices, such as the adoption of biodiversity stewardship as the primary mechanism for protected area expansion.

Implementing conservation interventions with counterfactual impact evaluation in mind can help to structure monitoring to allow more robust study designs going forward. Time series of outcome data in intervention as well-as non-intervention areas are the most critical enablers of robust studies of impact (Jones and Lewis 2015, Wauchope et al. 2021).

The limitations of quasi-experimental methods were highlighted by several methodological difficulties encountered in this study. All three studies used sets of confounding variables to control for differences between intervention and counterfactual observations that may bias effect estimates. In all three studies, great care was taken to identify relevant confounders through conceptual models and reviews of the literature on the key drivers of land use change, but it cannot be known whether these variables adequately control for the complex socio-ecological systems within which conservation interventions are implemented. No statistical methods exist to test for biases due to omitted variables, it is only possible to estimate sensitivity of effect estimates to unmeasured confounders (Rosenbaum 2007). It is also not always possible for matching methods to account for policies and practices that change concurrently with the implementation of a specific intervention, as was seen with the change to landowner incentive-driven protected area expansion in the analysis of the impact of the NPAES. Similarly, matching cannot account for potential characteristics of threatened ecosystems that have led to them being classified as threatened, that may not be present in non-threatened ecosystems.

Desbureaux (2021) demonstrated the sensitivity of effect estimates to study design and analysis decisions. In Chapter 4 it was found that effect estimates varied substantially for some ecosystems depending on matching variable selection. This study supports Desbureaux's recommendation that much more attention needs to be paid to the effects of design and analysis decisions in quasi-experimental studies, and particularly that the effects of matching covariate selection need to be more rigorously interrogated. Such robustness tests present another dimension of uncertainty around effect estimates in addition to sensitivity tests for omitted variables. The interplay between these two types of uncertainty has not yet been explored, and best practice for communicating these different dimensions of uncertainty is not yet in place.

Another unresolved challenge is how to justify design and analysis decisions in quasi-experimental studies when robustness tests show sensitivity to such decisions. For matching methods, the consensus is to test different methods and use the one that provides the best covariate balance, ideally with standardized mean differences less than 0.1 (Nguyen et al. 2017, Schleicher et al. 2020, Stuart 2010). However, for matching variable selection there are many proposed methods, but no clear consensus on which method is best. This study used prognostic balance to select variables, but further study is needed to determine whether this is a sufficiently robust method for variable selection.

Evaluations of spatially explicit conservation interventions are also well-known to be susceptible to biased effect estimates due to displacement or spillover effects. However, standard methods for testing for spillovers, which focuses largely on the areas immediately surrounding intervention areas (Ewers and Rodrigues 2008, Fuller et al. 2019) were found to be inadequate to detect wider, more complex displacement patterns within landscapes. As Chapter 4 showed, the complexity of such displacement patterns can increase significantly when there is interaction between different interventions being implemented within the same space. Pfaff and Robalino (2017) discuss the complex social and economic factors that may be affected by the implementation of spatial conservation interventions, and how that can manifest as different patterns of displacement of land conversion pressure but offer few solutions to how these can be dealt with in counterfactual study designs. Therefore, an important, but challenging direction for further study is to develop more sophisticated methods to account for complex, wider displacement effects within counterfactual study designs.

This study focused mostly on avoided loss of natural areas as a measure of impact. In Chapter 4 it was found that depending on how spatial interventions are configured, avoided loss does not necessarily scale up to a reduction in threats to biodiversity features such as ecosystems. There is therefore a need for more study of how spatial conservation interventions impact other elements of biodiversity, particularly species.

This study can also not claim to be an evaluation of the effectiveness of NEMBA, as there are many other regulations under this act that were not addressed. It is important to also assess the impact of regulations to manage the sustainable use of biological resources, regulations for the protection of species, and the effectiveness of biodiversity management plans.

As was pointed out in the policy recommendations, there is a particular need to study how the social and economic benefits of intact biodiversity can support the development of policies that can use economic incentives to reduce pressures on biodiversity. There is a need for a better understanding of the causal links between biodiversity conservation and social and economic outcomes, as well as the biodiversity impact of incentive-based policies compared to restrictive regulations. Results in Chapter 2 suggested that one incentive-based intervention that is already in place, biodiversity stewardship, may have better conservation impact than purchasing land for conservation because it is able to overcome the constraints of land availability for protection. This however needs to be confirmed through a counterfactual study comparing the biodiversity impacts of stewardship sites to land donated or purchased for conservation.

5.6. Final remarks

This study is a small contribution towards understanding the impacts of environmental policies and legislation in South Africa in the context of how much still remains unknown. I hope that it will catalyse a broader appreciation of the insights that can be gained from counterfactual approaches to the evaluation of policy effectiveness and stimulate much further study. May it also inspire a reappraisal of South Africa's ambitions for biodiversity conservation, and further exploration of policies and legislation to promote appreciation and wider sharing of the benefits of biodiversity.

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Appendix I

The article titled “An evaluation of the effectiveness of Critical Biodiversity Areas, identified through a systematic conservation planning process, to reduce biodiversity loss outside protected areas in South Africa” was published in the journal Land Use Policy 115: 106044 (2022).

Lize von Staden conceptualized the study, did the analyses, and wrote the article. Mervyn Lötter produced the conservation plans and land cover datasets used in the study and advised on the study design. All authors contributed to reviewing and editing drafts of the article.

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An evaluation of the effectiveness of Critical Biodiversity Areas, identified through a systematic conservation planning process, to reduce biodiversity loss outside protected areas in South Africa

Lize von Staden [a,b,*], Mervyn C. Lötter [c], Stephen Holness [d,e], Amanda T. Lombard [e]

[a] Department of Botany, Nelson Mandela University, Gqeberha, South Africa
[b] South African National Biodiversity Institute, Pretoria, South Africa
[c] Mpumalanga Tourism and Parks Agency, Lydenburg, South Africa
[d] Department of Zoology: Centre for African Conservation Ecology, Nelson Mandela University, Gqeberha, South Africa
[e] Institute for Coastal and Marine Research, Nelson Mandela University, Gqeberha, South Africa

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ABSTRACT

Systematic Conservation Planning (SCP) is a spatially explicit process used globally to prioritize conservation actions, but its effectiveness is difficult to quantify. In South Africa, terrestrial SCP processes are mainly used to identify important biodiversity areas outside of formal protected areas that are required to meet conservation targets. Environmental policy refers to these areas as Critical Biodiversity Areas (CBAs), and uses them to inform land use change decisions. Using Mpumalanga Province as a case study, avoided loss within CBAs is quantified using counterfactual matching methods. To contextualize the results, it is benchmarked against avoided loss achieved by protected areas during the same period. Significant reductions of 54–72% in land clearing were achieved in CBAs compared to other natural areas and were comparable to avoided loss achieved by protected areas in similar evaluations in other countries. Protected areas in Mpumalanga were found to be very effective (88% relative avoided loss) but are located in areas of lower land use change pressures than CBAs. Avoided loss was quantified as 1058 ha for Irreplaceable CBAs, 5285 ha for Optimal CBAs and 20,586 ha for protected areas. The consideration of biodiversity priorities in land use change decisions outside protected areas was found to be an effective complementary strategy to protected areas to avoid loss of biodiversity in areas typically not available for protected area expansion.

1. Introduction

Degradation and destruction of natural ecosystems is considered the leading cause of terrestrial biodiversity loss world-wide (Newbold et al., 2015). Establishment of protected areas is a commonly used conservation mechanism to avert ecosystem degradation and destruction. Studies have shown, however, that protected areas generally make small contributions to avoiding biodiversity loss (Geldmann et al., 2013), because they are established mostly in areas unsuited to other land uses (Joppa and Pfaff, 2009), while the most threatened biodiversity tends to be concentrated in areas of high human pressures (Hoekstra et al., 2005; Venter et al., 2016). Complementary conservation strategies are therefore needed to safeguard biodiversity in areas that are not available for protected area expansion, or where protected areas are not practical to implement (Dudley et al., 2018). Examples of alternative area-based conservation interventions include territories conserved by indigenous peoples and local communities (Carranza et al., 2014), payments for ecosystem services (PES, Costedoat et al., 2015), and land use zoning (Bruggeman et al., 2018).

South Africa uses maps of areas most important for the persistence of biodiversity (hereafter biodiversity priority maps) to guide the consideration of biodiversity in land use change decisions as a complementary area-based strategy to avoid extinction of species and collapse of ecosystems that are not yet well-represented within the protected area network (Cadman et al., 2010). Biodiversity priority maps are developed using systematic conservation planning (SCP), a spatially explicit approach for prioritizing conservation actions to secure the persistence of biodiversity features in situ (Margules and Pressey, 2000).

The SCP process identifies spatial biodiversity priority areas by setting quantitative conservation targets for species, ecosystem types,

* Correspondence to: South African National Biodiversity Institute, Private Bag X101, Pretoria 0001, South Africa.
E-mail address: L.vonStaden@sanbi.org.za (L. von Staden).

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and ecological processes in combination with other design criteria such as connectivity, avoiding implementation conflicts and minimizing implementation costs. Planning domains, which are typically provinces in South Africa, are divided into discrete spatial units known as planning units. Prioritization algorithms assign each planning unit a value indicating its importance for meeting the conservation targets and objectives set for the planning domain. Planning units that are not within existing protected areas, but are considered necessary to meet conservation targets either because they are irreplaceable, or because they contribute to the design criteria mentioned above, are then selected for classification as Critical Biodiversity Areas (CBAs, SANBI, 2017).

South African environmental regulations require that maps of biodiversity priorities are accompanied by guidelines detailing appropriate land uses compatible with each category of biodiversity priority included in the map. When maps of biodiversity priorities are formally gazetted as bioregional plans, the associated land use guidelines must be considered when land use changes are authorized (DEAT, 2009), and any land use decisions that are contrary to the guidelines may be legally challenged.

CBAs are areas that are mostly in a natural or near-natural state at the time of the conservation assessment, and associated management guidelines typically require that they be maintained in a natural or near natural state (SANBI, 2016). In other words, land use changes that would result in the destruction or degradation of these areas should preferably not be authorized.

What is not yet known, is how effective this system of SCP-based land use decision-making is in avoiding loss of biodiversity. Although SCP processes have been used to identify priority areas for conservation actions since the 1980 s (Watson et al., 2011; Adams et al., 2019a), a recent exhaustive review of the conservation literature found a lack of published evidence for conservation outcomes of SCP processes (McIntosh et al., 2018). Moreover, there are not yet any established methods for evaluating the conservation outcomes of SCP processes (Bottrill and Pressey, 2012; McIntosh et al., 2018).

The evaluation of the effectiveness of a conservation policy, mechanism or intervention requires the quantification of impact on a specific outcome attributed to the intervention, which is the difference between the observed outcome under the intervention and the outcome in the absence of the intervention (Ferraro and Pattanayak, 2006; Adams et al., 2019b). Such differences are most reliably estimated through randomized controlled trials, which is often not practical or ethical in conservation interventions (Pynegar et al., 2021).

An alternative approach is to use quasi-experimental methods to construct credible counterfactual estimates of conservation outcomes in the absence of an action (Ferraro and Hanauer, 2014). This method has been used to estimate the effectiveness of spatially explicit conservation interventions such as protected areas in avoiding biodiversity loss, for example Andam et al. (2008); Joppa and Pfaff (2010) and Honey-Rosés et al. (2011).

According to Bottrill and Pressey (2012), appropriate and informative evaluations of SCP need to be framed around the ultimate conservation goals of planning. In the South African conservation context, one of the most important goals of conservation planning is to avoid the loss of natural areas that are needed to meet conservation targets for species, ecosystems and ecological processes (Cadman et al., 2010). The aim of this study is therefore to quantify the avoided loss of biodiversity priority areas (CBAs) as a result of the implementation of CBA-associated land use guidelines, by adopting the spatially explicit counterfactual methods developed to evaluate the effectiveness of protected areas.

The first question this study aims to answer, is whether land use decisions guided by biodiversity priority maps have led to a reduction in loss of CBAs. Mpumalanga Province was chosen as a case study, as it had one of the first conservation plans specifically developed to inform land use decision making as required by environmental legislation (National Environmental Management: Biodiversity Act 10 of 2004). CBAs in Mpumalanga were matched to areas under similar land use pressure that are not needed to meet conservation targets, with land use changes in these non-prioritized areas representing a counterfactual where land use decisions do not need to consider biodiversity priority guidelines. This analysis provides an estimate of the effect size of the intervention, but to gauge its effectiveness, it needs to be contextualized through comparisons to alternative conservation interventions that are designed to achieve the same outcome (Rasolofoson et al., 2015; Robalino et al., 2015; Sims and Alix-Garcia, 2017).

The primary goal of protected areas is the same as land use guidelines associated with CBAs: to avoid the loss of natural areas to other land uses. In protected areas destructive land use changes are legally prohibited (Government of South Africa, 2004), and therefore protected areas provide a benchmark of the most secure form of land protection. Therefore, the effect size of protected areas in avoiding loss of natural areas is estimated using the same matching methods and over the same time period (2010–2020) as the implementation of SCP-based land use decision making in Mpumalanga Province, and used to contextualize the effectiveness of CBAs.

2. Material and methods

2.1. Study area and time frame of analysis

Mpumalanga is a land-locked province in eastern South Africa with an area of 76 520 km^2 (Fig. 1). The province's first SCP-based map of biodiversity priorities, known as the Mpumalanga Biodiversity Conservation Plan (MBCP) was developed in 2006, and associated land use guidelines were published for implementation in 2007 (Ferrar and Lötter, 2007). An updated map and land use guidelines, known as the Mpumalanga Biodiversity Sector Plan (MBSP) was published in 2014 (Lötter et al., 2014), based on a reassessment of spatial biodiversity priorities following improvements in available biodiversity data, changes in environmental regulations, and incorporating the effects of protected area expansion and land use changes since 2006.

The MBCP assessment was based on land cover data from 2000, which means that there is a considerable lag between the implementation of the plan and the baseline data. Methods of land cover classification improved significantly between 2000 and 2010, when a second land cover dataset was developed for the province, which means that it is difficult to distinguish real land cover changes between 2000 and 2010 from those that are due to methodological differences. Due to these complexities, it is not possible to evaluate the impact of conservation planning outcomes on land use changes in Mpumalanga Province before 2010, and the analysis is therefore restricted to the period 2010–2020.

One of the challenges with evaluation of the conservation outcomes of SCP processes is the often-protracted implementation of recommended conservation actions (Bottrill and Pressey, 2012), and therefore a longer time interval between the conservation assessment and evaluation is more likely to provide evidence of impact. The fact that conservation planning was already being applied in land use change decisions in Mpumalanga since 2007 means that there is no expected time lag in implementation that could undermine the causal link between planning and avoided loss of biodiversity priority areas.

2.2. Unit of observation, treatment, control and outcome variable

The study area was divided into 100 m x 100 m (1 ha) grid cells, with the centroid of each grid cell representing a unique sampling point, or unit of observation.

Sampling points within Critical Biodiversity Areas (CBAs) were evaluated as treatment observations. South African maps of biodiversity priorities recognise two types of CBAs. Irreplaceable CBAs represent ecosystems or areas of habitat for species for which the remaining intact areas are near or below the conservation target for the biodiversity feature. If such areas were to be lost, it means that the persistence of the biodiversity features present would be compromised (SANBI, 2016).

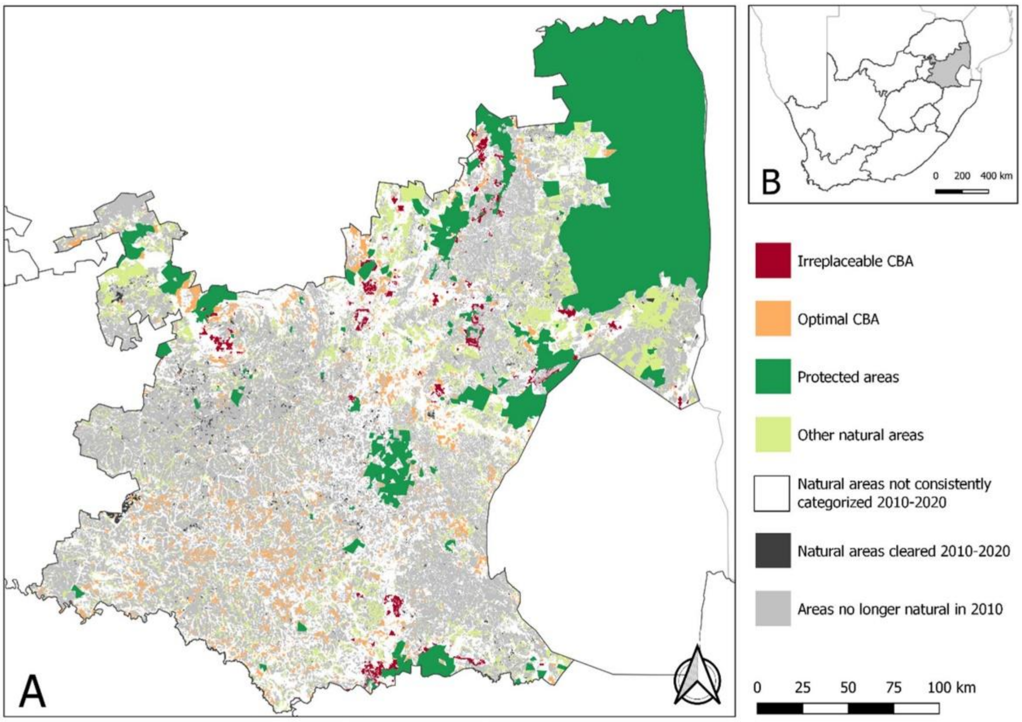

Fig. 1. A. Map of consistently categorized biodiversity priority categories in the two conservation plans developed for Mpumalanga Province, with an indication of natural areas that were cleared between 2010 and 2020. B. The location of Mpumalanga Province in South Africa.

Loss of these areas would therefore contribute to a loss of biodiversity.

The second type of CBAs are Optimal CBAs. These are areas where other options remain within the landscape for meeting the conservation targets for the biodiversity features present, but these areas represent the most efficient locations for meeting their targets. Loss of these areas would therefore not necessarily contribute to a loss of biodiversity if alternative areas with similar biodiversity features could be secured for conservation through for example biodiversity offsets. Since land use guidelines are designed around the potential biodiversity impacts of the loss of the different types of CBAs, the two types of CBAs were analysed separately.

Areas that are in natural, near-natural or semi-natural condition that are not required to meet conservation targets for ecosystem types, species or ecological processes are classified as other natural areas (ONAs) within biodiversity priority maps. These areas have no additional restrictions in terms of land use change authorisations, and therefore control observations were derived from sampling points within these areas.

Improved biodiversity data, as well as protected area expansion and loss of natural areas between the 2006 assessment and 2014 assessment, resulted in priority areas not being exactly aligned between the two plans. To ensure consistency in the evaluation of treatment and control samples, sampling was restricted to areas that were consistently classified as Irreplaceable CBA, Optimal CBA and ONAs in both the MBCP and the MBSP (Fig. 1).

For the comparative analysis, treatment observations were drawn from protected areas included within the 2014 MBSP. Control observations were selected from ONAs outside protected area buffers (see Section 2.7 on spillover effects for clarification).

The latest available land cover classification for the province is based on satellite imagery from 2020 (MTPA, 2020) and was used to estimate loss of natural areas to other land uses. Outcome was measured as a binary value indicating whether a particular unit of observation was lost to other land uses (outcome = 1) or remained in a natural condition (outcome = 0) between 2010 and 2020.

2.3. *Covariates*

Within observational studies, it is necessary to control for confounding variables that are likely to affect selection of observation units into treatment, or the observed outcome (Austin, 2011). In evaluations of avoided loss of natural areas, variables explaining development pressure are important covariates to consider, because development pressure is not evenly distributed across the landscape. Covariates reported in the literature as known predictors of development pressure that were included in this study were remoteness, distance from roads, terrain ruggedness and agricultural potential. In addition, covariates related to the local regulatory and economic context, such as mining pressure and development restrictions within protected area buffer zones were also included.

Bioregions were included as a covariate predicting selection into treatment. Bioregions are composite spatial terrestrial units of similar biotic and physical features at the regional scale, representing an intermediate vegetation classification between vegetation type and biome (Mucina and Rutherford, 2006). Specific land use types are more prevalent within some bioregions than others, for example plantation forestry in Mesic Highveld Grasslands, and different bioregions are unequally represented within the protected area network. Sampling points within more highly transformed bioregions that are also poorly protected are therefore more likely to be classified as CBAs.

Bioregion was preferred over the finer scale vegetation type, as conservation targets are set on vegetation types. Therefore, in some highly threatened vegetation types, all remaining natural areas may be required to meet their conservation targets, which means that it would not be possible to match control units to observations within these vegetation types. Variables that perfectly predict selection into treatment are known to reduce precision of treatment effect estimates and should therefore be avoided (Bergstra et al., 2019). A summary of all covariates considered in this study, including data structure and sources is provided in Table 1.

2.4. Matching

Matching methods are used to balance covariate differences between treatment and control samples in observational studies (Rubin, 1973), thereby controlling for bias due to observable confounders in the estimation of the treatment effect. Random samples of 50 000 points were taken from each of the treatment groups (two types of CBAs and protected areas). For protected areas, samples were restricted to observations that were in a natural condition according to 2010 land cover. Matches for each of the treatment groups were drawn from the full set of control observations in ONAs (772 491).

Matching was performed using the R package MatchIt (Ho et al., 2011), using the method of one-to-one nearest neighbour matching with replacement for the set of covariates listed in Table 1. Exact matching was required on categorical variables (see Table 1), while Mahalanobis distance was used for continuous covariates. No other common support restrictions or calipers were used.

Covariate balance of the matched samples were assessed according to standardized mean differences (SMDs), variance ratios and empirical cumulative density functions (eCDFs). Good matches are indicated by SMDs < 0.1, variance ratios close to 1 and maximum eCDFs close to 0.

Where treatment observations had to be discarded due to lack of common support, particularly due to requirements for exact matching on categorical covariates, the trimmed samples were examined for differences in sample means and variances from the original sample. This is to ensure that the treatment effects estimated for the trimmed samples remained generalizable to the treatment category it was derived from.

2.5. Estimation of treatment effect using post-matching regressions

Effect size was estimated as average treatment effect in the treated (ATT) by fitting binomial generalized linear models on the matched datasets with loss to other land uses as a binary outcome. Weights were included to account for repeated sampling of some control observations. The identity link function was used to quantify the ATT as the absolute difference between the outcome in CBAs/protected areas (treatment) and ONAs (control), therefore a negative coefficient indicates avoided loss. As all SMDs for the matched samples were below 0.1, no additional covariate adjustment was included in the binomial regressions (Nguyen et al., 2017).

Uncertainty was estimated using cluster robust standard errors (CRSE), which controls for dependence between observations within clusters (Austin, 2009), using a combination of matched pair identifiers and observation identifiers as the clustering variable.

After estimating the effect sizes, the avoided loss in hectares for each treatment category was calculated by using the % loss in the matched control samples to calculate its equivalent in hectares from the total area of each treatment category that was in a natural condition in 2010. Avoided loss is then the difference in between the hectares derived from the control sample and the observed total loss in the treatment category between 2010 and 2020.

Table 1
Covariates of development pressure used in matching analyses for CBAs and protected areas. * indicates categorical covariates where exact matching was required.

Covariate	Definition	Rationale
Remoteness	Distance in kilometres of sampling point from nearest built-up area as mapped in 2010 land cover dataset. Source: Mpumalanga Tourism and Parks Agency	Urban expansion is most likely in areas immediately adjacent to existing settlements, therefore sampling points nearer to built-up areas are likely to be under higher development pressure than more remote areas. Protected areas are more likely to be in more remote areas.
Distance from road	Distance in kilometres of sampling point from nearest road. Source: NGI Topo Data June 2020 http://www.cdngiportal.co.za/cdngiportal/	Development is more likely in accessible areas, therefore sampling points closer to roads are likely to be under higher development pressure than those further away from roads.
Land capability*	The Land Capability classification system defines the agricultural potential of an area. Land Capability is modelled in 15 classes indicating the suitability of land for rangelands and crop cultivation, with 1 indicating lowest suitability and 15 indicating highest suitability. Source: Agricultural Research Council – Institute for Soil, Climate and Water	Areas with higher agricultural suitability are likely to be under higher land conversion pressure than areas with low or poor suitability. Protected areas are generally located in areas of lower agricultural suitability.
Terrain ruggedness	An Index of Terrain Ruggedness was calculated from a digital elevation model following the method ofRiley et al. (1999). Higher values indicate increasing ruggedness. Source: 90 m resolution Shuttle Radar Topography Mission (SRTM) Digital Elevation Model	Development pressure is likely to be highest in areas with lowest ruggedness, as development costs may be higher in more rugged terrain.
Mining pressure*	Mining potential was quantified in four categories for South Africa, based on criteria related to size of known mineral deposits and their economic importance (Rouget et al., 2004). Source: National Spatial Biodiversity Assessment 2004 http://bgis.sanbi.org/nsba	Mining is of major importance to the South African economy, and Mpumalanga Province is rich in mineral deposits. Mining pressure is expected to be higher in areas with high mining potential.
Bioregion*	A categorical value indicating within which one of six bioregions present within Mpumalanga Province the sampling point is located. Source: Vegetation map of South Africa, Lesotho and Swaziland, 2018 version. http://bgis.sanbi.org/vegmap	Specific land use types are more prevalent within some bioregions than others. Planning units within highly transformed bioregions are more likely to be designated as CBAs.
Protected Area Buffer*	A binary value indicating whether the sampling point falls within a designated protected area buffer zone or not. Source: 2014 Mpumalanga Biodiversity Sector Plan http://bgis.sanbi.org/MBSP	Stricter development regulations apply within protected area buffer zones than elsewhere, regardless of whether the area falls within a CBA or not.

Relative effects were calculated following the method of Carranza et al. (2014), according to the formula C-T/C. Where C is the observed loss of natural areas in a sample of control observations matched to a treatment sample, and T is the loss in the treatment sample. Relative effects therefore allow for the comparison of effect sizes when they have been estimated against different background or baseline rates of land conversion.

2.6. Sensitivity analysis

Matching on observable covariates is expected to approximate the effect size estimates that would be derived from randomized controlled trials, but this relies on the assumption that there are no unobserved confounders affecting the probability of selection into treatment. It is not possible to directly assess this potential bias, but Rosenbaum (2002) developed a nonparametric method to quantify the magnitude of hidden bias that would be needed to change inferences about the treatment effect. A constant, Γ (gamma), representing the odds ratio of two observations with the same values for observable covariates but different probabilities of treatment assignment, is manipulated to determine how large it must be before inferences about the significance of the estimated treatment effect would be invalidated. The larger the value of Γ at the point where significance is overturned, the more robust the treatment effect estimate is to hidden bias. Sensitivity bounds on p-values for treatment effects were estimated for Γ values between 1 and 8 using the R-package rbounds (Keele, 2010).

2.7. Testing for spillover effects

It has been demonstrated that spatial conservation interventions often do not reduce environmental pressures, but merely displace them to adjacent areas where the interventions are not present (Ewers and Rodrigues, 2008; Heilmayr et al., 2020), a phenomenon termed spillover effects (Schleicher et al., 2020). While displacement of destructive land use changes from areas of high conservation priority to areas of low priority is a desired outcome of the land use change authorization system, areas of displacement may not represent a realistic counterfactual of land use change in the absence of the policy intervention. Therefore, potential spillover effects were assessed by comparing loss of ONAs that were consistently within 1 km of CBAs in both the MBSP and MBCP to ONAs consistently more than 1 km from CBAs in both plans. The same sampling and matching methods were used as for the CBA-ONA comparisons, and binomial generalized linear models were again used to quantify the difference in development pressure.

Development within protected area buffer zones is more strongly regulated than in natural areas beyond the buffers, to support the ecological integrity of protected areas (DEAT, 2010). Therefore, spillover effects are not anticipated in relation to protected areas, but sampling of control observations was restricted to areas outside protected area buffers, so as to provide a realistic counterfactual of avoided loss.

3. Results

3.1. Land cover changes in Mpumalanga Province between 2010 and 2020

Land cover changes between 2010 and 2020 in Mpumalanga Province indicate that 1.5% of all areas that were in natural condition in 2010 were lost to other land uses by 2020. Loss was predominantly to mining (38.4% of all loss), agriculture (21.9%) and expanding human settlements (16.1%, Fig. 2A).

Within SCP-defined biodiversity priority categories, loss was the lowest in protected areas (0.1%), and highest in ONAs (2.4%). Loss was 0.4% in Irreplaceable CBAs and 1.2% in Optimal CBAs. Loss of natural areas within Irreplaceable CBAs was mainly to plantations, while in Optimal CBAs it was to agriculture, mining and dams (Fig. 2B). Within protected areas, loss was mainly to dams, mining and agriculture, while

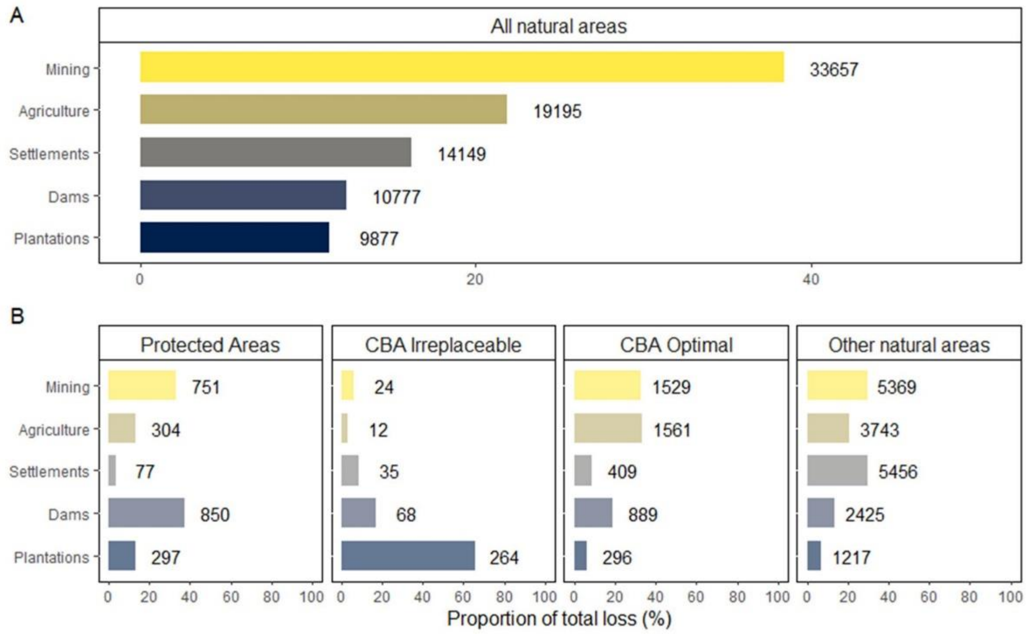

Fig. 2. A. Major drivers of land use changes across all natural areas in Mpumalanga Province between 2010 and 2020. B. Major drivers of land use changes within different biodiversity priority categories over the same period. Labels indicate the number of hectares lost in each category. Hectares for biodiversity priority categories do not add up to the value for all natural areas, as not all natural areas were consistently categorized between 2010 and 2020.

in ONAs, it was to settlements, mining and agriculture (Fig. 2B).

3.2. Matched samples

Before matching, the randomly sampled CBA observations were generally found to be in more remote and rugged areas further away from roads, more likely to be in areas of lower agricultural potential, but with higher mining potential than ONAs (Table 2). After matching, all SMDs for the covariates were below 0.08, all variance ratios were below 1.24, and all eCDF maximum differences were below 0.05 (Table 2).

The requirement of exact matching on categorical variables resulted in 338 treatment observations being excluded from the Irreplaceable CBA sample and 168 from the Optimal CBA sample for lack of suitable matches among the control observations. Comparisons of SMDs, variance ratios and eCDFs indicated no substantial differences in covariate values between the original randomized treatment samples and the remaining samples post-matching.

Matching was difficult for protected areas because the region of common support with ONAs was limited. The main reason for lack of common support was that remoteness ranged from 0.1 to 26.4 km for protected areas, while for ONAs it was limited to 0–10.2 km. The protected area observations that fell outside of the region of common support were concentrated within the Kruger National Park, South Africa's largest and one of its oldest protected areas. While remoteness is generally considered to be a driver of siting of protected areas (Joppa and Pfaff, 2009), in the case of the Kruger National Park it is a consequence of its size and age. The relatively high agricultural potential and low ruggedness of sampled observations inside the park suggest that the area would have been under high development pressure, had the land not been protected. In fact, 45% of unprotected land outside the park that is similar in terms of bioregion, ruggedness and agricultural potential is no longer in a natural condition. For these reasons, remoteness was excluded from the covariates used in matching the protected area sample to similar ONAs.

Before matching, protected area observations were generally in areas of lower agricultural and mining potential further away from roads than ONAs. Ruggedness was lower in the protected area sample (Table 2), mainly due to the contribution of the Kruger National Park, which protects large areas of lowland savanna plains on the eastern border of Mpumalanga Province. After matching, all SMDs were below 0.04, variance ratios were below 1.26 and eCDF maximum differences were below 0.04 (Table 2). Matching discarded 4263 protected area observations, mainly because no exact matches could be found for observations in the Mopane Bioregion, which is 100% within protected areas and protected area buffers in Mpumalanga Province. This did not substantially alter the value distributions of the other covariates compared to the original random sample.

3.3. Treatment effects

Effect size estimates reveal that CBAs and protected areas had statistically significant impacts in reducing land clearing compared to counterfactuals observed within ONAs (Table 3). The effects, as inferred from the coefficients of the fitted logistic regressions, were a 1.1% point difference in loss in Irreplaceable CBAs (CRSE 0.0027, $p < 0.0001$), 1.4% points in Optimal CBAs (CRSE 0.0020, $p < 0.0001$) and 1.09 in protected areas (CRSE 0.0009, $p < 0.0001$). The absolute effect sizes appear to be small, but is due to the relatively low rate of land conversion in Mpumalanga Province between 2010 and 2020.

Effect size estimates were robust to hidden bias, with critical Γ values ranging between 2.1 for Optimal CBAs and 7 for protected areas (Table 3). These values indicate that very strong predictors of treatment

Table 2
Covariate balance before and after matching for three treatments and potential spillover effects evaluated for Mpumalanga Province. Good matches are indicated by standardized mean differences (SMD) < 0.1, variance ratios close to 1 and maximum empirical cumulative density functions (eCDF) close to 0.

Treatment	Covariate	Before matching			After matching				
		Means treated	Means control	SMD	Means treated	Means control	SMD	Variance ratio	eCDF
Irreplaceable CBAs (N = 49 662)	Remoteness (km)	1.99	1.44	0.47	1.99	1.93	0.05	1.07	0.04
	Distance from roads (km)	1.32	0.84	0.45	1.32	1.27	0.05	1.10	0.03
	Ruggedness	53.54	24.97	0.68	53.34	49.98	0.08	1.23	0.03
	Agricultural potential[a]	6.12	7.50	0.70	6.13	6.13	0	1	0
	Mining potential[a]	1.07	0.55	0.43	1.06	1.06	0	1	0
	Bioregion[a]	3.03	2.45	0.68	3.02	3.02	0	1	0
	Protected area buffer[b]	0.48	0.24	0.48	0.47	0.47	0	–	0
Optimal CBAs (N = 49 832)	Remoteness (km)	1.56	1.44	0.11	1.56	1.54	0.02	1.07	0.01
	Distance from roads (km)	0.95	0.84	0.12	0.95	0.94	0.02	1.10	0.01
	Ruggedness	23.98	24.97	0.04	23.93	23.62	0.01	1.10	0.01
	Agricultural potential[a]	7.22	7.50	0.17	7.23	7.23	0	1	0
	Mining potential[a]	1.11	0.55	0.55	1.11	1.11	0	1	0
	Bioregion[a]	2.82	2.45	0.56	2.83	2.83	0	1	0
	Protected area buffer[b]	0.16	0.24	0.22	0.16	0.16	0	–	0
Protected areas (N = 45 737)	Distance from roads (km)	1.46	0.84	0.44	1.50	1.45	0.04	1.25	0.02
	Ruggedness	19.85	24.50	0.16	20.45	20.53	0.003	1.13	0.04
	Agricultural potential[a]	7.20	7.49	0.20	7.26	7.26	0	1	0
	Mining potential[a]	0.23	0.64	0.56	0.24	0.24	0	1	0
	Bioregion[a]	2.31	2.58	0.35	2.15	2.15	0	1	0
Spillovers – ONAs < 1 km from CBAs (N = 49 951)	Remoteness (km)	1.51	1.46	0.04	1.51	1.49	0.01	1.08	0.01
	Distance from roads (km)	0.88	0.88	0.004	0.87	0.87	0.01	1.09	0.01
	Ruggedness	26.82	24.14	0.10	26.81	26.68	0.005	1.05	0.01
	Agricultural potential[a]	7.23	7.60	0.22	7.23	7.23	0	1	0
	Mining potential[a]	0.88	0.35	0.53	0.88	0.88	0	1	0
	Bioregion[a]	2.73	2.27	0.62	2.73	2.73	0	1	0
	Protected area buffer[b]	0.13	0.31	0.55	0.13	0.13	0	–	0

[a] Categorical variables where exact matching was required.
[b] It is not possible to calculate variance ratios for binary variables.

Table 3
Effect size estimates for three spatial conservation interventions aimed at reducing loss of natural areas in Mpumalanga Province between 2010 and 2020. Percentages in brackets under observed and avoided loss indicate % of extent in 2010.

Treatment	Coefficient	Standard Error	p-value	Critical Γ at p = 0.05	Extent in 2010 (ha)	Observed loss (ha)	Avoided loss (ha)	Loss in matched ONAs (%)
Irreplaceable CBAs	-0.0111	0.0027	< 0.0001	3.2	95,603	403 (0.42%)	1058 (1.11%)	1.53
Optimal CBAs	-0.0139	0.0020	< 0.0001	2.1	390,073	4684 (1.20%)	5285 (1.35%)	2.56
Protected areas	-0.0109	0.0009	< 0.0001	7.0	1847,641	2279 (0.12%)	20,586 (1.11%)	1.24

categories are needed to overturn the significant estimates of avoided loss.

The estimated avoided loss in Irreplaceable CBAs is 1058 ha, which translates to a 72% relative reduction in the number of hectares that would have been lost in these highly valuable natural areas, had biodiversity priorities not been considered in land use change authorisations. Avoided loss in Optimal CBAs was 5285 ha (54% relative reduction) and for protected areas it was 20,586 ha (88% relative reduction).

3.4. Spillover test

No evidence was found for spillover effects within the immediate vicinity of CBAs. The difference in development pressure between ONAs within 1 km of CBAs and more than 1 km from CBAs was significant (0.79% points, CRSE 0.0013, $p < 0.001$), but rates of land clearing were lower within the immediate vicinity of CBAs (2%) than further away (2.8%). These results were more sensitive to hidden bias than the treatment effects (critical Γ 1.4), suggesting a higher likelihood that differences in rates of land clearing could be explained by unobserved confounders rather than proximity to CBAs.

4. Discussion

Systematic Conservation Planning is used globally to prioritize areas for conservation actions, but its effectiveness has been difficult to quantify. It has been argued that counterfactual evaluations of SCP are not feasible because the need to compare conservation outcomes between areas with and without maps of biodiversity priorities introduces too many social, political and biological complexities that are difficult to control for (Margoluis et al., 2009; Bottrill and Pressey, 2012). While this study is not able to evaluate the effectiveness of SCP as a conservation mechanism, it demonstrates that the way that South Africa implements SCP-based maps of biodiversity priorities in land use controls is an effective strategy to reduce biodiversity loss outside protected areas.

SCP-guided land use decisions led to significant reductions in loss of both Irreplaceable and Optimal CBAs. Consideration of biodiversity priorities in land use decisions is confirmed to be an effective conservation intervention, when contextualized among other spatially explicit interventions aiming to avoid biodiversity loss that were evaluated using similar counterfactual methods (Fig. 3).

Protected areas in Mpumalanga Province were also found to be extremely effective in avoiding biodiversity loss, when compared to CBAs as well as other spatially explicit conservation interventions (Fig. 3). These results are not unexpected, as most destructive land clearing is legally prohibited in protected areas, while the legal requirement for CBAs is only that land use guidelines must be considered when land use changes are authorized. Land clearing in protected areas was not completely avoided during the evaluation period, mainly due to the illegal expansion of mining into an adjacent protected area (Fig. 2B), but overall impacted a very small proportion (0.12%) of the protected area estate (Table 3).

An important finding was that when effectiveness was estimated in terms of absolute percentage point differences (Table 3), both Irreplaceable and Optimal CBAs performed better than protected areas. This is due to CBAs being located in areas of relatively higher land use pressure than protected areas, as inferred from observed rates of loss in

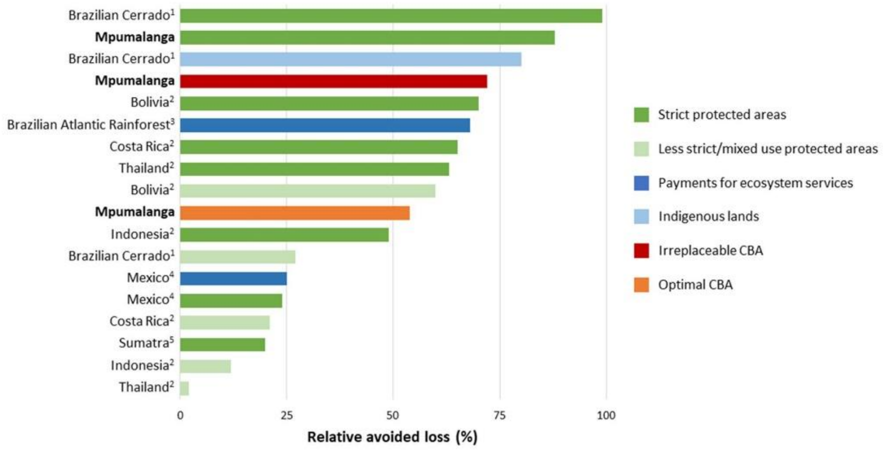

Fig. 3. Ranked effect size of CBAs and protected areas in Mpumalanga Province among similar counterfactual evaluations of spatial conservation interventions reported in the literature.
Data sources: [1]Carranza et al. (2014), [2]Ferraro et al. (2013), [3]Ruggiero et al. (2019), [4]Sims and Alix-Garcia (2017), [5]Gaveau et al. (2012).

matched control ONA samples for each treatment category (Table 3). Larger absolute estimates of avoided loss are consistently correlated with conservation interventions located in areas of higher land use pressures in counterfactual (Carranza et al., 2014; Pfaff et al., 2014) and simulated (Sacre et al., 2020) studies.

Critiques of the effectiveness of protected areas as conservation interventions often centre around the fact that they tend to be placed in areas of low land use pressure, and therefore these sites are likely to have remained in a natural condition, even if the protected areas were not in place (Ferraro and Pattanayak, 2006; Joppa and Pfaff, 2011). The results for protected areas in Mpumalanga are consistent with these concerns. The results for CBAs however indicate that land use change decisions can achieve significant reductions in loss of biodiversity priority areas in spite of higher land use pressures and less severe legal restrictions on land clearing than protected areas.

The comparison of relative effectiveness between protected areas and CBAs suggest that protected areas are still the most effective option where resources are available. Protected area expansion should therefore not be deprioritized in favour of reliance on land use decisions as a means to avoid biodiversity loss, but CBAs do provide an effective complementary mechanism to secure the persistence of biodiversity in areas where land may not be available for protected area expansion, or protected areas may not be feasible to implement.

The correlation between higher estimates of avoided loss and higher land use pressure has led to recommendations that decisions on where to implement conservation interventions should be guided by measures of threat (e.g. Sacre et al., 2020). While such an approach may ensure that conservation action is implemented where it is most urgently needed, there are also risks and potential unintended consequences when maximizing avoided loss becomes the primary aim of a conservation intervention (Vincent, 2016; Barnes et al., 2018).

Consistently prioritizing conservation over other land uses in areas of highest threat, which are also likely areas of highest economic growth potential, may give highest returns in terms of hectares spared, but is also most likely to provoke conflicts and backlash against conservation policies (Vincent, 2016). Observed examples are pre-emptive clearing of threatened ecosystems and critical habitat for threatened species earmarked for legislative embargos on land clearing (Lueck and Michael, 2003; Simmons et al., 2018).

A strength of SCP is that it avoids conflict with competing land uses as far as possible through the inclusion of cost surfaces in the assessment phase, as well as stakeholder engagement during the development of the plans (Naidoo et al., 2006; Lötter et al., 2014). The implementation of SCP-based land use decision making does not attempt to forestall economic development, because development is not legally prohibited within CBAs, but the distinction between Irreplaceable and Optimal CBAs allows for a flexible and responsive approach to transparently and defensibly weigh up the relative costs and benefits of conservation against economic gains on a case-by-case basis.

The reason South Africa's land use change authorization system is an effective mechanism to avoid biodiversity loss outside formal protected areas is that SCP makes land use decisions in favour of biodiversity persistence defensible. Greater absolute as well as relative avoided loss in Irreplaceable CBAs compared to Optimal CBAs demonstrate that the importance of Irreplaceable CBAs for the persistence of biodiversity is understood and considered within land use change authorizations in Mpumalanga Province.

Further evidence for the effectiveness of land use decision-making guiding destructive land use changes away from biodiversity priorities can also be found in the relative contributions of different types of land use change sectors to loss within CBAs compared to other natural areas. Mining is the leading cause of clearing of natural areas in Mpumalanga (Fig. 2A). Even though CBAs are in areas of higher mining pressure than ONAs (Table 2), mining was one of the lowest contributors to loss of Irreplaceable CBAs, while Optimal CBAs had proportionally similar losses to mining to ONAs (Fig. 2B).

It has been proposed that international biodiversity targets should be framed around avoided loss (Pressey et al., 2021). There is however a risk that targets aimed at maximizing avoided loss will fail to achieve biodiversity benefits in the same way that area-based protected area targets have failed, because it assumes that all hectares spared translates to equally valuable outcomes for biodiversity persistence (Vincent, 2016). There are not yet enough studies quantifying the impact of avoided loss of natural areas on specific elements of biodiversity, but one recent example illustrates that it does not necessarily translate to positive impacts on wildlife populations (Terraube et al., 2020).

SCP provides a powerful, yet simple measure of the importance of a particular area for the persistence of a range of biodiversity features, called irreplaceability. Irreplaceability is a measure of the likelihood of a site being required to meet a particular conservation objective (Margules and Pressey, 2000). South Africa's system of considering biodiversity persistence in land use change authorizations provides an example of how irreplaceability can be used to focus conservation actions and policies where biodiversity benefits such as avoided extinctions of species or avoided collapse of ecosystems are more likely to be tangible.

5. Conclusion

SCP-guided land use decision-making is an effective complementary conservation intervention to protected areas because it focuses conservation action on areas that matter the most for persistence of biodiversity, while avoiding potential conflicts with economic development as far as possible. Protected areas are more effective in terms of relative reductions in land clearing, and therefore they need to established where feasible, but the consideration of maps of biodiversity priorities outside protected areas can achieve near comparable levels of avoided loss in areas that are generally under higher development pressure, and where protected areas are less likely to be established.

South Africa's conservation community has invested over two decades into developing systematic conservation plans, and maps of biodiversity priorities with associated land use guidelines are now being implemented in land use decisions in all nine provinces (Botts et al., 2019). This is the first quantitative study that validates this strategy and provides evidence that legal support for the implementation of SCP can result in real-world benefits to biodiversity.

CRediT authorship contribution statement

Lize von Staden: Conceptualization, Methodology, Formal analysis, Writing – original draft. **Mervyn C. Lötter**: Methodology, Data curation, Writing – review & editing. **Stephen Holness:** Conceptualization, Writing – review & editing, Supervision. **Amanda Lombard**: Writing – review & editing, Supervision, Project administration.

Declarations of interest

none.

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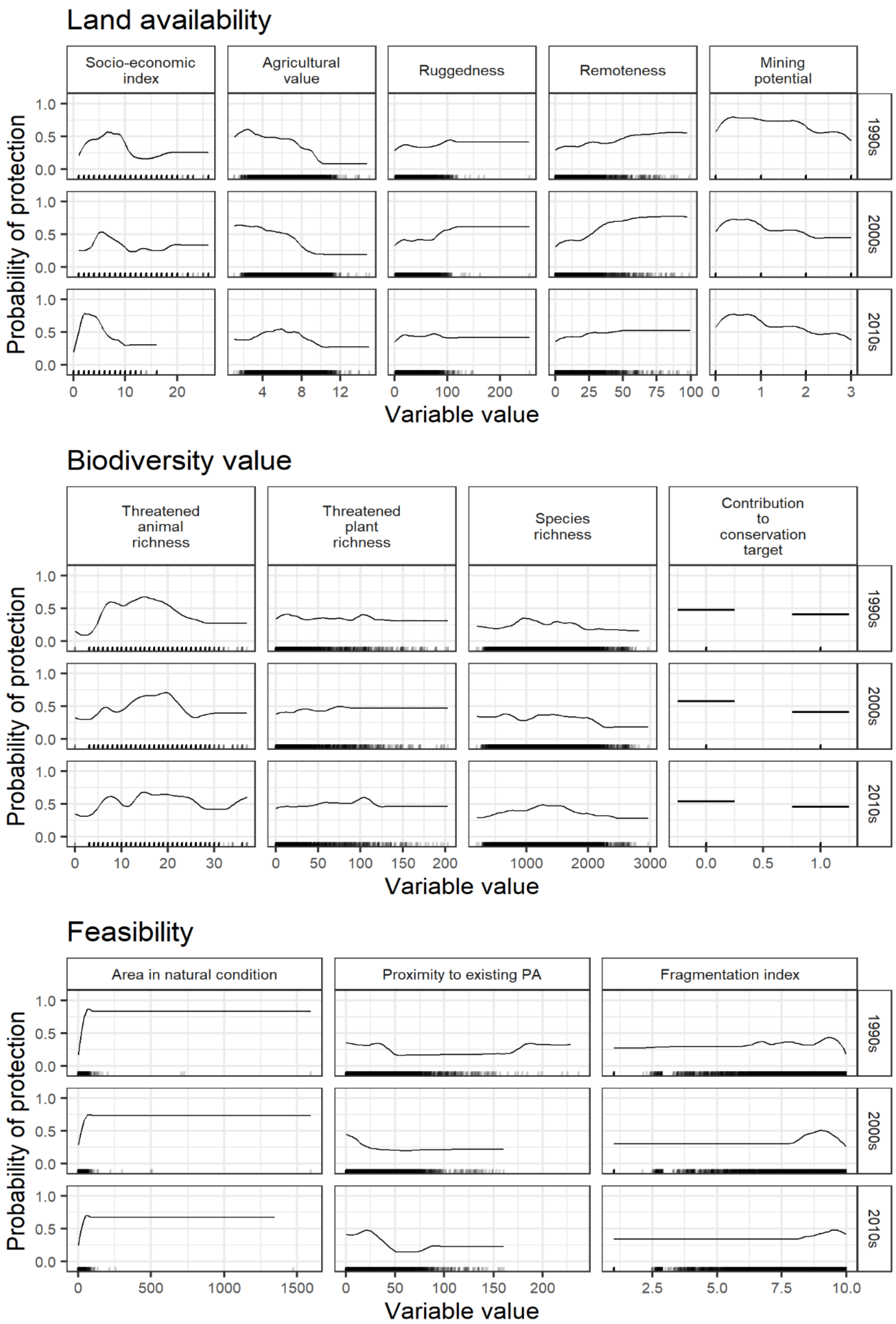
Appendix II

Supplementary materials for Chapter 2: An evaluation of the effectiveness of a protected area expansion strategy in guiding protected area expansion towards conservation targets.

SUPPLEMENTARY TABLE S1 Covariate balance for land parcel data before and after matching for each of the three decades evaluated. Good matches are indicated by standardized mean differences (SMD) < 0.1, variance ratios close to 1 and empirical cumulative density functions (eCDF) close to 0. CTT = land parcels contributing towards meeting ecosystem conservation targets. Caliper values are in raw units for the relevant variables.

		Before matching			After matching					
Decade	Covariate	Means CTT	Means not CTT	SMD	Means CTT	Means not CTT	SMD	Variance ratio	eCDF	Caliper
1990s	Agricultural potential	6.92	7.95	0.64	7.15	7.22	0.04	1.07	0.007	1
	Mining potential	0.34	0.27	0.10	0.10	0.10	0	1.00	0	
	Remoteness	6.78	1.45	0.58	3.80	3.77	0.01	0.98	0.003	
	Ruggedness	17.82	14.30	0.20	21.07	19.93	0.06	1.17	0.044	
	Socio-economic index	7.32	7.47	0.04	7.97	8.18	0.06	1.06	0.009	1
	Area in natural condition	2.96	0.16	0.23	1.62	1.23	0.07	0.97	0.007	10
	Fragmentation index	9.17	6.63	3.05	9.30	9.27	0.04	1.11	0.018	1
	Proximity to existing PA	18.6	10.04	0.41	7.16	6.51	0.09	1.00	0.006	
	Species richness	1147	1356	0.50	1410	1401	0.03	1.03	0.005	100
	Threatened animal richness	12	14	0.45	16	16	0.04	1.07	0.007	2
	Threatened plant richness	6	7	0.05	8	8	0.01	1.01	0.001	5
2000s	Agricultural potential	6.94	7.88	0.59	6.87	6.97	0.05	1.08	0.009	
	Mining potential	0.35	0.26	0.12	0.10	0.10	0.01	1.04	0.001	
	Remoteness	6.81	1.52	0.57	3.79	3.59	0.04	0.95	0.003	
	Ruggedness	17.45	15.20	0.13	24.7	23.89	0.03	1.09	0.033	
	Socio-economic index	7.36	7.36	0	7.14	7.47	0.09	1.10	0.084	
	Area in natural condition	2.96	0.20	0.23	1.76	1.39	0.07	0.90	0.007	
	Fragmentation index	9.17	6.71	2.95	9.22	9.19	0.04	1.10	0.016	
	Proximity to existing PA	15.33	8.62	0.37	5.47	4.88	0.09	1.03	0.006	
	Species richness	1142	1362	0.53	1433	1420	0.04	1.04	0.007	
	Threatened animal richness	12	14	0.42	15	15	0.06	1.14	0.012	
	Threatened plant richness	6	7	0.06	8	8	0.01	1.03	0.002	
2010s	Agricultural potential	6.90	7.89	0.62	6.81	6.84	0.01	1.09	0.008	

	Mining potential	0.35	0.27	0.12	0.08	0.09	0.03	1.07	0.003	
	Remoteness	7.36	1.85	0.57	4.06	4.24	0.03	1.03	0.003	
	Ruggedness	17.48	14.95	0.15	26.52	25.63	0.04	1.11	0.038	
	Socio-economic index	3.24	3.37	0.06	3.08	2.99	0.05	1.08	0.006	1
	Area in natural condition	3.03	0.16	0.23	1.53	1.24	0.06	1.04	0.006	10
	Fragmentation index	9.16	7.29	2.13	9.15	9.12	0.03	1.08	0.014	
	Proximity to existing PA	14.71	8.44	0.36	4.61	4.42	0.03	1.08	0.003	5
	Species richness	1129	1354	0.54	1458	1431	0.08	1.03	0.013	
	Threatened animal richness	12	14	0.40	15	15	0.02	1.05	0.011	
	Threatened plant richness	6	8	0.11	11	11	0.02	1.00	0.071	



SUPPLEMENTARY FIGURE S1 Partial dependence profiles of predictors of protected area expansion over three decades between 1990 and 2020. Steeper curves indicate strong independent contributions toward explaining protected area expansion, while flatter curves indicate weak predictors or predictors that interact with others.

Appendix III

Supplementary materials for Chapter 4: A counterfactual estimate of the contribution of an environmental policy to avoided loss of threatened ecosystems of the Grassland Biome, South Africa

Tests for spatial spillovers of land conversion pressure due to restrictions on land conversion within threatened ecosystems

Tests for displacement effects in the vicinity of threatened ecosystems were performed to guide the sampling of control observations during matching. Loss of natural areas between 2014 and 2020 within 1 km, 2 km and 5 km buffers around threatened ecosystems were compared with loss in grassland areas further away from threatened ecosystems to see if buffer zones had relatively higher losses. Random samples of 20 000 observations within buffer zones around threatened ecosystems were matched to similar observations outside buffer zones based on all the variables in Table 8. The method of one-to-one nearest neighbour matching without replacement with Mahalanobis distance on continuous covariates and exact matching on categorical covariates was used. Propensity score matching was also tested, but Mahalanobis distance provided better overall covariate balance (Supplementary Table S2). Differences in loss of natural areas between potential displacement areas around threatened ecosystems and non-threatened grasslands further away were tested by fitting binomial generalized linear models to the matched samples with displacement zone as a binary predictor and loss to other land uses as a binary response. For 1 and 2 km distances from threatened ecosystems, there were no significant differences in land conversion between 2014 and 2020, while loss was lower within 5 km from threatened ecosystems than further away (Supplementary Table S3).

SUPPLEMENTARY TABLE S2 Pre- and post-matching covariate balance for spillover tests. Tests were done on a range of distances from the edge of threatened ecosystems as well as between Other Natural Areas (ONAs) in threatened ecosystems compared with ONAs that are not in threatened ecosystems. Good balance is indicated by standardized mean difference (SMD) <0.1, variance ratios close to 1 and cumulative density functions (eCDF) close to 0.

		Before matching			After matching				
Spillover test	**Matching variable**	**Means within buffer zone around threatened ecosystem/ ONAs inside threatened ecosystems**	**Means outside buffer zone/ ONAs outside threatened ecosystems**	**SMD**	**Means within buffer zone around threatened ecosystem/ ONAs inside threatened ecosystems**	**Means outside buffer zone/ ONAs outside threatened ecosystems**	**SMD**	**Variance ratio**	**eCDF**
1 km	Local intactness	0.89	0.91	0.102	0.89	0.89	0.001	1.004	0.000
	Remoteness (km)	0.43	0.57	0.266	0.43	0.43	0.005	1.043	0.000
	Distance from road (km)	0.75	0.94	0.264	0.75	0.75	0.004	1.014	0.000
	Ruggedness	18.92	22.19	0.128	18.92	18.86	0.002	1.014	0.000
	Agricultural potential	6.88	6.57	0.211	6.88	6.89	0.003	1.007	0.000
	Population density	69.50	28.82	0.096	69.50	67.99	0.004	1.021	0.001
	Mining potential	0.74	0.44	0.300	0.74	0.73	0.002	1.004	0.000
	Dominant local land use	2.88	2.89	0.032	2.88	2.88	0.000	1.000	0.000
	Biodiversity priority category	1.86	2.13	0.287	1.86	1.86	0.000	1.000	0.000
	Province	3.23	2.47	0.372	3.23	3.23	0.000	1.000	0.000
2 km	Local intactness (%)	0.89	0.91	0.063	0.89	0.89	0.89	0.000	1.005
	Remoteness (km)	0.46	0.57	0.224	0.46	0.46	0.45	0.007	1.034
	Distance from road (km)	0.77	0.95	0.243	0.77	0.77	0.77	0.005	1.020
	Ruggedness	17.62	22.33	0.196	17.62	17.62	17.59	0.001	1.015
	Agricultural potential	6.91	6.56	0.238	6.91	6.91	6.91	0.001	1.006
	Population density	63.31	28.13	0.087	63.31	63.31	61.07	0.006	1.058
	Mining potential	0.74	0.43	0.308	0.74	0.74	0.73	0.001	1.005
	Dominant local land use	2.90	2.89	0.012	2.90	2.90	2.90	0.000	1.000

	Biodiversity priority category	1.93	2.13	0.221	1.93	1.93	1.93	0.000	1.000
	Province	3.17	2.45	0.348	3.17	3.17	3.17	0.000	1.000
5 km	Local intactness	0.90	0.91	0.034	0.90	0.90	0.000	1.005	0.000
	Remoteness (km)	0.50	0.58	0.137	0.50	0.49	0.010	1.058	0.001
	Distance from road (km)	0.80	0.96	0.207	0.80	0.80	0.008	1.028	0.001
	Ruggedness	17.95	22.63	0.191	17.93	17.92	0.001	1.014	0.000
	Agricultural potential	6.89	6.54	0.235	6.89	6.89	0.002	1.010	0.001
	Population density	53.07	27.10	0.084	53.08	50.86	0.007	1.026	0.002
	Mining potential	0.70	0.41	0.288	0.70	0.69	0.001	1.005	0.000
	Dominant local land use	2.90	2.89	0.023	2.90	2.90	0.000	1.000	0.000
	Biodiversity priority category	2.00	2.14	0.156	2.00	2.00	0.000	1.000	0.000
	Province	3.20	2.40	0.395	3.20	3.20	0.000	1.000	0.000
ONAs	Local intactness	0.68	0.85	0.534	0.68	0.68	0.000	1.004	0.001
	Remoteness (km)	0.26	0.40	0.312	0.26	0.26	0.007	1.023	0.000
	Distance from road (km)	0.68	0.76	0.100	0.68	0.68	0.009	1.046	0.001
	Ruggedness	12.69	19.59	0.369	12.69	12.62	0.004	1.028	0.001
	Agricultural potential	7.51	6.97	0.415	7.51	7.51	0.000	1.014	0.001
	Population density	141.35	48.17	0.125	141.35	136.38	0.007	1.074	0.003
	Mining potential	0.71	0.56	0.165	0.71	0.71	0.002	0.998	0.001
	Dominant local land use	2.33	2.82	0.453	2.33	2.33	0.000	1.000	0.000
	Province	4.08	2.78	0.626	4.08	4.08	0.000	1.000	0.000

Post-matching analysis of loss within different biodiversity priority categories indicated possible displacement effects within threatened ecosystems, from biodiversity categories where destructive land use changes are discouraged (Critical Biodiversity Areas and Ecological Support Areas), to areas where they are permitted (Other Natural Areas). Similar displacements are likely to also occur in non-threatened ecosystems. To test whether displacement of land conversion pressure disproportionately affects Other Natural Areas (ONAs) within threatened ecosystems, ONAs within threatened ecosystems were matched to ONAs within non-threatened ecosystems, using the same methods as for buffer distances around ecosystems. The difference in loss was quantified using a binomial generalized linear model with threatened ecosystem/non-threatened ecosystem as the binary predictor and loss to other land uses as a binary response. Loss in ONAs in threatened ecosystems was 2% higher than in ONAs in non-threatened ecosystems (Supplementary Table S3).

SUPPLEMENTARY TABLE S3 Estimated differences in loss of natural areas between spillover zones and areas under similar land conversion pressure in non-threatened grasslands.

Spillover test	Coefficient	Error	p-value	Effect
1 km	-0.00220	0.00195	0.26	No significant difference in land conversion
2 km	0.00090	0.00190	0.64	No significant difference in land conversion
5 km	-0.00425	0.00187	0.02	Significantly lower land conversion within 5 km of threatened ecosystem than further away
ONAs	0.0200	0.0031	<0.0001	There is significantly higher land conversion within ONAs inside threatened ecosystems than in ONAs not in threatened ecosystems.

SUPPLEMENTARY TABLE S4 Covariate balance pre- and post-matching for grassland ecosystems. Good balance is indicated by standardized mean difference (SMD) <0.1, variance ratios close to 1 and cumulative density functions (eCDF) close to 0.

		Before matching			After matching					
Ecosystem	**Matching variable**	**Means in threatened ecosystems**	**Means in non-threatened ecosystems**	**SMD**	**Means in threatened ecosystems**	**Means in non-threatened ecosystems**	**SMD**	**Variance ratio**	**eCDF**	**Caliper**
Bivane Montane Grassland	Agricultural potential	8.64	6.58	1.167	8.64	8.64	0.000	1.000	0.000	
	Distance from road (km)	0.97	0.94	0.048	0.97	0.97	0.000	1.000	0.000	
	Population density	7.00	30.05	0.116	7.00	7.72	0.004	0.000	0.001	
	Ruggedness	21.46	22.08	0.036	21.46	21.47	0.001	1.001	0.000	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	0.000	0.000	
	Dominant local land use	3.00	2.89	0.209	3.00	3.00	0.000	0.000	0.000	
	Biodiversity priority category	2.50	2.93	0.566	2.50	2.50	0.000	1.000	0.000	
Blesbokspruit Highveld Grassland	Local intactness	0.69	0.90	0.828	0.69	0.69	0.000	0.989	0.000	
	Province	2.06	2.50	1.052	2.06	2.06	0.000	0.989	0.000	
	Dominant local land use	2.26	2.89	0.562	2.26	2.26	0.000	0.989	0.000	
	Biodiversity priority category	2.65	2.93	0.332	2.65	2.65	0.000	0.989	0.000	
Blinkwater Valley	Agricultural potential	8.73	6.58	1.322	8.73	8.73	0.000	0.998	0.000	
	Distance from road (km)	0.44	0.94	1.797	0.44	0.44	0.002	0.993	0.000	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	1.000	0.000	
	Dominant local land use	4.00	2.89	2.178	4.00	4.00	0.000	0.000	0.000	
	Biodiversity priority category	3.34	2.93	0.449	3.34	3.34	0.000	0.998	0.000	
Blyde Quartzite Grasslands	Local intactness	0.93	0.90	0.130	0.93	0.93	0.009	1.015	0.001	
	Agricultural potential	6.10	6.58	0.221	6.10	6.09	0.003	1.034	0.002	
	Distance from road (km)	1.69	0.94	0.611	1.69	1.69	0.001	1.004	0.000	
	Mining potential	1.98	0.45	1.105	1.98	1.98	0.002	0.995	0.001	
	Province	5.00	2.50	1.408	5.00	5.00	0.000	0.000	0.000	
	Dominant local land use	3.04	2.89	0.742	3.04	3.04	0.000	1.000	0.000	
	Biodiversity priority category	1.34	2.93	2.517	1.34	1.34	0.000	1.000	0.000	

Boesmanspruit Highveld Grassland	Local intactness	0.80	0.90	0.473	0.80	0.80	0.001	0.998	0.000	
	Agricultural potential	8.06	6.58	2.110	8.06	8.06	0.003	0.997	0.000	
	Population density	28.00	30.05	0.013	28.00	27.78	0.001	0.951	0.003	
	Ruggedness	6.95	22.08	4.109	6.95	6.98	0.009	1.036	0.000	
	Province	2.14	2.50	0.572	2.14	2.14	0.000	1.000	0.000	
	Dominant local land use	2.07	2.89	0.826	2.07	2.07	0.000	1.000	0.000	
	Biodiversity priority category	2.02	2.93	1.135	2.02	2.02	0.000	1.000	0.000	
Bronkhorstspruit Highveld Grassland	Local intactness	0.80	0.90	0.469	0.80	0.80	0.000	0.992	0.000	
	Agricultural potential	8.57	6.58	2.689	8.57	8.57	0.000	0.991	0.000	
	Province	2.00	2.50	6.153	2.00	2.00	0.000	0.991	0.000	
	Dominant local land use	2.62	2.89	0.346	2.62	2.62	0.000	0.991	0.000	
	Biodiversity priority category	2.61	2.93	0.307	2.61	2.61	0.000	0.991	0.000	
Chrissiesmeer Panveld	Population density	10.11	30.05	1.654	10.11	10.11	0.000	0.967	0.000	
	Mining potential	1.37	0.45	0.992	1.37	1.37	0.000	0.967	0.000	
	Province	5.00	2.50	1.408	5.00	5.00	0.000	1.000	0.000	
	Dominant local land use	2.98	2.89	0.437	2.98	2.98	0.000	0.967	0.000	
	Biodiversity priority category	1.89	2.93	1.174	1.89	1.89	0.000	0.967	0.000	
Deneysville Highveld Grassland	Remoteness (km)	0.32	0.57	0.849	0.32	0.30	0.070	1.312	0.002	
	Local intactness	0.87	0.90	0.174	0.87	0.87	0.004	1.010	0.001	
	Agricultural potential	7.28	6.58	0.564	7.28	7.30	0.015	1.011	0.001	
	Distance from road (km)	0.27	0.94	2.812	0.27	0.27	0.010	1.191	0.001	
	Mining potential	1.03	0.45	3.433	1.03	1.04	0.089	0.376	0.004	
	Province	2.01	2.50	2.271	2.01	2.01	0.000	0.997	0.000	
	Dominant local land use	2.87	2.89	0.041	2.87	2.87	0.000	0.997	0.000	
	Biodiversity priority category	1.17	2.93	2.939	1.17	1.17	0.000	0.997	0.000	
Dullstroom Plateau Grasslands	Agricultural potential	6.40	6.58	0.113	6.40	6.40	0.000	0.991	0.000	
	Population density	11.99	30.05	0.147	11.99	11.98	0.000	0.988	0.000	
	Province	5.00	2.50	1.408	5.00	5.00	0.000	1.000	0.000	
	Dominant local land use	3.01	2.89	1.205	3.01	3.01	0.000	0.991	0.000	
	Biodiversity priority category	2.46	2.93	0.508	2.46	2.46	0.000	0.991	0.000	
Egoli Granite Grassland	Remoteness (km)	0.15	0.57	2.006	0.15	0.15	0.000	0.997	0.000	
	Local intactness	0.69	0.90	0.747	0.69	0.69	0.000	0.998	0.000	

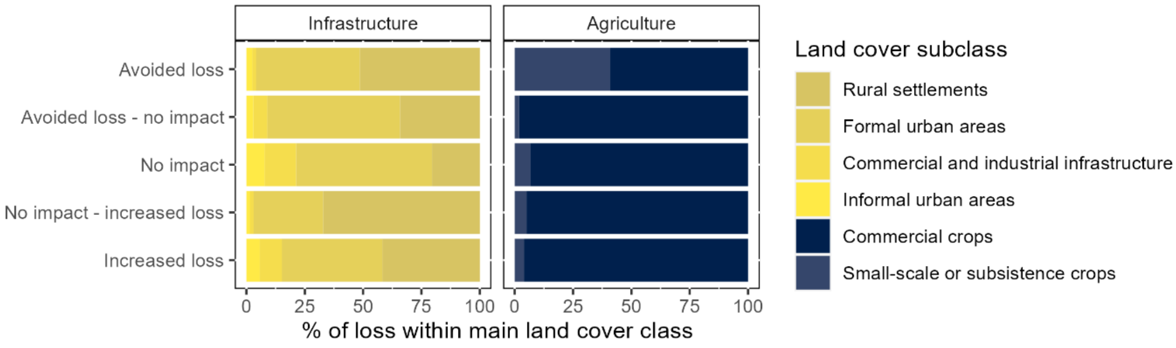
	Mining potential	0.01	0.45	2.807	0.01	0.01	0.000	0.996	0.000	
	Province	2.00	2.50	0.279	2.00	2.00	0.000	1.000	0.000	
	Dominant local land use	3.74	2.89	0.870	3.74	3.74	0.000	0.996	0.000	
	Biodiversity priority category	2.57	2.93	0.348	2.57	2.57	0.000	0.996	0.000	
Fort Metcalf Grasslands	Local intactness	0.98	0.90	1.007	0.98	0.98	0.000	1.000	0.000	
	Ruggedness	24.29	22.08	0.164	24.29	24.29	0.000	1.000	0.000	
	Mining potential	2.00	0.45	1.904	2.00	2.00	0.000	1.000	0.000	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	1.000	0.000	
	Dominant local land use	3.00	2.89	0.209	3.00	3.00	0.000	1.000	0.000	
	Biodiversity priority category	2.53	2.93	0.497	2.53	2.53	0.000	1.000	0.000	
Greytown North Grasslands	Local intactness	0.46	0.90	1.308	0.46	0.46	0.015	1.036	0.006	
	Distance from road (km)	0.28	0.94	2.317	0.28	0.27	0.021	1.028	0.001	
	Population density	68.06	30.05	0.404	68.06	63.83	0.045	1.116	0.009	
	Ruggedness	31.84	22.08	0.488	31.84	31.18	0.033	1.123	0.002	
	Mining potential	0.01	0.45	5.857	0.01	0.00	0.045	2.490	0.001	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	0.000	0.000	
	Dominant local land use	4.00	2.89	2.178	4.00	4.00	0.000	1.000	0.000	
	Biodiversity priority category	2.41	2.93	0.677	2.41	2.41	0.000	1.000	0.000	
Highover Nature Reserve and Roselands Farm Surrounds	Distance from road (km)	1.12	0.94	0.285	1.12	1.12	0.000	1.002	0.001	
	Population density	23.41	30.05	6.427	23.41	23.43	0.015	0.267	0.001	4
	Ruggedness	75.50	22.08	1.297	75.50	75.21	0.007	1.025	0.001	
	Mining potential	0.00	0.45	6.822	0.00	0.00	0.000	1.001	0.000	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	1.000	0.000	
	Dominant local land use	3.05	2.89	0.215	3.05	3.05	0.000	1.001	0.000	
	Biodiversity priority category	2.16	2.93	1.545	2.16	2.16	0.000	1.001	0.000	
Impendle Highlands	Local intactness	0.91	0.90	0.054	0.91	0.91	0.000	0.910	0.000	
	Mining potential	0.00	0.45	0.549	0.00	0.00	0.000	1.000	0.000	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	0.910	0.000	
	Dominant local land use	3.02	2.89	0.620	3.02	3.02	0.000	0.910	0.000	
	Biodiversity priority category	1.69	2.93	1.219	1.69	1.69	0.000	0.910	0.000	
	Local intactness	0.88	0.90	0.131	0.88	0.88	0.006	1.021	0.001	
	Agricultural potential	7.34	6.58	0.473	7.34	7.34	0.001	1.010	0.000	

Kaapsehoop Quartzite Grasslands	Distance from road (km)	0.64	0.94	0.609	0.64	0.64	0.001	0.997	0.000	
	Population density	10.91	30.05	66.159	10.91	10.89	0.075	0.005	0.001	
	Mining potential	2.62	0.45	2.229	2.62	2.62	0.000	0.999	0.000	
	Province	5.00	2.50	1.408	5.00	5.00	0.000	1.000	0.000	
	Dominant local land use	3.17	2.89	0.733	3.17	3.17	0.000	0.999	0.000	
	Biodiversity priority category	1.93	2.93	2.280	1.93	1.93	0.000	0.999	0.000	
Karkloof Forest Collective	Local intactness	0.88	0.90	0.119	0.88	0.88	0.003	0.976	0.002	
	Population density	25.23	30.05	0.694	25.23	25.84	0.088	0.978	0.001	
	Mining potential	0.05	0.45	1.767	0.05	0.03	0.088	1.528	0.005	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	1.000	0.000	
	Dominant local land use	3.26	2.89	0.830	3.26	3.26	0.000	0.980	0.000	
	Biodiversity priority category	1.75	2.93	2.030	1.75	1.75	0.000	0.980	0.000	
Klipriver Highveld Grassland	Local intactness	0.69	0.90	0.822	0.69	0.69	0.000	1.020	0.004	
	Agricultural potential	8.06	6.58	1.407	8.06	8.08	0.023	1.110	0.002	
	Population density	1945.23	30.05	0.797	1945.23	1928.12	0.007	0.996	0.005	
	Province	2.00	2.50	0.279	2.00	2.00	0.000	1.000	0.000	
	Dominant local land use	3.42	2.89	0.504	3.42	3.42	0.000	1.000	0.000	
	Biodiversity priority category	2.72	2.93	0.220	2.72	2.72	0.000	1.000	0.000	
Loskop Grasslands	Distance from road (km)	0.26	0.94	3.120	0.26	0.26	0.000	0.977	0.000	
	Population density	35.83	30.05	8.627	35.83	35.59	0.351	0.604	0.000	5
	Province	3.00	2.50	0.283	3.00	3.00	0.000	1.000	0.000	
	Dominant local land use	1.83	2.89	0.927	1.83	1.83	0.000	0.977	0.000	
	Biodiversity priority category	2.38	2.93	0.716	2.38	2.38	0.000	0.977	0.000	
Malmani Karstlands	Agricultural potential	5.15	6.58	0.690	5.15	5.15	0.000	1.001	0.000	
	Distance from road (km)	1.48	0.94	0.493	1.48	1.48	0.001	1.011	0.000	
	Population density	24.23	30.05	0.174	24.23	21.12	0.093	1.348	0.006	
	Ruggedness	70.52	22.08	1.386	70.52	70.44	0.002	1.004	0.000	
	Province	4.94	2.50	10.029	4.94	4.94	0.000	1.000	0.000	
	Dominant local land use	3.02	2.89	0.853	3.02	3.02	0.000	1.000	0.000	
	Biodiversity priority category	2.28	2.93	0.724	2.28	2.28	0.000	1.000	0.000	
Mauchesburg Alpine Grasslands	Local intactness	0.96	0.90	0.454	0.96	0.96	0.007	1.053	0.001	
	Distance from road (km)	1.39	0.94	0.443	1.39	1.38	0.006	1.050	0.002	

	Population density	9.39	30.05	0.571	9.39	10.44	0.029	1.006	0.001	5
	Ruggedness	61.72	22.08	1.104	61.72	61.95	0.006	1.034	0.001	
	Mining potential	2.05	0.45	1.375	2.05	2.05	0.003	0.992	0.002	
	Province	5.00	2.50	1.408	5.00	5.00	0.000	0.000	0.000	
	Dominant local land use	3.02	2.89	0.902	3.02	3.02	0.000	1.000	0.000	
	Biodiversity priority category	1.27	2.93	2.612	1.27	1.27	0.000	1.000	0.000	
Ngome Mistbelt Grassland and Forest	Local intactness	0.86	0.90	0.203	0.86	0.86	0.003	1.011	0.001	
	Agricultural potential	7.42	6.58	0.447	7.42	7.41	0.005	1.019	0.001	
	Distance from road (km)	0.98	0.94	0.049	0.98	0.97	0.018	1.101	0.002	
	Population density	22.95	30.05	0.773	22.95	22.44	0.056	0.890	0.001	
	Mining potential	1.44	0.45	0.932	1.44	1.44	0.001	1.002	0.000	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	1.000	0.000	
	Dominant local land use	3.09	2.89	0.684	3.09	3.09	0.000	1.000	0.000	
	Biodiversity priority category	2.59	2.93	0.401	2.59	2.59	0.000	1.000	0.000	
Oakland and Townhill Ridge	Local intactness	0.28	0.90	3.287	0.28	0.29	0.084	0.951	0.015	0.3
	Agricultural potential	8.81	6.58	1.769	8.78	8.73	0.043	0.945	0.004	
	Ruggedness	62.79	22.08	1.459	60.60	58.60	0.072	1.097	0.005	20
	Province	3.00	2.50	0.283	3.00	3.00	0.000	1.000	0.000	
	Dominant local land use	3.84	2.89	2.455	3.83	3.83	0.000	1.003	0.000	
	Biodiversity priority category	2.08	2.93	1.170	2.18	2.18	0.000	1.003	0.000	
Pietermaritzburg South	Agricultural potential	7.60	6.58	0.637	7.60	7.60	0.000	0.987	0.000	
	Population density	146.77	30.05	0.572	146.77	146.72	0.000	0.985	0.000	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	1.000	0.000	
	Dominant local land use	3.09	2.89	0.457	3.09	3.09	0.000	0.986	0.000	
	Biodiversity priority category	2.23	2.93	1.096	2.23	2.23	0.000	0.986	0.000	
Qudeni Mountain Mistbelt Forest and Grassland	Remoteness (km)	0.71	0.57	0.282	0.71	0.71	0.000	1.000	0.000	
	Ruggedness	74.49	22.08	1.396	74.49	74.49	0.000	1.000	0.000	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	0.000	0.000	
	Dominant local land use	3.00	2.89	0.209	3.00	3.00	0.000	0.000	0.000	
	Biodiversity priority category	2.17	2.93	1.395	2.17	2.17	0.000	1.000	0.000	
	Agricultural potential	8.45	6.58	1.569	8.45	8.45	0.000	0.983	0.000	
	Province	2.00	2.50	0.279	2.00	2.00	0.000	0.000	0.000	

Rietvleiriver Highveld Grassland	Dominant local land use	3.09	2.89	0.187	3.09	3.09	0.000	0.983	0.000	
	Biodiversity priority category	2.16	2.93	0.752	2.16	2.16	0.000	0.983	0.000	
Sekhukhune Mountainlands	Local intactness	0.97	0.90	0.734	0.97	0.97	0.014	1.037	0.001	
	Agricultural potential	5.42	6.58	0.553	5.42	5.42	0.000	1.001	0.001	
	Distance from road (km)	1.48	0.94	0.413	1.48	1.45	0.024	1.109	0.003	
	Population density	10.55	30.05	4.674	10.55	10.38	0.042	0.993	0.001	1
	Ruggedness	43.94	22.08	0.675	43.94	43.54	0.013	1.080	0.001	
	Province	4.61	2.50	4.325	4.61	4.61	0.000	1.000	0.000	
	Dominant local land use	3.00	2.89	0.209	3.00	3.00	0.000	1.000	0.000	
	Biodiversity priority category	2.42	2.93	0.651	2.42	2.42	0.000	1.000	0.000	
Sihleza	Local intactness	0.84	0.90	0.295	0.84	0.84	0.000	0.983	0.000	
	Agricultural potential	7.39	6.58	0.491	7.39	7.39	0.000	0.983	0.000	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	1.000	0.000	
	Dominant local land use	2.99	2.89	0.229	2.99	2.99	0.000	0.983	0.000	
	Biodiversity priority category	2.13	2.93	1.606	2.13	2.13	0.000	0.983	0.000	
Southern Weza State Forest	Distance from road (km)	3.62	0.94	1.938	3.62	3.59	0.022	1.021	0.002	
	Ruggedness	65.87	22.08	1.167	65.87	65.71	0.004	1.010	0.001	
	Province	3.00	2.50	0.283	3.00	3.00	0.000	1.000	0.000	
	Dominant local land use	3.07	2.89	0.670	3.07	3.07	0.000	1.000	0.000	
	Biodiversity priority category	2.54	2.93	0.633	2.54	2.54	0.000	1.000	0.000	
Stoffberg Mountainlands	Remoteness (km)	0.36	0.57	0.596	0.36	0.36	0.001	1.001	0.000	
	Local intactness	0.87	0.90	0.174	0.87	0.87	0.000	1.003	0.000	
	Distance from road (km)	0.59	0.94	0.698	0.59	0.59	0.004	0.994	0.000	
	Ruggedness	27.46	22.08	0.243	27.46	27.48	0.001	1.003	0.000	
	Mining potential	0.05	0.45	1.473	0.05	0.05	0.000	1.001	0.000	
	Province	5.00	2.50	1.408	5.00	5.00	0.000	0.000	0.000	
	Dominant local land use	2.85	2.89	0.082	2.85	2.85	0.000	1.000	0.000	
	Biodiversity priority category	2.71	2.93	0.267	2.71	2.71	0.000	1.000	0.000	
Tsakane Clay Grassland	Province	3.17	2.50	0.459	3.17	3.17	0.000	0.963	0.000	
	Dominant local land use	2.67	2.89	0.248	2.67	2.67	0.000	0.963	0.000	
	Biodiversity priority category	2.68	2.93	0.238	2.68	2.68	0.000	0.963	0.000	

Vaal-Vet Sandy Grassland	Remoteness (km)	0.30	0.57	0.682	0.30	0.30	0.003	1.014	0.000	
	Agricultural potential	7.28	6.58	0.696	7.28	7.28	0.000	1.003	0.000	
	Distance from road (km)	0.65	0.94	0.454	0.65	0.65	0.004	1.014	0.000	
	Population density	27.71	30.05	0.013	27.73	26.33	0.008	1.190	0.002	
	Ruggedness	5.88	22.08	4.799	5.88	5.88	0.001	1.043	0.000	
	Province	3.47	2.50	0.389	3.47	3.47	0.000	1.000	0.000	
	Dominant local land use	2.27	2.89	0.638	2.27	2.27	0.000	1.000	0.000	
	Biodiversity priority category	2.62	2.93	0.373	2.62	2.62	0.000	1.000	0.000	
Wakkerstroom-Luneburg Grasslands	Remoteness (km)	0.58	0.57	0.022	0.58	0.55	0.051	1.167	0.003	
	Local intactness	0.95	0.90	0.430	0.95	0.95	0.001	1.001	0.000	
	Distance from road (km)	1.30	0.94	0.334	1.30	1.26	0.035	1.126	0.003	
	Population density	10.72	30.05	1.142	10.72	9.74	0.058	1.357	0.002	
	Ruggedness	34.17	22.08	0.453	34.17	33.46	0.027	1.051	0.002	
	Mining potential	1.62	0.45	1.589	1.62	1.63	0.022	0.935	0.007	
	Province	4.88	2.50	5.053	4.88	4.88	0.000	1.000	0.000	
	Dominant local land use	3.00	2.89	0.209	3.00	3.00	0.000	1.000	0.000	
	Biodiversity priority category	2.42	2.93	0.507	2.42	2.42	0.000	1.000	0.000	
Western Highveld Sandy Grassland	Remoteness (km)	0.18	0.57	1.796	0.18	0.18	0.003	1.002	0.000	
	Distance from road (km)	0.69	0.94	0.342	0.69	0.68	0.006	1.055	0.000	
	Mining potential	0.05	0.45	1.243	0.05	0.05	0.003	0.990	0.000	
	Province	6.00	2.50	1.969	6.00	6.00	0.000	1.000	0.000	
	Dominant local land use	1.76	2.89	1.161	1.76	1.76	0.000	0.998	0.000	
	Biodiversity priority category	2.65	2.93	0.311	2.65	2.65	0.000	0.998	0.000	
Woodbush Granite Grassland	Mining potential	0.00	0.45	0.549	0.00	0.00	0.000	1.000	0.000	
	Province	4.00	2.50	0.846	4.00	4.00	0.000	0.000	0.000	
	Dominant local land use	3.18	2.89	0.641	3.17	3.17	0.000	0.852	0.000	
	Biodiversity priority category	1.93	2.93	2.207	1.94	1.94	0.000	0.852	0.000	



SUPPLEMENTARY FIGURE S2 Loss of natural areas to infrastructure development and agriculture in grassland threatened ecosystems broken down into subclasses. Ecosystems are grouped by the effectiveness of threatened ecosystem regulations, to assess whether different patterns of loss are correlated with differences in impact.

Variable selection for threatened ecosystem matching

When interventions are not randomly assigned, differences in outcome between units or areas that received the intervention and those that did not cannot be assumed to be solely due to the effect of the intervention. This is because factors that influence the outcome, known as confounding factors, may be different between areas receiving the intervention and those that did not. For example, threatened ecosystems are typically located in areas that are under the highest land conversion pressure. Therefore, estimating the impact of a policy to reduce land conversion pressure in threatened ecosystems needs to consider differences in land conversion pressure between areas that received the intervention and those that did not. Statistical matching was introduced as a method to control for confounding factors under circumstances where interventions are not randomly assigned (Rosenbaum and Rubin 1985). The method involves selecting untreated comparison units that are as similar as possible as those that received the intervention in terms of a set of confounding variables, thereby eliminating the biasing effects of these confounders on the estimation of the treatment effect.

This method however does not fully solve the problem of non-random treatment assignment, as the possibility remains that unmeasured or unobserved confounders may still bias effect estimates. This concern has led to a tendency for matching procedures to include as many potential confounders as possible (Weitzen et al. 2004, Harris and Horst 2016). There are many possible variables that are likely to explain differences in land conversion pressure (Lambin et al. 2001, Kolb et al. 2013). They are generally related to topography (slope, elevation), proximity to existing features (roads, cities, streams), biophysical characteristics (climate, soil), and socio-economic circumstances (population pressure, demand for natural resources). Studies of the impact of conservation policies designed to reduce loss of natural areas typically include a selection of these variables without further justification other than being known predictors of land conversion pressure. This approach was initially followed in this study, but a general set of matching variables consistently used for all grassland threatened ecosystems appeared to underestimate land conversion pressure in some threatened ecosystems. This resulted in some threatened ecosystems persistently having significantly higher levels of land conversion outcomes compared to matched non-threatened grasslands. Attributing higher land conversion within a threatened ecosystem to a policy aimed at reducing loss of threatened ecosystems seemed illogical, and these outcomes were therefore considered to be more likely due to omitted confounders. However, adding more matching variables did not resolve the problem, because it resulted in a different set of ecosystems appearing to have higher land conversion in response to threatened ecosystems regulations than matched non-threatened grasslands. These results suggested that there is no one-size-fits-all set of confounding variables for all threatened ecosystems, most likely due to drivers of land conversion being dependent on each ecosystem's inherent characteristics and location within the broader landscape. For example, for ecosystems within urban environments, local population density is a stronger predictor of land conversion pressure than the agricultural value of the land, while in rural areas, the reverse is often true. A study by Kolb et al. (2013) confirms the existence of ecosystem-specific drivers of land conversion. They found that the predictors of land conversion vary among different forest types in southern Mexico and recommend consideration of these fine-scale ecosystem-specific processes in land use change models.

No examples could be found in the conservation impact literature where different matching variables were used for individual spatial units receiving the same intervention, and the methods used in this study are therefore a somewhat radical departure from conventional approaches (for examples see Nolte et al. 2013, Shah and Baylis 2015, Shah et al. 2021). Support for the approach used here was however found in the medical literature, which is generally more advanced in the application of quasi-experimental impact evaluation methods. Specifically, the proliferation of large complex patient databases for medical insurance claims with potentially thousands of available confounding variables encouraged investigations into best practice for matching variable selection (Brookhart et al. 2010). These studies found that maximizing the number of matching variables can increase biases in effect estimates, and that careful selection of the most suitable variables is therefore important (Patrick et al. 2011, Ferri-García and Rueda 2022). Specifically, inclusion of variables that explain differences between treated and untreated units but are uncorrelated with the outcome — known formally as instrumental variables — introduced the most biases in effect estimates (Bhattacharya and Vogt 2007, Meyers et al. 2011, Patrick et al. 2011), while variables that are strongly correlated with the outcome, regardless of whether they also explain differences between treated and untreated units led to the least biased estimates (Brookhart et al. 2006, Patrick et al. 2011). Patrick et al. (2011) also found that matching variables selected a priori based on expert knowledge produced more biased effect estimates than systematic, data-driven methods for variable selection.

Following these findings, the correlation of predictor variables with land use change within individual threatened ecosystems was investigated. Each of the seven continuous or ordinal predictors of land conversion pressure was tested individually for each threatened ecosystem for correlations with loss of natural areas within the ecosystem. Binomial generalized linear models were fitted to all observations that were in a natural condition in 2014, with loss to other land uses by 2020 as the binary outcome. This analysis clearly demonstrated that the drivers of land use changes are not the same within different threatened ecosystems, as there were no consistent patterns in which variables were significantly correlated with loss of natural areas (Supplementary Table S5). Only local intactness was consistently negatively correlated with loss of natural areas in all threatened ecosystems, while other variables showed variable trends among different ecosystems (Supplementary Table S5). Therefore, selection of ecosystem-specific matching variables was considered justified.

The medical sciences also provide a range of published methods for systematic matching variable selection. Many types of multivariate outcome models have been proposed, with variable selection based on p-values (Patrick et al. 2011), stepwise model selection procedures (Hirano and Imbens 2001), variable importance rankings (Schneeweiss et al. 2017), or regularized regression (Shortreed and Ertefaie 2017). In many of these studies, thresholds for variable inclusion are however selected somewhat arbitrarily, and it is not certain whether proposed rules of thumb generalize beyond the specific datasets used in the studies. Comparative studies of different variable selection methods do not decisively point to a single superior approach (Greenland 2008, Brookhart et al. 2010). In addition to these limitations, model performance is known to be dependent on sample size, and the way it potentially influences variable selection is seldom examined (Markoulidakis et al 2022). Model selection procedures can also be compromised when the outcome of interest is rare, such as is the case with land cover change in the Grassland Biome (Peduzzi et al. 1996, Cepeda et al. 2003). Lastly, outcome-based variable selection methods have been critiqued for allowing results to influence study designs, which

SUPPLEMENTARY TABLE S5 Results of tests of correlations of individual predictors of land use change within threatened ecosystems with loss of natural areas in the ecosystem between 2014 and 2020. Values indicated under each predictor is the odds ratio derived from the binomial GLM coefficient. Odds ratios close to 1 indicate little or no correlation with loss of natural areas within the ecosystem. P-values: *** <0.001, ** <0.01, * <0.1, ns = not significant at p<0.1; zv = zero variance: variable had only one value for all observations in the ecosystem, and therefore a model could not be fitted.

	Predictors of land use change						
Ecosystem	Agricultural potential	Mining potential	Ruggedness	Remoteness	Distance from road	Local intactness	Population density
Bivane Montane Grassland	1.144 ns	zv	0.812 ns	0.128 ***	0.733 *	0.578 ***	zv
Blesbokspruit Highveld Grassland	1.203 ***	1.121 ***	0.870 ***	0.463 ***	0.634 ***	0.518 ***	1.157 ***
Blinkwater Valley	2.064 *	zv	1.242 ns	0.021 ns	0.452 **	0.828 ns	zv
Blyde Quartzite Grasslands	2.616 ***	2.712 ***	0.241 ***	0.003 ***	0.153 ***	0.398 ***	0.478 *
Boesmanspruit Highveld Grassland	1.284 ***	0.922 **	0.822 ***	0.390 ***	1.145 ***	0.538 ***	1.110 ***
Bronkhorstspruit Highveld Grassland	1.434 ***	1.096 *	0.913 *	1.065 ns	1.726 ***	0.779 ***	0.802 ***
Chrissiesmeer Panveld	1.150 ***	0.914 ***	0.909 ***	0.397 ***	0.814 ***	0.541 ***	1.011 ns
Deneysville Highveld Grassland	1.474 ***	1.148 *	0.920 ns	0.094 ***	0.337 ***	0.396 ***	0.705 ns
Dullstroom Plateau Grasslands	1.865 ***	1.149 ***	0.377 ***	0.020 ***	0.436 ***	0.436 ***	1.061 ***
Egoli Granite Grassland	1.296 ***	0.899 ***	0.793 ***	0.227 ***	0.439 ***	0.464 ***	1.433 ***
Fort Metcalf Grasslands	0.781 ns	zv	1.259 ns	0.220 ***	0.307 **	0.594 ***	0.757 ns
Greytown North Grasslands	1.465 ***	0.367 ns	0.617 ***	0.553 ***	0.201 ***	0.557 ***	0.599 ***
Highover Nature Reserve and Roselands Farm Surrounds	2.207 **	0.455 ns	0.091 ***	0.022 **	0.312 ***	0.314 ***	1.588 **
Impendle Highlands	1.442 ***	zv	0.441 ***	0.001 ***	0.805 ***	0.323 ***	0.875 *
Kaapsehoop Quartzite Grasslands	1.447 **	1.360 ns	0.479 ***	0.006 ***	0.561 **	0.278 ***	0.010 ns
Karkloof Forest Collective	0.950 ns	0.038 ns	0.753 ns	0.001 ***	0.843 ns	0.267 ***	0.750 *
Klipriver Highveld Grassland	0.840 ***	0.880 ***	1.131 ***	0.190 ***	0.381 ***	0.375 ***	1.427 ***
Loskop Grasslands	1.236 **	zv	0.583 ***	0.365 ***	0.461 ***	0.423 ***	1.106 ns
Malmani Karstlands	2.027 ***	0.652 ***	0.244 ***	0.0001 ***	0.118 ***	0.461 ***	1.278 ***
Mauchesburg Alpine Grasslands	1.417 ***	0.623 ***	0.478 ***	0.001 ***	0.374 ***	0.464 ***	0.713 ns
Ngome Mistbelt Grassland and Forest	1.300 ***	0.540 ***	0.669 ***	0.002 ***	0.444 ***	0.381 ***	0.702 ***
Oakland and Townhill Ridge	0.848 ns	zv	0.038 ***	0.949 ns	0.787 ns	0.590 ns	0.526 *

Pietermaritzburg South	1.083 ns	0.255 ns	0.521 ***	0.038 ***	1.026 ns	0.320 ***	0.865 *
Qudeni Mountain Mistbelt Forest and Grassland	1.331 ns	zv	0.348 ***	0.032 ***	0.231 ***	0.558 ***	0.772 ns
Rietvleiriver Highveld Grassland	1.024 ns	0.959 ns	1.142 ***	0.089 ***	0.490 ***	0.364 ***	1.392 ***
Sekhukhune Mountainlands	2.434 ***	0.933 *	0.072 ***	0.012 ***	0.169 ***	0.536 ***	1.381 ***
Sihleza	1.320 ***	zv	0.503 ***	0.030 ***	0.422 ***	0.302 ***	1.194 **
Southern Weza State Forest	0.973 ns	1.311 *	0.791 ns	2×10^{-7} ***	0.835 ns	0.289 ***	1.266 ***
Stoffberg Mountainlands	3.038 ***	1.033 ns	0.188 ***	0.061 ***	0.434 ***	0.407 ***	0.959 ns
Tsakane Clay Grassland	1.704 ***	0.880 ***	0.551 ***	0.054 ***	0.469 ***	0.325 ***	1.457 ***
Vaal-Vet Sandy Grassland	1.502 ***	1.217 ***	0.974 ***	0.055 ***	0.664 ***	0.380 ***	1.117 ***
Wakkerstroom-Luneburg Grasslands	2.382 ***	1.269 ***	0.197 ***	0.121 ***	0.552 ***	0.521 ***	0.896 ***
Western Highveld Sandy Grassland	1.077 ***	1.037 ***	0.941 ***	0.026 ***	0.554 ***	0.264 ***	1.029 ***
Woodbush Granite Grassland	1.420 ***	zv	0.135 ***	0.043 ***	0.135 ***	0.398 ***	0.765 *

may lead to methodological adjustments to produce more favourable outcomes (Rubin 2001, Stuart et al. 2013).

Stuart et al. (2013) proposed using prognostic scores as an additional means to assess balance of matched intervention and control samples. The prognostic score represents the baseline probability of the outcome in the absence of the intervention. It is estimated by modelling the outcome in the untreated group using the known predictors of the outcome, and then using the model to assign predicted outcomes to both the treatment and comparison groups. The prognostic score is not included among the matching variables but is used to compare similarity in the baseline probability of the outcome in the matched samples. If the mean prognostic scores are similar between the matched treatment and control samples, it gives an indication that the variables used in the matching procedure are creating samples with similar expected outcomes under non-intervention conditions. Stuart et al. (2013) showed that treatment and control samples that are well-balanced on the prognostic score have the least biased effect estimates. They also suggested that prognostic scores could be used to select variables for matching. Since the main challenge with matching threatened ecosystems to non-threatened grasslands was a lack of balance in baseline land conversion pressure between the samples being compared, using prognostic balance to select variables was considered a sensible approach. Prognostic variable selection also avoids intervention outcomes influencing study designs, as it only considers outcomes under non-intervention conditions (Stuart et al. 2013).

Prognostic scores were modelled on a sample of non-threatened ecosystem observations using a binomial generalized linear mixed-effect model with province and dominant local land use as the random effects, using all the remaining variables in Table 8. Biodiversity priority category was converted to an ordinal variable with categories ranked according to the severity of restrictions on land use change indicated within each category's associated land use guidelines. All covariates in the full model were found to significantly contribute towards explaining loss of natural areas in non-threatened grasslands (Supplementary Table S6).

SUPPLEMENTARY TABLE S6 Variable contributions towards explaining land cover change in a prognostic model constructed on non-intervention observations. Prediction accuracy 69.4%, no information rate 51.5%, p-value (accuracy > no information rate) <0.0001.

Predictor	Coefficient	Error	pvalue
Agricultural potential	0.1537	0.0186	<0.0001
Mining potential	-0.1039	0.0192	<0.0001
Ruggedness	-0.4758	0.0333	<0.0001
Remoteness	-1.2031	0.0752	<0.0001
Distance from road	-0.1255	0.0330	0.00014
Local intactness	-0.3881	0.0135	<0.0001
Population density	0.0459	0.0052	<0.0001
Biodiversity priority category	0.1556	0.0188	<0.0001

The data sample used to construct the model was stratified to ensure sufficient observations in each of the random effect groups. Where the number of available observations within the group was smaller than 1000, all observations were included. For groups with larger available samples, 500 observations within each of the outcome categories were randomly selected. This sampling approach was followed due to very strong imbalances in the outcome: >95% of observations remained in a natural condition between 2014 and 2020, and therefore fully randomized sampling often resulted in no observations indicating loss being sampled. This model was then used to predict baseline outcomes for the full set of treatment and control observations.

Then all possible combinations of the seven variables where exact matching was not required were used to create matched samples for each ecosystem. The three variables where exact matching was required were included in all matching models. After matching, the prognostic balance of each of the matched samples was calculated as the absolute standardized mean difference (ASMD) in mean prognostic scores between the samples. The variable combination that produced the smallest ASMD for each ecosystem was selected (Supplementary Table S7). The number of selected variables ranged between three and nine. For no ecosystem were all 10 variables selected (Supplementary Table S7).

Following the findings of Stuart et al. (2013), theoretically, this approach should select the matching model that would deliver the least biased effect estimate, but it is not possible to verify this approach with the data available in this study. Desbureaux (2021) demonstrated that when matching is used to control for confounders in observational studies, arbitrary methodological choices can lead to significantly different effect size estimates. Desbureaux (2021) recommends greater transparency regarding the impact of methodological choices on study results, by quantifying a range of possible outcomes under different methodological scenarios as an additional measure of uncertainty around effect estimates. Therefore, effect sizes were estimated for each ecosystem on each of the 128 matched samples resulting from the full range of variable combinations, and the effect estimate derived from the variable combination producing the smallest prognostic ASMD was examined relative to the full range of possible effect estimates.

Effect ranges for individual ecosystems were between 4.6 and 53 percentage points, with a mean range of 14.6 percentage points, suggesting that variable selection has a strong impact on point estimates of avoided loss. In all but four ecosystems, the full range of effect estimates included both positive and negative impacts. However, in most ecosystems, the most extreme effect estimates had the poorest prognostic balance (Supplementary Figure S3), indicating that these are likely to be biased.

Supplementary Figure S3 illustrates the correlations of prognostic balance with effect estimates through three example ecosystems. Malmani Karstlands is an example of an ecosystem where the robustness checks clearly confirm a positive impact of threatened ecosystem regulations. Only two matching variable combinations produced negative impacts, but these had relatively poor prognostic balance, and

SUPPLEMENTARY TABLE S7 Prognostic balance on samples matched on all 10 predictors of land conversion pressure compared with the variable combination that minimized prognostic balance for each of the 34 grassland threatened ecosystems. The metric used to assess balance was Absolute Standardized Mean Difference (ASMD) between the mean prognostic scores for the ecosystem and its matched sample. Supplementary Table S4 provides the list of selected variables for each ecosystem.

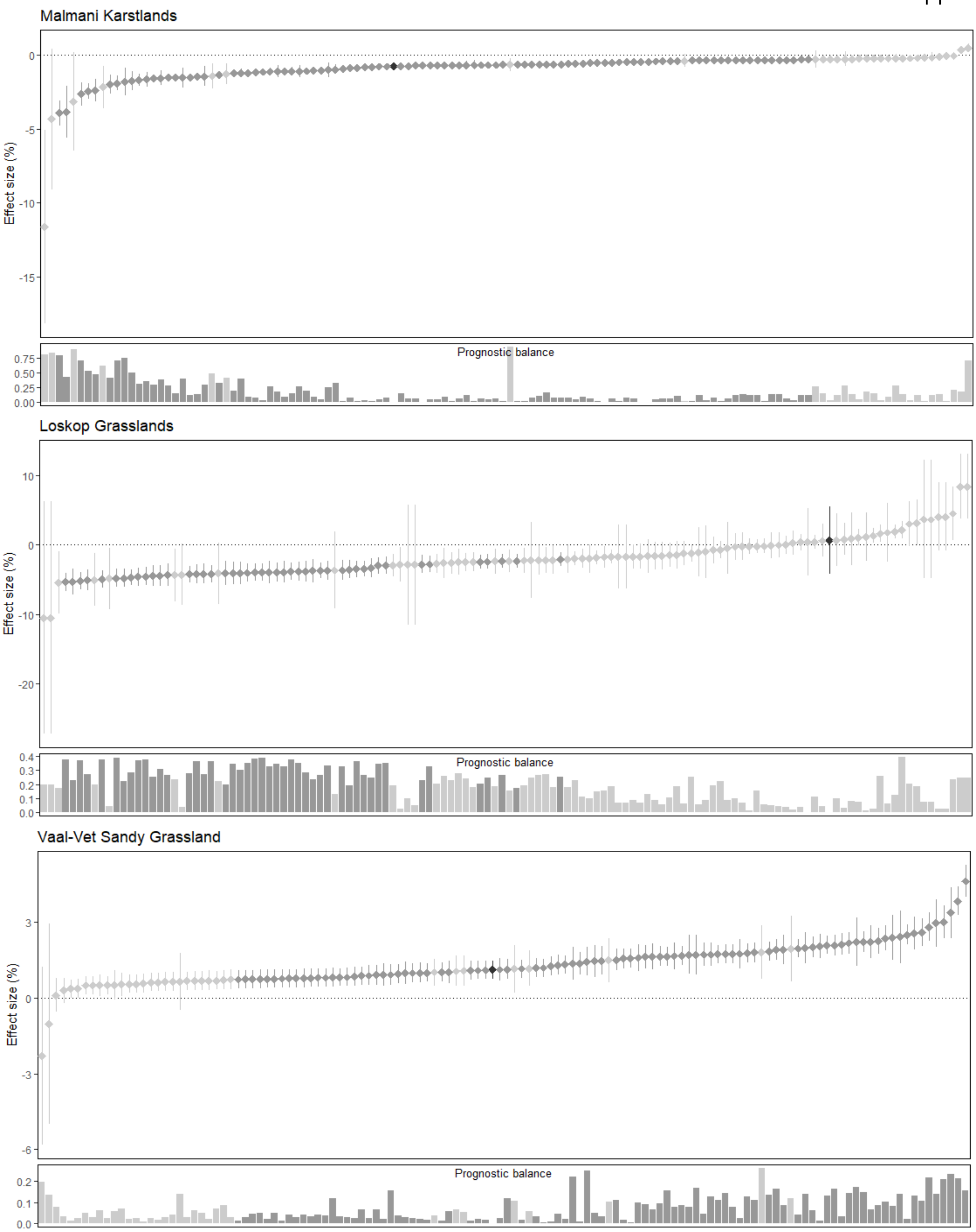
	Matching on all variables			Matching on selected variables			
Ecosystem	**Mean prognostic score for ecosystem**	**Mean prognostic score for matched sample**	**ASMD**	**Mean prognostic score for ecosystem**	**Mean prognostic score for matched sample**	**ASMD**	**Number of variables selected**
Bivane Montane Grassland	0.403	0.379	0.150	0.403	0.406	0.016	7
Blesbokspruit Highveld Grassland	0.609	0.557	0.278	0.609	0.606	0.017	4
Blinkwater Valley	0.251	0.696	4.818	0.251	0.254	0.034	5
Blyde Quartzite Grasslands	0.158	0.173	0.082	0.158	0.157	0.004	7
Boesmanspruit Highveld Grassland	0.410	0.424	0.078	0.410	0.409	0.004	7
Bronkhorstspruit Highveld Grassland	0.588	0.562	0.153	0.588	0.588	0.000	5
Chrissiesmeer Panveld	0.497	0.484	0.076	0.497	0.497	0.001	5
Deneysville Highveld Grassland	0.410	0.416	0.034	0.410	0.410	0.001	8
Dullstroom Plateau Grasslands	0.385	0.339	0.194	0.385	0.385	0.001	5
Egoli Granite Grassland	0.563	0.556	0.030	0.563	0.562	0.000	6
Fort Metcalf Grasslands	0.262	0.245	0.102	0.262	0.261	0.004	6
Greytown North Grasslands	0.750	0.579	0.942	0.750	0.585	0.907	8
Highover Nature Reserve and Roselands Farm Surrounds	0.230	0.332	0.444	0.230	0.233	0.011	7
Impendle Highlands	0.321	0.302	0.087	0.321	0.308	0.060	5
Kaapsehoop Quartzite Grasslands	0.242	0.209	0.201	0.242	0.241	0.002	8
Karkloof Forest Collective	0.360	0.345	0.075	0.360	0.358	0.010	6
Klipriver Highveld Grassland	0.639	0.634	0.022	0.639	0.639	0.000	6
Loskop Grasslands	0.488	0.582	0.385	0.488	0.488	0.002	5
Malmani Karstlands	0.148	0.125	0.127	0.148	0.148	0.000	7
Mauchesburg Alpine Grasslands	0.118	0.099	0.121	0.118	0.118	0.002	8
Ngome Mistbelt Grassland and Forest	0.386	0.352	0.158	0.386	0.387	0.006	8

Oakland and Townhill Ridge	0.624	0.667	0.150	0.624	0.628	0.014	6
Pietermaritzburg South	0.432	0.409	0.118	0.432	0.433	0.004	5
Qudeni Mountain Mistbelt Forest and Grassland	0.129	0.155	0.216	0.129	0.130	0.006	5
Rietvleiriver Highveld Grassland	0.586	0.557	0.173	0.586	0.591	0.030	4
Sekhukhune Mountainlands	0.198	0.183	0.068	0.198	0.198	0.001	8
Sihleza	0.334	0.395	0.326	0.334	0.337	0.013	5
Southern Weza State Forest	0.123	0.172	0.388	0.123	0.123	0.003	5
Stoffberg Mountainlands	0.419	0.394	0.097	0.419	0.419	0.000	8
Tsakane Clay Grassland	0.543	0.494	0.246	0.543	0.539	0.022	3
Vaal-Vet Sandy Grassland	0.462	0.468	0.031	0.462	0.462	0.001	8
Wakkerstroom-Luneburg Grasslands	0.277	0.283	0.031	0.277	0.277	0.002	9
Western Highveld Sandy Grassland	0.492	0.494	0.009	0.492	0.492	0.002	6
Woodbush Granite Grassland	0.333	0.249	0.352	0.333	0.333	0.001	4

are therefore possibly biased. Prognostic balance also indicates that variable combinations producing the strongest positive impact estimates are less likely to be unbiased, and therefore the true effect of threatened ecosystem regulations for Malmani Karstlands is probably a small positive impact. Loskop Grasslands is an example of an ecosystem where the robustness checks indicate great uncertainty in effect estimates, with effect sizes ranging from 10.5% avoided loss to 8.4% increased loss. Many variable combinations produced significant positive impact estimates, but these generally have poor prognostic balance. Variable combinations producing the best prognostic balance on matches resulted in both positive and negative impact estimates. It is therefore most likely that threatened ecosystem regulations had no significant impact on Loskop Grasslands.

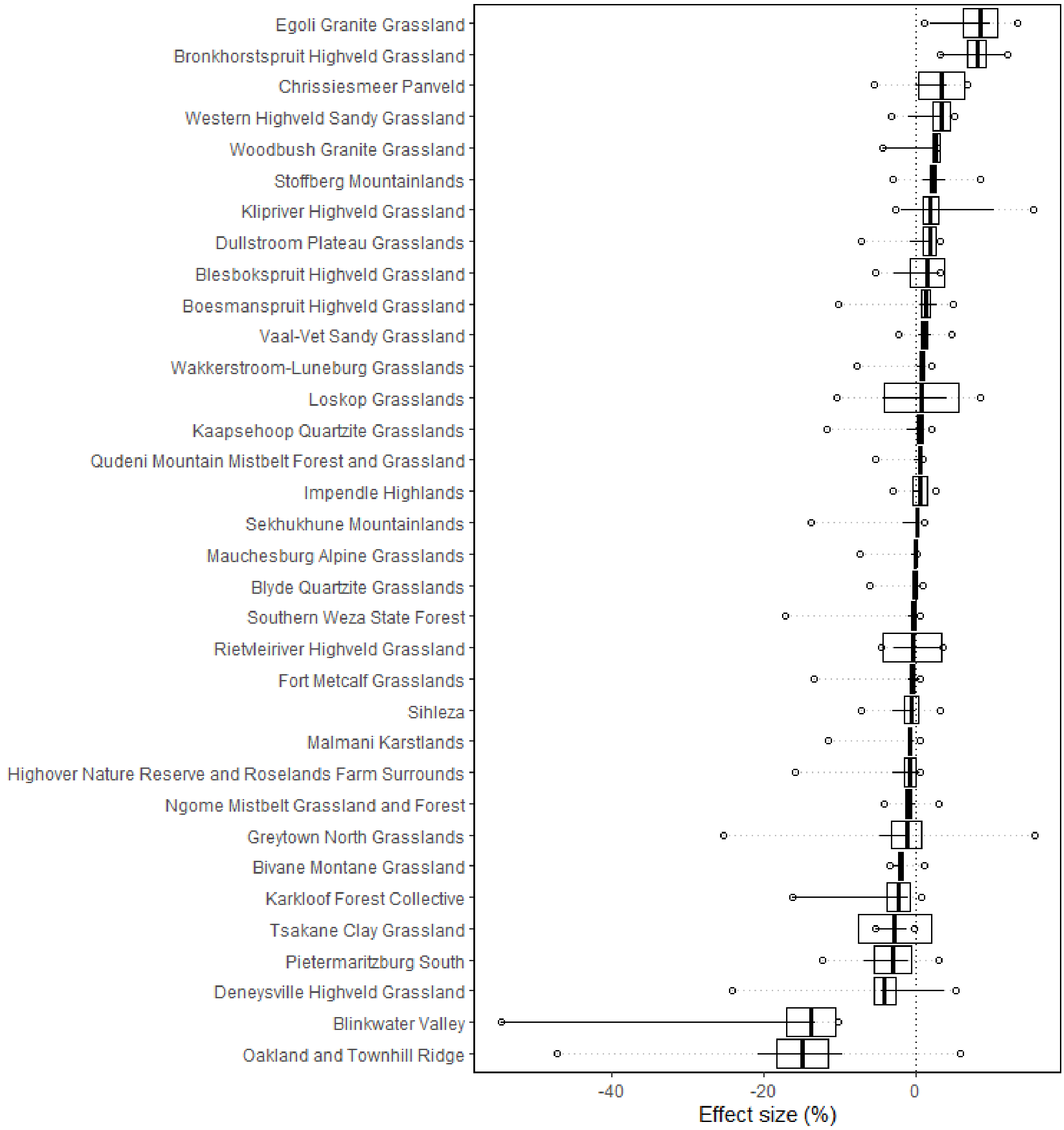
The robustness checks also confirmed that for a small number of threatened ecosystems, higher loss within the ecosystem than in matched non-threatened grasslands is less likely due to methodological shortcomings and probably represents a true effect (Supplementary Figure S4). See for example the robustness checks for Vaal-Vet Sandy Grassland in Supplementary Figure S3 which shows that nearly all matching variable combinations result in significant negative impacts of increased loss relative to the counterfactual. In these ecosystems, matched samples that are well balanced on the prognostic score still produced effect estimates of increased loss relative to the counterfactual, indicating that the apparent negative effects are not due to matching variables failing to find matches under similar baseline land conversion pressure.

These conclusions of course assume that the prognostic model is correctly specified. Linear models of land cover change are typically less accurate than more sophisticated modelling methods that consider stochastic processes and spatial structures in the data (Kolb et al. 2013, Mas et al. 2014, Overmars et al. 2003), but such approaches were beyond the scope of this study. Stuart et al. (2013) found that even misspecified prognostic models produced prognostic balances that were well



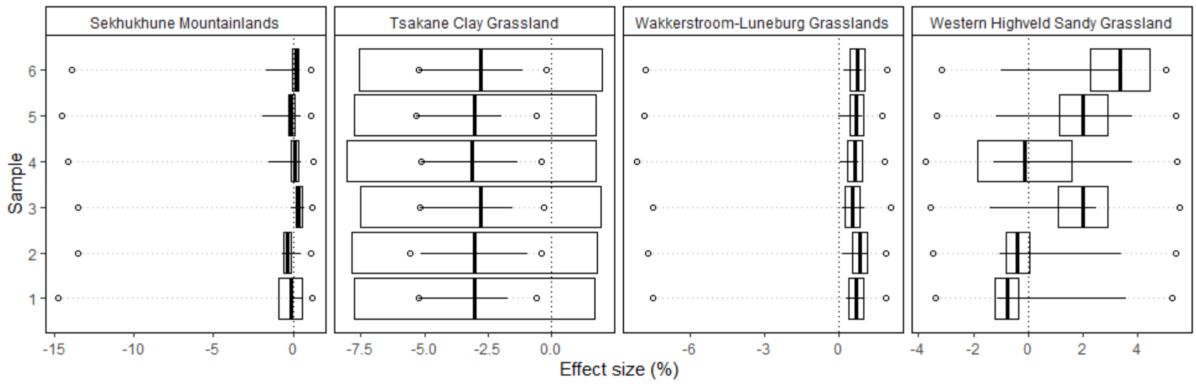
SUPPLEMENTARY FIGURE S3 The range of effect estimates for three example ecosystems obtained through matching on different variable combinations. For each ecosystem, the upper vertical line plot represents the effect estimate and its standard error, with the corresponding bars below indicating the prognostic balance of the matched sample. Light grey indicates non-significant effect estimates, while dark grey indicates significant effects. The effect highlighted in black is from the matching variable combination selected based on best prognostic balance. The horizontal line at zero on the Effect size axis indicates the threshold between positive impact (effect estimates below zero) and negative impact (effect estimates above zero).

correlated with reductions in bias in effect estimates, but do not indicate to what degree the prediction accuracy of their misspecified prognostic models were reduced.



SUPPLEMENTARY FIGURE S4 The range of effect estimates for each threatened ecosystem derived from different combinations of matching variables. Circles indicate the full range of estimated effects, with solid lines indicating the range of effects found within the matched samples with the top 10% best prognostic balance. Boxes indicate the effect estimate and error of the matching variable combination that had the best prognostic balance.

The impact of random sampling of treatment observations for larger ecosystems on matching model selection and effect estimates was also investigated. For selected ecosystems, the matching model selection procedure was run six times on different random samples, and the range of estimated effects and selected matching model compared. Random sampling had a negligible effect on the range of effect estimates derived from the full set of matching variable combinations (Supplementary Figure S5). For some ecosystems, such as Tsakane Clay Grassland and Wakkerstroom-Luneburg Grasslands, the same variable combination consistently had the best prognostic balance across multiple random samples, resulting in very low variability in the point effect estimate for the ecosystem. In other ecosystems however, a range of variable combinations had the best prognostic balance, depending on the sample. This resulted in somewhat larger variability in the effect estimate, but not as large as the effect of variable selection. For example, in Western Highveld Sandy Grassland, the effect estimates on different samples ranged from 0.8% avoided loss to 3.3% increased loss (Supplementary Figure S5).



SUPPLEMENTARY FIGURE S5 The effect of random sampling on matching model selection and effect estimates for selected threatened ecosystems. The symbology is as for Supplementary Figure S4.

For ecosystems where sampling had an impact on matching model selection, the best variable combination in each sample was recorded, and its ranking on prognostic balance in other samples investigated. It was found that these variable combinations were consistently present within the matched samples with the top

10% best ranked prognostic balance across different random treatment samples. The sampling tests also revealed that the range of effects indicated by the best 10% of variable combinations is relatively consistent across samples (solid horizontal lines in Supplementary Figure S5), even when the point effect estimate is not.

Based on these findings, a general assumption was made that point effect estimates based on the selection of a single best matching model was probably not realistic for all threatened ecosystems, because as with Western Highveld Sandy Grassland, it cannot be certain where within the range of effects suggested by the 10% best variable combinations the true effect is. Therefore, for each ecosystem, in addition to a point estimate of effect size, an impact category adjusted for uncertainty due to random sampling and matching model selection was also assigned. A challenge with assigning this category purely on the range of effects suggested by the 10% best variable combinations is that while these may suggest clear positive or negative impacts, the error estimates and p-values sometimes suggest that the null hypothesis that the difference in loss of natural areas between the threatened ecosystem and its matched control sample is due to chance cannot be rejected. See for example Tsakane Clay Grassland in Supplementary Figure S5, where the effect sizes derived from the coefficients are consistently negative (indicating avoided loss), but the error is also consistently large, resulting in p-values >0.05. Therefore, only for ecosystems where both the best 10% effect range and p-values on point estimates agreed on either avoided loss or increased loss were these impact categories assigned to ecosystems, while for the rest, intermediate categories comprising the sensitivity of effect estimates to sampling and matching model selection were created (Supplementary Table S8).

SUPPLEMENTARY TABLE S8 Rules for assigning impact categories to threatened ecosystems adjusting for uncertainty due to sampling and matching model selection. The selected model is the matching variable combination with the lowest prognostic balance. The 10% range is the variable combinations ranked within the lowest 10% prognostic balance. The ordinal ranks were used in meta-regression of factors explaining differences in impact between ecosystems.

Selected model		10% range		Impact category	Ordinal rank
Coefficient	p-value	Smallest coefficient	Largest coefficient		
-	<0.5	-	-	Avoided loss	5
-	<0.5	-	+	Avoided loss – no impact	4
-	>0.5	-	-		
-	>0.5	-	+	No impact	3
+	>0.5	-	+		
+	<0.5	-	+	No impact – increased loss	2
+	>0.5	+	+		
+	<0.5	+	+	Increased loss	1