The IUCN Red List for Ecosystems: How does it compare to South Africa's approach to listing threatened ecosystems?

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Plagiarism declaration

I, Maphale Stella Matlala know the meaning of plagiarism and declare that all of the work in the dissertation (or thesis), save for that which is properly acknowledged, is my own.

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1.1 Abstract

In the early 2000s, South Africa extended the concept of the International Union for Conservation of Nature (IUCN) Red List of Threatened Species for its application to ecosystems. These efforts were made in the absence of internationally-accepted standards for assessing threats to ecosystems. South Africa Ecosystem Threat Status (SA ETS) standards are purposed to aid biodiversity monitoring efforts and have progressed into the legislated national listing of the threatened ecosystems. The gazetted list of threatened ecosystems is ratified to inform policy development, conservation, and land-use planning tools. In 2014, the IUCN published a framework for the Red List of Ecosystem (RLE), which has become the internationally-accepted norm. It makes sense for South Africa to adopt this new framework, however, changing and/or updating the national assessment framework needs to be explained and defended, as there are implications relevant to conservation decisions and land-use planners. As such, comparing the IUCN RLE and SA ETS frameworks is an important part of the process. Here I review the philosophy and purpose of the SA ETS and IUCN RLE frameworks, compare, and contrast their scientific basis. The overarching conceptual similarities include: (i) the definition of the assessment unit, (ii) spatial and functional criteria, and (iii) the ordinal risk categories. Similarly, there are differences between the two frameworks which include the: (i) nested sub-criteria that measure change over different timeframes, (ii) absolute threshold values, and (iii) integration of uncertainty in assessments. Whilst these are all important differences, they only matter in the South African context if the overall assessment outcomes differ substantially as they could disrupt the implementation of the EIA regulations, as well as the conservation and land-use planning initiatives. Therefore, this thesis aims to investigate these differences quantitatively and interrogate their basis to better understand the implications of adopting the IUCN RLE standards and their efficacy as a conservation tool in South Africa.

1.2 Introduction

South Africa is ranked by the United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC) as the third most biologically diverse country in the world (Lombard, 1995). This complicates trade-offs between the country's economic needs and the conservation of biodiversity-rich areas (Reyers *et al*., 2001). To harmonize these competing activities, compromises are inevitable and have become a way of life. In South Africa, ecosystem-focused biodiversity monitoring efforts have become a powerful tool used to mainstream biodiversity considerations into policy development, and land-use planning (Botts *et al*., 2020). These efforts have rapidly gained traction internationally as an effective approach for preserving ecological processes and patterns (Bland *et al*., 2019).

For decades conservation practices were largely species-focused, due to the lack of a global and hierarchical classification system that defines the diversity and transformation of ecosystems at multiple spatial and temporal scales (Faber-Langendoen *et al*., 2014; Boitani *et al*., 2015; Nicholson *et al*., 2014; Keith *et al*., 2015a). Despite the broad acceptance of the species approach, there are concerns about it being used in isolation as the primary tool to inform biodiversity conservation and environmental management efforts. These concerns largely stem from this approach overlooking broad-scale ecological processes and patterns which are equally important in sustaining species populations within the system (Franklin, 1993; Cowling *et al*., 2004; Keith *et al*., 2015a). Other important limitations are that: (i) there are far too many (approximately 374 000) known plant species (Christenhusz & Byng, 2016) to completely and comprehensively implement the species-by-species approach, (ii) a large number of these species are data deficient (Rebelo *et al*., 2011) and (iii) it is costly and time-consuming exercise. What complicates matters more is the lack of readily available analytical tools to conduct species assessments (Bachman *et al*., 2011). These limitations have resulted in the assessments relying heavily on expert field knowledge (Franklin, 1993; Rodrigues *et al*., 2006). It has become increasing evident that a broader-oriented conservation approach is key in preserving a richer array of biological diversity and will also complement the conservation work done at the species level (Bland *et al*., 2017).

Until recently, there have been no internationally accepted and consistent scientific standards for how this can be done (Keith, *et al*., 2013; Rodriguez *et al*., 2015). In the early 2000s, South Africa and others (including Australia, United Kingdom, and New Zealand) each independently extended the concept of the International Union Conservation of Nature (IUCN) Red List of Threatened Species to apply to ecosystems (see, Driver *et al.*, 2005; Joint Nature Conservation Committee, 2007; Bonifacio & Pisanu, 2013; Davis *et al*., 2016). The ecosystem assessment standards are purposed to aid biodiversity monitoring efforts and have progressed into the legislated national listing of the threatened ecosystems in South Africa. South Africa's gazetted national list of threatened ecosystems is ratified to inform policy development, conservation, and land-use planning tools (RSA, 2011; Botts *et al*., 2020). In 2014, the IUCN published a framework for the Red List of Ecosystems (RLE), which has become the norm internationally (Rodriguez *et al*., 2015; Bland *et al*., 2017). It makes sense for South Africa to adopt this new framework, however, changing and/or updating the assessment framework needs to be explained and defended, as there are implications relevant to conservation decisions and land-use planners. As such, comparing the IUCN RLE and South Africa (SA) Ecosystem Threat Status (ETS) frameworks is an important part of the process.

The focus of this thesis is to assess the origins and history of the IUCN and South Africa's approach to assessing threats to ecosystems. In this chapter [Chapter 1] I review the key concepts including the scientific basis and criteria to understand the purpose and philosophy of the SA ETS and IUCN RLE frameworks. In Chapter 2, I compare and contrast the SA ETS and IUCN RLE assessment outcomes of ecosystems susceptible to only spatial threats. Finally, in Chapter 3, I test whether the IUCN RLE is a good proxy for the distribution of threatened species in South Africa.

1.3. South Africa recognizing the need for a system that assesses threats to the ecosystem

Findings from a White Paper focused on the *Conservation and Sustainable Use of South Africa's Biological Diversity* (1997) brought attention to inadequate biodiversity conservation practices outside protected areas. Prior to 2000, biodiversity conservation legislation gave little to no attention to biodiversity outside protected areas. Where conservation legislation existed, even for protected areas, it was poorly enforced. This is despite protected areas being ostensibly established to safeguard biodiversity and help achieve national and international conservation goals (DEAT, 1997; Alexander, 2009). In response to the White Paper (1997), the National Environmental Management: Biodiversity Act (no.10 of 2004) (hereafter, Biodiversity Act) and National Environmental Management: Protected Areas Act (no. 57 of 2004) introduced a suite of legal tools to promote effective management and conservation of indigenous biodiversity both inside and outside protected areas. These include biodiversity agreements and management plans, bioregional plans, biodiversity stewardship programmes, and national lists of threatened ecosystems and species. The protected areas mandate is to ensure that there is a representative, ecologically viable sample of all ecosystem types within the reserve network (Alexander, 2009). Furthermore, the Biodiversity Act mandated the South African National Biodiversity Institute (SANBI) to monitor and report periodically on the country's biodiversity status.

In the absence of international RLE standards, SANBI in collaboration with various stakeholders developed a set of assessment criteria to assess the declines of terrestrial ecosystems, which have been ratified in national legislation (Botts *et al*., 2020). This makes South Africa one of the first countries to use monitoring data to inform the development of legal tools that protect broad-scale ecological features (Driver *et al*., 2005; Driver *et al*., 2012). Since ecosystem threat status assessment requires clearly defined and delineated ecosystem types, the SA ETS assessments rely on and were made feasible by the simultaneous development of a wall-to-wall national map of ecosystem types; namely the National Vegetation Map of South Africa (Mucina & Rutherford, 2006). The delineation and recognition of ecosystem types were also crucial in guiding protected area expansion to achieve the goal of protecting a representative and ecologically viable sample of all ecosystem types within the reserve network (Department of Environmental Affairs, 2016).

1.3.1. SANBI championing biodiversity conservation efforts

SANBI's initial attempt at reporting on the state of South Africa's biodiversity is presented in the National Spatial Biodiversity Assessment (NSBA) 2004. This report has a strong emphasis on spatial aspects of biodiversity such as priority areas, and protected areas. Subsequently, the NSBA 2004 informed the development of the first National Biodiversity Strategy and Action Plan (NBSAP; DEAT, 2005). The NBSAP is a twenty-year strategy South Africa established to fulfil its obligations as a signatory to the Convention on Biological Diversity (Raimondo, 2015). It consists of a list of strategic objectives that align with the international agenda and national development goals that promote sustainable use of the country's prized terrestrial and aquatic biodiversity (DEAT, 2009). Its main emphasis is on mainstreaming biodiversity considerations into policies and other decision-making tools that contribute notably to the country's economic development growth (Government of South Africa, 2015).

Over the years, the underlying biodiversity data in South Africa has matured and become more readily available. These advances prompted the decision to broaden the NSBA scope such that both spatial and broader themes such as climate change, and alien species are taken into consideration as they have an impact on the ecological state of biodiversity. This led to the development of the National Biodiversity Assessment (NBA) 2011 and 2018 reports. The NBA is purposed to provide a scientifically defensible summary report on the state of South Africa's biodiversity across terrestrial, inland aquatic, estuarine, coastal and marine environments on a periodic interval (Driver *et al*., 2012; Skowno *et al*., 2019a).

1.4. The role of the listed ecosystems in environmental policy and land-use planning

South Africa, led by SANBI, developed a national list of threatened terrestrial ecosystems based on national criteria, identifying different symptoms of ecosystem change that can lead to irreversible loss of natural habitat (Driver *et al*., 2012). Publishing the list under the government gazette created direct links to policy-related tools such as the Environmental Impact Assessments (EIA) regulations as outlined in the National Environmental Management Act (no.107 of 1998, amended in 2002 and 2004). The EIA regulations are aimed at discouraging land-intensive activities in gazetted ecosystem types. However, these regulations only apply to Critically Endangered and Endangered ecosystem types, leaving the Vulnerable and Least Concern with no additional protections (RSA, 2011).

In terms of Critically Endangered and Endangered ecosystems, the EIA regulations pose strict rules to guard against the removal of at least 300 m^2 of indigenous natural vegetation. Therefore, any proposed development activities within these threatened ecosystems trigger the need to comply with the EIA regulations (NEMA-Act no.107 of 1998). Since most of the country's development activities occur at the municipal level, the Local Government Municipal Systems Act 32 of 2000 requires that the listed ecosystems be featured in planning tools (e.g. Integrated Development Plans and Spatial Development Frameworks) to inform spatial planning initiatives (RSA, 2011). Many of these legal tools overlap to some degree, nonetheless, they represent the degree to which South Africa has invested its efforts and resources in mainstreaming biodiversity considerations. More importantly, they provide different avenues for a wide array of biodiversity issues to be considered and addressed timely and at various phases of the project cycle from planning, operation, and completion.

1.5. The rise of the IUCN Red List of Ecosystems

For years there have been ongoing formal and informal discussions in the international arena regarding the need to develop a standardised, broad-oriented assessment framework that would promote the conservation of ecological processes and patterns, including species' interactions with each other and the physical environment (Franklin, 1993; Rodríguez *et al*., 2012; Keith *et al*., 2013). This concept was formally proposed by Edward Maltby, the first chair of the Commission on Ecosystem Management (CEM) at the first IUCN World Conservation Congress in 1996. Almost a decade later, the CEM and the IUCN secretariat showed broad support for the development of a standardised RLE framework (Rodríguez *et al*., 2012). In October 2008, the RLE Committee for Scientific Standards was established under the umbrella of the IUCN to formally lead the development of an international framework for the RLE (Nicholson *et al.*, 2009).

As part of the initial attempts towards this development process, Nicholson *et al*., (2009) reviewed 12 existing regional RLE frameworks, from which key similarities were identified. The frameworks assess the risk of ecosystem decline based on: (i) the loss of natural habitat, (ii) limited natural extent, and (iii) decline in ecological processes, with some adopting the use of thresholds and risk categories to rank ecosystem declines. These concepts laid a foundation for the development of the IUCN RLE standards version 2.0, officially adopted on the 21st May 2014 (Rodriguez *et al*., 2015). The IUCN RLE framework consists of a set of criteria and risk categories to assess how close an ecosystem is to collapsing. Collectively, the assessment criteria and categories ensure that the assessments are (i) conducted systematically across different environments and geographic areas, (ii) transparent and repeatable, (iii) easily understood by policymakers and the general public, and (iv) consistent with and complementary to the IUCN Red List of Threatened Species (Bland *et al*., 2017).

The systematic application of the IUCN RLE criteria and categories addresses national and regional challenges caused by inconsistent application of the listing methods. For example, in Australia, a multi-jurisdictional approach for fulfilling various local and national environmental management obligations (including the RLE) was adopted. While this implies that the responsibilities are shared across different states and federal governments, it also became the source of confusion amongst cross-jurisdictional contributors to the RLE assessments. The administrative burden for maintaining the RLE lists also increases (Hawke, 2009). Acknowledging that the benefits of having multi-jurisdictional frameworks are outweighed by their drawbacks, the states and federal governments in Australia have shown broad support for the alignment with the IUCN RLE framework. This decision is intended to harmonise the historically disparate listing methods across multiple jurisdictions and increasing the level of efficiency across all jurisdictions (Bonifacio & Pisanu, 2013; Bland *et al*., 2019).

For the conceptual and operational rigour of the resulting assessments, the IUCN has developed tools such as the *redlistr package* https://github.com/red-list-ecosystem/redlistr and published peer-review scientific papers and case studies to aid the process of building capacity within the assessment teams (Bland *et al*., 2018). The availability of these resources (accessible on https://iucnrle.org/resources/capacity-building/) has increased interest across the globe with many more countries including Colombia, adopting the IUCN RLE standards (Etter *et al*., 2017). Their alignment is encouraging as it strengthens the global capacity for monitoring and reporting on the world's biological diversity (Rodriguez *et al*., 2015). Furthermore, it aids progress towards establishing a global list of threatened ecosystems by 2025 (Bland *et al*., 2017).

1.6. The scientific basis of SA ETS versus the IUCN RLE approach

The review of the origins of the SA ETS and the IUCN RLE frameworks suggests that they both aid biodiversity monitoring efforts. More importantly, they both inform the development of various high-level tools that are purposed to reduce or avert species extinction and ecosystem collapse (RSA, 2011; Keith *et al*., 2013). South Africa's alignment with the IUCN RLE standards forms the basis of the next sections of this chapter which compare and contrast the two frameworks based on their scientific basis and assessment criteria.

Table 1: Comparison of the underlying ecological and theoretical concepts adopted in the South Africa Ecosystem Threat Status (SA ETS, source: RSA, 2011) and the International Union for Conservation of Nature Red List of Ecosystems assessment frameworks (IUCN RLE, source: Bland et al., 2017).

Ecological and theoretical concepts	**South Africa Ecosystem Threat Status (SA ETS)**	**the International Union for Conservation of Nature Red List of Ecosystems assessment frameworks (IUCN RLE)**
Assessment units	Vegetation types: used synonymously to ecosystems defined to be a dynamic complex of animal, plant, and micro-organism communities and their non-living environment (Tansley, 1935).	Ecosystem: complexes of organisms and their associated physical environment, within an area.
End-point of decline	Irreversible loss of natural habitat: not formally defined.	Collapse: forms part of the IUCN RLE categories triggered when it is "virtually certain that the defining biotic and abiotic features are lost from all occurrences, and the characteristics of the native biota are no longer sustained."
Uncertainty		The existence of uncertainty in the outcomes of the assessments is well acknowledged in the RLE assessments. To ensure that these assessments remain consistent across environments and repeatable over time, a confidence bounds method has been proposed in estimating uncertainty in the assessment outcomes.
Time frames	Historic declines: assessed using a reference date of 1750, representing the earliest onset of global natural resource exploitation.	Historic declines: uses a reference date of 1750 which represents the earliest onset in which the industrial exploitation of natural resources commenced. Past, present, and future declines: 50-year time period is used to diagnose ecosystem dynamics and trends with reasonable certainty.
Spatial criteria	Irreversible loss of natural habitat (criterion A1): lists ecosystems that have experienced declines in geographic distribution. Rate of loss of natural habitat (criterion B): identifies ecosystems with high rates of spatial decline.	Reduction in geographic distribution (A): lists ecosystems that have experienced declines in geographic distribution.

	Limited extent and imminent threat (criterion C): lists ecosystems that are small in spatial extent and experiencing ongoing declines.	Reduction in geographic distribution (criterion B): lists ecosystems that are naturally small in spatial extent and experiencing ongoing declines.
Functional criteria	Ecosystem degradation and loss of integrity (A2): ecosystem undergoing significant changes in structure, function, and composition. Threatened plant species associations (criterion D1): lists ecosystems based on the number of threatened species it supports. Threatened animal species associations (criterion D2): lists ecosystems based on the number of threatened species it supports. Habitat fragmentation (criterion E)	Environmental degradation: (criterion C): lists ecosystems experiencing degradation. Disruption in biotic processes (criterion D): lists ecosystems that are undergoing functional declines due to disruption of biotic processes.
Targets and thresholds	Biodiversity targets: ranges between 16% and 36% of the intact ecosystem's total spatial extent. The individual targets represent at least 75% of the remaining plant and genetic diversity occurring within each ecosystem type. South Africa used the Species Area-Relationship technique to calculate conservation targets. Thresholds: fixed absolute values which are expert and policy-driven, therefore lack ecological relevance.	Thresholds: fixed absolute values that are expert and policy-driven, therefore they lack ecological relevance.
Risk categories	Critically Endangered (CR): an ecosystem has undergone severe degradation of ecological structure, function, or composition as a result of human intervention. Endangered (EN): an ecosystem has undergone degradation of ecological structure, function, or composition as a result of human intervention, although they are not CR ecosystems. Vulnerable (VU): has a high risk of undergoing significant degradation of ecological structure, function, or composition as a result of human intervention, although they are not CR or EN ecosystems.	Critically Endangered (CR): An ecosystem is at an extremely high risk of collapse when the best available evidence suggests that it triggers the thresholds of one or more of the assessment criteria for critically endangered. Endangered (EN): an ecosystem at a very high risk of collapse when the best available evidence suggests that it meets one or more of the assessment criteria for endangered. Vulnerable (VU): an ecosystem at a high risk of collapse when the best available evidence suggests that it meets one or more of the assessment criteria.

1.6.1. Assessment units

South Africa uses the National Vegetation Types described by Mucina & Rutherford, (2006) as proxies for ecosystem types. Both the IUCN and South Africa definitions of an ecosystem (Table 1) are conceptually analogous as they include key elements that mirror the Tansley (1935) definition. Both definitions delineate ecosystem types based on the: (i) characteristic of biotic elements; (ii) abiotic environment; (iii) processes and interactions between species and their physical environment; (Mucina & Rutherford 2006; Keith e*t al.*, 2013). The two definitions are broad enough to be adopted under any ecosystem classifications despite the Mucina & Rutherford, (2006) definition established solely for the delineation of terrestrial vegetation or ecosystem types. It is worth noting that South Africa also made alternative delineations of ecosystems that did not match the National Vegetation map. These special ecosystems included animal habitat (e.g. Rhino and Cranes) and very small forest patches with known biodiversity which are regarded as priority areas for meeting explicit biodiversity targets as defined in a systematic biodiversity plan, etc. Although the delineation of these ecosystems deviated from the conceptual framework, it was a compromising approach that was undertaken to include provincial and local scale data in the listing of ecosystems (RSA, 2011).

1.6.2. The endpoint of ecosystem decline

The endpoint of ecosystem decline (hereafter, ecosystem collapse) is a relatively new concept in conservation biology, referring to a state when an ecosystem starts losing vital ecological features that provide distinct identity (Keith *et al*., 2013; Nicholson *et al*., 2014). The concept has received increasing attention, with several articles reviewing its practicality (Boitani *et al.*, 2014; Nicholson *et al*., 2014; Sato & Lindenmayer, 2017). Its origins likely stem back to the Australian threat category 'Presumed extinct', which refers to the irreversible loss of natural habitat such that an ecosystem type is unlikely to recover its species composition, structure, and ecological processes back to its reference state (for example, the conversion of a woody ecosystem type to pasture grasslands) (Nature conservation Act 1980, p.25). Analogous to species extinction, the concept of 'presumed extinct' was later re-introduced by Keith *et al.*, (2013) as 'ecosystem collapse' (Table 1). It forms part of the risk categories adopted under the IUCN RLE framework and invoked once an ecosystem is completely modified to a point where key ecological features and characteristics providing its distinct identity have entirely disappeared (Table 1; Bland *et al.,* 2017). South Africa uses the term 'irreversible loss', informally and synonymously to ecosystem collapse for natural habitat experiencing spatial decline but lacks a clear ecological and formal definition (RSA, 2011).

The IUCN definition of ecosystem collapse has received criticism from Boitani *et al*., (2014), who argues that it is subjected to ambiguous interpretations and uncertainty as ecosystems are dynamic and change over time and in space *(*Boitani *et al.,* 2014; Keith, 2015a; Sato & Lindenmayer, 2017). However, Bland *et al*., (2017) argue that the concept of collapse is ecosystem-specific. As such, the intentions of the decision-makers or the assessors influence the decision to list an ecosystem in the collapsed category. For example, the intention may be to trigger conservation and management responses that would prohibit further conversion of the remaining small patches of indigenous vegetation or restore the degraded areas within the ecosystem classified as collapsed (Dudley *et al*., 2014). To eliminate ambiguity in the assessments, Bland *et al*., (2018) recommends: (i) a clear description of the reference and collapsed state, (ii) clearly defined collapse state and recovery transitions, (iii) the use of vital and distinct features as indicators, and (iv) adoption of quantitative thresholds.

Often, a spatially collapsed ecosystem is easily detectable using historical data in association with tools such as remote sensing. Visual changes in vegetation structure, configuration, and geographic distribution signal a warning to assessors that an ecosystem is on the verge of collapse or has collapsed. For example, an ecosystem type that has been entirely replaced by anthropogenic land cover such as open cast mining activities has collapsed. These methods may be less proficient at detecting ecosystem degradation, or ecosystems that have been replaced by novel ecosystems resembling natural habitat (e.g. planted pastures may resemble grasslands) (Bland *et al*., 2017).

1.6.3. Uncertainty

Uncertainty is an old and well-documented concept that has received growing interest in conservation biology over the years (Regan *et al*., 2003; Nicholson *et al.*, 2009; Rocchini *et al.*, 2013; Keith *et al*., 2013; Blackburn *et al.*, 2014; Nicholson *et al*., 2014). Its sources are diverse, particularly in ecological assessments, and are presented by the: (i) adopted absolute threshold values, (ii) definition of the reference and collapsed state of ecosystems, and (iii) technical methods used to map ecosystem distribution which could potentially over-or under-estimate the remaining natural extent due to pixels misclassification (Bland *et al*., 2017). The IUCN developed a technique that incorporates various sources of uncertainty to estimate confidence limits that quantify its magnitude in the assessment outcomes (Butchart *et al*., 2007;

Bland *et al*., 2017). For example, Keith *et al*., (2013, Supplementary material) assessed the risk of the Aral Sea collapse between 1960 and 2005. Its initial surface area was estimated to be 67 499 km^2 and it declined to 17 382 km^2. Between 1976 and 1989, the Aral Sea experienced some spatial decline and was thought to have collapsed when the area estimates were below 55 700 km^2 but it could also have plausibly collapsed when it declined to 39 734 km^2. To capture this uncertainty, the authors calculated the risk category under criterion A based on 39,734 km^2 estimates (resulting in the listing in the Collapse category) as well as based on 55,700 km^2 (also listed in the Collapse category), so the plausible bounds were Collapse-Collapse. In this case, the Aral Sea declined below the absolute threshold values, so the status category is Collapsed regardless of the threshold used.

The IUCN RLE has also introduced the Data Deficient category as an additional measure to deal with uncertainty in the assessment outcomes. This category is triggered when there is inadequate data to apply any of the IUCN RLE sub-criterion that assesses the risk of collapse exacerbated by either spatial and/or functional threatening processes (Bland *et al*., 2017). Despite the data limitations, uncertainty estimates remain to be an integral part of the risk assessments as they increase the credibility and confidence of the assessment outcomes, as well as the quality of rendered decisions (Bland *et al*., 2017). In cases where there are conflicting objectives (e.g. anthropogenic land-use activities versus biodiversity conservation efforts), uncertainty estimates in the RLE assessment outcomes can guide decision-makers in achieving optimal solutions or trade-offs between the alternative uses of the natural areas (Ascough *et al*., 2008). Both the inclusion of confidence limits and the Data Deficient category in the RLE assessments serve as an acknowledgment that the extent and severity of anthropogenic land cover change and the disruption of the ecological processes and functions on natural habitats have not yet been fully understood. Unfortunately, this is an important component that is overlooked in the SA ETS assessments (see, Driver *et al*., 2005; Driver *et al.*, 2012; Skowno *et al*, 2019b).

1.6.4. Timeframes

The IUCN RLE criteria assess the risk of ecosystem collapse over three temporal points (historic: baseline date of 1750, present: over the past 50 years, and future: over any 50-year window including the past, present, and future; Keith *et al*., 2013). A 50-year period is presumed to be adequate to diagnose past and future trends with some level of certainty while the 1750 reference date represents the earliest global inception of natural resource exploitation.

This historic reference date is not cast in stone, as such, the IUCN allows for its adjustment for countries or regions with sufficient and credible evidence to suggest that the anthropogenic land cover changes commenced prior- or subsequent to this date (Keith *et al*., 2013).

The Vegetation Map of South Africa represents the historic extent of the country's ecosystem types (e.g. Mucina & Rutherford, 2006). The earliest attempts to produce such a were that of Pole Evans in 1936. However, his mapping approach lacked consistency as some parts of the map were delineated at a fine-scale, while others were coarse. For example, he identified three grassland types and three savanna types (Mucina & Rutherford, 2006) but failed to provide information on the diversity of ecosystem types in the Eastern Cape province (Lubke *et al*., 1986). A great milestone was achieved by John Acocks when he mapped the extent of 70 ecosystem types in South Africa, Lesotho, and Swaziland with a great level of detail (Mucina & Rutherford, 2006). This map laid a foundation for the maps that were later published (e.g. Mucina & Rutherford, 2006). The improved wall-to-wall ecosystem map is used as a reference state in the SA ETS assessments to assess natural habitat loss to anthropogenic land-use (RSA, 2011). Although the earliest recorded attempts of ecosystem mapping date as far back as 1936, there is no credible evidence of when anthropogenic land-use commenced. As such, South Africa adopted the recommended IUCN reference date of 1750 to assess historic declines (Skowno *et al*., 2019b).

1.6.5. Assessment criteria

The SA ETS and IUCN RLE classification systems consist of a range of criteria designed to classify threatening processes influencing the risk of ecosystem collapse (RSA, 2011; Keith *et al*., 2013). Several of these criteria are directly comparable between the two systems. These include spatial criteria that assess the risk of collapse based on the declines in natural extent (SA ETS criterion A1 and IUCN RLE criterion A3) and limited natural extent (SA ETS criterion C and IUCN RLE criterion B). The other comparable criteria assess the risk of collapse based on functional declines (Table 1). Despite these conceptual similarities, SA ETS assessments are largely focused on spatial symptoms of collapse with selected ecosystem types also being assessed for functional declines due to the lack of nationwide ecosystem degradation data (Skowno *et al*., 2019b). On the other hand, only a few criteria are incomparable as they are unique to each framework. These criteria assess how close each type is to collapsing by: (i) integrating multiple threat interactions into simulation models to estimate the probability of collapse (IUCN RLE criterion E) (Bland *et al*., 2017) and (ii) identifying priority areas (e.g.

special ecosystems comprising of small forest patches with known biodiversity) for meeting biodiversity targets as defined by a systematic biodiversity plan (SA ETS criterion F) (RSA, 2011; see, the Appendix A for a full range of the IUCNRLE and SA ETS assessment criteria).

1.6.6. Biodiversity targets and decision thresholds

Quantitative target- and threshold-based approaches play a prominent role in guiding environmental management and conservation responses to broad-scale ecological features such as ecosystems (Huggett, 2005). Their adoption in both the SA ETS and IUCN RLE assessments is aimed at measuring how far advanced an ecosystem is from triggering the collapsed state or being irreversibly lost. The absolute threshold values are different between the two systems (see, Appendix A). However, the difference in their technical application is subtle. For example, in the SA ETS assessments, the biodiversity targets and fixed absolute values are used to assign an ecosystem type to any one of the ordinal risk categories (Critically Endangered (CR), Endangered (EN), or Vulnerable (VU)) depending on the proportion of an ecosystem type remaining in a pristine condition (RSA, 2011). On the contrary, the IUCN RLE thresholds assign an ecosystem type into any one of these ordinal categories based on the natural extent lost, with the exception of criterion B which focuses on the remaining natural extent (Bland *et al*., 2017).

The national biodiversity targets adopted in the SA ETS framework were calculated for selected ecosystem types based on species-area relationships (SAR) technique and the rest were estimated based on experts' knowledge. Nonetheless, this technique relies on plant species inventory data to estimate the extent of each ecosystem that is to be retained in a natural condition to conserve the majority of its species diversity (Desmet & Cowling, 2004). Estimated biodiversity targets range between 16% and 36% (Driver *et al*., 2012), and are intended to preserve at least 75% of the remaining plant and genetic diversity occurring within each ecosystem (Rouget *et al*., 2004). While the 75% aligns with the global target represented in three (Targets 5, 7, and 8) of the CBD Aichi Targets (CBD, 2011) they only conserve a single occurrence of the 75%, of the remaining plant and genetic diversity (Raimondo, 2017), hence they are not necessarily viable.

Unlike South Africa, the IUCN relies only on fixed absolute thresholds to determine ecosystem threat status across all different realms (Bland *et al*., 2017), possibly due to limitations on the data available or shortcomings of the methods currently available for estimating biodiversity targets, including the SAR. For example, the SAR technique estimates targets based on species richness. However, this technique generally ignores uncertainty in the estimated species richness per area and also disregards the ecological processes and patterns (Karenyi *et al*., 2016). It also fails to consider the patchiness of species within habitats as it assumes that habitat types are connected and homogenous (Keith *et al*., 2013). South Africa also uses absolute threshold values in conjunction with the biodiversity targets to assess the risk of collapse. The process or technical method employed to set adopted threshold values is poorly documented making it difficult to replicate or even interrogate.

Internationally, the IUCN is criticized for adopting thresholds that lack ecological relevance (Boitani *et al*., 2015). It offers limited guidance on the processes undertaken to set these RLE absolute threshold values (Bland *et al*., 2017), and thus failing to uphold one of its requirements that seek to promote transparency in the assessments. In response, Keith *et al*., (2015b) argue that the adopted fixed absolute values are decision thresholds which "specify the conditions required to invoke a decision". Even with this argument, these thresholds remain to be poorly motivated making it difficult to interrogate their basis.

1.6.7. Risk categories

The SA ETS and IUCN RLE risk categories are both analogous to the IUCN RLS classification system and were redefined to promote their practicality in the context of ecosystem assessments (Mace & Lande, 1991; RSA, 2011; Nicholson *et al*., 2014). For this study, the primary focus is only on the four ordinal risk categories which are Critically Endangered (CR), Endangered (EN), Vulnerable (VU), and Least Concern (LC). These categories carry considerable weight in South Africa's biodiversity policies and conservation planning spheres (e.g. EIA and Protected Areas Expansion strategies). The first three ordinal categories (CR, EN, and VU) are collectively referred to as 'threat categories' in the South African context (RSA, 2011) while the IUCN refers to all categories in the RLE framework as 'risk categories'. These terminologies are often used interchangeably and serve to rank the risk of ecosystem collapse (Table 1; Bland *et al*., 2017). However, to standardize the terminology in this thesis, these categories will be referred to as 'risk categories'. Depending on data availability and credibility, an ecosystem is assessed against a series of criteria and assigned a threat status if it triggers any of the absolute threshold values listed under each of the three ordinal risk categories, otherwise, it is assigned an LC category. The LC category highlights ecosystems that are either intact or have experienced declines that are too little to trigger the threshold of any of the other risk categories (Mace & Lande, 1991; Keith *et al*., 2013).

1.7. Discussion

The IUCN RLE framework has become the norm internationally and South Africa acknowledges the importance of aligning its ecosystem threat status assessments with these standards. South Africa became one of the latest countries to formally adopt the IUCN RLE standards (presented in the NBA 2018) and its spatial criteria were systematically applied across all 458 terrestrial ecosystem types (Skowno *et al.*, 2019b). As in Australia and other parts of the world, the strongest motivations for the alignment are to (i) reduce confusion and inconsistencies during reporting to national and international structures, (ii) increase the legitimacy of the ecosystem threat assessments by basing them on a body of sound scientific literature, and (iii) for the threatened endemic ecosystems to be recorded under the IUCN RLE registry to award them international recognition (Skowno *et al*., 2019a).

Based on this review, there are overarching conceptual similarities between SA ETS and IUCN RLE frameworks as they both share common ancestry (IUCN RLS). Firstly, the definition of an ecosystem in both frameworks includes common elements such as organisms and their interactions with their physical environment. These elements capture the dynamic nature of ecosystems whilst remaining applicable across different realms. Secondly, the IUCN RLE and SA ETS frameworks both consist of spatial and functional criteria to assess ecosystem declines. These criteria represent different symptoms that can threaten ecosystems which may result in their irreversible loss. Thirdly, both frameworks use ordinal risk categories to rank the risk of ecosystem collapse (Driver *et al*., 2012; Bland *et al*., 2017). The adoption of the IUCN Red List of Threatened Species risk categories in national ecosystem frameworks (e.g. in countries like Australia, New South Wales, and South Africa) suggests that they are not only relevant and operational in the context of threatened species assessments but also in the assessments of broad-scale ecological features such as ecosystems (Bonifacio & Pisanu, 2013; New South Wales Government 1995, 2010; RSA, 2011). In addition, their broad acceptance suggests that they can easily be understood by policymakers and the general public (Bland *et al*., 2017).

Similarly, there are obvious differences between the national and international frameworks which include the introduction of concepts such as: (i) uncertainty which is important in increasing credibility and confidence of the assessment outcomes and also improving the quality of rendered decisions, (ii) ecosystem collapse which measures how close an ecosystem is to be wiped-out or disappearing (RSA, 2011; Bland *et al.*, 2017). Other differences include the nested IUCN RLE sub-criteria that assess ecosystem change over time (past, present, and future), the fixed absolute thresholds values (RSA, 2011; Bland *et al*., 2017). Whilst these are all important differences highlighting areas that require in-depth attention to harmonise the two frameworks, they matter more in the South African context if the overall assessment outcomes differ substantially as they could disrupt the implementation of the EIA guidelines, as well as in conservation and land-use planning initiatives. As such, the focus of the next chapter of this thesis is to examine how the SA ETS (criterion A1: irreversible loss of natural habitat & criterion C: limited extent and imminent threat) and IUCN RLE (criterion A: reduction in geographic distribution & criterion B: restricted geographic distribution) spatial criteria are quantitatively comparable when applied systematically across 458 South Africa's terrestrial ecosystem types and interrogate the basis of their differences if there any.

2 A critical appraisal of the IUCN RLE approach for South Africa

2.1 Abstract

South Africa is globally recognised for its well-established biodiversity conservation practices, which are guided by national policies and legislations. One important piece of legislation that has a bearing on achieving many of the national and international targets is the gazetted national list of threatened terrestrial ecosystems. It is ratified to inform the development of many of the legally binding and non-binding tools used to mainstream biodiversity considerations into land-use planning, conservation, and other policy spheres. With the publication of the International Union for Conservation of Nature (IUCN) Red List of Ecosystems (RLE) framework, the country needs to weigh up the pros and cons of abandoning the South Africa Ecosystem Threat Status (SA ETS) framework and conforming to the IUCN RLE standards. Both the IUCN RLE and SA ETS frameworks share a common ancestry, but there are differences in some of the key concepts and thresholds which may result in overestimation or underestimation of the risk of collapse. Here I investigate and quantitatively interrogate the basis of these differences. The outcomes will put into perspective the implications of this alignment. Taken together, the results reveal that the proportions of matching assessment outcomes are high (428 of 458) when the threat statuses are split (Critically Endangered (CR) and Endangered (EN) versus Vulnerable (VU) and Least Concern (LC)) in accordance with the policy uptake but relatively low per individual risk category (CR: 14, EN: 14, VU: 14, LC: 339). The IUCN RLE system down-listed 16 ecosystems to the VU category that the SA ETS system classified as EN. Such shifts raise concerns as they will disrupt the implementation of EIA guidelines and the establishment of land-based conservation programmes. Failure for such ecosystems to be considered for conservation response will accelerate their rate of collapse as the anthropogenic land cover change continues to intensify. To mitigate some of the challenges presented by the alignment with the IUCN RLE, the South African National Biodiversity Institute in collaboration with various key partners are exploring the possibility of extending the concept of "species of special concern" to ecosystems. Its practical applicability to ecosystems may result in an additional criterion being introduced for use in South Africa, outside of the IUCN RLE standards. The objective of such a criterion in the context of ecosystems will be to list ecosystem types that fail to trigger the IUCN RLE criteria, however, there is credible evidence to suggest that they need a particular conservation and management response.

2.2 Introduction

South Africa developed national legislation (National Environmental Management Biodiversity Act, hereafter, Biodiversity Act) that is focused on biodiversity conservation at the ecosystem level (Department of Environmental Affairs, 1997). To implement this legislation and deliver on both national and international biodiversity commitments, the country developed a headline indicator 'Ecosystem Threat Status' to systematically assess and monitor threats to ecosystems using robust criteria. The South Africa Ecosystem Threat Status (SA ETS) indicator was established in the absence of globally accepted standards (Botts *et al*., 2020). In 2014, the International Union for Conservation of Nature (IUCN) adopted the range of criteria that classify threats to ecosystems and categories that rank the risk collapse as global standards for the Red List of Ecosystems (RLE). Subsequently, the IUCN published RLE guidelines in 2016 and 2017 and has since become the norm internationally (Bland *et al*., 2016; Bland *et al*., 2017). Therefore, the country needs to weigh up the pros and cons of adjusting its national headline indicator (SA ETS) to conform to the IUCN RLE standards. Understanding the associated policy and conservation implications of this shift is an important step considering the extensive use of the gazetted national list of threatened ecosystems for reporting purposes and also the development and implementation of many of the legally- and non-binding conservation tools (Figure 1). The outcomes are expected to strengthen the policy uptake of the IUCN RLE standards in South Africa (Botts *et al*., 2020).

The development of the IUCN RLE standards is an indication that the world is leaning towards ecosystem-focused conservation approaches as an effective mechanism for conserving a richer array of ecological features (Bland *et al*., 2019). Having adopted this approach more than a decade ago (Driver *et al*., 2005), South Africa strengthened its environmental policies and legislative frameworks by publishing the list of threatened ecosystems under the government gazette. This allows for the gazetted list to be entrenched into various legal tools to reduce the rate of biodiversity loss (Figure 1). In addition, the country aligned its conservation strategies to several global policies (Figure 1) including the Convention on Biological Diversity (CBD) and the Sustainable Development Goals (SDG; Skowno *et al*., 2019a). At least four of the CBD Aichi Targets (5, 10, 14, and 15) and one of the Sustainable Development Goals (Goal 15) contribute notably to biodiversity conservation at the ecosystem level (Osborn *et al*., 2015). The alignment resulted in synergies emerging between different public and private sectors, jointly and notably contributing towards fulfilling the national and international agenda and goals (Pierce *et al*., 2005; Government of South Africa, 2015).

The Biodiversity Act is the principal legislation that champions the management and conservation of the country's biodiversity and has introduced a suite of legal tools to avert further biodiversity loss outside protected areas (RSA, 2011). Critically Endangered (CR) and Endangered (EN) ecosystems are generally small and fragmented, as such, fail to feature in the expansion strategy of large protected areas (Government of South Africa, 2010). However, they are protected through other land-based mechanisms of protected area networks such as stewardship programmes (Figure 1).

Other legal tools include the Environment Impact Assessment (EIA) regulations which are one of the most powerful regulations used to inform land-use authorisation processes. Their relevance to the listed ecosystems is captured under Activity 12 in the Listing Notice 3 of the National Environmental Management Act (NEMA) EIA regulations. These regulations state that any spatial activity that involves clearance of at least 300 m^2 of the indigenous natural vegetation within CR or EN ecosystems triggers the need to conduct a basic EIA assessment. In special cases where an authorisation to develop is granted, additional stringent and costly conditions are imposed to mitigate the negative impacts of the development. These might include the implementation of biodiversity offsets which are the last resort in the mitigation hierarchy. Biodiversity offsets are intended to compensate for biodiversity loss by restoring degraded ecosystems after all measures to avoid, reduce, or remedy loss have been exhausted. As such, they have the potential to achieve win-win outcomes by enabling developers to "secure and maintain the license to operate" whilst achieving conservation objectives (Brownlie & Botha, 2009). However, the effectiveness of biodiversity offsets as a conservation tool has been put under scrutiny due to its concept of compensating ecological losses by accepting gains with high uncertainty elsewhere. To make matter worse, there is no regulatory system to ensure there is compliance with the mitigation hierarchy requirements (Bull *et al*., 2013). When Vulnerable (VU) and Least Concern (LC) ecosystems are involved, less stringent EIA rules apply as their risk of collapse is relatively low (RSA, 2011).

Whilst the EIA regulations deal with authorisations of licenses to operate and the associated environmental conditions, land-use planning tools such as bioregional plans identify areas sensitive to development on land and seascapes (Figure 1). To ensure the broad uptake of these legal tools, bioregional plans published in terms of Section 40 of the Biodiversity Act are linked to multi-sectorial planning tools such as Spatial Development Frameworks (Figure 1; DEA, 2011). These planning legal tools are adopted to accelerate mainstreaming efforts of biodiversity considerations at a municipal level (Alexander, 2012).

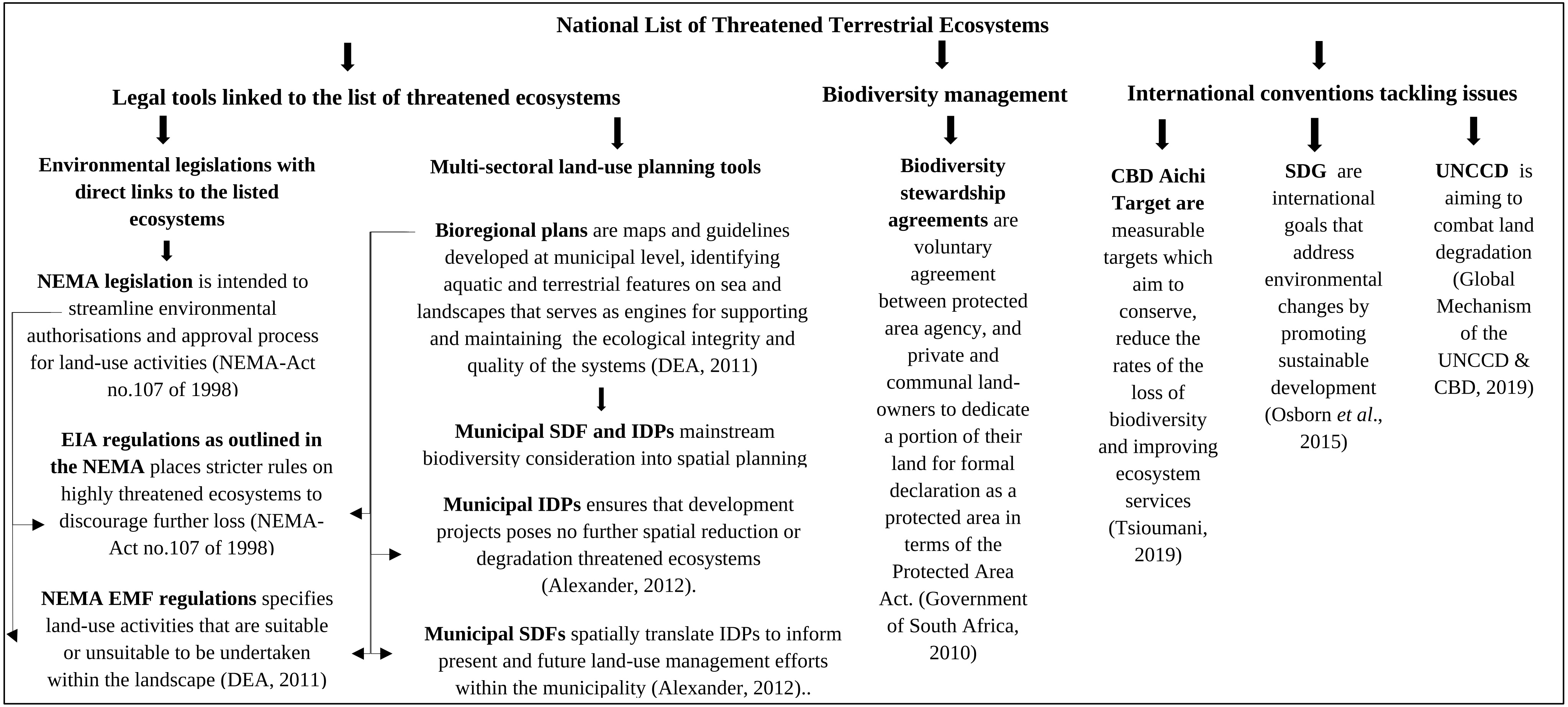

Figure 1: Three pieces of policy-related tools that have direct ties with the list of threatened ecosystems include the National Environmental Management Act (NEMA) legislation, Environmental Impact Assessment (EIA), and Environmental Management Framework (EMF) regulations. Bioregional plans which inform national regulations are also directly linked to several spatial planning tools that are used to mainstream biodiversity considerations into land-use plans at the municipal level. These include Spatial Development Frameworks (SDFs) and Integrated Development Plans (IDPs). Land-based mechanisms that offer protection to threatened ecosystems include Stewardship programmes. South Africa is a signatory to several international treaties including the Convention on Biological Diversity (CBD), Sustainable Development Goals (SDG), and United Nations Convention to Combat Desertification (UNCCD) which all consist of a list of biodiversity targets or goals that are developed to monitor progress towards addressing biodiversity concerns at an ecosystem level.

The direct links the gazetted national list of threatened ecosystems has with national legislations and spatial planning tools puts into perspective the influential role it plays in South Africa (Figure 1). As such, it will not be surprising if the transition from national to international standards of assessing threats to ecosystems is met with resistance due to the potential conservation implications that may arise. To put this into context, the IUCN RLE standards have the potential to either validate or undermine the work accomplished thus far in protecting and averting further biodiversity loss in natural environments. This is likely to be the case in scenarios where ecosystem assessment outcomes differ substantially in the context of policy and land-use planning. For example, ecosystems gazetted under the Biodiversity Act as either CR or EN might become down-listed (i.e. assigned to either VU or LC category) under the IUCN RLE system. Such shifts are concerning as they may result in previously restricted development activities within economic productive landscapes being allowed. Ecosystems that will be most impacted by this change are those that have not yet been protected through land-based mechanisms of large protected area networks (McIntosh *et al*., 2018).

As South Africa joins the long list of more than 100 countries including Colombia and Myanmar (Bonifacio & Pisanu, 2013; Etter *et al*., 2015; Bland *et al*., 2019; Murray *et al*., 2020) that are complying with the IUCN RLE standards and the benefits of this shift have also become evident. That is, South Africa will obtain a holistic global picture of how well it is performing in averting biodiversity loss at an ecosystem level by comparing its ecosystem assessment outcomes with other countries across different regions (Bland *et al*., 2017). In addition, the IUCN Red List makes it easier for countries to secure funding from international donors to strengthen their biodiversity conservation efforts, and address knowledge and data gaps through focused research (Betts *et al*., 2020). However, the robustness and adequacy of the IUCN RLE as a conservation tool has not yet been tested at a national scale. This is the case in many regions, particularly in countries like South Africa that have shifted from national to international standards. Here I compare and contrast the assessment outcomes generated based on the IUCN RLE and SA ETS criteria that are related only to the spatial distribution of ecosystems (i.e. reduction in geographic distribution and limited extent) to better understand the degree to which the assessment outcomes are comparable, investigate potential differences, and interrogate their basis.

2.3 Methods

The IUCN RLE and SA ETS assessments were conducted systematically across 458 terrestrial ecosystems using the national land cover-based habitat modification dataset for 1990 and 2014 and the national ecosystem (vegetation) map (Table 2). Since the last ETS assessments reported in the National Biodiversity Assessment (NBA) 2011 (Driver *et al*., 2012), the national vegetation map has undergone spatial refinements which were primarily influenced by the availability of credible fine-scale data and improved satellite imagery (Dayaram *et al*., 2019). Some of these changes include: (i) alteration of the ecosystem boundaries, (ii) merging of existing ecosystems types and introduction of new ones, (iii) the alignment of the estuarine and coastal ecosystem maps with the vegetation map, and (iv) the removal of the wetland types from the vegetation map as they are comprehensively dealt with in the national wetlands map (Dayaram *et al*., 2019). Although these improvements were only done on a small part of the ecosystem map, they resulted in an area increase for some types while others experienced a decrease (Dayaram *et al*., 2019).

Another milestone achieved is the availability of national land cover maps for two time-points (1990-2014) which led to the development of the land cover-based habitat modification (hereafter, habitat modification) layer (Skowno, 2018). The habitat modification layer consists of three classes: natural extent lost pre-1990, natural extent lost post-1990, and remaining natural extent (Table 2). These classes align with IUCN RLE guidelines for assessing historic and projecting future declines catalyzed by anthropogenic land-use activities. The historic declines are estimated based on the 1750 reference date which represents a period prior to the widespread of anthropogenic land conversion. Ecosystem declines were also projected 26-years into the future based on the absolute rate of declines between 1990 and 2014 to cover the 50-year window (i.e. from 1990 to 2040) as recommended by the IUCN RLE (Equation 1). This timeframe is long to diagnose changes in ecosystem geographic distribution with some reasonable level of certainty (Bland *et al*., 2017).

Credibility and the quality of the resulting risk categorisations rely heavily on the input data. For this reason, the best and readily available land cover datasets from a few provinces and metropolitan municipalities were also used to improve the reliability of the assessment outcomes. These additional land cover (hereafter, supplementary) maps were delineated at the fine-scale resolution with some being more recent than others (Table 2). Since the supplementary datasets are not multi-temporal, they were not used to project future declines but rather to assess the historical change of ecosystem distribution.

Table 2: Input layers used for the assessment of terrestrial ecosystem threats.

Input datasets	Description	Sources
National vegetation map presented in vector format	The map includes 458 vegetation types classified based on the broad floristic distinctness of the groups across South Africa, Lesotho, and Swaziland.	Mucina & Rutherford, 2006, curated by the South African National Biodiversity Institute (SANBI)
Two time points national land cover map presented in a raster format	The land cover-based habitat modification map (1990-2014) provides full coverage of South Africa, Lesotho and Swaziland were resampled at 30m resolution. The two-time points dataset consists of three land change classes: lost pre-1990 (ecosystem natural extent lost before 1990), lost post-1990 (ecosystem natural extent lost after 1990) and natural (ecosystem extent remaining in a natural condition). The natural class is a combination of intact areas and those that are in a near-natural (degraded) condition. The near-natural areas are areas that were either ploughed, cultivated, or cleared from the early 1960s but recovered over time.	2013/14 GEOTERRAIMAGE (GeoTerraImage, 2015) modified by SANBI to produce a change NLC with two-time points (1990 and 2014) (Skowno, 2018)
Cities (City of Cape Town, Nelson Mandela Bay metro, and all municipalities in Gauteng province)	Land cover product from the City of Cape Town (polygon feature geodatabase), Nelson Mandela Bay Metro (polygon shapefile), and Gauteng province (2.5m resolution) were combined and classified into two land cover classes: natural and modified	Sourced from Cape Nature provincial conservation agencies in the Western Cape and Gauteng province. Modifications on this dataset were led by SANBI's Biodiversity Research and Monitoring division (see, Skowno, 2018 for detailed description)
KwaZulu-Natal (KZN)	2011 provincial raster land cover product (20m resolution) with two land cover classes: natural and modified	Sourced from Ezemvelo KZN Wildlife provincial conservation agency (Jewitt, 2012). Modifications on this dataset were led by SANBI's Biodiversity Research and Monitoring division (see, Skowno *et al*., 2019b for detailed description)
Mpumalanga (MP)	2017 provincial raster land cover product with two classes: natural and modified (10m resolution)	Sourced from Mpumalanga provincial conservation agencies in Mpumalanga province. Modifications on this dataset were led by SANBI's Biodiversity Research and Monitoring division (see, Skowno *et al*., 2019b for detailed description)

The habitat modification and the supplementary datasets were combined in geographic information system software ArcGIS 10.4 (with the Spatial Analyst extension; ESRI, Redlands, USA) to create a composite land cover map. Environmental settings in Geoprocessing were fixed to regulate and standardise all outputs by (i) re-projecting spatial datasets to Africa Albers Equal Area conic projection; the coordinate system best suited for Southern African landmasses (Silberbauer, 1997). It uses two standard parallels (set to -24 and -33 respectively) to reduce some of the distortions (Gericke & du Plessis, 2012), (ii) resample pixel resolution to 30m^2, and (iii) rasterize the vegetation map to match the grid size of the habitat modification layer.

The overall classes of the composite land cover map were determined using a hierarchical decision rule. The supplementary land cover classes were given precedence only in their respective regions. For example, if Mpumalanga land cover classifies pixels under this province as “not natural” and the habitat modification dataset assigns the same pixels to the “natural” class, then the Mpumalanga class is awarded precedence. This is because supplementary land cover classes are assumed to be more accurate in their respective regions as some are mapped at a finer-scale than the habitat modification dataset (Table 2). This rule was applied in cases where habitat modification and supplementary land-clover pixel classification disagreed. Subsequently, the proportion of each ecosystem type remaining in a natural and not natural condition (i.e. extent lost pre- and post-1990) was quantified using *Tabulate Area*, within the *Spatial Analyst* toolbox in ArcGIS.

2.3.1 Execution of the IUCN RLE and SA ETS rules for spatial criteria

Outputs from the GIS analysis were imported into R statistical software version 3.6.3 (R Development Core Team, 2020) and merged to create a single data frame. This data frame consists of the following fields: ecosystem name, conservation targets, natural extent 1750 (km^2), natural extent lost pre-1990 (km^2), and natural extent lost post-1990 (km^2). The area fields were used to estimate the historic declines (i.e. from 1750 to 2014) and also project declines 26 years into the future (Equation 3) to cover the recommended 50-year window (i.e. starting from 1990 to 2040). This was followed by the application of the IUCN RLE and SA ETS rules to assess the risk of collapse. Since the SA ETS assessments report on the risk of collapse based on how much of the natural extent is remaining (RSA, 2011), while the IUCN RLE approach is focused on how much of the natural extent is lost (Bland *et al*., 2017), for comparability sake and eliminate confusion, the assessments were conducted based on how much is lost.

2.3.1.1 Reduction in geographic distribution (SA ETS - criterion A1 and IUCN RLE criteria A)

South Africa's approach of assessing threats to ecosystems under SA criterion A1 (irreversible loss of natural habitat) includes the use of biodiversity targets and thresholds which measure how close an ecosystem is to collapsing (Table 3). Biodiversity targets range between 16% and 36% (Driver *et al*., 2012), and are intended to preserve a minimum of 75% of the remaining plant and genetic diversity supported by each of the 458 ecosystem types (Rouget *et al*., 2004). Ecosystem types that trigger any of the absolute threshold values and/or biodiversity targets are declared threatened and assigned to any of the three ordinal risk categories (CR, EN, or VU), otherwise, they are assigned to the LC category (Table 3).

Unlike SA ETS criterion A1, which only focuses on historic declines, the IUCN RLE criterion A includes sub-criteria that back-cast and project the risk of ecosystem collapse over 50 years (sub-criterion A1: past, sub-criterion A2a: future and sub-criterion A2b: any 50-year period including the past, present, and future). Back-casting in red-listing assessments is referred to as "retrospectively adjusting earlier Red List categorisations using current information" (Butchart *et al*., 2007) while the projection refers to the extrapolation of ecosystem declines into the future based on the credible and readily available information (IUCN Standards and Petitions Committee, 2019). This is in addition to sub-criterion A3 that identifies ecosystems that have experienced historic spatial declines from 1750 to 2014 (Bland *et al*., 2017). Although SA ETS criterion A1 and IUCN RLE criterion A both assesses change in ecosystem geographic distribution, SA ETS criterion A1 is more analogous to IUCN RLE sub-criterion A3 as they developed to measure the historic shrinkage in the area influenced by anthropogenic land-use activities (RSA, 2011; Bland *et al*., 2017).

The availability of the habitat modification layer generated using two time-points land cover maps allowed only historic and future (sub-criteria A2b and A3, respectively) spatial declines to be estimated systematically with some degree of certainty. As more land cover datasets are acquired in the future, it will become possible to conduct assessments against RLE sub-criteria A2a and A1. To estimate ecosystem decline over any 50-years, absolute rates of decline were first calculated between 1990 and 2014 (Equation 1). These estimates were projected into the future (2040) and then converted into percentages (Equation 3). Ecosystem threat statuses were determined by assessing these percentages (2040) against A2b thresholds (Table 3) and classifying only those that triggered any of the thresholds as threatened.

$$Absolute\ rate\ of\ decline^{1} = \frac{NaturalExtent_{t2} - NaturalExtent_{t1}}{Year_{t2} - Year_{t1}} \quad \text{(Equation 1)}$$

where

t_1=1990 and t_2= 2014

$$Extent\ lost_{2040} = NaturalExtent_{t1} - (Absolute\ rate\ of\ decline * 50) \quad \text{(Equation 2)}$$

$$Ecosystem\ lost\ by\ 2040 = \frac{NaturalExtent_{t1} - ExtentLost_{2040}}{NaturalExtent_{t1}} * 100 \quad \text{(Equation 3)}$$

Table 3: Description of SA ETS criterion A1 and the IUCN RLE criterion A and the associated thresholds used to identify ecosystems experiencing aggressive spatial declines between 1750 and 2014.

Criteria	Sub-criterion	Critically Endangered (CR)	Endangered (EN)	Vulnerable (VU)
South African criterion A1: Irreversible loss of natural habitat RSA, 2011)	Historic declines	Percentage of natural habitat loss ≥ (100% - biodiversity target)	Percentage of natural habitat loss ≥ (100% - (biodiversity target - 15%))	Percentage of natural habitat loss ≥ 40%
IUCN RLE criterion A: Reduction in geographic distribution (Bland *et al*., 2017)	A2b: any 50-year period	≥ 80% of Natural habitat loss	≥ 50% of Natural habitat loss	≥ 30% of Natural habitat loss
	A3: historic declines	≥ 90% of Natural habitat loss	≥ 70% Natural habitat loss	≥ 50% of Natural habitat loss

2.3.1.2 Restricted geographic distribution (SA ETS - criterion C and IUCN RLE – sub-criterion B1 and B2)

The SA ETS criterion C and IUCN RLE criterion B are designed to identify ecosystems that have small geographic distribution and are susceptible to functionally and/or spatially explicit threats (RSA, 2011; Bland *et al*., 2017). As outlined in the Biodiversity Act, ecosystems with restricted geographic distribution assessed under SA ETS criterion C cannot be listed under the CR category because they are considered to have not "necessarily already undergone severe degradation of ecological structure, function or composition" catalysed by anthropogenic pressures. As such, the threshold for this category has remained undeveloped. Although the primary objective of these two criteria (SA ETS criterion C and IUCN RLE criterion B) is similar, their technical methods of identifying ecosystems with restricted geographic

[1] Absolute rate of decline (ARD) is a term introduced by the IUCN in the RLE assessments. It represents a fixed or constant fraction of the ecosystem's natural extent (km^2) lost each year producing a linear pattern (Bland *et al*., 2017).

distribution differ. The SA ETS criterion C identifies ecosystems with a limited extent based only on the original natural area remaining (RSA, 2011). Meanwhile, IUCN RLE criterion B uses two spatial metrics: sub-criterion B1 (Extent of Occurrence-EOO) and sub-criterion B2 (Area of Occupancy-AOO). These spatial metrics are conceptually and technically mirrored from the IUCN Red List of Threatened Species (RLS, Mace & Lande, 1991).

In both the IUCN RLE and RLS assessments, the EOO and AOO represent the geographic extent or range (in the context of species) and remain the key component of criterion B in both frameworks (Gaston & Fuller, 2009; Lee *et al*., 2019). Typically, EOO estimates are always greater than AOO (Gaston & Fuller, 2009). The IUCN RLE sub-criterion B1 measures the EOO by estimating the area of the minimum convex polygon. As such, its estimates are not a physical representation of the ecosystem's extent but rather encompass all known spatial occurrences of an ecosystem type. Sub-criterion B2 measures the AOO using a grid approach to count the total number of 10×10 km grid cells containing the natural patches of an ecosystem type (Gaston & Fuller, 2009; Bland *et al*., 2017). Following the IUCN RLE guidelines, grid cells that contain less than 1% of ecosystem patches were excluded as they are considered to "contribute to negligible risk-spreading effect" as opposed to that of larger patches (Bland *et al*., 2017). These make the IUCN RLE technical methods of identifying narrowly distributed ecosystems more complex compared to that adopted by the SA ETS system.

A 120 m^2 resolution ecosystem remnants map (circa 2014) was created in ArcGIS 10.4 using the *con tool* within the *Spatial Analyst* toolbox. A *con tool* applies a conditional logic on the composite land cover map (i.e. created by combining habitat modification and supplementary land cover maps) to extract ecosystem patches remaining in a natural condition. The outputs from these GIS analyses (i.e. composite land cover and remnants maps) were both imported into R version 3.6.3 to apply SA ETS and the IUCN RLE rules. Given the complexity of criterion B, Lee *et al*., (2019) developed a software package in R [redlistr] and an R script to aid its systematic and repeatable application. These IUCN tools are freely available on the Comprehensive R Archive Network (CRAN, https://cran.r-project.org/package=redlistr). They are developed to systematically estimate the EOO and AOO across terrestrial, inland aquatic, estuarine, coastal, and marine environments. These estimates were then assessed against sub-criteria B1 and B2 size thresholds to assess the risk of collapse (Table 4).

Ecosystems failing to meet any of these size thresholds were assigned to the LC category, otherwise, they were considered to be limited in area or extent and assigned to any of the three ordinal risk categories (CR, EN, or VU). The listing of ecosystems in these ordinal risk categories is depended on whether they are under threat. Both South Africa and the IUCN do not give recommendations on the acceptable quantitative threshold for imminent threat and ongoing declines, respectively. The current notion is that they can be set to suit the intentions of the decision-makers and the goals of the assessment. Aligning with the threshold adopted for the national assessments (represented in the National Biodiversity Assessment 2018, technical report for the terrestrial realm), an annual rate of decline threshold of 0.4% was adopted to identify ecosystems that are under imminent threat (SA ETS) and those that are experiencing continuing declines due to spatially explicit threats (IUCN RLE). In addition, the *ggplot2* package (Wickham, *et al.*, 2020a) in R version 3.6.3 was used to plot the annual rates of declines against the ecosystem extent (IUCN RLE sub-criterion B1 and B2) to illustrate the implications of setting the thresholds for ongoing decline high or low (Figure 3).

$$Annual\ rate\ of\ decline^{2} = \frac{\left(\frac{Area_{t2}}{(Area_{t2}+AreaNatural_{t2})}\right)}{n} * 100 \qquad \text{(Equation 4)}$$

where

$Area_{t_2}$ = natural extent lost post-1990

$Area\ Natural_{t_2}$ = area remaining in a natural state post-1990

n = number of years between 1990 and 2014

Table 4: Description of SA ETS criterion C and the IUCN RLE criterion B and the associated thresholds used to identify ecosystems with a limited extent.

Criterion	Sub-criterion	Critically Endangered (CR)	Endangered (EN)	Vulnerable (VU)
Criterion C: Limited extent and imminent threat (RSA, 2011)			Ecosystem extent ≤ 30 km²	Ecosystem extent ≤ 60 km²
Criterion B: Restricted geographic distribution (Bland *et al.*, 2017)	**B2** - Extent of a minimum convex polygon (km²) enclosing all occurrences (EOO) AND an observed or inferred continuing decline in spatial extent.	≤ 2 000 km²	≤ 20 000 km²	≤ 50 000 km²
	B2 - The number of 10×10 km grid cells occupied (AOO) AND an observed or inferred continuing decline in spatial extent.	Grid count ≤ 2	Grid count ≤ 20	Grid count ≤ 50

[2] Annual rate of decline represents an annual percentage of natural habitat lost at the beginning of the decline for the past 24 years (i.e. 1990-2014).

2.3.2 Confusion Matrix

The confusion matrix is a widely used method to assesses the performance of a classifier using binary and multi-class data. Typically, the rows of a confusion matrix represent the actual (true) class and the column represents the predicted class (Pontius & Santacruz, 2014). In this case, the matrix serves to compare the two risk assessment approaches. Therefore, instead of assessing the classification accuracy, the goal is to assess the agreement of the assessment outcomes generated by IUCN RLE and SA ETS systems (Pontius & Santacruz, 2014). From these tabulated classes, matching ecosystem assessment outcomes between the IUCN RLE and SA ETS systems can be inferred as the sum of diagonal in the matrix, while the estimates below and above the diagonal line represent the mismatches (Table 5, Santra & Christy, 2012).

Table 5*: An illustrative example of a 2x2 confusion matrix*

Classification method 1	Classification method 2	
	Class 1	Class 2
Class 1	Agreement on class 1	Disagreement
Class2	Disagreement	Agreement on class 2

2.3.2.1 Measures of classification agreement

There are several classification performance measures derived from the confusion matrix that estimate the degree to which the classifier is efficient. These include accuracy, sensitivity, and specificity (Sokolova & Lapalme, 2009; Tharwat, 2018). Accuracy represents a fraction of the classes that are in agreement and is sensitive to imbalances (Tharwat, 2018; Delgado & Tibau, 2019). The class imbalance is a common challenge that occurs when the proportions of observations are not equal across all classes, resulting in a skewed distribution. As such, the classifier will likely be biased towards the classes with more observations (Brodersen *et al*., 2010). Several measures have been introduced to tackle the effect of imbalance in the data. These include the use of balance accuracy measure which is an average of the classes that are in agreement between the IUCN RLE and SA ETS systems (Luque *et al*., 2019). Sensitivity and specificity are independent measures of accuracy. That is, sensitivity presents a fraction of observations correctly identified for the first class, while specificity represents the fraction of the observations in the second class (i.e. the sum of the other three categories) correctly identified, etc (McPherson *et al*., 2004; Allouche *et al*., 2006). For example, the sensitivity estimate of the first class (CR) is calculated by dividing the total number of ecosystems which are consistently listed as CR between the two systems by the overall number of ecosystems (i.e. both consistently and inconsistently listed) assigned to this class. Similarly, specificity for the CR class is calculated by dividing the number of ecosystems with mismatching classification by the overall number of inconsistently listed types.

To determine the proportion of ecosystems listed consistently and inconsistently between the SA ETS and IUCN RLE systems, 4x4 and 2x2 matrices were computed using *confusionMatrix* function within the *caret* package (Kuhn, 2008). The 4x4 matrix represents each of the four risk categories (CR, EN, VU, and LC) while the 2x2 matrix represents the split between the top two (CR and EN) versus the bottom two (VU and LC) categories in accordance with the policy uptake (i.e. NEMA EIA regulations) in South Africa. The corresponding classifier's performance measures for both the 4x4 and 2x2 matrices were also generated using this function.

2.4 Results

2.4.1 Overall IUCN RLE and SA ETS assessment outcomes

Both the IUCN RLE and SA ETS systems estimated a total number of 27 ecosystems to be on a brink of collapse (CR) which make up 0.4% and 0.6% respectively of the remaining natural area. The difference in the remaining natural area suggests that some of these ecosystems are unique to each system. The total number of ecosystems listed by the two systems in the EN category (IUCN RLE: 36 and SA ETS: 38) and their estimated contribution to the natural habitat remaining (3%) are not too different (Table 6). Despite the uniqueness of some of the listed ecosystems, both systems are in agreement that at least 63 ecosystems are at immediate risk of collapse (CR or EN). These are ecosystems that will likely disappear in the absence of immediate preventative measures that prohibit or discourage land-intensive activities to take place. As such, they need to be conserved through the listing process to reduce their risk of collapse (Skowno *et al*., 2019a).

A total number of 31 ecosystems listed under the VU category following the IUCN RLE standards contribute only 4% to the remaining natural habitat, while 51 ecosystems classified as VU following the SA ETS standards make up 8% of the habitat remaining in a pristine state (Table 6). Ecosystems in a vulnerable state are considered to have not yet experienced aggressive declines. However, their declines signal a warning that they are at a tipping point from being at immediate risk of collapse in the absence of conservation initiatives that would arrest further spatial declines (Driver *et al*., 2012).

Table 6: The summary of the assessment outcomes per risk category of 458 terrestrial ecosystems assessed for the spatial decline and their remaining natural area (km^2) per risk category. The percentages estimates represent the original natural area remaining that covers South Africa's total surface area and also the contribution of ecosystems in each risk category to the area remaining natural.

	IUCN RLE				SA ETS			
Threat Status	Number of threatened ecosystems	Natural extent remaining (km^2)	% of the original natural habitat remaining	% of remaining natural habitat in South Africa	Number of threatened ecosystems	Natural extent remaining (km^2)	% of the original natural habitat remaining	% of remaining natural habitat in South Africa
CR	27	4215.82	0.33	0.42	27	5851.75	0.46	0.59
EN	36	29254.46	2.31	2.93	38	30419.13	2.41	3.05
VU	31	41589.58	3.29	4.17	51	84191.12	6.66	8.45
LC	364	921791.65	72.91	92.47	343	876389.49	69.32	87.92
Total	458	996851.50	78.85	100.00	458	996851.50	78.85	100.00

The IUCN RLE and SA ETS systems suggest that the arid regions of the country (particularly the Northern Cape) are largely in an intact condition. This has been attributed to a low anthropogenic land-use footprint in these regions (Skowno *et al*., 2019b). Both systems also suggest that the threatened ecosystems are widespread across eight of nine provinces (KwaZulu-Natal, Limpopo, Western Cape, Eastern Cape, Free State, Gauteng, North West, and Mpumalanga) in South Africa. The metropolitan municipalities which serve as the economic hubs of South Africa are generally affected by the intensified anthropogenic land-use activities (Greenberg, 2010), resulting in threatened ecosystems concentrating in these parts of the country (Figure 2). The IUCN RLE and SA ETS systems are in agreement that the Western Cape, particularly the lowlands region of the Fynbos biome supports the high concentration of threatened ecosystems.

Both the IUCN RLE and SA ETS systems estimated a total number of 27 ecosystems to be CR (Table 6). However, the difference in their spatial distribution (Figure 2) suggests that many of these ecosystems listed by the IUCN RLE system are limited in extent. This explains the small footprint of the CR ecosystems in some regions such as the southeast of the country or within the KwaZulu-Natal province. Other threats including alien invasion have added pressure on ecosystem types widespread across the east coast region of KwaZulu-Natal impacting savanna and grassland types (Jewitt *et al*., 2015) as well as the lowlands region of the Fynbos biome (Rouget *et al*., 2003). The impacts of degrading processes are not restricted only to these regions but rather a nationwide threat to biodiversity. However, its extent and severity on a national scale are inadequately assessed in the recently published RLE assessments due to data limitations (Skowno *et al*., 2019a).

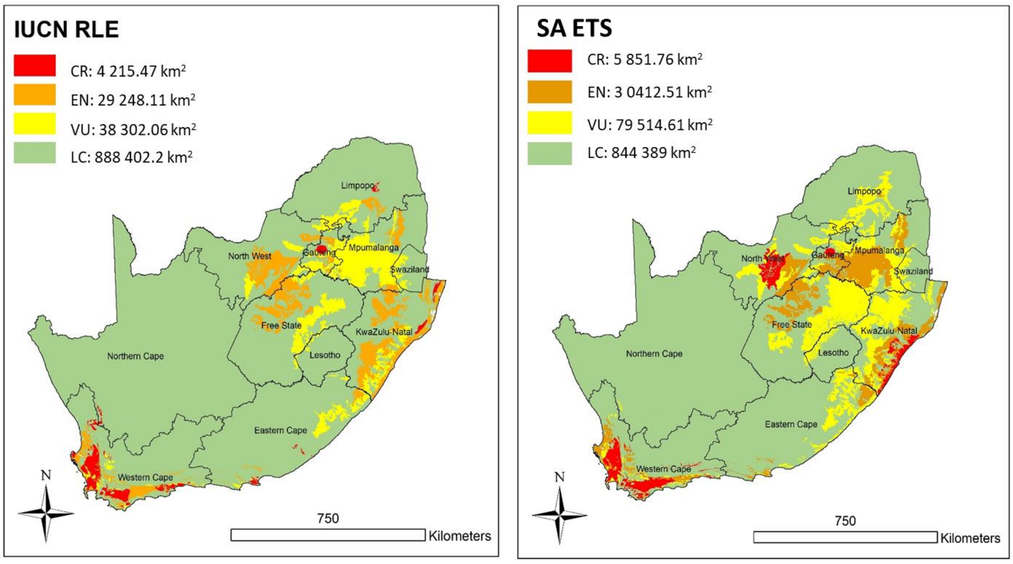

Figure 2: The distribution of threatened terrestrial ecosystems in South Africa and the estimated natural extent remaining as per IUCN RLE (a) and SA ETS assessment standards (b). Figure 2b suggests that the ecosystems listed as threatened (CR, EN, and VU) cover a large extent of South Africa's surface area while Figure 2a suggests that they are less widespread in their geographic distribution.

2.4.2 Classification performance of a classifier

2.4.2.1 Multi-class (4x4) confusion matrix and their performance measures

The classifier of a 4x4 confusion matrix (Table 7) identified a total number of 381 (out of 458) consistently listed ecosystems between the SA ETS and IUCN RLE systems, with a substantial number (339) of these ecosystems assigned to the LC category. From a total number of 27 ecosystems listed as CR by the SA ETS system, 13 are inconsistent listed (i.e. down-listed to EN by the IUCN RLE system) between the two systems. The SA ETS system listed a total number of 38 ecosystems in the EN category, only 14 of these ecosystems were also classified as EN by the IUCN RLE system. Of the others, eight were up-listed to the CR category, and 16 down-listed to the VU category by the IUCN RLE system. From a total of 51 ecosystem types that were assigned to the VU category by the SA ETS system, 37 have mismatching threat status (i.e. two ecosystems up-listed and 25 down-listed by the IUCN RLE system). Two ecosystem types experienced a dramatic shift in threat status. These ecosystems were assigned to the LC category by the SA ETS while the IUCN RLE system classified them as Critically Endangered.

Sensitivity estimates are high for the low-risk class (LC) at 93% and low for the high-risk class (EN) at 39%, while specificity estimates are over 90% for all four classes. The balanced accuracy is high for the LC (95%) class and relatively low for the other threat class, particularly the EN (66%) class (Table 7). These performance measures represent the correct classifications between the two systems and the classifier's accuracy in its estimation. In summary, the inconsistently listed ecosystems, particularly the 16 that are down-listed from EN to VU category by the IUCN RLE system (Table 7) will likely disrupt conservation initiatives. This is because the VU ecosystems fail to trigger conservation response as they still have a reasonable proportion of their natural habitat remaining. As such, the existing land-based conservation plans that may already be featuring these types will likely require some revisions to take into consideration the adjustments presented by the IUCN RLE standards.

Table 7: Multi-class (4 x 4) confusion matrix and the associated classifier performance measures. The estimates on the diagonal line represent the number of ecosystems that are listed consistently by the IUCN RLE and SA ETS systems. The estimates above the diagonal line represent ecosystems that are up-listed by the SA ETS system while the estimates below the diagonal line represent ecosystems that are up-listed by the IUCN RLE system.

		IUCN RLE				
	Risk categories	CR	EN	VU	LC	Total
SA ETS	CR	14	13	0	0	27
	EN	8	14	16	0	38
	VU	3	9	14	25	51
	LC	2	0	1	339	342
	Total	27	36	31	364	458

Performance measures	Statistics by Category			
Statistics by Class	Class: CR	Class: EN	Class: VU	Class: LC
Sensitivity	0.51852	0.38889	0.45161	0.9313
Specificity	0.96984	0.94313	0.91335	0.9681
Balanced Accuracy	0.74418	0.66601	0.68248	0.9497

2.4.2.2 Binary (2x2) confusion matrix and performance measures

The classifier correctly identified a total number of 428 of 458 ecosystems that are listed consistently between the IUCN RLE and SA ETS systems. The majority of these ecosystems (379) have a lower risk of collapse (VU or LC) while the remaining 49 ecosystems are under immediate risk (listed as either CR or EN, Table 7). From a total number of 65 ecosystems listed in the two top risk categories (CR or EN) by the SA ETS system, 16 of these ecosystems are down-listed by the IUCN RLE system. Meanwhile, the SA ETS system down-listed 14 of 63 ecosystem types classified as either CR or EN by the IUCN RLE system (Table 8).

Sensitivity and specificity are estimated to be 78% and 96% respectively, while the balanced accuracy of the classifier is estimated at 87% (Table 8). Although the number of ecosystems that are listed consistently between the IUCN RLE and SA ETS systems is high, there is still a reasonable proportion with mismatching classification. These are ecosystems worth being concerned about as their misclassification suggests that some types will either lose or gain conservation responses. Consistently classified ecosystems, particularly those listed in either CR or EN category pose no policy disruptions in terms of the implementation of the EIA guidelines. That is, the EIA rules and procedures remain applicable irrespective of which assessment method has been applied or adopted.

Table 8: A binary (2 x 2) confusion matrix and the associated classifier performance measures. The estimates on the diagonal line represent the number of ecosystems that are matching, estimates above the diagonal line represent ecosystems that are up-listed by the SA ETS system, and below are those that are up-listed by the IUCN RLE system.

	Risk categories	IUCN RLE		
		CR or EN	VU or LC	Total
SA ETS	CR or EN	49	16	65
	VU or LC	14	379	393
	Total	63	395	458
Classifier performance measures	statistics estimates			
Sensitivity	0.7778			
Specificity	0.9595			
Balanced accuracy	0.8686			

2.4.3 The implications of setting annual rates of decline high or low

The scale at which ecosystems are delineated plays a role in the ecosystem assessments particularly under SA ETS criterion C and the IUCN RLE criterion B. The country's terrestrial ecosystems are delineated at a fine-scale. As such, their majority qualify to be assessed under the IUCN RLE criterion B (sub-criterion B1-Extent of Occurrence and sub-criterion B2-Area of Occupancy) but not all are experiencing ongoing spatial declines. In the absence of a universal quantitative threshold to represent ongoing declines, South Africa adopted a 0.4%/year threshold (Equation 4) which is considered to be a reasonable representation of the annual rate of decline (Skowno *et al.*, 2019b). As such, this threshold was also adopted in this thesis to filter out the narrowly distributed ecosystems that are impacted by progressing declines in recent years (between 1990 and 2014).

Although the 0.4%/year threshold may be considered a national threshold in South Africa, there still lacks ecological relevance to justify its adoption. As such, a set of arbitrary threshold values (0.3%/year, 0.4%/year, and 0.5%/year) representing ongoing declines were tested to better understand how sensitive terrestrial ecosystems are to the risk of collapse under any of these thresholds. Figure 3 suggests that a reasonable number of ecosystem types that have a remaining natural habitat of less or equal to 2 000 km^2 (CR) and those that are less or equal to 20 000 km^2 (EN), in addition to those that occupy more than 1% of the 10×10 km grid cell area (CR: 1 grid cell and EN: ≤ 5 grid cells) have experienced high annual rates of spatial declines in the last 24 years (1990-2014). Figure 3(a) and (b) suggest that the 0.3%/year threshold is more robust while the 0.5%/year threshold (Figure 3(e) and (f)) is less robust in identifying narrowly distributed ecosystems with progressing spatial declines (Figure 3(e) and (f)). More notably, Figures 3(c) and (d) suggest that South Africa adopted a conservative threshold (0.4%/year) to filter these ecosystems. In summary, countries adopting the absolute threshold-based approach to represent ongoing declines must do so with caution due to the influential role it has on the final listing of the ecosystem under criterion B.

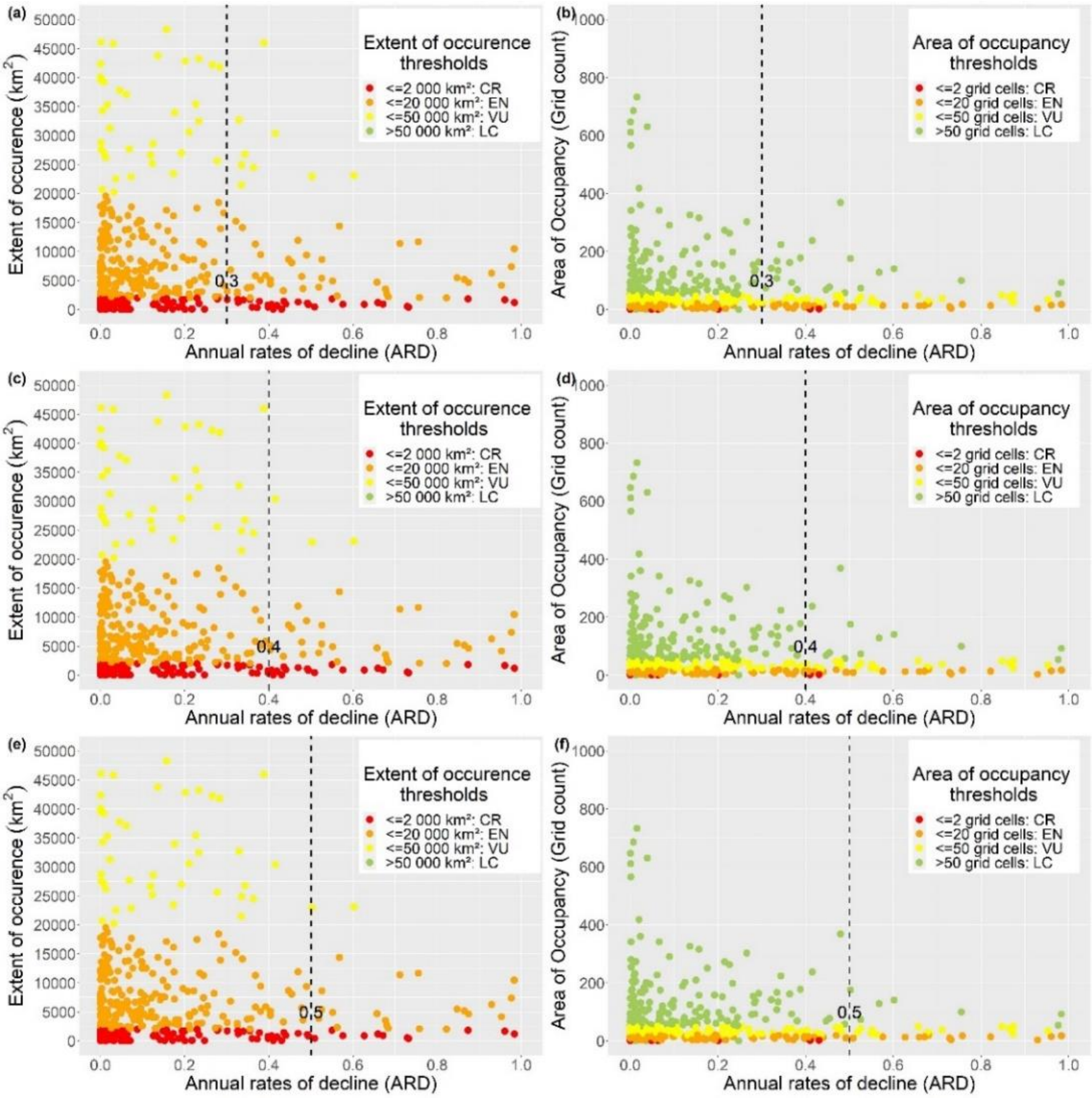

Figure 3: Visual representation of the implications of setting the annual rate of decline high or low, with the dotted vertical line representing the absolute rates of decline fixed at 0.3%, 0.3%, and 0.5. All Figures representing the Extent of Occurrence and Area of Occupancy suggest that as the annual rates of decline threshold increases, fewer ecosystems qualify to be listed as threatened under sub-criteria B1 and B2, and vice-versa.

2.4.4 Criteria and sub-criteria that drive the assessments

Different components of the IUCN RLE criteria A and B are responsible for the majority of the IUCN RLE listings. For example, when considering individual sub-criteria, sub-criterion B1 is solely responsible for listing the majority (22) of ecosystems under the CR category. This is not surprising considering the large thresholds and the area estimates calculated using the minimum convex polygon (Bland *et al*., 2017). The IUCN RLE sub-criteria A3, B1, and B2 are also influential in listing many of the threatened ecosystems under the Endangered category. These are ecosystems that have experienced historic declines and those that are restricted in their distribution. Criterion A (A2b and A3) and sub-criterion B2 have played an influential role in the listing ecosystems under the VU category, however, the bulk (54) of these ecosystems have experienced historic declines (Table 9).

For the SA ETS listing (Table 9), criterion A1 is solely responsible for classifying the majority (115) of terrestrial ecosystems as threatened. The high number of threatened ecosystems in this criterion is attributed to the decision thresholds in conjunction with the biodiversity targets which are tailor-developed for South Africa's small terrestrial ecosystems (Dayaram *et al*., 2017). More importantly, Table 9 results highlight the critical role each sub-criterion plays independently in identifying ecosystems experiencing various manifestations of change over different time frames (Bland *et al*., 2017). Therefore, the RLE assessments should be based on the full range of criteria to list all ecosystems experiencing spatial and functional change so as to maximise conservation impacts and arrest biodiversity loss.

Table 9: Criteria responsible for the listing of ecosystems under the IUCN RLE and SA ETS systems. The IUCN RLE sub-criteria A2b (potential future change) and A3 (historic change) assess the reduction in spatial extent, sub-criteria B1 (extent of occurrence), and B2 (area of occupancy) identifies ecosystems with restricted extent. Similarly, SA ETS sub-criterion A1 lists ecosystems that have experienced a reduction in geographic distribution while sub-criterion C classifies ecosystems with restricted geographic distribution.

Threat Status	IUCN RLE criteria						SA ETS criteria	
	A2b	A3	A2bA3	B1 (EOO)	B2 (AOO)	B1B2	A1	C
CR	1	9	9	22	3	22	27	0
EN	4	24	24	33	22	33	37	8
VU	24	54	54	3	22	4	51	2
LC	429	371	371	400	411	399	343	448
TOTAL	458	458	458	458	458	458	458	458

2.5 Discussion

South Africa's alignment with the IUCN RLE standards has the potential to either validate or undermine the ongoing conservation and management efforts that are intended to arrest biodiversity loss in natural environments. The risk of this occurring is presented by the shift in assessment outcomes between the IUCN RLE and SA ETS systems (McIntosh *et al*., 2018). As such, the robustness and adequacy of the IUCN RLE as a conservation tool need to be tested to understand the pros and cons of this alignment from the policy and land-use conservation planning perspectives. In this chapter, I compared and contrasted the assessment outcomes generated based on the IUCN RLE and SA ETS spatial criteria (i.e. reduction in geographic distribution and limited extent) to better understand the degree to which they are comparable, investigated potential differences, and interrogated their basis.

Taken together, the results reveal that the proportions of matching assessment outcomes are high when the risk categories (CR and EN versus VU and LC) are split in accordance with their policy uptake (i.e. NEMA EIA regulations) but relatively low per individual class or risk category. The matches are encouraging considering that the SA ETS and IUCN RLE systems are developed independently, and there are conceptual and technical differences in their application. More importantly, the matching risk categories increase the legitimacy of the assessment outcomes and confidence in the quality of the conservation interventions rendered. Therefore, the transition from SA ETS to the IUCN RLE system could be executed without major disruptions to some of the conservation interventions including the EIA guidelines for these ecosystems (NEMA-Act no.107 of 1998). However, the ecosystems that are down-listed by the IUCN RLE system, particularly the 16 types that have transitioned from EN to VU category (Table 7) raise concerns. More so, because they are likely to disrupt the implementation of EIA guidelines and the establishment of land-based conservation programmes (i.e. stewardship sites through protected area expansion strategies) that may already be in the pipeline (Government of South Africa, 2010).

In South Africa, ecosystems assigned to the VU category fail to trigger conservation responses as they are considered to still have a reasonable proportion of their original natural habitat remaining. Instead, they serve as a warning bell to the conservationist about their possibility of being at immediate risk of collapse in the future under the 'business as usual scenario' (Driver *et al*., 2012). Failure for these ecosystems to be featured in policy and spatial planning tools (Figure 1) exposes them to anthropogenic land cover change that will further accelerate their risk of collapse. The ecosystems down-listed by the IUCN RLE system could also set South Africa back in terms of achieving both its national and international agenda and goals (Figure 1). An improved understanding of the basis of these misalignments serves as an important step that could potentially map a way forward on how to address the associated land-use conservation implications.

The IUCN size threshold adopted under criterion B (Table 4) suggests that the international perception of the small ecosystem is quite different from that used in South Africa. This is not surprising given that the IUCN RLE system is developed at a global scale and it is intended to be applicable across diverse environments (Bland *et al.*, 2017). The practicality of the size thresholds becomes questionable, particularly in mega-diverse countries like South Africa where species composition influences the delineation of ecosystem types (Dayaram *et al.*, 2017). Equally, South Africa's conceptualisation of what constitutes an ecosystem to be regarded as small is poorly motivated and the thresholds are trivial (RSA, 2011). This is an important source or driver of the mismatches in the ecosystem threat status (Table 9).

The practicality and efficacy of the international standards could also be improved by the development of a hierarchical ecosystem classification system that includes an indicative spatial mapping scale range (Dahdouh-Guebas *et al.*, 1998). Each of the prescribed mapping scales is to be accompanied by area or decision thresholds for each of the assessment criteria. Using the spatial mapping scale range as a guiding tool, assessors can select the decision thresholds that align with the scale at which an ecosystem map is delineated nationally, regionally, etc. Such improvements will address the generalised notion that 'one size fits all which is not necessarily the case in ecosystem assessments (Boitani *et al.*, 2015).

Clearer guidance on what constitutes ongoing spatial and functional decline and associated uncertainty is of the utmost importance considering how influential the concept of progressing loss in the listing of threatened ecosystems under criterion B (Figure 3; Bland *et al.*, 2017). The current perception is that quantitative thresholds for ongoing declines can be set to achieve the primary objectives of the assessments and intentions of the decision-makers. Such ambiguity will result in the over- or under-estimation of ecosystem status, resulting in incomparable assessments between countries and regions. Acknowledging that the application of the full suite of the ecosystem assessment criteria is highly dependent on data availability and credibility, threatened species data can be used to bridge some of these data gaps as it directly relates to ecosystem risk of collapse. For instance, Skowno *et al.*, (2019b) defensibly used threatened species to assess ongoing functional declines. That is, threatened species data was used to identify narrowly distributed ecosystems with very high levels of biotic disruption from overgrazing, invasive species, and poor fire management which resulted in additional ecosystems being listed as threatened. Nonetheless, it is important to note that the purpose of this study is not to discredit the use of the IUCN RLE system but rather to participate and contribute towards addressing potential issues that may hinder its effectiveness and broad uptake. More importantly, I acknowledge that these comparison assessments are based on selected IUCN RLE and SA ETS criteria, as such, a different picture is likely to surface when the full suite of assessment criteria are considered.

Lastly, to mitigate some of the challenges presented by the alignment with the IUCN RLE, the South African National Biodiversity Institute in collaboration with various key partners are exploring the possibility of extending the concept of "species of special concern" to ecosystems. Its practical applicability to ecosystems may result in an additional criterion being introduced for use in South Africa, outside of the IUCN RLE standards. The objective of such a criterion in the context of ecosystems will be to list ecosystem types that fail to trigger the IUCN RLE criteria, however, there is credible evidence to suggest that they need a particular conservation and management response. This concept is not entirely new in South Africa as it has been successfully implemented at the species level. The 'species of special concern' was developed as an additional criterion to the IUCN Red List of Threatened Species (RLS) system to list species of cultural, ecological, or economic significance that failed to be listed under the IUCN RLS risk categories using modern criteria (Victor & Keith, 2004). The primary objective of this criterion is to inform conservation and management prioritisation efforts of natural habitats that may otherwise be overlooked the through RLE listing process (SANBI & UNEP-WCMC, 2016). Since this mirrored criterion will be developed independently and locally, it enables the country to accomplish its national agenda and goals whilst remaining compliant with the IUCN RLE standards (Victor & Keith, 2004). Such strategic solutions have the potential of increasing the broad uptake of the IUCN RLE standards across a wider group audience.

3 Is the IUCN Red List of Ecosystems a good proxy for the distribution of threatened species in South Africa?

3.1 Abstract

South Africa lacks national legislation that protects species based on their threat status despite their majority being at high risk of extinction due to habitat loss. Threatened species in the country are mostly protected through ecosystem-focused conservation efforts that are informed by the list of threatened ecosystems. However, the adequacy of these conservation interventions is questionable since plant species tend to respond to fine-scale disturbances. Furthermore, South Africa has incomplete data on land degradation to adequately apply all RLE criteria and assess different symptoms influencing ecosystem change. As such, I hypothesize that the majority of plant species susceptible to degrading processes are concentrated within the Least Concern (LC) ecosystem types. To test this hypothesis, I investigate whether the RLE is a good proxy for the distribution of threatened species in South Africa by examining the overall spatial relationship between: (i) threatened ecosystems and threatened plant species, and (ii) threatened ecosystems and plant species susceptible to land degradation. Two databases representing 2 837 threatened plant species with their associated threatening processes and 458 threatened terrestrial ecosystems assessed for spatial declines were considered for this analysis. The results reveal that the LC ecosystems support the majority (2 367) of threatened species while ecosystems at immediate risk of collapse support a reasonable number of threatened species (Critically Endangered (CR): 724 and Endangered (EN): 826). These counts represent the total number of unique species per ecosystem risk category since some species occur in multiple ecosystem types. The majority (1 151) of plant species occurring within the LC ecosystems are susceptible to degrading processes while ecosystems at immediate risk of collapse (CR: 97 and EN: 137) support a relatively low number of these species. The small and fragmented natural habitats with the CR ecosystems have the highest species density per unit area (km^2) and are largely clustered within the Fynbos and Succulent Karoo biome. Many of South Africa's threatened plant species will benefit from ecosystem-focused conservation efforts, however, the majority will be overlooked. These findings highlight the importance of implementing the full suite of the existing conservation tools (e.g. identification of stewardship sites, Critical Biodiversity Areas, and Ecological Support Areas) effectively such that threatened and unprotected ecological features may be strategically targeted for conservation initiatives, bending the curve of biodiversity loss.

3.2 Introduction

Approximately one-fifth of the world's known plant species are at risk of extinction (Maschinski & Albrecht, 2017). However, there is still a knowledge gap for a large number of species which suggests that the extinction risk may well be underestimated (Davies *et al*., 2011). Based on the current trajectory, it is undoubtedly clear that the global community has yet again failed to reach many of the international targets (e.g. Sustainable Development Goals (SDG) and Convention on Biological Diversity (CBD) Aichi Targets) set for 2020 (Cooper & Noonan-Mooney, 2013).

The 20th century represents an era in which species-focused approaches emanated globally (Franco, 2013). The primary objectives of these approaches include improving knowledge gaps and informing conservation efforts to avert imminent extinctions of threatened species (Hutto *et al*., 1987). Quantifying the number of species affected by each of the observed threats is pivotal for putting into perspective the severity of the impacts and establishing broad trends. However, these efforts are largely rooted in locating the sites or species spatially (Heywood, 2015). This is an exponentially vast amount of field-based work that is often hindered by financial and political constraints. Without a dramatic shift to overcome these constraints or either opting for alternative cost-effective conservation measures, it would be unrealistic to expect that all threatened species will individually receive some form of statutory protection (Prugh *et al*., 2010).

The CBD endorsed the ecosystem approach as its primary framework for action following the rapid proliferation of discussions that emanated in the early 2000s. This framework tackles three of the CDB strategic objectives holistically: (i) conservation, (ii) sustainable use of biodiversity, and (iii) equitable sharing of the benefits (Waylen *et al*., 2014). Not only is the ecosystem-focused approach important for aiding efforts to achieve sustainable development goals, but also minimises financial costs and balances the inherent limitations associated with species-focused approaches (Hutto *et al*., 1987). That is, it reduces further declines for multiple threatened species simultaneously, whilst ensuring that ecological processes and patterns are maintained in equilibrium across all ecosystems (Hutto *et al*., 1987; Mace *et al*., 2006; Keith *et al*., 2013). It is equally important to recognise that species and ecosystems tend to be sensitive to disturbances at a different spatial scale. As such, considering one approach over the other may not completely achieve the goal of reducing or even bringing to halt the rates of biodiversity loss (Carignan & Villard, 2002). Therefore, it is highly recommended that both species- and ecosystem-focused conservation and management efforts be implemented in parallel to enhance long-term positive changes (Bland *et al*., 2017).

South Africa has a long history of ecosystem-based conservation efforts and developed national legislation (i.e. National Environmental Management Act) with a set of regulations that outlines its conservation and management practices. These regulations prohibit clearance of at least 300 m^2 of indigenous natural vegetation within ecosystems at immediate risk of collapse (i.e. either Critically Endangered or Endangered) under the South Africa Ecosystem Threat Status (SA ETS) framework. However, the country does not have similar legislation for protecting species based on their threat status despite habitat destruction being one of the primary drivers of their extinction (Brook *et al*., 2008; Prugh *et al*., 2010; Evans *et al*., 2011). The absence of such legislation is partly attributed to the former national framework (SA ETS) consisting of a criterion that assesses ecosystem threat status based on the number of threatened species they contain. The threatened species criterion is established under the assumption that species threat status mirrors the conditions of the natural habitats that sustain them. This is a cost-effective approach that the country adopted to indirectly conserve some of its threatened species (RSA, 2011).

The International Union for Conservation of Nature (IUCN) recently published a global framework (known as the Red List of Ecosystems-RLE) for listing ecosystems at risk of collapse (Rodriguez *et al*., 2015; Bland *et al*., 2017). South Africa became one of the latest countries to adopt these international standards, replacing the nationally developed (i.e. SA ETS) framework (Skowno *et al*., 2019a). The IUCN RLE does not utilise threatened species information to assess the risk of ecosystem collapse (Bland *et al*., 2017), which could potentially have a bearing on how well South Africa's threatened species are protected by conservation and management interventions informed by the gazetted list of threatened ecosystems. This poses major concerns considering the heavy utility of the list of threatened ecosystems in conservation and land-use planning, and the absence of the national legislation for threatened species in South Africa (RSA, 2011).

In addition to habitat destruction, land degradation has equally devasted both terrestrial and aquatic species (Brook *et al*., 2008; Prugh *et al*., 2010; Evans *et al*., 2011; Etter *et al*., 2015). These threatening processes interact and affect species in complex ways that have not yet been fully understood (Vanbergen & Insect Pollinators Initiative, 2013; Craig *et al*., 2017). At the ecosystem level, Plesník *et al*., (2011) argue that land degradation is "understood and consequently recognised more intuitively than based on the well-developed criteria applied during ecosystem assessment". The absence of such criteria hindered earlier attempts to assess the extent and severity of degradation ecosystems systematically across the landscape (Skowno *et al*., 2019b). Moreover, many countries in regions such as Sub-Sahara Africa lack scientifically robust and consistently mapped baseline data for land degradation, which has also made it impossible to reliably distinguish between degraded and the remaining natural habitats (Vogt *et al*., 2011; Gibbs & Salmon, 2015; Hobbs, 2016; Nkonya *et al*., 2016; Skowno *et al*., 2018). On the other hand, South Africa's land degradation data is incomplete for most parts of the country, restricting the full application of the RLE criteria. As such, ecosystems that are experiencing severe functional changes tend to be assigned a lower risk category unless they are also threatened by anthropogenic land cover change. Such a misclassification downplays the risk of collapse (Skowno *et al*., 2019b), reducing the spatial overlap between threatened species and ecosystems.

Acknowledging these drawbacks, South Africa developed a suite of legal conservation tools to tackle biodiversity loss from different angles and spatial scales which makes threatened species beneficiaries. These include: (i) stewardship programmes that select sites based on a well-established criteria which amongst others targets habitats of unprotected threatened species (SANBI, 2018), (ii) Threatened or Protected Species (TOPS) impacted by illegal harvesting, trade and/or hunting are protected nationally in terms of National Environmental Management: Biodiversity Act No. 10 of 2004 and internationally through Convention on International Trade in Endangered Species (CITES) (Raimondo, 2015), (iii) Key Biodiversity Areas (KBA) identify sites comprising of vital threatened species habitats that are of global importance in contributing towards persistence of biodiversity over a long period (Eken *et al*., 2004), (iv) Critical Biodiversity Areas (CBAs) are identified through systematic conservation planning to target unprotected inland and aquatic features, including threatened species and special of special concern that are critical for conserving biodiversity and maintaining ecological functions of the landscape while the Ecological Support Areas (ESAs) provide connectivity and linkages between CBAs to avoid further fragmentation of the landscape (SANBI, 2017).

South Africa has a long history of assessing threats to species. These efforts are presented in a well-established threatened species database that also includes detailed information on the individual threatening processes and their localities on the landscape (SANBI, 2020). The availability of such a database presents a unique opportunity for the country to look at land degradation through species lenses while efforts to address data gaps at the ecosystem level are ongoing (Skowno *et al*., 2019a). Such innovative approaches will shine a light on how well ecosystem-focused conservation efforts indirectly offer protection to threatened species. At the same time, the findings will also stimulate discussions around finding alternative solutions to address conservation gaps and/or strengthen the effectiveness of the existing tools. This is an important and necessary step considering that threatened species are also conserved through the listing process due to the lack of national legislation for threatened species (RSA, 2011). In this study, I test the degree to which the IUCN RLE ecosystem threat levels can be used as a proxy for the distribution of threatened species in South Africa by investigating (i) whether the IUCN RLE system will provide adequate protection of the country's biodiversity, as indicated by the overlap between threatened ecosystems and species of conservation concern and (ii) how well-threatened ecosystems spatially overlap with species threatened by loss of habitat loss versus other threatening processes including land degradation?

3.3 Methods

A database on South Africa's list of threatened plant species with the associated threatening processes was obtained from the South African National Biodiversity Institute (SANBI) Threatened Species Programme. This database consists of 2 837 unique plant species at high risk of extinction (CR: 448, EN: 869, and VU: 1 520), assessed following the IUCN Red List of Threatened Species standards. Each threatened plant species was linked to the ecosystem type(s) it occurs in using a combination of processes and tools including georeferencing and spatial analysis, followed by expert verification processes (SANBI, 2020). A species was declared endemic if 90% or more of its localities are predominantly within a specific ecosystem type (Powrie, 2017).

Another key input data is South Africa's list of threatened terrestrial ecosystems. The 458 terrestrial ecosystem types are delineated based on a well-established and hierarchical classification system (Mucina & Rutherford, 2006) and assessed for spatial declines using a land cover-based habitat modification layer for the two-time points (1990-2014) and fine-scale land cover data from selected provinces. The risk of collapse was measured based on two of the IUCN RLE criteria (criterion A: Reduction in geographic distribution and Criterion B: Restricted geographic distribution) that are related to change in the geographic distribution of ecosystems. These ecosystems were then assigned to any of the four ordinal risk categories (Critically Endangered (CR), Endangered (EN), Vulnerable (VU), and Least Concern (LC)) depending on how close each type is from collapsing (Bland *et al*., 2017; see Chapter 2 for detailed assessment process). Both threatened species and ecosystems databases were imported and merged in R statistical software version 3.6.3 (R Development Core Team, 2020). A screening process was undertaken to remove duplicates and ecosystem types without records of threatened species. This resulted in a total of 8 125 records representing unique associations between each species and each of the 345 ecosystem types being retained. A reasonable number of these species occur in multiple ecosystem types and are also threatened by a combination of threatening processes (SANBI, 2020).

3.3.1 Quantifying the spatial relationship between threatened ecosystems and threatened plant species and their associated densities

The hypothesis that the IUCN RLE is a good proxy for the distribution of threatened species was tested by investigating their spatial relationship using the *dplyr* package (Wickham *et al*., 2020b). This was accomplished by tabulating the number of threatened species occurring in each of the 345 ecosystem types listed in any of the four risk categories (CR, EN, VU, or LC). Note that the counts represent the total number of unique species per ecosystem risk category since some species occur in multiple ecosystem types that are also assigned to the same risk category (Young & Desmet, 2016). These counts were then presented in stacked barplots using *ggplot2* package (Wickham, *et al.*, 2020a) to illustrate the general spread of threatened species across ecosystems listed in these risk categories (Figure 4a). Lastly, a total of 1 764 threatened endemic species were extracted and also presented in a stacked barplot to determine whether a similar pattern replicates across the ecosystem risk categories (Figure 5a).

The Least Concern ecosystems generally have a large natural extent remaining while Critically Endangered and Endangered ecosystems have small fragmented natural patches (Skowno *et al*., 2019b). Habitat fragmentation between natural patches has the potential to either increase or decrease species abundance (Tischendorf *et al*., 2005). To test this hypothesis, I calculated the density of threatened plant species by dividing the total number of threatened species in each risk category by the overall remaining natural extent (km^2) of ecosystems assigned to each of the four risk categories (CR, EN, VU, and LC). The estimates were tabulated in a contingency table and plotted (Figure 4b) using *ggplot2* package (Wickham, *et al*., 2020a). This process was repeated with threatened endemic species to determine if the distribution pattern is replicated (Figure 5b).

3.3.2 Quantifying the spatial overlap between threatened ecosystems and species impacted by degradation versus other threats processes and their densities

South Africa's Threatened Species Programme recorded and assigned a total of 26 unique threats impacting all organisms into 11 threat categories (Table 10). The availability of such data presented an avenue to investigate the spatial overlap between threatened ecosystems and species that are susceptible to degrading processes versus other threatening processes including habitat loss. I chose this approach because of the shortcomings surrounding South Africa's land cover-based habitat modification dataset. This dataset does not include the full range of the land cover classes (e.g. natural, near-natural, fair, and poor condition, and not-natural condition). It only includes two reliably mapped areas representing the remaining natural habitats and anthropogenic cover types. This is because South Africa lacks reference data that could be used to reliably distinguish between degraded and natural habitats. As such, degraded areas tend to be assigned to the natural class, overestimating the remaining natural extent (see, Skowno, 2018 for a detailed description of the land cover-based habitat modification classes).

A filtering process was undertaken to remove duplicates on the composite data based on the species name and individual threats. Each of the individual threats was assigned to either one of the three broad threat categories, namely, land degradation, habitat loss, or any other threatening processes. An exception was awarded for mining & quarrying activities since its symptoms influence both habitat loss and land degradation (Table 10). If the majority of a single species' threats are degrading, then land degradation is declared to be the principal threat of that specific species, else habitat loss or other threatening processes is recorded as the primary threat. Species were split according to their susceptibility to either single or multiple broad threatening processes. A total number of 2 252 plant species are predominantly driven to extinction by either one of these broad threatening processes while 585 plant species are impacted by a combination of these broad pressures. The number of species susceptible to these broad threatening processes and their associated density was tabulated using the *dplyr* package and presented in a stacked bar plot to illustrate the spread across different ecosystem risk categories.

Generally, small isolated patches of natural habitat have high plant species density (Vance & Nevai, 2007). These habitats may already be susceptible to a variety of broad threatening processes which also exacerbate the extinction risk of species they host (Brook *et al*., 2008; Prugh *et al*., 2010). Knowledge of these ecosystems is key in guiding strategic conservation and management plans that will avert further loss of natural habitats and encourage species persistence (Guy *et al*., 2019). Arbitrary thresholds for threatened plant species density (i.e. greater than 0.15, 0.31, 0.62 species per unit area (km^2)) were used to classify the 345 ecosystems into two categories supporting high threatened plant species density and those that support low threatened plant species density. This was achieved by assessing the sensitivity of species density supported by each ecosystem type against the three arbitrary thresholds. A total number of 46 ecosystems have species densities of more than 0.15 species per unit area (km^2) while 24 have densities greater than 0.31 species per unit area (km^2) and only 12 have density more than 0.62 species per unit area (km^2). The data frames generated using each of the arbitrary thresholds were intersected to identify ecosystems that have been consistently selected across each of the three thresholds class and have high densities of threatened plant species.

Table 10: Description of all organism's threatening processes and their classification into threat categories as per South Africa's Threatened Species Programme standards (SANBI, 2020). The individual threats were reclassified into either one of the three broad threat categories (i.e. habitat loss, land degradation, and other threatening processes) based on how their symptoms change the natural habitats. The symptoms of mining & quarrying activities cause both spatial and functional declines of the natural habitat, as such, it is recorded under both broad threat categories.

Threat categories	Description	Broad threat categories
Agriculture	- Annual & perennial non-timber crops - Wood & pulp plantations	Habitat loss (anthropogenic pressures)
Oil & gas drilling	- Mining & quarrying	
Residential & commercial development	- Commercial & industrial areas - Housing & urban areas - Tourism & recreation areas	
Transportation & service corridors	- Including roads & railroads - Utility & service lines	
Agriculture	- Livestock farming & ranching	Land degradation (pressures that cause environmental degradation, & disruption of biotic processes and patterns)
Biological resource use	- Gathering terrestrial plants - Logging & wood harvesting	
Climate change & severe weather	- Droughts - Habitat shifting & alteration - Storms & flooding	
Invasive and other problematic species	- Invasive non-native/alien species/diseases - Problematic native species/diseases - Problematic species/disease of unknown origin	
Natural system modifications	- Dams & water management/use - Fire & fire suppression - Other ecosystem modifications	
Oil & gas drilling	- Mining & quarrying	
Pollution	- Agricultural & forestry effluents - Air-borne pollutants, domestic & urban wastewater - Industrial & military effluents	
Human intrusions & disturbance	- Recreational activities	Other threatening processes

3.4 Results

3.4.1 Threatened plant species occurring across South Africa's terrestrial ecosystems listed in four risk categories

South Africa's terrestrial ecosystems capture threatened plant species disproportionately across different risk categories. The majority of threatened plant species (2 367 of 2837) are concentrated within the Least Concern ecosystem types (Figure 4a). These ecosystems make up the bulk of the remaining natural habitat in the country (921 791.65 km^2). Vulnerable ecosystem types, which have 41 589.58 km^2 natural extent remaining, have 641 threatened species; Endangered ecosystem types, which have 29 254.45 km^2 natural habitat remaining, have 826 threatened species; and Critically Endangered ecosystem types, which have only 4 215.82 km^2 natural habitat remaining, have 724 threatened plant species. These counts represent the total number of unique species per ecosystem risk category, eliminating any element of duplication. Notably, Critically Endangered ecosystems support a low number of threatened plant species but have a substantially higher density of species per unit area (km^2) than ecosystem types within the lower risk categories. Meanwhile, the Least Concern ecosystem supports a higher number of threatened species, but the species density is lower than that of threatened ecosystems (Figures 4a and 4b). This is to be expected since the natural patches of Critically Endangered ecosystems are small and fragmented (Skowno *et al*., 2019b), as such, reduces species dispersal rates between habitats and increases the turnover between communities (Prugh *et al*., 2008; Slingsby, 2011).

South Africa also has exceptionally high plant endemism or species that are localised to individual ecosystem types (Skowno *et al*., 2019a). These make up the bulk (62%) of threatened plant species (Figure 4c). The majority (1 515) of endemic species are widespread across the Least Concern ecosystem types. Despite the diversity of species found in these ecosystems, their density per unit area is relatively low in comparison to that supported by the threatened ecosystems (Figure 4d). Large natural habitats tend to have high habitat heterogeneity (Arroyo-Rodriguez *et al*., 2009; Schuler *et al*., 2017) which may explain the substantially high species diversity supported by the Least Concern ecosystem types (Figures 4b and 4d). Therefore, the hypothesis that species richness and density are correlated must be made with caution as the increase in the number of sampled individuals per habitat and the habitat size influences the direction of this relationship (Slingsby, 2011).

In summary, while many of these species will indirectly benefit from highly focussed conservation efforts that are informed by the list of threatened ecosystems. Meanwhile, the majority will likely continue being exposed to threatening processes that exacerbate their extinction rates in the absence of additional measures that provide strategic conservation responses to these species (RSA, 2011). More importantly, the uneven distribution of threatened species across the ecosystem risk categories puts into perspective how valuable the Least Concern types are for supporting and ensuring the persistence of threatened endemic plant species. As such, alternative solutions are needed to ensure that the roll-out of strategic conservation responses also includes some of the Least Concern ecosystems, particularly those that support high threatened species populations. More so considering that the functional declines in the RLE assessments are inadequately assessed. As more of these ecosystems become up-listed, the spatial overlap between threatened ecosystems and species will likely be more pronounced.

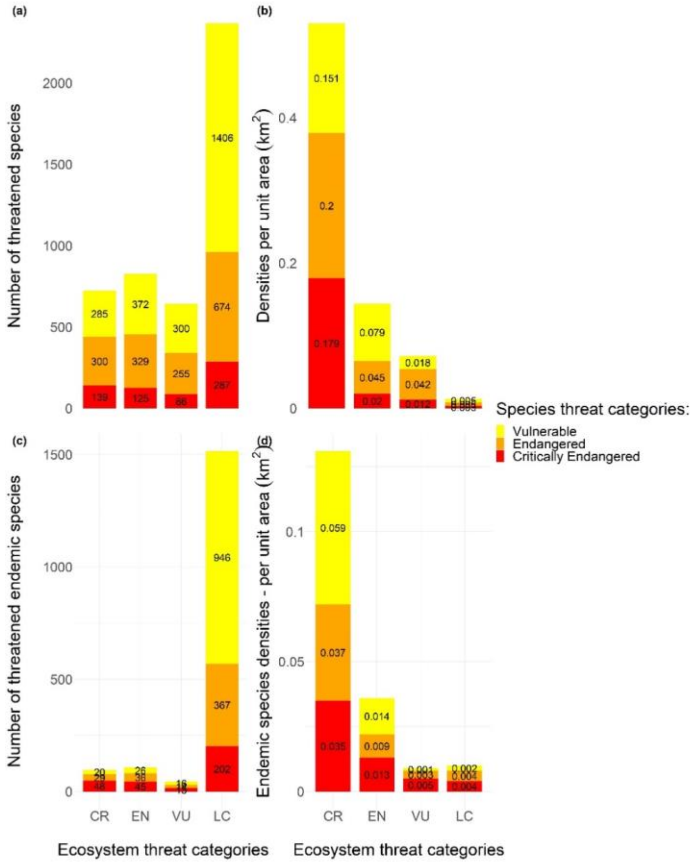

Figure 4: The total number of the overall threatened plant species and threatened endemic plant species (assessed by the Threatened Species Programme following the IUCN RLS standards) distributed across terrestrial ecosystems listed in the four risk categories (Critically Endangered – CR, Endangered – EN, Vulnerable – VU, and Least Concern – LC) following the IUCN RLE standards (Figure 4a and 4c), and their densities per unit area-km² (Figure 4b and 4d).

3.4.2 Broad threatening processes that exacerbate the extinction rates of plant species widespread across threatened ecosystems

Range of threatening processes impact species at different levels which influences their extinction rates (Carignan & Villard, 2001). The extinction risk of the majority (2 254 of 2 837) of South Africa's threatened plant species is exacerbated by either one of the broad threatening processes (e.g. habitat loss, land degradation, or other threats) (Figure 5a). Many of these species (1 495) are widespread across the Least Concern ecosystem types with 1 151 of them being impacted by degrading processes while 343 are susceptible to habitat loss. Ecosystems at immediate risk of collapse support a low number (CR: 178 and EN: 265) of species threatened by either habitat loss or land degradation pressures. The distribution of threatened species density per unit area (km^2) is skewed towards the highly threatened ecosystems (Figure 5a). These are ecosystems that have limited spatial extents and are predominately isolated whilst the spatially intact types support low species density per unit area (km^2; Figure 5b).

On the other hand, a considerable number (583 of 2 837) of the country's plant species are susceptible to multiple broad threatening processes that not only shrink but also degrade the natural habitat(s) that sustain them (Figure 5c). The majority (316) of these double impacted species are endemic or localized to a single ecosystem type. The distribution pattern of the diversity and density of these double impacted plant species (Figures 5c and 5d) across ecosystems listed in the four risk categories (CR, EN, VU, and LC) remain consistent with that of species impacted by a single broad threatening process (Figures 5a and 5b). In summary, the high concentration of species susceptible to degrading processes found within the least threatened ecosystems suggests that these types are also undergoing functional changes. Therefore, the hypothesis that threatened ecosystems are a good proxy for the distribution of threatened species is to be made with caution particularly when there is incomplete data to comprehensively assess all threats to ecosystems. More importantly, the results reveal that the RLS information has the potential to improve the RLE assessments which could lead to some of the down-listed ecosystems being up-listed (Skowno *et al*., 2019b).

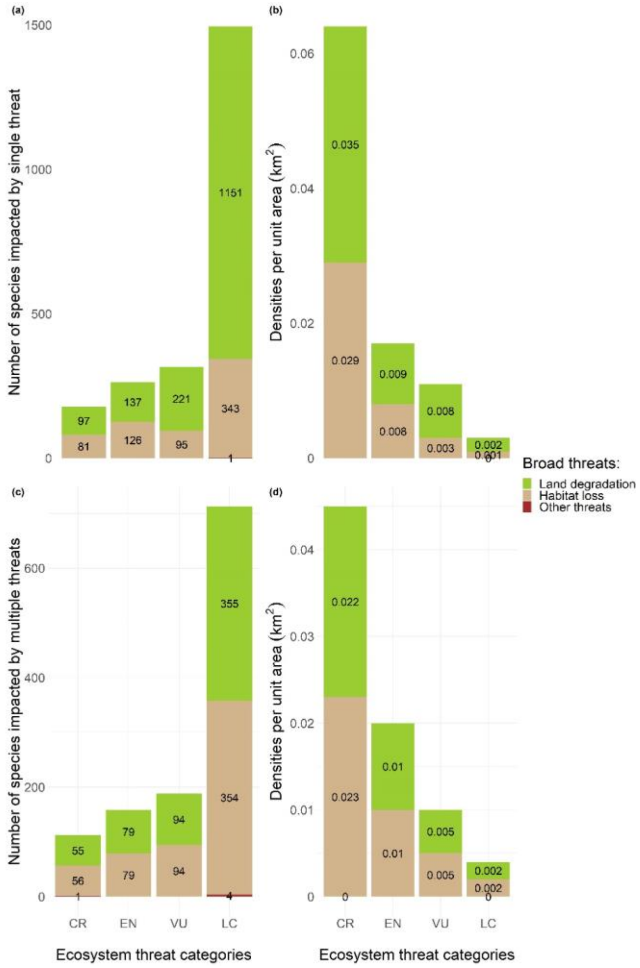

Figure 5: The distribution of threatened plant species susceptible to a single (5a) and multiple pressure (5b) across natural patches of ecosystems listed in the four risk categories (Critically Endangered – CR, Endangered – EN, Vulnerable – VU, and Least Concern - LC) and their associated densities per unit area (Figure 5b and 5d).

3.4.2.3 Ecosystem types with high densities of species threatened by spatial and functional threats

Threatened plant species density is unevenly distributed across South Africa's terrestrial ecosystems (Figure 4 and 5). This can be attributed to several factors including the fragmentation which reduces the size of the remaining natural habitat and their proximity to each other, resulting in the declining dispersal rates (Andren, 1994). From a total of 345 ecosystems supporting threatened species, only 12 of these types were consistently listed under each of the three arbitrary thresholds (0.16, 031, and 0.62) classes while the majority (333) changed classes. These results suggest that the species densities are very sensitive to the chosen arbitrary thresholds. The Critically Endangered Piketberg Quartz Succulent Shrubland has the highest density (9.662) of threatened species per unit area (km^2). This ecosystem type is found in the Succulent Karoo biome, an international hotspot recognised for its high diversity of endemic plant species. In the recently published national state of South Africa's biodiversity report, the findings reveal that this biome has experienced increased rates of habitat loss of 0.017%/year between 1990 and 2014 (Skowno *et al*., 2019b). Habitat destruction has not only had devasting impacts on many of the ecosystems but also the species supported by this biome. Moreover, degrading processes have also added pressure on species supported by Piketberg Quartz Succulent Shrubland, exacerbating their extinction risk (Table 11).

Fynbos biome is also globally recognised for its exceptionally high concentration of diverse endemic plant species and is one of the most threatened biomes in South Africa (Skowno *et al*., 2019a). This biome also supports 11 of the 12 ecosystems that have a high density of threatened species per unit area (km^2), with only one of these types being at a lower risk of collapse (i.e. Hangklip Sand Fynbos). The Critically Endangered Peninsula Shale Renosterveld has the highest density (4.88) of threatened species per unit area (km^2) in this region whilst other ecosystem types have densities ranging between 0.664 to 1.601 species per unit area (km^2). In recent years (1990 to 2014), the Fynbos biome has experienced increased rates (0.15%/year) of habitat loss (Skowno *et al*., 2019b) which has also impacted six of these ecosystems (Elgin Shale Fynbos, Agulhas Sand Fynbos, Citrusdal Shale Renosterveld, Cape Flats Sand Fynbos, Hangklip Sand Fynbos, and Peninsula Shale Renosterveld) at varying degree including the species they host. The other four ecosystems (Potberg Ferricrete Fynbos, Lourensford Alluvium Fynbos, Peninsula Granite Fynbos, and Peninsula Shale Fynbos) support plant species that are largely susceptible to degrading threats (Table 11).

Lastly, the Endangered Alexander Bay Coastal Duneveld ecosystem type occurring within the Dessert biome also has a high threatened species density (0.951) per unit area (km^2). Plant species widespread across the remaining natural patches of this ecosystem type are susceptible to both degrading processes and habitat loss (Table 11). This biome has experienced only 0.009%/year rates of habitat loss between 1990 and 2014 (Skowno *et al*., 2019b). In summary, many of South Africa's ecosystems with a high density of threatened species are also threatened. This suggests that the species supported by these ecosystems are already or will indirectly benefit from ecosystem-focused conservation interventions. More importantly, their abundance in these ecosystems signals a need for improved effective and focused restoration and management responses to mitigate the negative impacts presented by degrading threats. On the other hand, the high species densities found within the Least Concern Hangklip Sand Fynbos also signal the need for this ecosystem type to be up-listed or conserved through other measures to restrict fine-scale land-use activities that threatened species occurring in this ecosystem. This is an important step since the country lacks national legislation that protects species based on their threat status (RSA, 2011).

Table 11*: Ecosystems with high densities of threatened plant species per unit area (km^2).*

Ecosystem type	Biome	Natural Extent (km^2)	Ecosystem threat status	Number of threatened species per ecosystem type	Density per unit area (km^2)	Principal threats
Agulhas Sand Fynbos	Fynbos	127.3851	VU	91	0.714	Habitat loss
Alexander Bay Coastal Dune-veld	Desert	2.1033	EN	2	0.951	Habitat loss & Land degradation
Cape Flats Sand Fynbos	Fynbos	132.5448	CR	171	1.29	Habitat loss
Citrusdal Shale Renosterveld	Fynbos	13.257	CR	16	1.207	Habitat loss
Elgin Shale Fynbos	Fynbos	82.809	CR	55	0.664	Habitat loss
Hangklip Sand Fynbos	Fynbos	56.3625	LC	90	1.597	Habitat loss
Lourensford Alluvium Fynbos	Fynbos	7.9956	CR	11	1.376	Land degradation
Peninsula Granite Fynbos	Fynbos	34.3755	EN	42	1.222	Land degradation
Peninsula Shale Fynbos	Fynbos	6.0273	VU	7	1.161	Land degradation
Peninsula Shale Renosterveld	Fynbos	3.6882	CR	18	4.88	Habitat loss
Piketberg Quartz Succulent Shrubland	Succulent Karoo	0.621	CR	6	9.662	Habitat loss & Land degradation
Potberg Ferricrete Fynbos	Fynbos	19.9818	VU	32	1.601	Land degradation

3.5 Discussion

Ideally, RLE assessments should be based on a full range of criteria to capture different ways spatial and functional symptoms impact ecosystems (Bland *et al*., 2017). However, this is rarely the case because many countries including South Africa have incomplete degradation data to comprehensively undertake such assessments (Tully *et al*., 2015; Skowno *et al*., 2019a). The limitation of this in the RLE assessments is that it underestimates the risk of collapse, reducing the spatial overlap between threatened species and ecosystems. As such, the conservation footprint becomes restricted to ecosystems that have made it to the list and the fine-scale features that spatially overlap with these types (Skowno *et al*., 2019b).

Taken together, the results reveal that not all ecosystem types undergoing spatial declines entirely reflect the status of threatened plant species they contain. Many of these threatened plant species overlap with ecosystems at immediate risk of collapse (CR and EN). Such species will indirectly benefit from broad-scale conservation interventions that are informed by the list of threatened ecosystems. These include the rigorous EIA procedures and guidelines that restrict spatial declines of indigenous vegetation within CR and EN ecosystems, but not from functional declines (NEMA-Act no.107 of 1998). Meanwhile, the majority of plant species threatened by either habitat loss and/or land degradation occurring within the least threatened ecosystems (Figure 5) will likely continue experiencing increasing extinction rates in the absence of strategic conservation responses. This is because the spatial declines within Least Concern ecosystems are considered to either be minimal or stable to trigger conservation response. As such, anthropogenic land-use activities are permitted to take place without any restrictions to promote the country's economic growth (Ferrar & Lötter, 2007).

The high concentrations of plant species impacted by habitat loss found within the Least Concern ecosystems suggest that fine-scale spatial disturbances are not adequately captured by the habitat modification layer (see, Skowno, 2018). This also alludes that plant species and ecosystems respond to spatial disturbances at a different scale (Carignan & Villard, 2001). More importantly, the spatial overlap between the majority of threatened plant species and the Least Concern ecosystems raises concerns about the efficacy of the IUCN RLE as a conservation tool in South Africa to combat the rate of biodiversity loss. This is a valid concern considering that South Africa: (i) suffers from incomplete land degradation data which has restricted the application of the full range of the IUCN RLE criteria, (ii) uses the list of threatened ecosystems extensively to inform its conservation and management practices, and (iii) lack national legislation to protect species based on their risk of extinction.

These concerns are valid however, they are based on incomplete RLE assessments. Encouragingly, many of these concerns can be addressed through the effective implementation of existing conservation tools such as stewardship sites, KBA, CITES, CBAs, and ESAs, allowing threatened and unprotected ecological features to be strategically targeted for conservation initiatives (RSA, 2011). The effective implementation of these legal tools in parallel with those that are informed by the gazetted list of threatened ecosystems will cover the majority of threatened species and expand the conservation footprint that will bear long-term positive impacts. South Africa also used threatened species data in its recently published RLE to assess functional declines on selected ecosystems as it relates directly to the risk of collapse. This resulted in some ecosystems being up-listed, increasing the spatial overlap between threatened ecosystems and species. By doing so, the country remained completely compliant with the IUCN RLE standards whilst improving the credibility of the assessments (Skowno *et al*., 2019b). It is worth emphasizing that the use of threatened species data in the RLE assessments does not entirely resolve all data challenges restricting the implementation of all the IUCN RLE criteria in the country. As such, long-term solutions that allow all terrestrial ecosystems to be adequately assessed for functional declines remain pivotal in improving the effectiveness of the RLE as a conservation tool and bending the curve of biodiversity loss.

4 Synthesis

The publication of the International Union for Conservation of Nature (IUCN) Red List of Ecosystems (RLE) standards is an important development that has received broad acceptance globally. More than 100 countries across the globe including South Africa and Myanmar have adopted the IUCN RLE standards as their national framework for assessing the risk of collapse (Bonifacio & Pisanu, 2013; Skowno *et al*., 2019a; Murray *et al*., 2020). The strongest motivations for the alignment include: (i) elimination of confusion and reducing the administrative burden for maintaining multiple lists of threatened ecosystems, (ii) increased legitimacy of the ecosystem threat status assessment by basing them on a body of sound scientific literature, (iii) comparable assessments across different environments and countries across the globe, (iv) for the threatened national ecosystems to be recorded under the IUCN RLE registry (Skowno *et al*., 2019a). furthermore, the IUCN Red List makes it easier for countries to secure funding from international donors to achieve national biodiversity conservation objectives, and address knowledge and data gaps through focused research (Betts *et al*., 2020).

The IUCN RLE standards only became available after many countries including South Africa and Australia each independently tailor-developed national indicators or standards for assessing threats to ecosystems (RSA, 2011; Bonifacio & Pisanu, 2013). These indicators are developed to aid biodiversity monitoring efforts and many have progressed into the legislated national list of the threatened ecosystems (Bland *et al*., 2019; Botts *et al*., 2020). In South Africa, the gazetted list of threatened ecosystems is ratified to inform policy development and land-use planning tools that mainstream biodiversity considerations into economic development activities (RSA, 2011). Considering the strong links between the gazetted list of threatened ecosystems and many of the policy and spatial planning tools, the change and/or update to the IUCN RLE standards may disrupt conservation and land-use plans. As such, there is a need to interrogate and holistically understand the implications that may emanate from this shift, hence the importance of this study.

This study was aimed to compare the South Africa Ecosystem Threat Status (SA ETS) and the IUCN RLE standards conceptually and quantitatively. The efficacy of the IUCN RLE as a conservation tool in South Africa was also tested by investigating whether the risk status of terrestrial ecosystems adequately reflects the risk status of the plant species they contain.

The findings from this study will allow South Africa to participate and contribute notably towards improving the IUCN RLE standards, increasing its practicality and broad acceptance across the globe.

4.1 Summary of the key findings

Both the IUCN RLE and SA ETS standards have overarching conceptual similarities (e.g. the definition of the assessment unit, spatial and functional criteria, and risk categories) as they both share the common ancestry (IUCN Red List of Threatened Species). Equally, there are key differences (e.g. the conceptualization of small ecosystems, the decision thresholds, and the nested sub-criteria that measure change over different timeframes) that explain the misalignments in the ecosystem threat status between the two systems. To participate and contribute towards improving the efficacy of the IUCN RLE as the global standards for assessing the risk of ecosystem collapse, I propose that the RLE Committee for Scientific Standards (CSS) consider the following: (i) flexible area or decision thresholds for sub-criterion B1 (Extent of Occurrence) that are informed by ecosystem mapping scale range (ii) provide clearer guidance on what constitutes ongoing spatial and functional decline and associated uncertainty. Locally, the South African National Biodiversity Institute in collaboration with various key partners is exploring the possibility of extending the concept of "species of special concern" to ecosystems to mitigate some of the conservation concerns.

4.1.1 Different perceptions of what constitutes ecosystems with limited extent

South Africa has high species diversity, which influences the fine-scale delineation of terrestrial ecosystem types (Dayaram *et al*., 2017). The mapping scale of terrestrial ecosystem types informed the decision on the thresholds acceptable to represents ecosystems that are narrowly distributed. However, these thresholds are trivial and remain poorly motivated (RSA, 2011). Based on the size thresholds (IUCN RLE criterion B and SA ETS criterion C), it is clear that South Africa's perception of a limited natural extent of ecosystems is different from what is perceived internationally (Table 4). This explains the substantial number of ecosystems that qualify under the IUCN RLE sub-criterion B1. Therefore, I propose that the RLE CSS consider the adoption of a hierarchical ecosystem classification system which includes indicative mapping spatial scale range (Dahdouh-Guebas *et al*., 1998), each with its independent area or decision thresholds. Using the mapping scale range as a guiding tool, assessors can select the decision thresholds that align with the scale at which ecosystems are delineated nationally, regionally, etc. Such improvements will address the general notion in the RLE assessments that 'one size fits all' and strengthen the policy uptake of the IUCN RLE standards (Boitani *et al*., 2015).

4.1.2 Ongoing spatial and functional decline

Clearer guidance on what constitutes ongoing spatial and functional decline and associated uncertainty is key considering the importance of this concept and the influential role it carries in the listing of threatened ecosystems under criterion B (Bland *et al*., 2017). The current perception is that quantitative thresholds for ongoing declines can be set to achieve the primary objectives of the assessments and intentions of the decision-makers. Such ambiguity will result in the over- or under-estimation of ecosystem status, resulting in incomparable assessments between countries and regions.

4.1.3 The use of threatened species to bridge data gaps

The findings from this study emphasized the importance of assessing the risk of collapse based on the full range of the IUCN RLE criteria since each criterion captures different symptoms exacerbating the risk of collapse. Although this is encouraged to avoid underestimating the risk of collapse, consequently, narrowing conservation efforts (Bland *et al*., 2017), such assessments are highly data-dependent. That is, the availability and credibility data dictate which criterion is to be implemented. Therefore, alternative solutions such as the use of threatened species data can be used to bridge some of these data gaps as it directly relates to the risk of collapse (Skowno *et al*., 2019b),

4.1.4 Expanding conservation impacts at the ecosystem level

The comparison of the IUCN RLE and SA ETS assessment outcomes suggests that there is a reasonable number of ecosystems listed inconsistently between the two systems. Although some of these ecosystems have failed to trigger the listing under the IUCN RLE spatial criteria, they present evidence of spatial and/or functional declines that are concerning to provincial conservation agencies. To find practical solutions to bridge conservation gaps, the South African National Biodiversity Institute in collaboration with various key partners is exploring the possibility of extending the concept of “species of special concern” to ecosystems. The objective of such a criterion will be to list ecosystem types that fail to trigger the IUCN RLE criteria, however, there is credible evidence to suggest that they need a particular conservation and management response. Its practical applicability to ecosystems may result in an additional criterion being introduced for use in South Africa, outside of the IUCN RLE standards.

4.2 Future research

A major shortcoming in this study is the lack of land degradation data which restricted the use of the full range of IUCN RLE criteria. Therefore, comprehensive assessments using the full range criteria (Bland *et al*., 2017) are imperative to adequately test the efficacy of the IUCN RLE standards as a conservation tool for ecosystems in different regions. There is also scope to investigate what is the appropriate ecological threshold to represent ongoing spatial and functional declines of ecosystems.

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Appendices

Appendix A: Comparison of the full range of the IUCN RLE and SA ETS criteria with their associated biodiversity targets and thresholds where available.

IUCN RLE criteria	SA ETS criteria
A: Reduction in geographic distribution • **A1:** Over the past 50 years (CR: ≥ 80%, EN: ≥ 50%, VU: ≥ 30%) • **A2a:** Over the next 50 years (CR: ≥ 80%, EN: ≥ 50%, VU: ≥ 30%) • **A2b:** Any 50-year period including the past, present, and future (CR: ≥80%, EN: ≥50%, VU: ≥30%) • **A3:** Historical decline since approximately 1750 (CR: ≥ 90%, EN: ≥ 70%, VU: ≥ 50%)	**A1: Irreversible loss of natural habitat** (CR: remaining natural habitat ≤ biodiversity target, EN: remaining natural habitat ≤ biodiversity target + 15%, VU: remaining natural habitat ≤ 60% of the original area of an ecosystem) ***AND*** **B:** Rate of loss of natural habitat
B: Restricted geographic distribution • **B1:** Extent of Occurrence (CR: ≤ 2000, EN: ≤ 20 000, VU: ≤ 50 000) • **B2:** Area of Occupancy (CR: ≤ 2, EN: ≤ 20, VU: ≤ 50) • **B3:** A very small number of threat-defined locations (VU: threat-defined locations that are fewer than 5)	**C: Limited extent and imminent threat** (EN: ecosystem extent ≤ 30 km^2 and imminent threat, VU: ecosystem extent ≤ 60 km^2 and imminent threat)
C: Environmental degradation • **C1:** The past 50 years based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity (CR: 80% extent & 80% relative severity, EN: 50% extent & 50% relative severity, VU: 30% extent & 30% relative severity) • **C2a:** The next 50 years, based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity **OR** • **C2b:** Any 50-year period including the past, present, and future, based on a change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity	**A2: Ecosystem degradation and loss of integrity** (CR: ≥ 60% of ecosystem significantly degraded, EN: ≥ 40% of ecosystem significantly degraded, VU: ≥ 20% of ecosystem significantly degraded)

(CR: 80% extent & 80% relative severity, EN: 50% extent & 50% relative severity, VU: 30% extent & 30% relative severity) • **C3:** Since 1750 based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity (CR: 90% extent & 90% relative severity, EN: 70% extent & 70% relative severity, VU: 50% extent & 50% relative severity)	
D: Disruption of biotic processes or interactions • **D1:** The past 50 years based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity (CR: 80% extent & 80% relative severity, EN: 50% extent & 50% relative severity, VU: 30% extent & 30% relative severity) • **D2a:** The next 50 years, based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity **OR** • **D2b:** Any 50-year period including the past, present and future, based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity (CR: 80% extent & 80% relative severity, EN: 50% extent & 50% relative severity, VU: 30% extent & 30% relative severity) • **D3:** Since 1750 based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity (CR: 90% extent & 90% relative severity, EN: 70% extent & 70% relative severity, VU: 50% extent & 50% relative severity)	**D1: Threatened plant species associations** (CR: ≥ 80 threatened plant species, EN ≥ 60 threatened plant species, VU ≥ 40 threatened plant species) **D2: Threatened animal species associations** **E: Fragmentation**
E: Quantitative analysis that estimates the probability of ecosystem collapse (CR: ≥ 50% within 50 years, EN: ≥ 20% within 50 years, VU: ≥ 10% within 100 years	
	F: Priority areas for meeting explicit biodiversity targets as defined in a systematic biodiversity plan (CR: very high irreplaceability and high threat, EN: very high irreplaceability and medium threat, VU: very high irreplaceability and low threat)

Appendix B: List of abbreviations and acronyms

Definition	Acronym
Absolute rate of decline	ARD
Commission on Ecosystem Management	CEM
Committee for Scientific Standards	CSS
Convention on Biological Diversity	CBD
Convention on International Trade in Endangered Species	CITES
Critical Biodiversity Areas	CBA
Critically Endangered	CR
Ecological Support Areas	ESA
Endangered	EN
Environmental Management Framework	EMF
Integrated Development Plans	IDPs
International Union Conservation of Nature	IUCN
Key Biodiversity Areas	KBAs
Least Concern	LC
National Biodiversity Assessment	NBA
National Biodiversity Strategy and Action Plan	NBSAP
National Environmental Management Act	NEMA
National Spatial Biodiversity Assessment	NSBA
Red List of Ecosystems	RLE
Red List of Threatened Species	RLTS
South Africa Ecosystem Threat Status	SA ETS
South African National Biodiversity Institute	SANBI
Spatial Development Frameworks	SDFs
Sustainable Development Goals	SDG
Threatened or Protected Species	TOPS
United Nations Convention to Combat Desertification	UNCCD
United Nations Environment Programme World Conservation Monitoring Centre	UNEP-WCMC
Vulnerable	VU