# Identifying and prioritising important areas for biodiversity and conservation in Mozambique

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## Author declaration

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## Abstract

It has long been recognised that biodiversity has been heavily impacted by ongoing human activities which has led to massive losses of species and habitats. The consequences of this are profound, as it compromises the functioning of ecosystems services, such as the provision of clean air, water and food, and exacerbates poverty levels and inequality, particularly in developing countries where climate change events are frequent and severe. Nearly the entire world recognises that this trend needs to change now and the inspiration comes from the Convention on Biological Diversity's vision of "living in harmony with nature by 2050". A key part of achieving this vision involves establishing conservation areas to halt biodiversity loss. However, there is growing recognition that, as well as creating protected areas (PAs), we need to account for other area-based conservation measures (OECMs), with a focus on putting local communities at the forefront of conservation initiatives to enhance their well-being and livelihoods. This approach is largely welcomed as conservation will be used as a tool to promote social cohesion and higher potential to achieve national and local targets. There is also a recognition that these new conservation areas need to focus on sites that are important for biodiversity, as many species and ecosystems are under-represented in current PA network.

This thesis consists of four chapters that use spatial conservation prioritisation analysis to identify important sites for biodiversity and people at the national and transfrontier levels in Mozambique. I tested the Important Plant Area (IPA) and the Key Biodiversity Area (KBA) criteria and provided a novel way of delineating their boundaries. Building from this, I applied systematic conservation planning to identify the best places to establish new OECMs in the Maputaland transfrontier biodiversity hotspot then I scaled this up to a national level of Mozambique based on meeting conservation targets. Next, I described a process I co-led to identify KBAs for the whole of Mozambique. Finally, I used national KBA and IPA data to identify priority areas for conservation at a national level, assessing the effectiveness of the current conservation area network for meeting biodiversity targets. This thesis provides insights into how Mozambique's network of PAs and OECMs could be developed but implementing these plans will require governments and civil society to develop and implement context-specific projects that are led by local communities.

## Keywords

Biodiversity conservation, Key Biodiversity Areas, Important Plant Areas, systematic conservation planning, spatial prioritisation, protected areas, OECMs, community-based conservation, ecotourism, Marxan with Zones, Maputaland.

## Covid-19 Impact Statement

H.A. Matimele would like his examiners to note the impact that the COVID-19 pandemic had on his thesis.

The lockdown measures since March 2020, in response to the COVID-19 pandemic, had a significant negative impact on his PhD work. He found himself isolated and remote from his supervisors, family and friends. Matimele's family were also under lockdown in Mozambique, which increased pressure as he had to provide help remotely, spending many hours on WhatsApp calls to provide assistance with schoolwork for the children. He also contracted COVID-19 and experienced serious illness. Over the whole lockdown period, it was difficult to gather data from collaborating institutions and stakeholders in Eswatini, Mozambique, and South Africa. Due to this situation, he experienced moments of anxiety and stress leading to a limited ability to remain focused on my work.

Despite all the difficulties, his supervisors and the wider DICE community have been keen to get him through this situation. We had regular virtual meetings on a weekly basis to discuss work matters, but also to socialise, which made him feel cared for and supported. We made a few changes to one of the data chapters, where it was not possible to collect the necessary information in person. The University granted a COVID-19 further extension with no additional fees being charged. For that reason, he expresses his sincere gratitude to the University of Kent.

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# 1 Introduction

## 1.1 Biodiversity context

Biodiversity is being lost at an unprecedented rate (Williams et al. 2021; Adam et al. 2022; Jaureguiberry et al. 2022). Estimates suggest that the present human-induced rate of species extinction has reached a level more than three orders of magnitude higher than known natural extinction rates (Veach et al. 2017; Munstermann et al. 2022). This fact is of major concern and is particularly impacting the tropics, where the world's terrestrial biodiversity is concentrated (Bax & Francesconi 2019). Much of this decline has been through habitat loss, with estimates showing that about 1.6 million $km^2$ of tropical forest cover had been lost in the twenty-year period up to 2010 (Achard et al. 2014). Moreover, a twelve-year period analysis found that the tropics were losing 2,101 $km^2$ of forests per year within the total global forest cover loss of 2.3 million $km^2$ (Hansen et al. 2013). In addition, many species are impacted by over-exploitation, with thousands of species threatened by logging, hunting, fishing and over-harvesting (Maxwell et al. 2016). Such degradation of natural ecological communities has an impact on the ecosystem services essential for human well-being, such as water supply, climate regulation, carbon storage, biodiversity richness (Hasan et al. 2020; Orth et al. 2020).

In view of the continuing environmental degradation, site-scale conservation initiatives are pivotal in our efforts to halt and reverse the current trend (Pimm et al. 2014; Rodrigues et al. 2014; Mace et al. 2018). Given this urgency, scaling up conservation will require governments to step up at national level (Bicknell et al. 2017; Metcalfe et al. 2022). As a first step, countries need to establish plans that stipulate measurable goals (Maron et al. 2021), then set targets and identify relevant stakeholders who can contribute to implementing the plan and engage with people on the ground. This thesis describes research that is designed to inform such national plans, focusing on the country of Mozambique as a case study of a biodiversity rich, low-income nation (Figure 1-1). In this chapter, I begin by describing protected areas (PAs) and the other main policy instruments for site-based conservation and then explain some of the most widely used approaches for identifying areas that are important for biodiversity. Next, I provide background on conservation in Mozambique and the Maputaland Centre of Endemism, which partly falls within Mozambique and is the focus of a number of conservation initiatives. I finish by outlining the structure of the rest of this thesis, briefly describing the four Maputaland- and Mozambique focused studies.

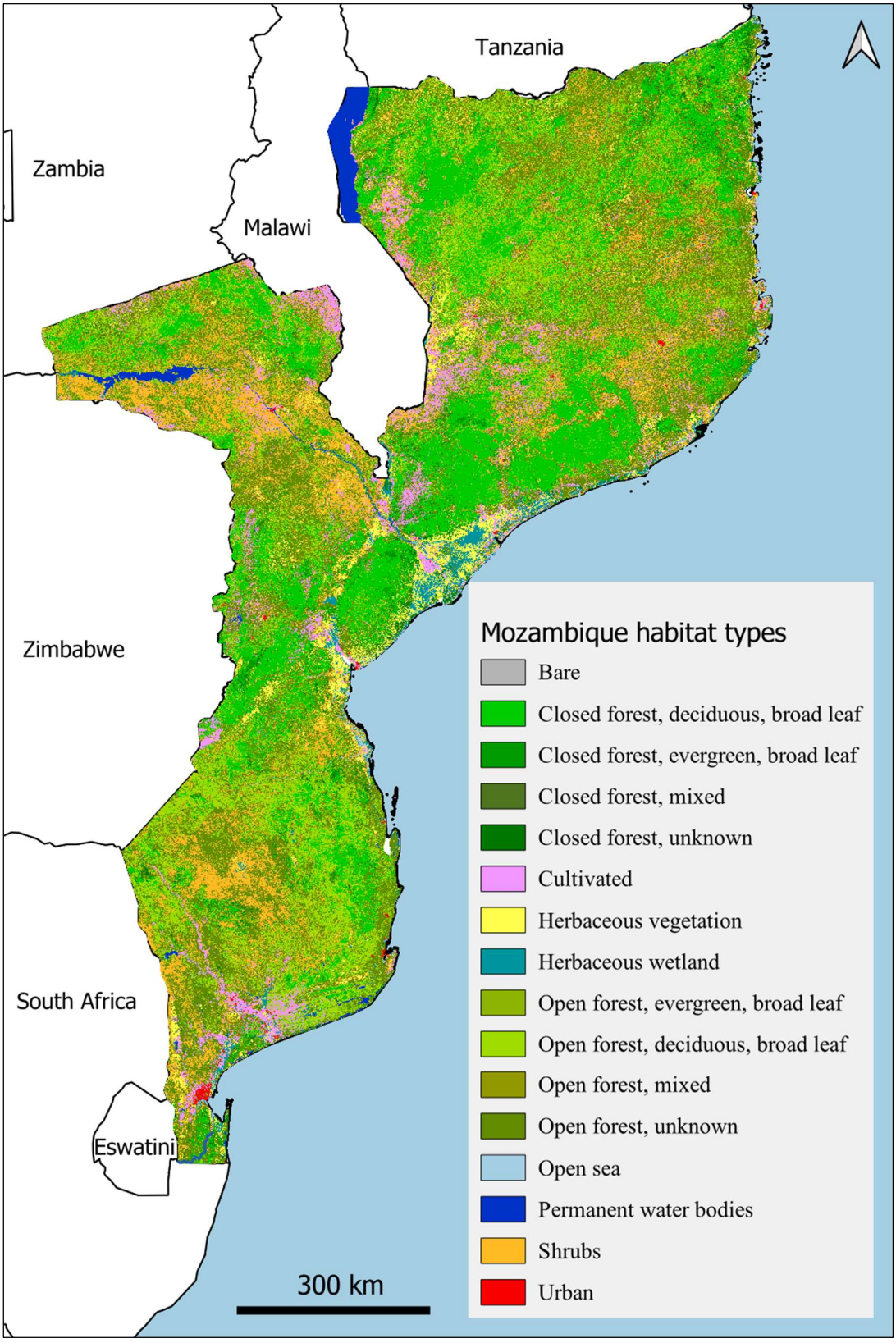



***Figure 1- 1: Distribution of broad habitat types in Mozambique.***

## 1.2 Protected Areas and OECMs

Establishing PAs is one of the key strategies to minimise threats to the environment and thereby limit further loss of biodiversity (Watson et al. 2014; Veach et al. 2017; Bhola et al. 2021). In the past decade, nations have been expanding or establishing new PAs to cover about 15% of the land (Bax & Francesconi 2019). This is just below the 17% that parties to CBD have committed to achieving by 2020 under the Aichi Target 11 (CBD 2010). In light of these efforts, research suggests that a network of PAs need to be appropriately designed and adequately managed, so that it can deliver desirable outcomes in safeguarding biological elements whilst providing vital ecosystem services to human beings (Saura et al. 2017; Visconti et al. 2019). However, it has also been suggested that while PAs can be effective at conserving biodiversity, often this comes at a cost to local communities (Watson et al. 2014). This is because in the past PAs management strategies have focused on restricting public access, depriving neighbouring communities and negatively affecting their livelihoods (Ref). To address this, parties to CBD have developed the Kunming-Montreal Global Biodiversity Framework to inform changes in conservation. Under Target 3 of the Global Biodiversity Framework, signatories committed to conserve at least 30% of the terrestrial and marine realms by 2030. This target specifies that these conservation area networks should be "effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures" (CBD 2022).

Target 3 is important because it is designed to overcome many of the limitations with current PA networks relating to poor management effectiveness (Geldmann et al. 2019), and ecological representativeness (Bax & Francesconi 2019). It also specifies that conservation area networks should not be restricted to PAs, but that other effective area-based conservation measures (OECMs) can play a role. OECMs are sites that are managed in ways that achieve positive and sustained outcomes for biodiversity without being primarily managed for conservation (MacKinnon et al. 2020). While it will be necessary to identify additional areas to meet Target 3, the process of identifying OECMs, to some extent, will involve recognising and documenting existing conservation areas. This includes areas that were established by communities many years before and management was based on locally-developed rules and practices (Gurney et al. 2021). Such recognition will put these local people at the centre of governance of their areas, potentially bringing people together to work towards achieving a common goal (Obura et al. 2021).

These conservation programmes that integrate local communities and their traditional knowledge with scientific management are thought to be among the most effective (Berkes 2009). This is especially relevant in Africa, where some communities have thrived over the millennia by adhering to beliefs that connect humans physically and spiritually to nature (Wilson & Primack 2019). In accordance

with these beliefs, the collection of animal and plant products are stipulated by customary law systems. These laws also initiated Africa's first protected areas, for example, areas of spiritual significance or those where kings and traditional chiefs were the only people given hunting rights (Hurt & Ravn 2000). Thus, incorporating traditional beliefs into programmes designed to expand PA and OECM networks is likely to garner more support and produce better conservation and development outcomes (Alves-Pinto et al. 2021). Among other advantages, such well-designed community-led conservation projects will also have a key role in climate change mitigation and adaptation and can support local livelihoods through ecotourism and sustainable use of natural resources (Watson et al. 2014; Cumming & Allen 2017).

## 1.3 Identifying important areas for biodiversity

Another component of the Global Biodiversity Framework Target 3 is that PAs and OECMs should include "areas of particular importance for biodiversity (CBD 2022). This is critical because existing protected area networks fail to represent many important species and ecosystems (Butchart et al. 2015), often because they are placed on land with low agricultural value and so over-represent montane, desert and tundra biomes (Hoekstra et al. 2005). In response, conservation scientists and practitioners have developed different ways to identify important areas for biodiversity, and the three most widely known are Important Plant Areas (IPAs), Key Biodiversity Areas (KBAs) and systematic conservation planning.

### 1.3.1 Important Plant Areas

It has been estimated that one fifth of plant species are threatened with extinction, many of which are not properly described, or their description provided insufficient data (Antonelli et al. 2020). However, it is thought that plant conservation has not been adequately targeted and is poorly funded (Corlett 2016). This makes plant conservation an enormous global issue, which could be largely addressed through establishment of PAs systems that account for plant data and are managed effectively for these species (Havens et al. 2014). In light of this, Plantlife International developed the IPA approach in the early 2000s, which was then updated over a decade later (Darbyshire et al. 2017). The IPA approach was inspired by the Important Bird Area methodology (IBA), which was successfully used to establish important sites for birds (BirdLife International 2014). The IPA methodology is designed to identify "the most important places in the world for wild plant and fungal diversity that can be protected and managed as specific sites" (Marignani et al. 2014; Darbyshire et al. 2017). The IPA methodology consists of three higher-level quantitative criteria based on (A) the presence of threatened species, (B) exceptional botanical richness and (C) threatened habitats. Within these criteria are eight sub-criteria, with associated thresholds, focusing on biological elements relevant for plants and fungi. This approach can be applied nationally to determine "the presence of

threatened species, exceptional botanical richness and threatened habitats" (Darbyshire et al. 2017), thereby aligning it with the internationally recognised measures of dealing with threats to biodiversity, such as the IUCN Red List of Threatened Species.

The IPA methodology was designed to be a clear and transparent way for national stakeholders to use the best available data to identify a national network of IPAs. The Plantlife International database includes over 2,000 sites that have been identified (Plantlife International 2018) but other countries, including Mozambique and South Africa (Darbyshire et al. 2023), have applied the criteria so this figure is an under-estimate. By identifying these networks of IPAs, national stakeholders are better equipped to inform conservation planning initiatives so that areas important for plants are not missed. Moreover, the presence of IPAs can inform decisions on associated social and economic development, in agreement with the mitigation hierarchy (Chan et al. 2020).

### 1.3.2 Key Biodiversity Areas

The KBA methodology was designed to identify sites that provide a significant contribution to the global persistence of biodiversity (IUCN 2016). In a similar manner to the IUCN Red List, the KBA methodology consists of five higher-level quantitative threshold criteria ranging from A to E, together with11 sub-criteria and associated thresholds, focusing solely on ecological elements such as species and ecosystems. These higher-level criteria are: (A) threatened biodiversity; (B) geographically restricted biodiversity; (C) ecological integrity; (D) biological processes; and (E) irreplaceability through quantitative analysis (IUCN 2016). Criterion A is based on the presence of a significant proportion of the global population of a species or ecosystem type being assessed as threatened globally under the IUCN Red List (either Critically Endangered, Endangered or Vulnerable). Criterion B identifies areas containing a globally significant proportion of range-restricted individual or co-occurring species, assemblages of species, and ecosystem types. Criterion C identifies areas holding the largest proportion of intact ecological communities across the world. Criterion D establishes sites having the presence of a significant proportion of the global population of a species during one or more life history stages or processes, or during periods of environmental stress, or providing a breeding site for the greatest number of the global population size. Criterion E identifies sites with the highest irreplaceability for the global persistence of biodiversity based on appropriate quantitative analysis of irreplaceability (IUCN 2016).

The KBA methodology provides an umbrella approach used in a number of initiatives for identifying sites of global importance across different taxonomic groups and ecosystems (KBA Standards and Appeals Committee 2022). It was developed with strong stakeholder engagement to ensure that relevant aspects of the different existing criteria-threshold approaches were accounted for, making the

KBA approach a rigorous and repeatable methodology. Worldwide, over 16,000 sites have been identified (KBA Partnership 2022).

### 1.3.3 IPAs, KBAs and GBF Target 3

Identifying IPAs and KBAs can play a key role in biodiversity conservation because it provides baseline information on biodiversity and its distribution (IUCN 2016). The first step in these approaches involves identifying 'trigger' species and ecosystem types, namely those that meet the criteria based on their global distribution, threat status or characteristics. The second step is to assess the trigger elements at the national level, identifying sites that contain globally or regionally important populations or ranges of these species and ecosystem types (Lim et al. 2023). The information from IPAs and KBAs can then form the basis for monitoring and vigilance programmes to ensure the important biodiversity they contain persists (Smith et al. 2019). In addition, IPA and KBA data can be taken into consideration when designing development plans and environmental impact assessments. For conservation, IPA and KBA data can be particularly relevant for informing initiatives aimed meeting the Global Biodiversity Framework Target 3. However, having IPA or KBA status does not imply how a site should be managed or governed, with research showing that designating them as OECMs is often more appropriate than gazetting them as PAs (Donald et al. 2019). Moreover, while these sites are likely to be priorities for conservation management, this will depend on local threats and funding levels. Thus, these important sites for biodiversity should ideally be included in landscape or national-level prioritisation exercises, which include data on a wide range of biodiversity, threat and socio-economic and social data (Smith et al. 2019).

### 1.3.4 Systematic conservation planning

One way to produce these more holistic plans for guiding conservation action is to adopt a systematic conservation planning approach, which is a transparent, defensible and inclusive process for identifying priority areas for conservation (Margules & Pressey 2000). It generally involves working with stakeholders to develop a shared vision, translating these into a set of objectives or targets and then using spatial prioritisation software to select a site or sites that meet these objectives (Groves & Game 2016). The spatial prioritisation process uses complementarity-based algorithms to identify sites that work best in combination, meeting targets in a cost-effective way (Kaim et al. 2017) and balancing biodiversity needs with other competing land-uses (Veach et al. 2017). In the past systematic conservation planning has been criticised as an academic pursuit that rarely led to implementation (Knight et al. 2008). However, more recent research has shown that this approach has been used to design marine, terrestrial and freshwater ecological networks in many countries (Sinclair et al. 2018). A key part of this success has been ensuring that the planning process involves the

relevant implementation agencies and accounts for the local context (Smith et al. 2009), providing broader benefits that include improved fundraising and institutional knowledge (Bottrill et al. 2012).

## 1.4 Conservation in Mozambique

### 1.4.1 History and current trends

The history of conservation areas in Mozambique before the colonial era is not well documented but is likely to have consisted of a number of sacred sites, as well as sites assigned for hunting or protecting other natural resources (Soto 2008). The history of formal actions aimed at safeguarding biodiversity in Mozambique began in the colonial era, with some regulations dating from the late 19$^{th}$ Century (Hedberg & Hedberg 1968). During those early days, two different initiatives existed, the first focusing on setting aside areas for trophy hunting, and the second focusing on plants of economic value, mainly for the sustainable harvesting of timber. However, there was no formally recognised organisation overseeing these initiatives (Hedberg & Hedberg 1968). This changed in the 1940s, when the Portuguese administration established most of the protected areas (PAs) that still exist today (Gomes e Sousa 1968). The PAs were set up under six categories: National Parks, Integrated Natural Reserves, Partial Reserves, Special Reserves, Forest Reserves, and Zones under the regime of Special Vigilance (Lindsey & Bento 2012). However, independence in 1975 was followed by a civil way that engulfed the country from 1976 to 1992, producing devastating consequences across different sectors including biodiversity conservation (Monadjem et al. 2010).

The impact of the civil war on conservation was exacerbated by an increase in the human population along the coast, resulting in a huge transformation of the landscape for settlement, farming, and infrastructure. Moreover, during the war most PAs had very limited or no management (Soto 2008) and were often used as refuges by local communities fleeing their homes to stay alive. In addition, some areas under conservation were occupied by governmental military bodies or by RENAMO (*Resistência Nacional de Moçambique*) fighters. For that reason, illegal hunting and uncontrolled logging was widespread (Dunham et al. 2010). This only changed in 1992, when the Mozambican Government signed a peace agreement with RENAMO. Since that time, the Mozambican Government has focused on rehabilitating and establishing a network of PAs that now cover 24% of the country (MITADER 2019). The Government manages these PAs, which include National Parks, National Reserves, Areas for Environmental Protection, through the National Administration of Protected Areas that was established in 2011 under the Ministry of Land and Environment. There are also a number of other conservation areas that are managed by different stakeholders, including local communities, government agencies and the private sector. These include Forest Reserves, controlled Hunting Blocks, Game Farms and Community-based conservation areas (MITADER 2019). The

forest reserves are managed by the Forestry Department, under the Ministry of Land and Environment but have low enforcement and limited resources.

Recently, the national government introduced an Integrated Landscape Management approach. This consists of bridging initiatives aimed at improving rural development in areas such as financing, infrastructure and land tenure security. These initiatives are expected to help achieve the management of natural resources in a sustainable way. In addition, it is expected that they will conserve biodiversity and restore degraded habitats, establishing conservation areas to be managed by communities and the private sector (Pringle 2017). These schemes are being implemented in extensive areas, incorporating different land usage associated with human activities including agriculture, forestry, conservation and urban development. Apart from implementing local measures to strengthen conservation actions after the civil war, Mozambique committed itself in 1992 to a number of international conventions, including the Convention on Biological Diversity in 1994 (Resolution 2/94) and then its supplementary agreement, the Nagoya Protocol in 2014. In response to these conservation instruments, Mozambique has implemented reforms to some areas of the national legal framework. An example is the current Conservation Areas Law, in place since 2014 (Law 16/2014 of 20$^{th}$ June), being the first legal instrument in the country that refers to "no net loss of biodiversity" and calls for the provision of compensation for negative impacts resulting from activities in PAs and their buffer zones.

### 1.4.2 Funding for conservation in Mozambique

Despite all the effort towards improving the status of PAs, conservation success in Mozambique is hindered by several factors, including limited funding coupled with lack of expertise and staff. For example, according to a 2018 World Bank report, between 2008 and 2014, the Mozambican Government only allocated between 20 to 26 million US dollars to manage the PAs network, an average of US$34/km$^2$. This places Mozambique far behind other countries in the region as Zimbabwe, Kenya and South Africa allocate US$187/km$^2$, US$2,500/ km$^2$ and US$2720/ km$^2$ respectively. The argument that country-scale rate of biodiversity decline depends on conservation spending levels (Waldron et al. 2017) has several implications for Mozambique. In order to address this issue, the Mozambican Government has been engaging with several stakeholders at national and international level to arrange for the co-management of the PAs. The resulting collaboration has led to 81% of PA funding coming from international donors and 9% from incomes generated by the PAs themselves. However, these funds are not equally dispersed, with over 50% of the total being allocated to two PAs, namely Gorongosa and Limpopo National Parks. The outstanding funds are allocated, in limited amounts, to some of the remaining PAs, leaving funding for Forest Reserves and other conservation areas at a low level.

To help overcome the difficulties associated with ensuring financial sustainability, in 2011 the Mozambican Government supported the establishment of a Foundation for Biodiversity Conservation (BIOFUND). This is a non-profit organisation aimed at helping conservation throughout the country by mobilising, managing, and allocating funds for conservation action. BIOFUND works closely with the National Administration of Protected Areas that oversees the work of entities in charge of co-managed PAs (e.g. the Carr Foundation in Gorongosa National Park, the Wildlife Conservation Society in Niassa National Reserve and Peace Park Foundation co-managing PAs in the southern region of Mozambique). In addition, the Government established the National Sustainable Development Fund in 2016. The main aim of this fund is to raise financial resources in order to address environmental problems and promote development in rural areas. The National Sustainable Development Fund oversees initiatives funded by donors, which are now believed to be worth in excess of US$400 million.

## 1.5 Conservation in Maputaland

### 1.5.1 Maputaland biodiversity context

Maputaland is a Centre of Endemism (van Wyk 1996) comprising an area of 19,180 km$^2$ covering parts of Eswatini, Mozambique and South Africa. Maputaland is the northern part of the Maputaland-Pondoland-Albany global biodiversity hotspot (Mittermeier et al. 2004) and is thought to be the second richest biodiversity hotspot in Southern Africa after the Cape Floristic Region (Bandeira *et al*. 2014). The region consists of a network of ecosystems, comprising a wide range of vegetation types thought to have originated during the Pleistocene (du Randt 2018). It is situated below the southern edge of the tropics in a transitional area of Africa where many tropical plant and animal species reach the southernmost limit of their range (Bandeira et al. 2014). This Centre of Endemism is said to contain over 200 endemic plant species or infraspecific taxa, with the total number of plant species found within the Centre believed to be at least 2500 (van Wyk 1996). It is also important for a wide range of animal species, containing for example 60% of South Africa's bird species and 25% of South Africa and Eswatini's butterfly species (du Randt 2018). The importance of Maputaland is recognised globally, as it holds the UNESCO iSimangaliso Wetland Park World Heritage Site, five RAMSAR sites, ten important bird areas (Smith et al. 2008) and within the Mozambique section, three IPAs (Darbyshire et al. 2023) and three KBAs (WCS et al. 2021).

The Maputaland Centre of Endemism is also experiencing a high number of threats (Nel et al. 2017). The most important of these are habitat degradation resulting from the expansion of commercial and small-scale agriculture due to the pressure of human population growth (Jewitt 2018). The majority of the people living in this area rely on subsistence farming for sustaining their livelihoods (Martins &

Shackleton 2017). However, the yield from these household farming systems is inadequate because of the nutrient-poor soils of the Maputaland (Martins & Shackleton 2018). An immediate alternative source of income is based on charcoal production from the native woody species, which is found particularly in Mozambique to supply the nearby cities of Matola and Maputo (Tokura et al. 2020). With the ongoing expansion of urban and industrial areas, coupled with increased access to Maputaland along the recently completed bridge over the Maputo Bay and the tarred road through to Ponta d'Ouro in Mozambique, there will be further loss of natural vegetation and an increase in habitat destruction. Therefore, there is a pressing need for the relevant stakeholders to focus on planning for landscape management and strengthening of the existing network of conservation areas to ensure persistence of biodiversity and sustainable livelihoods (Smith et al. 2008).

### 1.5.2 Systematic conservation planning in Maputaland

Systematic conservation planning in Maputaland was pioneered early in the 2000s (Smith et al. 2006) focusing solely on the section that falls in KwaZulu-Natal, South Africa. This study produced the first conservation priority map which was then used to guide development by Ezemvelo KwaZulu-Natal Wildlife, the provincial government agency responsible for conservation. Two years later, an initiative involving Mozambique, Eswatini and South Africa, applied systematic conservation planning to the whole of Maputaland for the first time (Smith et al. 2008). The results of this study informed the establishment of a corridor between Maputo Special Reserve in Mozambique and Tembe Elephant Park in South Africa, adding 24,770 ha of cross boundary conservation area into the existing network of Conservation Areas (Smith et al. 2008). However, these studies were primarily based on vegetation type and animal species data, so many important plant species were not considered in the prioritisation process.

### 1.5.3 Protected areas and the Transfrontier Conservation Areas in Maputaland

Within the whole Maputaland region a network of PAs is already in place (Smith et al. 2008). The first PAs were established in the late 1800s, particularly in the section of Maputaland falling within KwaZulu-Natal in South Africa. South Africa was the first country in Africa to implement formal biodiversity protection to minimize the anthropogenic impacts posed by the high number of early colonial settlements (Wilson & Primack 2019). As a result, a dozen PAs exist in the South African section alone (Jewitt et al. 2015). By contrast, the number of PAs established within the portion of Maputaland found in Mozambique and Eswatini is about a third of those in KwaZulu-Natal (Nhancale & Smith 2011). These PAs were established in the mid-20$^{th}$ Century and include the Licuati Forest Reserve and the Maputo National Park, both in Mozambique. With respect to the Eswatini section, it was only in 1980 that the first protected area, Mlawula Nature Reserve, was officially proclaimed. The Mlawula Nature Reserve is situated in the Lubombo Mountains and forms part of the Lubombo

Conservancy, together with communally-managed Shewula Nature Reserve and privately-managed Mbuluzi Game Reserve (Roques et al. 2003).

Although the network of PAs has increased considerably in the whole of Maputaland, it still under-represents many important ecosystems and species (Smith et al. 2008). Plants are particularly poorly represented, even though Maputaland is known to contain unique sand forest ecosystems that support many endemic or near-endemic plant species where new species are still being discovered (Boon & Wyk 2023). This is exacerbated by the lack of management of PAs set solely for plant conservation, particularly in the Licuati Forest Reserve in Mozambique (Tokura et al. 2020).

Maputaland also contains the Lubombo Transfrontier Conservation Area (TFCA). These transfrontier conservation areas, initially known as Peace Parks (Kansky et al. 2021), straddle political boundaries between two or more countries, and run through landscape where nature and culture are managed jointly (Ntuli et al. 2019). TFCAs help protect extensive conservation areas and ecological corridors by encouraging knowledge exchange between countries, supporting cooperative efforts in conservation and enforcement, and promoting the inclusion of conservation considerations into land-use planning. Currently, the TFCA in the Maputaland region focuses on three areas, linking the mountains in the west to the coastal ecosystems in the east (Peace Parks Foundation 2017; Ezemvelo KZN Wildlife 2018). With the establishment of these TFCAs, it is envisaged that they will provide socio-economic advantages through programs that facilitate strengthening of responsible eco-tourism, sustainable use of natural resources and increased investment possibilities for investors and donors (Wilson & Primack 2019).

## 1.6 Thesis outline

Inspired by the fact that signatories to the CBD, including Mozambique, have recently committed to ambitious time-bound conservation targets, here I address research gaps in the field of spatial conservation prioritisation to help inform transfrontier- and national- level conservation planning. The following main chapters of this thesis are:

**Chapter 2** uses remote sensing data segmentation as a tool to delineate ecological boundaries for IPAs and KBAs in Maputaland, where working with experts and stakeholders helped to identify and map important species for analysis.

**Chapter 3** uses the systematic conservation planning approach to identify the best places to establish new OECMs in Maputaland, based on meeting conservation targets, minimising impacts on farming and maximising opportunities for community-based ecotourism.

**Chapter 4** applies the Key Biodiversity Area methodology to Mozambique, identifying sites for a wide range of taxonomic groups in the marine, terrestrial and freshwater realm. Taxon experts and stakeholders' contribution informed the identification of important species for the analysis and assisted the KBA delineation process.

**Chapter 5** employs systematic conservation planning methodology to identify where to establish new OECMs in Mozambique, based on meeting conservation targets and minimising impacts on agriculture, and carrying out sensitivity analyses to identify the most suitable analyses parameters.

**Chapter 6** gives an overview of the results of this thesis. Here I discuss the thesis findings, outline the contributions to this field of knowledge and present future research ideas, together with recommendations for policy and practitioners.

# 2 Developing a new boundary delineation method for important sites for biodiversity


## Abstract

Site-scale conservation initiatives are pivotal in our efforts to halt the current trend of biodiversity loss. As stipulated in the Kunming-Montreal Global Biodiversity Framework, there needs to be strategic expansion of the network of protected areas (PAs) and other effective area-based conservation measures (OECMs) which will contribute to the expected conservation area coverage target of 30% of the globe by 2030. Two widely used approaches entail identifying Key Biodiversity Areas (KBAs) and Important Plant Areas (IPAs), but these remain untested in terms of producing consistent and reproduceable boundary delineation. In this study we tested a novel application of segmentation methodology for the drawing of boundary delineation focusing on the Maputaland Centre of Endemism in Southern Africa. We applied the methodology to three different scenarios which firstly do not account for the existing management units (MUs) in the form of PAs and/or OECMs, secondly account for MUs and thirdly account for MUs and political borders. Our findings show that segmentation is an effective methodology in drawing boundary delineation in a repeatable manner and it focuses on the ecological requirements of the KBA and IPA trigger elements.

## 2.1 Introduction

Parties to the Convention on Biological Diversity (CBD) have committed under Target 3 of the Kunming-Montreal Global Biodiversity Framework to safeguard at least 30% of the marine and terrestrial realm by 2030, particularly areas that are important for biodiversity and ecosystem services (CBD 2022). This provides an opportunity for the 196 signatory countries to improve their protected area (PA) networks, many of which over-represent ecosystems with low economic value and so fail to conserve important species, habitats, and ecological processes (Joppa & Pfaff 2009; Barber et al. 2012; Butchart et al. 2015; Delso et al. 2021). As a first step in filling these conservation gaps, several conservation groups have developed approaches to identify important sites for biodiversity (Rogers et al. 2010; Pimm et al. 2014). Two of the most widely used are the Key Biodiversity Areas (KBAs) approach, developed by the International Union for Conservation of Nature (IUCN 2016), and the Important Plant Areas (IPAs) approach, developed by Plantlife International (Darbyshire et al. 2017). The KBA methodology identifies important sites for maintaining global biodiversity (IUCN 2016) and over 16,000 sites have been identified (KBA Partnership 2022). The IPA methodology identifies important sites that can contribute to the protection of wild plant and fungal diversity worldwide (Darbyshire et al. 2017) and over 2,000 sites have been identified (Plantlife International 2018).

The KBA and IPA approaches, which were developed by a large number of experts, are based on methodologies that are repeatable and applicable in any location across a range of taxonomic groups globally. Both methodologies are based on a criteria-threshold approach, where important sites contain at least a threshold amount of one or more species, habitat or ecosystem type meeting one or more of the criteria. The KBA approach has five higher-level criteria: (A) threatened biodiversity; (B) geographically restricted biodiversity; (C) ecological integrity; (D) biological processes; and (E) irreplaceability through quantitative analysis (IUCN 2016). Within these criteria are 11 sub-criteria focusing on specific biological elements. The IPA methodology consists of three higher-level quantitative criteria based on (A) the presence of threatened species, (B) exceptional botanical richness and (C) threatened habitats. Within these criteria are eight sub-criteria focusing on biological elements relevant for plants and fungi (Darbyshire et al. 2017). The threshold amount of a species, habitat or ecosystem type needed to trigger a KBA or IPA then depends on which sub-criteria is being applied. This methodology also means that the KBA or IPA assessment allows for identification on a site-by-site basis, without the need for data on other sites within a planning region. For species, this generally involves using population estimates, point locality data or distribution maps to identify sites that contain enough of at least one species to meet any of the relevant thresholds.

Identifying KBAs and IPAs provides essential baseline data on globally and regionally important biodiversity, which can be used for monitoring and informing processes to create new PAs and Other Effective Area-Based Conservation Measures (OECMs). This is why both methodologies have a strong focus on repeatability and comparability to inform decision making at the national and international level (Darbyshire et al. 2017; KBA Standards and Appeals Committee 2022). There is one aspect, however, where the process of identifying KBAs and IPAs is much more open to interpretation. This relates to how these sites are delineated, as previous experience has shown that KBA delineation can have a significant impact on replicability (Eken et al. 2004), and this is also likely for IPAs. Here we present a new approach to reduce this problem, based on combining remote sensing and species distribution data to delineate important sites, and apply it to a centre of endemism in southern Africa.

The problem of site delineation repeatability stems from two issues. First is that there is no one agreed approach for delineating the key habitat patches for the trigger species identified in the KBA and IPA assessment, which can lead to different experts interpreting the data in very different ways. The second issue stems from both the KBA and IPA methodologies requiring that new sites must consider the existing management units (Darbyshire et al. 2017; KBA Standards and Appeals Committee 2022). These management units can be PAs, indigenous territories, private lands, concessions, administrative boundaries, or any other defined area with shared governance (IUCN 2016). This ensures that each KBA and IPA is managed by a coherent management group, and most commonly involves matching the site to an existing PA boundary. The problem is that these management unit boundaries rarely match the distribution of the trigger species. This can lead to the reporting of KBA and IPA areas which inflates figures on extent of land or sea of high biodiversity value, commonly when a large PA is given KBA or IPA status even though it only contains a small patch of trigger species habitat (Knight et al. 2007). In addition, for habitat patches that overlap with PAs, it is rare for boundaries to be modified to include areas outside of the PA. Thus, areas outside of the management units are often ignored, and this is a particular problem in trans-frontier planning, where habitat in neighbouring countries is excluded from the assessment process.

Of these two problems, the first is potentially easier to address, as recent work has shown that a widely used approach in remote sensing is also relevant for identifying important sites for biodiversity (Lötter 2015). This involves classifying satellite images into a set of homogeneous segments based on their spectral similarity, which is known to reflect ecological similarity as well (Kettig & Landgrebe 1976; Blaschke et al. 2000; Bock et al. 2005; Blaschke 2010). Once these habitat segments have been identified, it is then possible to identify which contain enough of each trigger species to meet the KBA and IPA population size thresholds. However, this process still involves setting a number of parameters, so it is important to test the approach before recommending its widespread use. To do

this, we focused on the Maputaland Centre of Endemism, a region straddling three countries in southern Africa that contains many threatened and endemic species. We concentrated particularly on amphibian, freshwater fish, invertebrate, plant and reptile species. This was partly because of data availability, however we excluded Maputaland's birds and mammals because they have much broader ranges, and so the process of identifying KBAs for these species would be based on identifying the existing management units that contain the largest number of individuals or percentage of range (WCS et al. 2021).

We tackled the first problem of finding a repeatable approach to delineate the key habitat patches by using segmentation, which is a processing technique that divides an image into multiple partitions, where each partition consists of a set of pixels used to identify regions of similar landcover (Raut et al. 2009). This allowed us to investigate the second issue of how accounting for different management units influences the final set of KBAs and IPAs identified. Another advantage of focusing on Maputaland is that it covers parts of Eswatini, Mozambique and South Africa and this means we can investigate the impact of identifying KBAs and IPAs based on transnational and national assessments. Here we investigate the impacts of using different parameters and scenarios to identify priority sites for biodiversity in Maputaland by: (i) identifying the threatened and geographically-restricted amphibian, freshwater fish, invertebrate, plant and reptile species found in Maputaland and using the available point locality data to model their distributions; (ii) dividing the region into a series of habitat patches based on remote sensing segmentation techniques and identifying which of these meet the KBA and IPA thresholds, and (iii) comparing the number and extent of sites identified based on three different national and transnational scenarios that account for existing management units.

## 2.2 Methods

### 2.2.1 Defining the Study Area

The Maputaland Centre of Endemism (van Wyk 1996), hereafter referred to as Maputaland, is a region of 19,180 $km^2$ in eastern Eswatini, southern Mozambique and north-eastern South Africa (Figure 2-1). Maputaland consists of the Lubombo Mountains in the west, with a maximum elevation of 600 m, and a coastal plain that has a maximum altitude of c. 150 m (van Wyk & Smith 2001). It forms the southernmost extent of coastal lowlands of tropical eastern Africa that stretch north to Somalia and have experienced little climate fluctuation throughout the Pleistocene Era, leading to high levels of speciation and endemism. This explains why Maputaland is home to over 2,500 native plant species, of which over 200 are endemic (van Wyk 1996; van Wyk & Smith 2001), and why it forms part of the Maputaland-Pondoland-Albany global biodiversity hotspot (Smith et al. 2008).

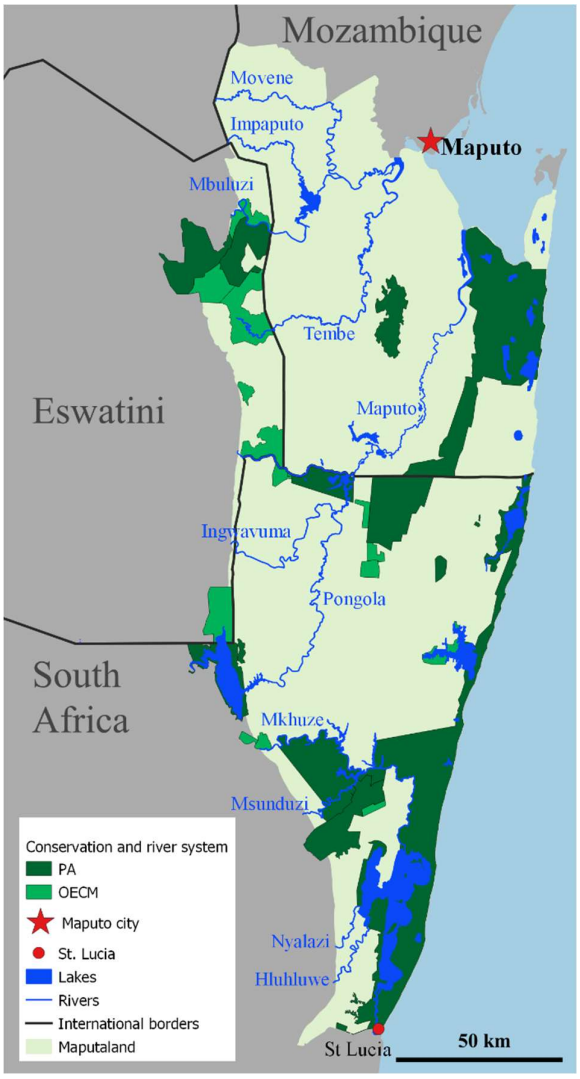



***Figure 2- 1: Maputaland's network of protected areas and OECMs (other effective area-based conservation measures), rivers and lakes. Maputo city is the capital city of Mozambique.***

### 2.2.2 Selecting the trigger species

The KBA and IPA approaches involve identifying taxa that meet their associated criteria, which are then labelled as "trigger elements" or "trigger species". Our analysis was only based on threatened or geographically restricted species (Table S2-1), because we lacked data on the other assessment criteria. To assess which species met the KBA threatened species criteria, we collected information on their global conservation status. This was based on the IUCN Red List of Threatened Species (IUCN 2022), and we assessed the species against the Red List threat criteria, identifying those that are Vulnerable, Endangered or Critically Endangered. Producing the list of KBA geographically restricted species also used data from the IUCN Red List of Threatened Species but some species have not been assessed, and for these we had to rely on other sources for species ranges. For animals we used expert opinion and a number of species-specific articles. For plants we used the list of Maputaland endemics

and near-endemics identified by previous authors (van Wyk 1996; van Wyk & Smith 2001; Matimele 2016; Darbyshire et al. 2019). The IPA approach covers species, subspecies and varieties and, in addition to those of global importance, it also identifies these taxa against sub-criteria based on regional importance (Table S2-1). In most cases, these latter sub-criteria were not relevant because most of the potential trigger species are endemic, so anything that is nationally important is also globally important. However, we used the Red List of South African Plants (SANBI 2022) to identify regionally threatened species when assessing sub-criterion A(ii).

### 2.2.3 Collating the trigger species distribution data

To produce distribution maps of each trigger species we first needed to collate the relevant point location data. For the amphibians, freshwater fish, invertebrates and reptiles we collected data from relevant museums and government organisations, including Ezemvelo KZN Wildlife, the South African Institute for Aquatic Biodiversity and individual experts. For the plants, we mostly used data from (Matimele 2016) and (Darbyshire et al. 2019), as well as information from the National Herbarium of Mozambique, the Eswatini plant database, the Eduardo Mondlane University Herbarium, the Buffelskloof Nature Reserve Herbarium, the Pretoria National Herbarium, the Natal Herbarium, and the Royal Botanic Gardens, Kew. Some herbarium records from the 19$^{th}$ and early 20$^{th}$ Century did not include locality data but we were still able to geo-reference 325 of these specimens based on the locations listed on their labels. We accessed data from the Global Biodiversity Information Facility but did not include it in the analysis, as there were several inconsistences in the georeferenced points and records were mostly duplicates of those from our herbaria and museum sources.

We were also able to conduct fieldwork to supplement the sparse data on the trigger plant species. Between November 2019 and January 2020, the field surveys took place in three specific areas of Maputaland region (Figure 2-4), namely: 1) the Lebombo Mountains, 2) the central corridor of the region and 3) the coastal belt. The Lebombo Mountains covers the area from Pambane (-25.817452°/ 31.962415°) in Namaacha district, Mozambique, entering Eswatini at Goba Frontier and then through South Africa to Phinda Game Reserve (-27.813487°/ 32.338381°). Sampling was conducted in several places along the Lebombo Mountains, covering at least a third of the region. The central corridor was sampled from the Licuati Forest Reserve, Mozambique (-26.381213°/ 32.486192°) through Tembe National Park to Mphomphomeni area (-27.094801°/ 32.393711°), South Africa. The coastal belt ranged from Maputo National Park (-26.390008°/ 32.865657°), Mozambique, entering South Africa through Sodwana (-27.549181°/ 32.666583°) to as far south as Cape St. Lucia Lighthouse (-28.515216°/ 32.399497°). The total sampling undertaken covered a quarter of the Maputaland region across the three countries. The sampling approach followed standard methods according to the terrain,

including walking and driving through areas to gather material from a full range of accessible habitats (Timberlake et al. 2009). Sampling areas were selected by targeting the historical distribution of endemic and near-endemic plant species shown in the records held at the National Herbarium of Mozambique, The Eduardo Mondlane University Herbarium in Mozambique, Buffersкloof Nature Reserve Herbarium, the Pretoria National Herbarium, the Moss Herbarium of the University of Witswatersrand in South Africa, the Eswatini Herbarium and the Royal Botanic Gardens, Kew in the UK. In addition, the field survey aimed at recording new localities of the distributions of the endemic and near-endemic species. The specimens were collected, pressed and dried and their location recorded using a Geographical Positioning System (GPS) device. They were then taken to the National Herbarium of Mozambique for identification using taxonomic keys, and then lodged after confirmation by comparing the newly identified material to older material of the same species. Through this fieldwork a total of 121 plant species were recorded, confirming their presence in previously surveyed sites but also identifying eight new locations for 22 species.

### 2.2.4 Producing the species distribution maps

We used the Maximum Entropy models (MaxEnt) to model the extent of suitable habitat for each trigger species (Phillips et al. 2006). The underlying algorithm predicts environmental suitability for a species and the likelihood of its occurrence in a suitable habitat as a function of given environmental variables (Phillips et al. 2006; Wulff et al. 2013). This model is designed for presence-only data (Elith et al. 2011; Merow et al. 2013; Guillera-Arroita et al. 2014; Grimmett et al. 2020) and is therefore particularly suited to poorly studied tropical regions where species absence data is not available (Phillips et al. 2006).

Running the MaxEnt algorithm involves the inclusion of environmental variables that are thought to be directly or indirectly related to the distribution of the species (Wulff et al. 2013). We worked with variables describing the terrain, climate and soil characteristics, as all of these are hypothesised to be important for shaping the distribution of the different plant communities in Maputaland (van Wyk 1996; Elith et al. 2011). The terrain data we used include a 90 m resolution elevation, slope, solar irradiation and hill shading data derived from the SRTM digital elevation model v4.1 (Javis et al. 2008). We used 30 seconds (1km) resolution data on mean annual temperature, annual precipitation, annual temperature range, temperature annual and precipitation range, temperature of the coldest month, temperature of the warmest month, precipitation during the wet quarter and precipitation BIOCLIM data from the WorldClim database (Hijmans et al. 2005). We also used 250 m resolution data on calcium, phosphorus, phosphorus nutrients, potassium, potassium nutrients, sand and total nitrogen nutrients from the global soil grid (Poggio et al. 2021).

As the inclusion of correlated variables can lead to over-fitted models with limited use (Kriticos et al. 2012), we evaluated the level of correlation between the variables using the package "usdm"(Naimi et al. 2014) in R version 4.1.1. This was done by calculating the variance inflation factor (VIF) (Soultan et al. 2019), which evaluates the extent to which the variance increases in response to correlated predictors (Akinwande et al. 2015). Thereafter, using the function "vifstep", we excluded variables with VIF greater than 5 (Akinwande et al. 2015). We ran MaxEnt using the R package "dismo" (Hijmans & Elith 2011), building models for each species based on the 13 environmental variables refined by vifstep (Table 2-1). The data was split into two sets, with 75% used for training and 25% for testing. We used k-fold cross validation for species with limited number of records (Anguita et al. 2012). In addition, we set the number of background points to be 10,000 with a buffer of 100 km from the species records and 5,000 for the number of iterations to ensure sufficient time for convergence (Yackulic et al. 2013).

We used the area under the receiver operating characteristic curve (AUC) to assess the performance of each model (Phillips et al. 2006). Despite some criticism of the use of AUC to test performance, several studies have shown that this parameter is effective in determining how well the models perform with presence-only data (Phillips et al. 2006; Wulff et al. 2013; van Proosdij et al. 2016; Katz & Zellmer 2018). By running MaxEnt with presence-only data, an important assumption is that sampling is random, and that detection probability is constant throughout the sites of a defined landscape (Yackulic et al. 2013). As AUC ranges from 0 to 1, models with high values (> 0.60) represent the best performance achieved for each species (Elith et al. 2011).

***Table 2-1: Environmental variables with variance inflation factor (VIF) < 5 that were used in the MaxEnt algorithm for predicting species distributions in Maputaland.***

| Variable | Description | VIF |
|---|---|---|
| Bio_3 | Isothermality | 1.89 |
| Bio_7 | Temperature annual range | 3.49 |
| Bio_13 | Precipitation of wettest month | 2.44 |
| Bio_15 | Precipitation seasonality | 3.06 |
| Calcium | Calcium | 4.16 |
| Depth to bedrock | Depth to bedrock | 1.42 |
| Elevation | Elevation | 4.37 |
| Phosphorus | Phosphorus | 1.56 |
| Phosphorus | Phosphorus nutrients | 2.06 |
| Potassium | Potassium | 1.69 |
| Potassium nutrients | Potassium nutrients | 3.01 |
| Sand | Sand | 1.91 |
| Total N nutrients | Total Nitrogen nutrients | 1.71 |

To build the models, we used the 13 environmental variables (Table 2-1) across the different taxonomic groups. To represent habitat suitability, MaxEnt outputs contain values ranging from 0 to 1. We used "spec-sens" (sum of sensitivity and specificity) in R to set threshold values to classify every model into presence and absence (suitable and unsuitable habitat) map.

We found that the MaxEnt model performance for species with smaller datasets was not adequate, therefore, for species with fewer than 10 records, we used only these point data for their distribution maps. For the other species, MaxEnt produced a 1 km resolution species distribution model as a layer presenting potentially suitable habitat based on terrain, climate and soil. Thus, for both the point data and SDM maps, we then had to exclude potential habitat covered by unsuitable landcover types. We did this by identifying which of the 45 natural and anthropogenic landcover types used in the Maputaland landcover map were unsuitable for each species, based on the literature and expert opinion. We then used QGIS to combine the SDMs with data from the Maputaland landcover map, thus excluding the unsuitable habitat. Finally, we removed polygons smaller than 4 $km^2$ (in accordance with the size of a square grid of an Area of Occupancy (AOO) under the IUCN Red List of threatened species (IUCN Standards and Petitions Committee 2022), using the Maputaland landcover map and the "sf" package (Pebesma 2018) in R.

### 2.2.5 Identifying the trigger species localities

Identifying KBAs and IPAs for species involves assessing whether a site contains a significant proportion of their global population, based on a set of criteria and associated thresholds (Table S2-1). The population size assessment can be based on number of mature individuals, area of occupancy, extent of suitable habitat, range and number of localities. These parameters can be estimated at both local and global scale but require a consistent dataset of biological elements. Our data et was sparse and mostly consisted of GPS records, so to produce a consistent, comparable and repeatable analysis, we based our assessment on the number of localities in which each species was found. In defining a locality we followed IUCN guidelines to infer the extent of a discrete population in accordance with the degree of habitat fragmentation and the known dispersal capabilities of the species. (IUCN 2022). We used the minimum spanning tree approach (MST) to calculate the number of localities (Figure 2-2) for each assessed species (Kruskal 1956). This is a rigorous, transparent, and repeatable methodology (Zsebők et al. 2018) that has been widely used in ecology (Dalang & Hersperger 2012; Zsebők et al. 2018; Thornhill et al. 2018). In conservation it has been used as part of IUCN Red List assessments (Richards 2019) and to identify Alliance for Zero Extinction sites (Ricketts et al. 2005), which contain the only populations of Endangered and/or Critically Endangered species.

To identify and count the number of localities for each species, we first used QGIS to produce distance matrices between all their recorded point locations. We then used Kruskal's algorithm

(Kruskal 1956) in the R optrees package (Fontela 2015) to perform a minimum cost spanning tree analysis. From this, we could count the total number of localities and calculate the number needed to trigger a site as a KBA or IPA based on their associated thresholds. For example, for a threatened species found at ten localities, a site containing one of those localities would be enough to be classified as a KBA based on Criterion A and as an IPA based on Criterion A. All the species in our study had such restricted distributions that even one locality in a site was enough to meet the KBA and IPA thresholds for all the associated criteria.

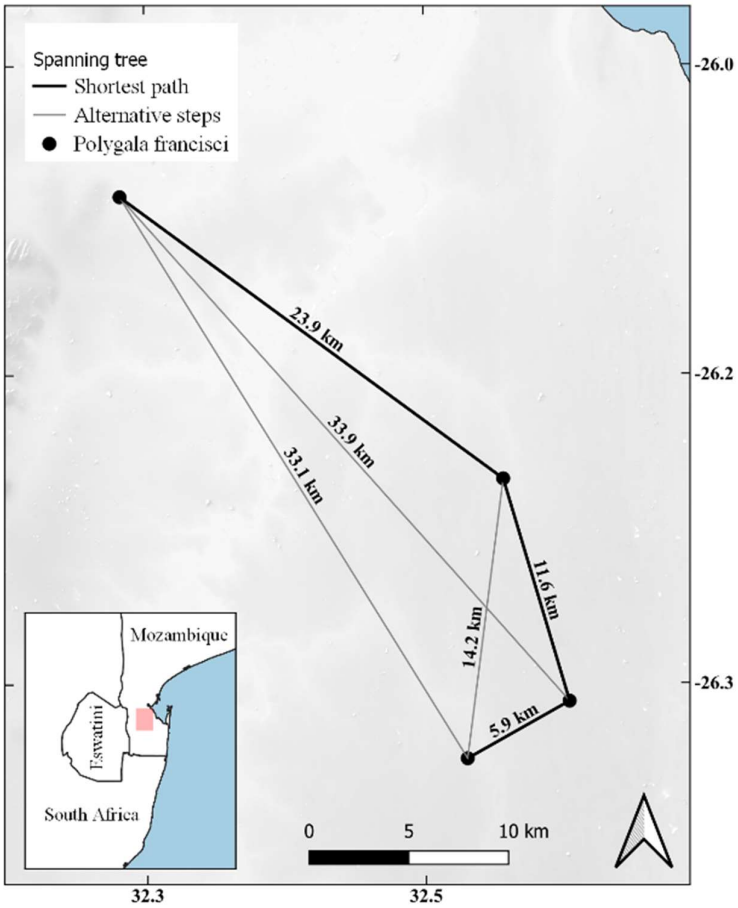



***Figure 2- 2: An example of a minimum cost spanning tree showing the species distribution points that make up two localities for a plant species (*Polygala francisci*). This involves producing a distance matrix containing the length of each line between all points and identifying those points that are within the mean distance value of arcs that form the shortest path. For example, 13.8 km is the mean value between the arcs of the minimmum spanning tree (black lines).***

### 2.2.6 Boundary delineation for KBAs and IPAs

The first step in delineating KBAs and IPAs was to identify patches of similar habitat types. We did this using a process known as segmentation, which uses remote sensing data to identify patches of pixels with similar spectral characteristics (Senthilkumaran & Rajesh 2009). We used the Segment Mean Shift function in ArcGIS Pro to delineate spectrally similar habitat segments based on a 2020

Landsat 8 OLI satellite image of the Maputaland region. This involved setting spectral, spatial and minimum segment size parameters for which we tested a number of different values until the outputs identified habitat segments that reflected those shown in the Maputaland landcover map, settling on values of 18, 10 and 1500 respectively. We applied various combinations of settings and weightings, and we assessed the results visually until the outcome produced results that matched the purpose and scale of our analysis. The output was a raster layer that was then converted into a vector format (Figure 2-3). Habitat segments smaller than 10 ha were merged with the larger segments with which they shared the longest common boundary. We also removed habitat segments that consisted of urban areas, commercial agriculture, subsistence agriculture and mining areas based on the Maputaland landcover map.




***Figure 2- 3*: Satellite image with the segmented units overlaid, displaying the segments as polygons, with the source image visible underneath.**

Next, we used the Select by Location Tool in ArcGIS Pro to identify for each trigger species whether it was likely to be found in each habitat segment. This tool allows the user to vary the search distance parameter, therefore we tested three different values: search distance = 0 m meaning we only assumed that a species was found in a habitat segment if it contained at least one point record; search distance = 100 m and search distance = 200 m. meaning we assumed that a species was found in a habitat

segment if a point record was within 100 m or 200 m of the point record. For each search distance output, we then used the Dissolve Tool in ArcGIS to combine neighbouring habitat segments that contained one or more trigger species, forming a set of habitat blocks that were the basis of the three KBA and IPA identification scenarios outlined below:

**1. Ecological Unit Scenario.** Some neighbouring habitat blocks were part of larger habitat patches that had been fragmented by roads or farmland. As KBAs and IPAs are supposed to represent these larger patches, we therefore defined these important sites by using ArcGIS to group blocks that were within 200 m of each other. We also used the Dissolve Tool in ArcGIS to merge any small patches of unsuitable habitat into the site. The minimum area of KBAs and IPAs is generally set based on the existing PAs found in the assessment region, therefore we deleted those sites that were smaller than 10 $km^2$ because the smallest PA in Maputaland is 13 $km^2$ (Figure 2-1).

**2. Management Unit Scenario.** This used the outputs of Scenario 1 and combined them with the existing PAs and OECMs (Langhammer et al. 2007). For the PAs and OECMs in Eswatini that are adjacent to one another, we assumed that they represented one management unit, as there are plans to manage them as a single unit in the future (although at present they are managed by different groups).

**3. International Boundary Unit Scenario.** This used the outputs from Scenario 2 but divided them based on the political boundaries, so that each KBA or IPA was only found in one country.

## 2.3 Results

### 2.3.1 Details of the trigger species

We found that 105 species triggered one or more KBA or IPA criteria (Table S2-2), of which 37 are globally threatened, 24 regionally threatened and 96 (91%) are endemic or near-endemic to Maputaland. The KBA trigger species were made up of 2 amphibians, 2 freshwater fish, 36 invertebrates, 45 plants and 9 reptiles. The IPA criteria were triggered by 55 plant species, which included all of the 45 KBA trigger species, with an additional 10 that met the criteria based on being important subspecies or varieties.

The number of distribution point records per species ranged between 1 and 95, with a median of 4. The number of records differed between the three countries, with fewer fauna records in Mozambique compared to South Africa and Eswatini (Figure 2-4). In addition, more points were recorded on the coastal plain compared to the Lubombo Mountains in the west. For every species assessed, each locality on its own was enough to meet the global population thresholds, so that any site containing that locality could trigger KBA and/or IPA status.

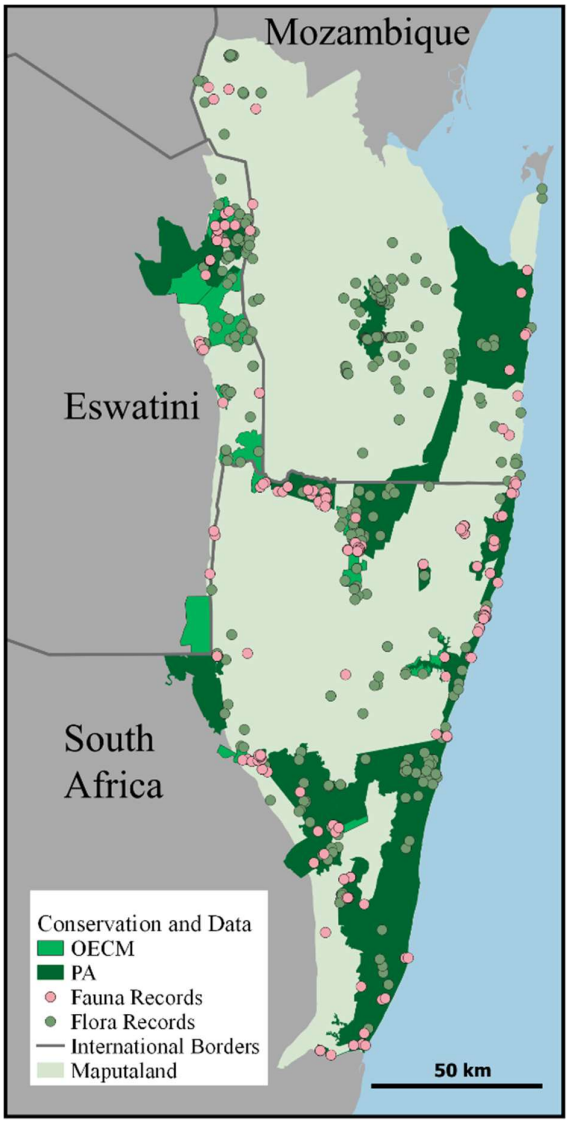



***Figure 2- 4: Distribution data of the fauna and flora taxa assessed for KBA and IPA throughout Maputaland.***

### 2.3.2 Segmentation analyses

The segmentation analysis divided Maputaland into 3,503 habitat segments, with an area range of 10 ha to 41,692 ha and a median of 241 ha. Based on the three Search Distance scenarios, the total area of the habitat patches containing localities for any of the trigger species ranged between 5,646 km$^2$ for Scenario 1 to 6,955 km$^2$ for Scenario 3, with the median patch size ranging between 18.5 and 23.3 ha (Figure 2-5). This meant the percentage of Maputaland identified as containing important habitat for the trigger species was 29.4% for Scenario 1; 33.6% for Scenario 2 and 36.2% for Scenario 3.

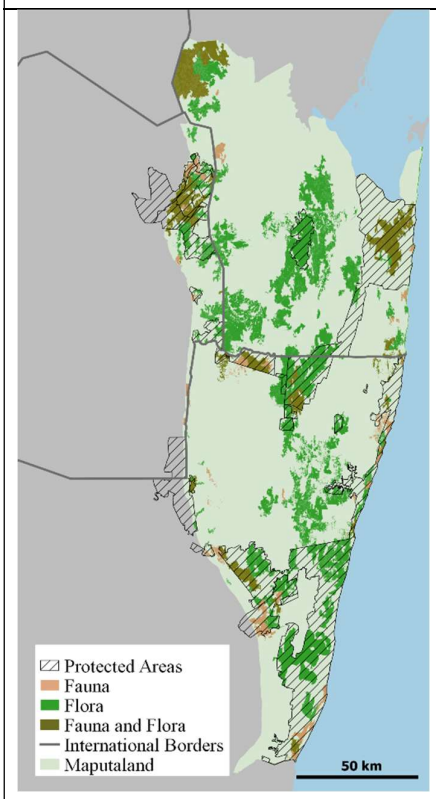
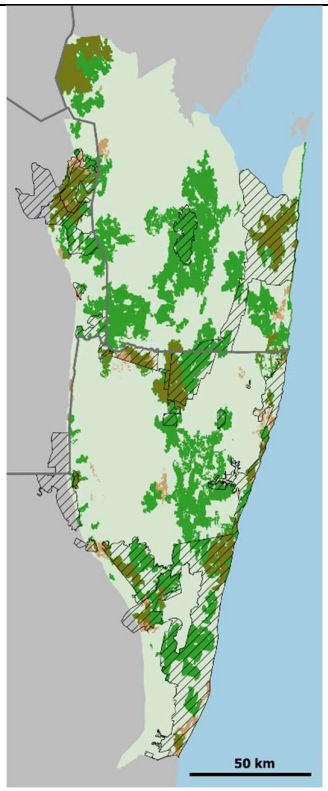
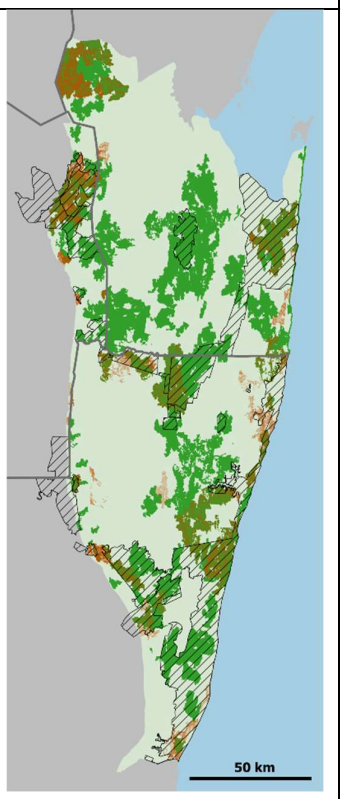
| Search distance = 0 m | Search distance = 100 m | Search distance = 200 m |
|---|---|---|
|  |  |  |
| Total area = 5,646 km$^2$<br>Median = 18.5 | Total area = 6,442 km$^2$<br>Median = 20.4 | Total area = 6,955 km$^2$<br>Median = 23.3 |

***Figure 2- 5: Network of segmented polygons selected for containing distributional records of taxa under KBA and IPA analyses for three search distances. The maps show areas selected based on fauna, flora, and both fauna and flora data set.***

### 2.3.3 Boundary delineation

The Ecological Units Scenario shows that the results varied with search distance, ranging from 20 sites with a median area of 211 km$^2$ covering 27% of the region for the 0 m search distance, to 16 sites with a median area of 261 km$^2$ covering 30% of the region for the 200 m search distance (Figure 2-6). Out of the 20 sites identified for 0 m search distance, 12 contain one species found in no other sites and 14 contain species found at only one other site.

The Management Units Scenario again shows that the results varied with search distance, ranging from 13 sites with a median area of 459 km$^2$ covering 39% of the region for the 0 m search distance, to 12 sites with a median area of 569 km$^2$ covering 42% of the region for the 200 m search distance (Figure 2-6). Out of the 13 sites identified for 0 m search distance, eight contain one species found in no other sites and eight contain species found at only one other site.

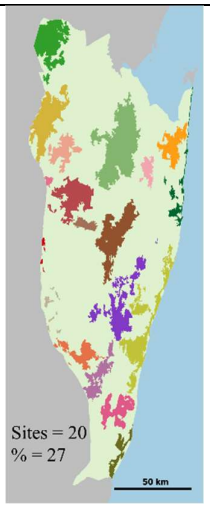
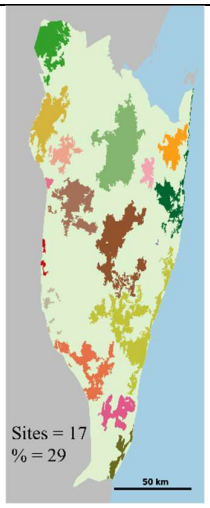
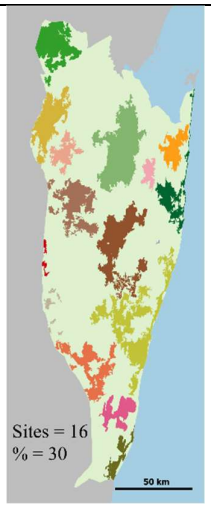
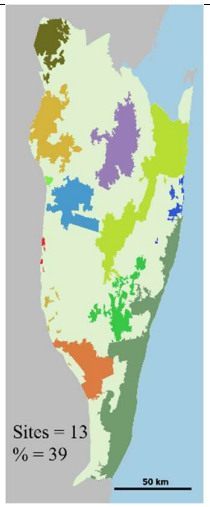
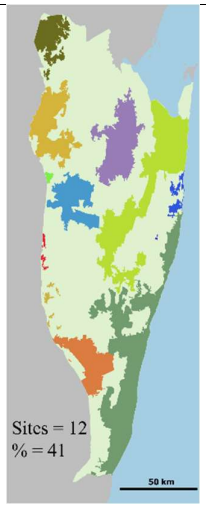
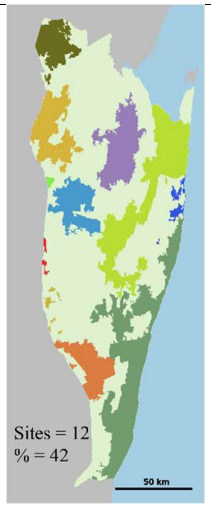
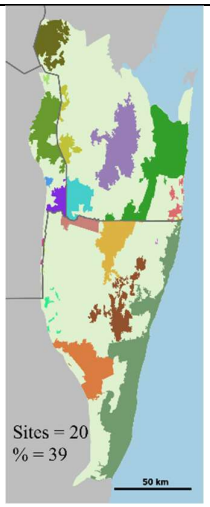
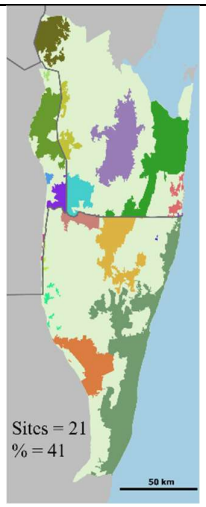
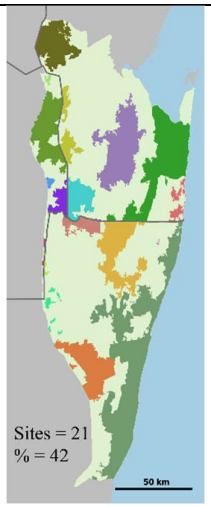
| Scenarios | Search dist. = 0 m | Search dist. = 100 m | Search dist. = 200 m |
|---|---|---|---|
| Ecological Units |  |  |  |
| Management Units |  |  |  |
| International Boundary Units |  |  |  |

***Figure 2- 6: Potential important sites for Maputaland's biodiversity based on three search distances under the following scenarios: (A) Ecological Unit Scenario; (B) Management Unit Scenario and (C) International Boundary Unit Scenario.***

The International Units Scenario involved dividing up the sites identified in the Management Units Scenario based on political boundaries, which led to a small number of sub-divided sites being removed because they failed to meet the minimum size threshold. However, this had a marginal impact on the total area of the resultant sites. Once again, the results varied with search distance, ranging from 20 sites with a median area of 176 $km^2$ covering 39% of the region for the 0 m search distance, to 21 sites with a median area of 153 $km^2$ covering 42% of the region for the 200 m search distance (Figure 2-6). Out of the 20 sites identified for 0 m search distance, 12 contain one species found in no other sites and 13 contain species found at only one other site.

### 2.3.4 The impact on the landscape of different delineation scenarios

Our analyses show that patterns of the potential KBA or IPA change under different delineation scenarios (Figure 2-7). From the first scenario (Ecological Unit Scenario) to the second (Management Unit Scenario), the different potential KBAs or IPAs either remained the same or merged to form larger sites, whereas from the second scenario to the third scenario (International Boundary Unit Scenario), they either remained the same or split to form smaller sites (Figure 2-7). Irrespective of the search distance used, six sites remained as single sites throughout the scenarios (diagram A in Figure 2-7), two sites were split in Scenario 2 (diagram B), and three sets of sites merged in Scenario 2 and then split in Scenario 3 (diagrams E and G).

| Merge/split examples | Search distance (m) | | |
|---|---|---|---|
| | 0 | 100 | 200 |
| A: remains single site | 6 | 6 | 6 |
| B: splits into two | 2 | 2 | 2 |
| C: splits into three | 0 | 1 | 1 |
| D: merges two | 1 | 0 | 0 |
| E: merges two then splits again into two | 1 | 1 | 1 |
| F: merges two and splits to three | 1 | 0 | 0 |
| G: merges three and splits to two | 2 | 2 | 2 |

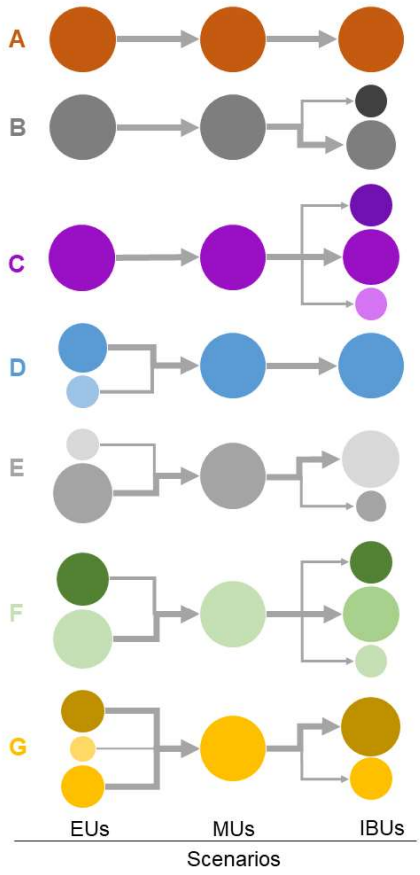



***Figure 2- 7: An illustration of how KBAs/IPAs in Maputaland (represented by circles) merged or split depending on the different delineation scenario. Scenarios are: 1) Ecological Unit Scenario EUs; 2) Management Unit Scenario MUs; 3) International Boundary Unit Scenario IBUs. The width of the connecting paths between the circles shows the related ratio of the area between them. The larger the width, the higher is the proportion area from one circle to another.***

## 2.4 Discussion

### 2.4.1 Identifying the trigger species

Our analysis focused on the threatened and geographically restricted amphibians, freshwater fish, invertebrates, plants and reptiles found in Maputaland. We did not include birds and mammals because these species are wider ranging, therefore KBAs for these species are generally identified by selecting which existing PAs contain large enough populations to meet the thresholds (WCS et al. 2021). We excluded other taxonomic groups because of a lack of data and we will discuss the potential impacts of this below. Only assessing species based on their threat and geographically restricted status may also have had impacts but this is unlikely, as most of the other criteria focus on habitat-based metrics. The exception is KBA Criterion D, which identifies sites holding a significant proportion of the global population of a species during one or more of its life history stage or process, but we have no evidence that this is the case for any terrestrial species found in Maputaland.

One of the advantages of applying the KBA and IPA methodologies is that they provide an objective and repeatable approach for identifying globally and regionally important species, especially those that are not listed as threatened (Lim et al. 2023). This is well illustrated for Maputaland, where previous work to identify important species depended in part on local expertise and therefore missed a number of geographically restricted species (Smith et al. 2008). We also found the encouraging result that all of the plant species that met the KBA criteria were also identified as important by the IPA criteria. suggesting that most sites identified as a KBA for plants are also likely to be an IPA and vice versa. This is because of the similarity in their criteria (IUCN 2016; Darbyshire et al. 2017). For example, KBA Criterion A1a and A1b are equivalent to IPA Criterion A(i) but with slight differences in the thresholds (Table S2-1). The main difference is that the IPA criteria also include regionally threatened species and subspecies, which identified an additional eight trigger species in Maputaland.

It is also important to note that each species assessment was based on the existing Red List threat status data and lists of endemic and near-endemic species. This means that their KBA and IPA trigger species status could change over time, which is quite possible given that many of these species are poorly known. In particular, there are already examples where species that were thought endemic to Maputaland have subsequently been recorded in neighbouring regions. Thus, it is likely that some species that are currently seen as important based on the geographically restricted criteria will lose that status, although this may be counter-balanced by the discovery of new species within this biodiversity hotspot.

### 2.4.2 Mapping the trigger species

Our knowledge about the distributions of Maputaland's threatened, endemic and near-endemic species remains limited, which in turn makes it difficult to confidently pinpoint important sites for representing them in the region. To address this, we used habitat suitability modelling techniques (Guisan & Zimmermann 2000) to identify the extent of suitable habitat (ESH) for each species. ESH is defined as being the area with the best ecological conditions for a certain species, and it is one of the KBA parameters used to assess whether a site holds a threshold proportion of a species global population size (IUCN 2016). An important part of this process was using remote sensing segmentation techniques to identify habitat segments that contain each species locality. Segmentation is already widely used for mapping vegetation communities (Munyati et al. 2013; Munyati 2018), providing a robust and repeatable process (Massada et al. 2012). Our analysis is the first of its kind to assess whether segmentation could be used to inform KBA and IPA boundary delineation and we found it produced satisfactory results. To validate the models, we worked closely with a variety of experts throughout the process (MacMillan & Marshall 2006; Leblond et al. 2014). This is in line with

the KBA and IPA processes, which should be built on strong stakeholder engagement as part of the decision support process (IUCN 2016; Darbyshire et al. 2017)

When interpreting our results, however, it is important to recognise that our analysis was impacted by the "rare species modelling paradox" (Lomba et al. 2010), where the species of highest conservation concern are also the most difficult to model (Guisan et al. 2006; Elith et al. 2011; van Proosdij et al. 2016; Chiffard et al. 2020). This issue may be less of a problem in Maputaland, as the region has distinct biogeographic boundaries and therefore the distribution of many species is quite constrained by the underlying geological and climatic conditions (Smith et al. 2008). That said, the patchiness of the field data still makes it likely that the models were influenced by false absences, increasing the likelihood that species with non-identical habitat requirements were predicted to have similar distributions because they were recorded together at the same handful of sites. This may explain why many of the sites we identified in our analysis contained a similar set of species, often duplicating these species in each site. One reason for this is that the KBA and IPA approaches are applied on a case-by-case basis, so that meeting only one of the thresholds is enough to qualify as an important site (IUCN 2016; Darbyshire et al. 2017). Care is therefore needed when interpreting data in this situation. Fortunately, this is addressed by both the KBA and IPA methodologies, which include steps where the candidate sites in a region are checked by experts (Darbyshire et al. 2017; KBA Standards and Appeals Committee 2022). The ultimate solution is to collect better data, and building on this study, we think future fieldwork should focus on areas and taxa that are less well known. Our study also shows the benefits of using transnational datasets to identify KBAs and IPAs, as our model accuracy depended on using the larger, more representative data provided by combining data from all three countries.

We also investigated the impacts of changing the search distance within which we assumed a habitat patch was suitable habitat for a species, based on its distance from a known locality of that species. This had important impacts, as increasing the search distance from 0 m to 200 m increased the area of Maputaland containing important habitat for all the species from 29.4% to 36.2% of the region, and increased the median patch size of this habitat from 18.5 ha to 23.3 ha. Combining the habitat patches for all the species also showed that most of the land highlighted was important for plant species, with little important habitat for animals being found outside the existing PAs (Figure 2-4). This might be because Maputaland is particularly important for plant endemism and diversity, but it is also probably because we were able to collect new plant distribution data as part of this study. This further highlights the need for better data on a wider range of taxa.

### 2.4.3 Delineating the KBAs and IPAs

We compared three scenarios for delineating KBAs and IPAs based on the initial results from combining the locality and habitat segmentation outputs. We also investigated how using the habitat maps based on different search distance values influenced the results. For the latter, we found that basing KBA and IPA delineation on habitat maps that used higher search distance values identified more land as important, reflecting the pattern from the original input maps. However, the increase was less pronounced than in the original input maps, so for example the results from Scenario 1 ranged between 27% and 30% of Maputaland based on the habitat maps where the search distance parameter was 0 m and 200 m, respectively. This was because our process for identifying the important sites included a step where we subsumed small patches of unsuitable habitat into any site within which they were entirely contained. This step produced sites that included some unsuitable habitat, but they were less fragmented and more suitable for KBA and IPA status, so should still be seen as sites that best reflect the distributions of the species they were selected to represent (Bisquert et al. 2015).

With the KBA and IPA methodologies based on identifying manageable units, part of the delineation process involves considering the existing PAs and OECMs. This is apparent in the second scenario, which was achieved through another repeatable step of combining the Scenario 1 with Maputaland's PAs and OECMs. This had a significant impact on the amount of land that was selected, increasing the area coverage of the Maputaland to between 39% and 42%. The difference between the coverage results from Scenario 1 and Scenario 2 shows that incorporating management units can include large sections that do not contain trigger species habitat. This can create three issues: (i) reporting the area of land with KBA or IPA status is likely to inflate estimates of high biodiversity value land; (ii) reporting the number of trigger species found in a KBA or IPA will give the impression that the whole site is important for those species, even when the associated species are found in discrete patches within the site; and (iii) reporting data at the KBA/IPA level provides less precise data for informing work on ensuring effective management to maintain the trigger species.

However, there is a good reason why the KBA and IPA methodologies consider management units. This helps to ensure that each site is the responsibility of a particular management group, making it more likely that they will be officially recognised and their management mainstreamed into existing governance systems (Plumptre et al. 2019). Thus, we need approaches that account for the mismatches between Scenario 1 and 2 that we have highlighted. One solution would be for conservation agencies to define different management zones within large PAs or OECMs reflecting KBA or IPA boundaries. This would then make it possible for KBAs or IPAs to incorporate the specific zone segments, rather than the whole PA or OECM. Basing management decisions on the actual size of trigger species habitat patches is also important because, as our results show, many trigger species have important parts of their range outside of existing management units. For that

reason, caution is needed when identifying KBAs based on existing management units only, and future re-zoning could include adding more land to fully cover each KBA when appropriate.

Scenario 3 illustrates another set of issues that appear when the delineation accounts for political borders. Our results show that some of the areas that straddle political borders are divided into smaller, less viable polygons. It is widely recognised that political borders fragment ecosystems, posing a significant threat to biodiversity (Liu et al. 2020). In theory, the identification of KBAs should overcome this, as the process is based on global data and identifying globally important sites. Thus, this process should galvanise nations to work closely when establishing KBAs and IPAs, particularly in trans-frontier regions. However, the KBA guidelines state that boundary delineation should not go beyond political borders unless there is already a recognised cross-border conservation area, or one will be in place in the near future (IUCN 2016). This illustrates that there is another benefit of developing Transfrontier Conservation Areas (TFCAs) in areas of high biodiversity value (Wolmer 2003; Hanks 2003), so that the KBAs and IPAs in these regions can also cross these international boundaries, reflecting the natural patterns of biodiversity rather than political boundaries. But this is problematic for two reasons. First, the TFCA processes often take many years to establish, but there is a need to identify KBAs and IPAs more rapidly. Secondly, even in places like Maputaland that are already TFCAs, KBA and IPA identification tends to take place at the national level and so focuses on one country at a time. By contrast, our analysis shows it is possible to use the available data to highlight the trans-frontier locations and areas of extent that are important for their biodiversity. We would therefore suggest that this should be taken into consideration when updating any guidelines.

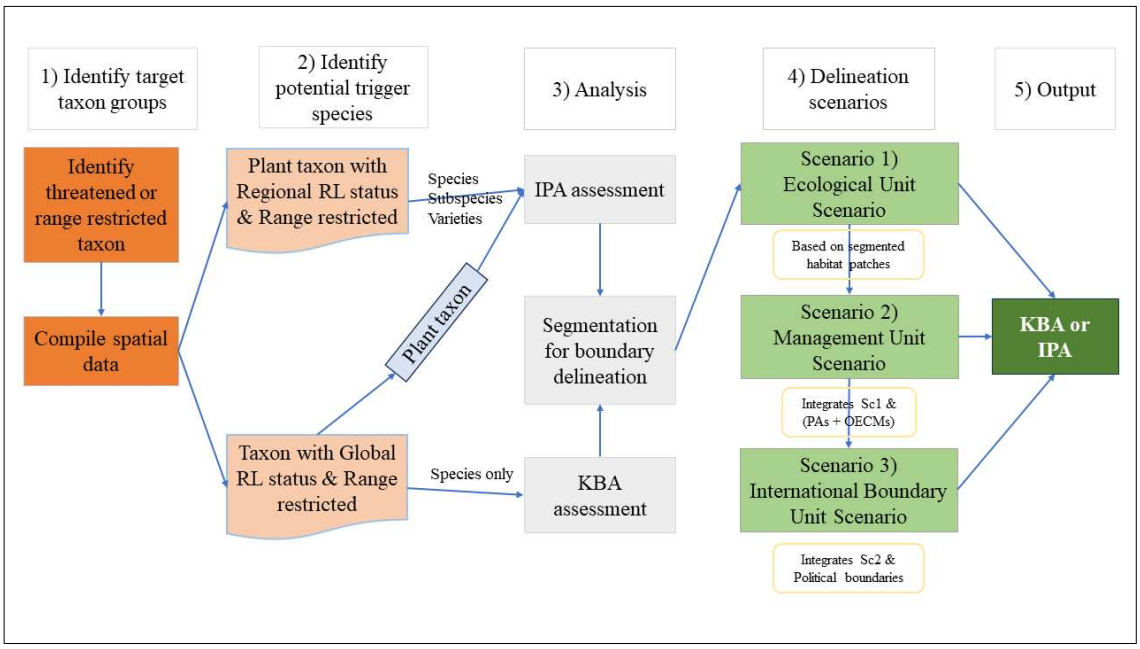



***Figure 2- 8: Overview of the KBA and IPA delineation stages based on a segmentation approach.***

### 2.4.4 Incorporating the KBAs and IPAs into policy and practice

The KBAs and IPAs we identified in Maputaland are important sites for biodiversity but they are not automatically conservation priorities (Smith et al. 2019). This is because identifying priority areas for conservation needs to account for a wider range of biodiversity elements, as well as a range of social and socio-economic data. This is particularly relevant here because some of the sites we identified contain the same set of species and so might not all be seen as priorities for conservation action. KBA and IPA should, therefore, be included in a conservation prioritisation process based on the systematic conservation planning approach. Systematic conservation planning uses complementarity-based algorithms (Margules & Pressey 2000), so would select the KBA and IPA elements alongside other features that satisfactorily meet conservation targets for establishing a functional landscape (Iwamura et al. 2018). The approach is context specific, being guided by determining which benefits a particular action could bring, how much it would cost and ultimately the likelihood of its success (Moilanen 2012; Kukkala & Moilanen 2013; Groves & Game 2016). This process can therefore inform decisions on the allocation of resources to be provided by national governments, donors and conservation agencies, for supporting initiatives such as PA expansion and management, including monitoring and vigilance (Joseph et al. 2009; Bolam et al. 2019). All of these aspects suggest that to be effective, this prioritisation process also needs to be carried out at the trans-frontier level.

# 3 Designing an ecological network for conservation and ecotourism in a biodiversity hotspot


## Abstract

Expanding the global network of conservation areas will depend on creating new community managed protected areas (PAs) and Other Effective Area-Based Conservation Measures (OECMs). These new types of conservation area often have different goals and funding mechanisms when compared to traditional state PAs, so there is a need to account for these differences when designing conservation landscapes. Here we show how the systematic conservation planning approach can be used to design such an ecological network for Maputaland, a trans-frontier conservation area in southern Africa. We used Marxan with Zones to identify where best to locate two types of conservation area to complement the existing PA network, based on meeting targets for 45 landcover types and 212 species. Zone 1 conservation areas are designed to protect species vulnerable to over-harvesting. The cost of including sites in this zone was based on its risk of being impacted by anthropogenic pressures, so that less vulnerable sites were preferentially selected where possible. Zone 2 conservation areas are designed to be less intensively managed and funded through community-based ecotourism, so only meet targets for the landcover types and species that are less vulnerable to over-harvesting. The cost of including sites in this zone was based on its suitability for ecotourism, as measured using Analytical Hierarchy Process modelling. The results identified priority areas for both zones in Eswatini, Mozambique and South Africa and will be particularly important for designing trans-frontier conservation ecotourism initiatives and protecting a number of endemic plant species.

## 3.1 Introduction

We are in the middle of a global environmental crisis, with recent work highlighting the dramatic loss of biodiversity, and the interdependency of healthy ecosystems, climate change mitigation, poverty alleviation and human wellbeing (IPBES 2019). Protected areas (PAs) are a widely used policy instrument for tackling biodiversity loss, as they are designed to reduce habitat loss and the over-exploitation of species (Watson et al. 2014). Signatories to the Convention on Biological Diversity have therefore committed through the Kunming-Montreal Global Biodiversity Framework, to increase the percentage of the terrestrial realm managed for conservation to 30% by 2030. These international commitments are not simply to increase the amount of land in PAs, however, as Target 3 also specifies that conservation areas should be effectively and equitably managed, include important areas for biodiversity and ecosystem services, and form networks that are ecologically representative and well connected (CBD 2021). This is vital because many PA networks currently fall short of these aspirations by having low levels of management effectiveness (Coad et al. 2019), failing to represent important biodiversity elements (Butchart et al. 2015) and reducing equitability (Allan et al. 2022).

One way to address these limitations and meet the Global Biodiversity Framework Target 3 is to continue to create new state-managed PAs. However, this is unlikely to be achieved in many countries, as it becomes politically more difficult to gazette under-protected ecosystems (Kuempel et al. 2016), it is therefore increasingly recognised that other approaches should play a larger role. In particular, there is a new focus on creating OECMs (other effective area-based conservation measures), which are defined as "a geographically defined area other than a PA, which is governed and managed in ways that achieve positive and sustained long-term outcomes for the in-situ conservation of biodiversity" (IUCN-WCPA Task Force on OECMs 2019). Most OECMs differ from PAs by having conservation as a secondary management objective, such as, for example, sacred groves or sites designated for watershed protection (MacKinnon 2020). Thus, in many cases the process of creating OECMs will involve recognising and documenting existing sites, but there is also scope for expanding the area of land under conservation by working with landowners to designate new areas and adopt new management approaches (IUCN-WCPA Task Force on OECMs 2019).

Developing a network of OECMs to complement the existing PAs with sites that fill the gaps in biodiversity representation and ecological connectivity will generally be a complicated process, as it involves working with a number of stakeholders and balancing several conflicting requirements. One means of informing such decisions is to adopt the systematic conservation planning approach (Margules & Pressey 2000), which is an operational model for identifying and implementing the conservation of priority areas. The first steps in any systematic conservation planning project focus on framing the scope, and consist of identifying and involving stakeholders, and elaborating the context

for the conservation areas (Groves & Game 2015). This is followed by a spatial conservation prioritisation that involves translating the broad goals of the projects into objectives, often underpinned by targets on how much of each conservation feature (species, ecosystems, etc.) should be represented in the conservation area network (Smith et al. 2021).

Systematic conservation planning has been used to inform the development of conservation area networks globally (Sinclair et al. 2018). Many of these analyses did not explicitly incorporate data on how the priority areas they identified should be managed, but more recently there has been a trend to account for management zones when developing ecological networks. This has been pioneered in marine conservation (Klein et al. 2010), where it is common for different zones to permit different types of fishing and other activities. Other studies have looked at management zoning for different conservation, restoration and agricultural land-uses (Adams et al. 2016; Law et al. 2017). Such an approach is also highly relevant for developing conservation area networks of PAs and OECMs, especially as some OECMs are likely to have lower levels of management intensity (Alves-Pinto et al. 2021) and so may not contain species that are sensitive to over-harvesting or direct persecution. Here we present the first systematic conservation planning analysis that uses this zoning approach to identify priority areas for new OECMs, accounting for biodiversity patterns, management intensity and the socio-economic factors that underpin conservation decision making.

Our analysis focuses on the Maputaland Centre of Endemism, an area of 19,180 $km^2$ that falls within Eswatini, Mozambique and South Africa and forms part of the Maputaland-Pondoland-Albany global biodiversity hotspot (Van Wyk 1996; Mittermeier et al. 2004). The region is a prime ecotourism destination but is also home to some of southern Africa's poorest communities, many of whom depend on subsistence agriculture for their livelihoods. This is why the three national governments have developed a trans-frontier conservation initiative, based on tackling poverty and biodiversity loss through expanding the ecotourism sector. This involves improving infrastructure, training local people, and expanding the conservation area network. More than a quarter of Maputaland is already in state- and privately-managed PAs but previous analyses have shown that much of the important biodiversity is not conserved (Smith et al. 2008; Darbyshire et al. 2023), and is threatened by the expansion of agriculture and the over-harvesting of natural resources. Thus, it is widely recognised in the region that more conservation areas are needed, but with a focus on creating new community-managed OECMs that would create employment through ecotourism.

To help achieve the broad conservation and development goals that are common across many biodiversity hotspots, we used the Marxan with Zones software package (Watts et al. 2009a) to identify where new types of community-based OECMs are best located in Maputaland. Our analysis focused on three scenarios that were based on whether OECMs containing species that are threatened

by over-harvesting would need additional resources to ensure they are effectively managed. Building on the existing systematic conservation planning system (Smith et al. 2008), we added new spatial models on species distributions, suitability for agricultural and suitability for ecotourism. We then measured how well the current network of conservation areas meets biodiversity representation targets and carried out spatial conservation prioritisation analyses based on three management effectiveness scenarios. From these outputs, assisted by comparison with results from the earlier study, we were able to assess the importance of accounting for management effectiveness when developing OECM networks, and the impact of including recent biodiversity distribution data and relevant socio-economic metrics into spatial conservation planning.

## 3.2 Methods

The systematic conservation planning approach involves a number of steps consisting of defining the context, producing a list of biodiversity elements that should be represented in the conservation area network (these can be important species, ecosystem types, etc., and are collectively known as 'conservation features'), collecting the relevant biodiversity and cost data, setting targets, dividing the planning region into a number of planning units and using spatial prioritisation software to identify the best sets of planning units for achieving the conservation objectives (Groves & Game 2015). For this study we used the Marxan with Zones software package (Watts et al. 2009a), as it allows the user to specify several management zones and then identifies which planning units should be assigned to which zone, based on meeting the conservation targets, minimising the planning unit costs and maintaining connectivity. As part of this process, the user can specify a zone-specific cost for each planning unit and a zone-specific contribution per conservation feature. This could involve, for example, specifying that a planning unit contains 20 ha of riverine forest but that amount should only contribute to the target if the planning unit is assigned to a conservation zone.

### 3.1.1. Study region and conservation context

In 2000 the governments of Eswatini, Mozambique and South Africa launched the Lubombo Trans-frontier Conservation Area (TFCA) initiative to conserve biodiversity and tackle poverty in Maputaland. The region already contains a number of state- and privately-managed PAs, as well as a number of conservation areas (CAs) that are communally- or privately-managed (Figure 3-1). Some of the CAs in Eswatini are already recognised as OECMs and many of the others meet the OECM definition, although they have not gone through the official documentation process.

The three governments have also recognised the importance of building on previous systematic conservation planning work that helped guide the previous development of Maputaland's PA network (Smith et al. 2008). There is a need to update the approach to match the new context, and in this

analysis we identified priority areas for OECMs instead of PAs. We also adopted a zoning approach, identifying important areas for conservation based on three different types of management. Zone 1 identified areas most suitable for higher intensity management OECMs (referred to as the "High Intensity Zone" hereafter), which contain species that are vulnerable to over-harvesting or direct persecution and so for the management to be effective need higher levels of intervention (e.g. more frequent law enforcement patrols and/or species-specific management projects). Zone 2 identified areas most suitable for lower intensity management OECMs (referred to as the "Low Intensity Zone" hereafter), which do not contain these species vulnerable to over-harvesting or persecution and so can be managed using lower-intensity habitat management techniques. Zone 3 identified areas that are not needed to meet the conservation targets (referred to as the "Other Zone" hereafter).

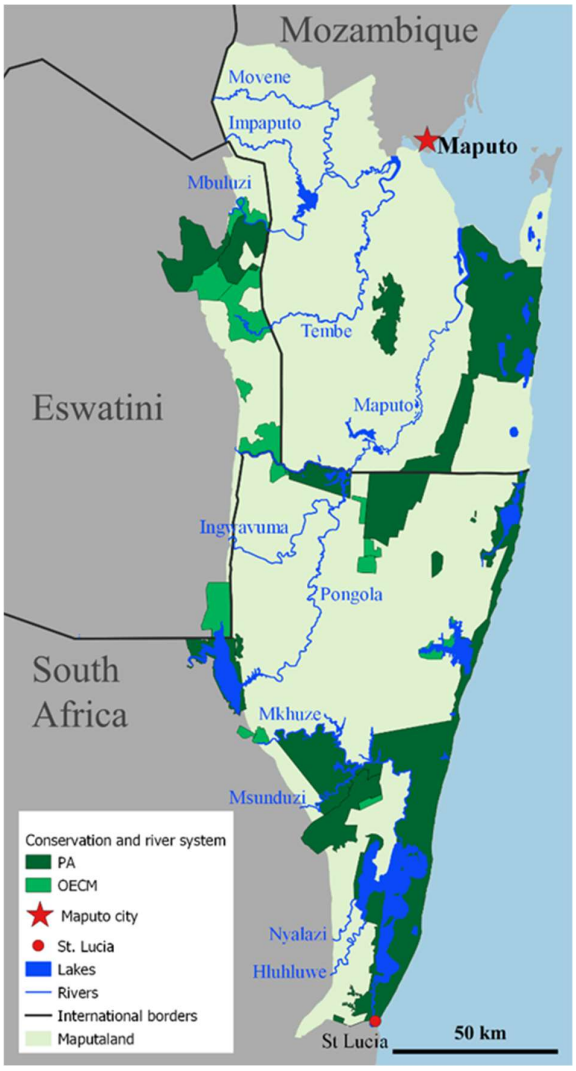



***Figure 3-1: Maputaland showing the region's rivers, protected areas and conservation areas.***

### 3.1.2. Selecting and mapping the conservation features

We convened an expert group of ecologists and conservation scientists from academia, government and NGOs to develop a list of landcover types and species that should be used as conservation features in the conservation planning system. This built on previous work based on 44 landcover types and 53 species (Smith et al. 2008), using the same landcover types but developing a more systematic approach to selecting species, choosing all amphibians, birds, freshwater fish, insects, invertebrates, mammals, plants, and reptiles with available data that are threatened, based on the IUCN Red List or national red lists, and/or endemic or near-endemic to Maputaland. We also identified which species are threatened by over-harvesting or direct persecution in Maputaland, based on information from regional and national red lists and feedback from our expert group, and which species are threatened by over-harvesting or direct persecution globally, based on the global IUCN Red List threat data.

To map each of the landcover types, we updated an existing 25 m resolution landcover map that was based on Landsat ETM and ASTER satellite images (Smith et al. 2008). This involved using the ENVI 5.1 software to identify patches of urban, commercial agriculture and subsistence agriculture landcover type based on Landsat 8 imagery, which were then overlayed over the previous landcover map to update the areas that had been transformed in the previous 15 years. This approach was also used to map all the landcover types in the north-western section of the planning region which were then added to the 2008 map to match political boundaries. The distribution maps of the species were based on national, regional, and international natural history collections, plus data from the government conservation agencies in each country and field surveys for plant species. Where there was sufficient data, we produced a distribution map for each species based on Maximum Entropy (MaxEnt) models (Phillips et al. 2006), but this was not possible for species with fewer than 10 records, and for these we simply mapped their distributions based on the point data. Finally, we identified which landcover types were not suitable for each species, based on the literature, information from the IUCN Red List and feedback from our expert group, and used QGIS to remove patches of the unsuitable landcover types to produce the final species distribution maps.

### 3.1.3. Setting the conservation feature targets

We used four approaches to set targets for each of the conservation features, building on the previous Maputaland analysis (Smith et al. 2008) and methods developed by Ezemvelo KwaZulu-Natal Wildlife, the state conservation agency responsible for conservation planning in the South African section of Maputaland. For the landcover types, setting targets involved estimating the original area of each, identifying those that most closely matched the vegetation types used in the KwaZulu-Natal analysis, and multiplying the total area by a proportion value estimated to be the amount required to represent a single occurrence of 75% of the plant species occurring within the vegetation type (Jewitt

2018). The process of setting species targets was based chiefly on the system used in KwaZulu-Natal (Pfab et al. 2011). For species assessed as threatened under Criterion B of the IUCN Red List, we set their targets as 100% of their distribution in Maputaland. For species assessed under Criterion A of the IUCN's Red List, together with species assessed as Near Threatened (NT) or Least Concern (LC), we first calculated their global target based on conserving 10,000 mature individuals and 11 localities (instead of "locations", using the definition of the IUCN Standards and Petitions Committee (2022), given the limited information about threats), with a minimum of 4 $km^2$ per each locality (a 2x2 grid cell for AOO threshold set under the IUCN Red List) (IUCN Standards and Petitions Committee 2022). We produced the final target for each species by multiplying their global target by the estimated proportion of their global range in Maputaland. Finally, we used the targets developed by Smith et al. (2008) for 5 large mammal species (black rhino, hippopotamus, African savanna elephant, wild dog and lion) that are the focus of ambitious re-introduction programmes in Maputaland.

### 3.1.4. Developing the planning units and cost layers

The Maputaland conservation planning system entails dividing the planning region into 21,632 planning units. These units were created based on a series of 1 $km^2$ hexagons, which are known to produce efficient and connected results (Nhancale & Smith 2011), that were then joined in QGIS with the borders of the PAs and CAs to ensure they accurately represented their boundaries in the spatial prioritisations. The planning units were then imported into the CLUZ conservation planning plug-in for QGIS (Smith 2019), with the PAs given the status of locked-in to the High Intensity Zone and the CAs given the status of locked-in to the Low Intensity Zone, thus ensuring they were automatically assigned to these zones in the subsequent Marxan with Zones analyses. Repeating the approach used by Smith et al. (2008), we also identified planning units unsuitable for conservation if more than 25% of their area consisted of commercial agriculture or urban areas, or if more than 80% of their area consisted of commercial and/or subsistence agriculture. These planning units were locked-in to the Other Zone. We then used CLUZ to import the distribution data and calculate the amount of each conservation feature found in each planning unit.

We used a different planning unit cost metric for each of the conservation zones. For the High Intensity Zone, we repeated the approach used by Smith et al. (2008), modelling risk of agricultural transformation to identify those areas that are most valued as farmland because they are most likely to be cleared for agriculture. We used a logistic regression analysis to identify distance to existing landcover, the soil quality and elevation, which best explains land clearance patterns in the last 14 years, as shown by the 2006 and 2020 Maputaland landcover maps. Through modelling, we then produced a 25 m resolution map (Figure 3-2a). For the Low Intensity Zone, we used a 25 m ecotourism potential map (Figure 3-2b) that was produced following a multi-criteria decision analysis

approach (Dharmaratne 2020). This was based on nine important criteria within four groups: (i) Landscape – landcover and natural attractions; (ii) Biodiversity – bird diversity, mammal diversity, and distance to PAs; (iii) Accessibility – distance to tarred roads, distance to non-tarred roads, and distance to towns; and (iv) Socio-Cultural – cultural attractions. To calculate the planning unit cost for each zone, we first calculated the sum of the agricultural risk and ecotourism potential pixels in each planning unit. We then used a linear transformation to both metrics, so their sets of values ranged between 0.002 and 100. Because we wanted to ensure that it was always more expensive for a planning unit to be assigned to the High Intensity Zone, we added 100 to all the High Intensity Zone values. We set all the Other Zone planning unit cost values as 0.001, so that it was always less expensive for Marxan with Zones to assign a planning unit to this zone if it was not needed to meet the conservation targets or maintain connectivity.

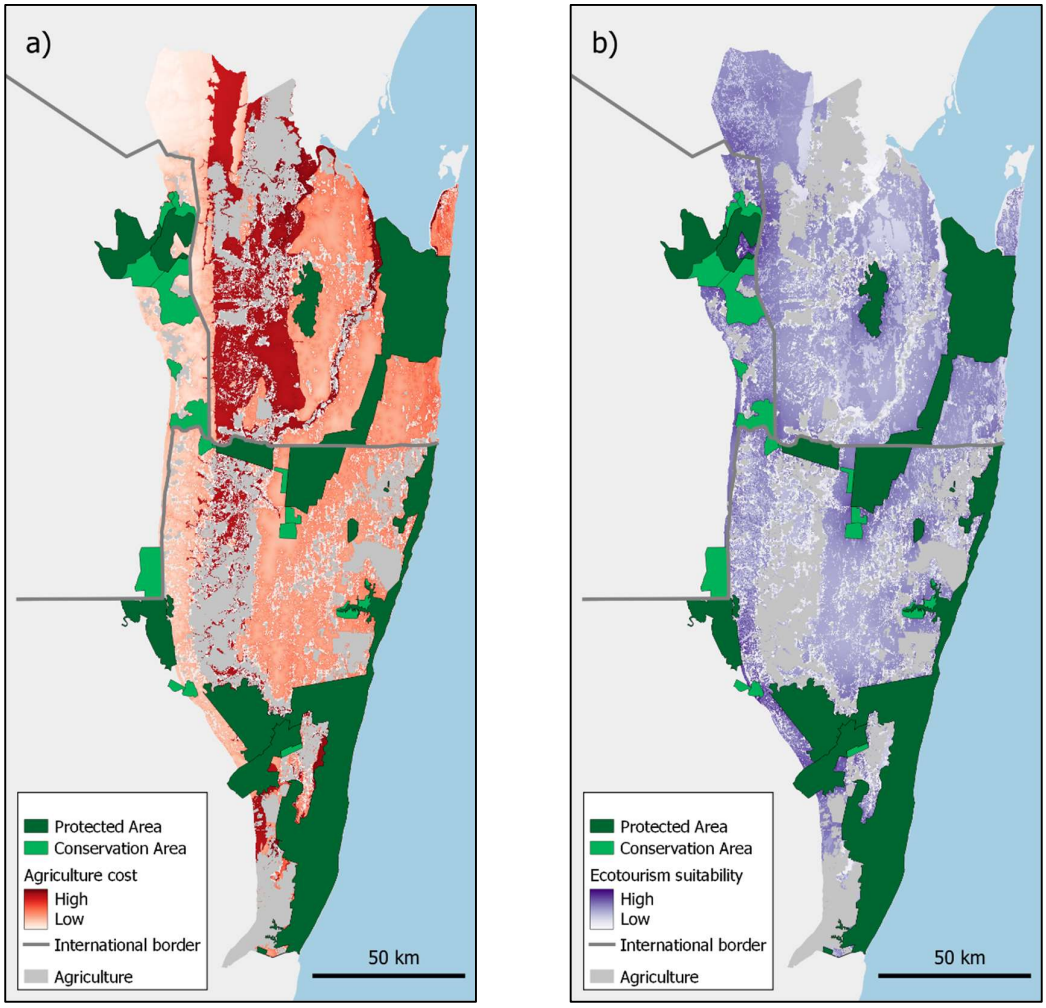



***Figure 3-2: (a) Agricultural opportunity cost map; (b) Ecotourism potential cost map for Maputaland***

### 3.1.5. Spatial prioritisation

Once all the data had been imported into CLUZ, we used it to identify how well the existing PA and CA networks met the targets. We then used the Marxan with Zones software to carry out spatial prioritisations based on the three scenarios to identify where would be best to conserve to meet the targets. Scenario 1 assumed that all the species used in the analyses could be effectively conserved in the Low Intensity Zone. Scenario 2 assumed that species identified as threatened by over-harvesting or direct persecution in Maputaland could only be effectively conserved in the High Intensity Zone. Scenario 3 assumed that species identified as threatened by over-harvesting or direct persecution globally could only be effectively conserved in the High Intensity Zone.

Marxan with Zones uses a simulated annealing algorithm to identify near-optimal solutions, based on meeting the targets, minimising planning unit costs and maintaining connectivity (Watts et al. 2009a). This involves running the software a number of times, with each run involving a number of iterations. Each run generally produces a slightly different result, so Marxan with Zones identifies the 'best' output based on the one with the lowest cost and the 'selection frequency' output, which counts the number of times each planning unit is assigned to each zone in each of the run results. The user can influence the fragmentation levels of the selected planning unit by adjusting the Boundary Length Modifier (BLM) to increase the cost of having high fragmentation levels, and/or adjusting the zone boundary cost modifier, which reduces the cost of zones with similar management objectives sharing a boundary (Watts et al. 2009b). We ran a number of calibration exercises to determine the parameters to use in the main Marxan with Zones analyses and determined that setting species penalty values of 1.5 ensured all the targets were met, and that analysis based on 1,000 runs of 500 million iterations produced the most efficient results. Through this calibration process, we also found that setting a BLM value of 0.05 and a zone boundary cost modifier of 0.5 between Zone 1 and Zone 2 identified portfolios where none of the resultant zones were overly fragmented or extensive.

## 3.3 Results

### 3.1.6. Details on the PAs and important species

Maputaland contains 11 PAs and 19 CAs, in total covering 5,238 km$^2$, or 27.3% of the region, an increase of 1,408 km$^2$ or 7.3% of the region since 2006 (Figure 3-1). The combined area of agriculture and urbanisation is 28.6% of the planning region, with a mean annual habitat loss of 0.39% per year in the last 14 years. For the conservation features used in our analysis, we identified 45 landcover types and 212 species from seven taxonomic groups. For the species, 30.2% are globally threatened, 46.2% regionally threatened and 52.4% endemic or near-endemic to Maputaland (Table 3-1). Many of these species are threatened by over-harvesting and direct persecution, with 19.8% threatened in

Maputaland and 21.2% threatened globally; only one species is threatened in Maputaland but not threatened globally (Table 3-1).

***Table 3-1: Details of species used as conservation features in the Maputaland conservation planning system. Globally and regionally threatened species are those classified as Critically Endangered, Endangered or Vulnerable in the Global IUCN Red List and Regional Red List for southern Africa respectively. Near-endemic species are defined as those with 50% of their range falling within Maputaland. Data on whether the species are threatened by use (over-harvesting), or direct persecution come from the IUCN global Red List or regional Red List.***

| Taxon | Number of species | Globally threatened | Regionally threatened | Endemic/Near-endemic | Globally threatened by use or persecution | Regionally threatened by use or persecution |
|---|---|---|---|---|---|---|
| Amphibians | 4 | 1 | 1 | 3 | 0 | 0 |
| Birds | 33 | 11 | 29 | 0 | 13 | 12 |
| Freshwater Fish | 8 | 8 | 7 | 4 | 0 | 0 |
| Invertebrates | 38 | 7 | 5 | 34 | 0 | 0 |
| Mammals | 16 | 11 | 8 | 1 | 11 | 11 |
| Plants | 103 | 26 | 47 | 61 | 19 | 18 |
| Reptiles | 10 | 0 | 1 | 8 | 2 | 1 |
| **% of all species** | | 30.2 | 46.2 | 52.4 | 21.2 | 19.8 |

### 3.1.7. Gap analysis and spatial prioritisation

Maputaland's PAs meet targets for 19 of the 45 landcover types and 141 of the 212 species. Adding the CAs to the PAs meets an additional 2 landcover type targets and 9 species targets (Figure 3-3).

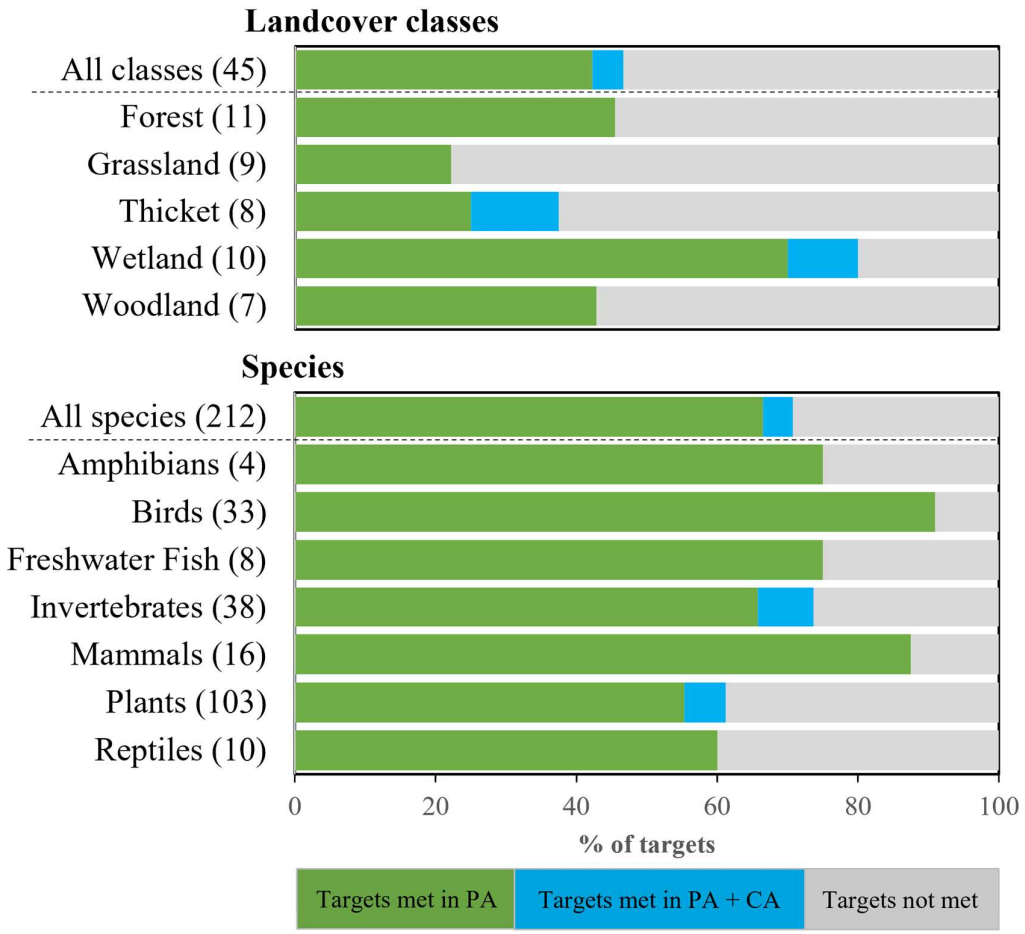



***Figure 3-3: Gap analysis showing proportion of conservation features (species and landcover types) meeting targets within the existing network of protected areas (PAs) and conservation areas (CAs) in Maputaland.***

The best portfolios produced by the program from all three scenarios identified priority areas for conservation in similar areas, with large patches in the Lubombo Mountains in the west, around Licuati Forest Reserve, Maputo Special Reserve, Tembe Elephant Park and Lake Sibiya (Figure 3-4). The combined area of the two OECM zones in the three scenarios were also similar, with 4723.8 $km^2$ for Scenario A (High Intensity Zone area = 0, Low Intensity Zone area = 4723.8 $km^2$), 4974.1 $km^2$ for Scenario B (High Intensity Zone area = 1362.0 $km^2$, Low Intensity Zone area = 3612.1 $km^2$) and 5107.3 $km^2$ for Scenario C (High Intensity Zone area = 1,733 $km^2$, Low Intensity Zone area = 3374.3 $km^2$). All of the High Intensity Zones identified in Scenario B and Scenario C are in the west of the planning region.

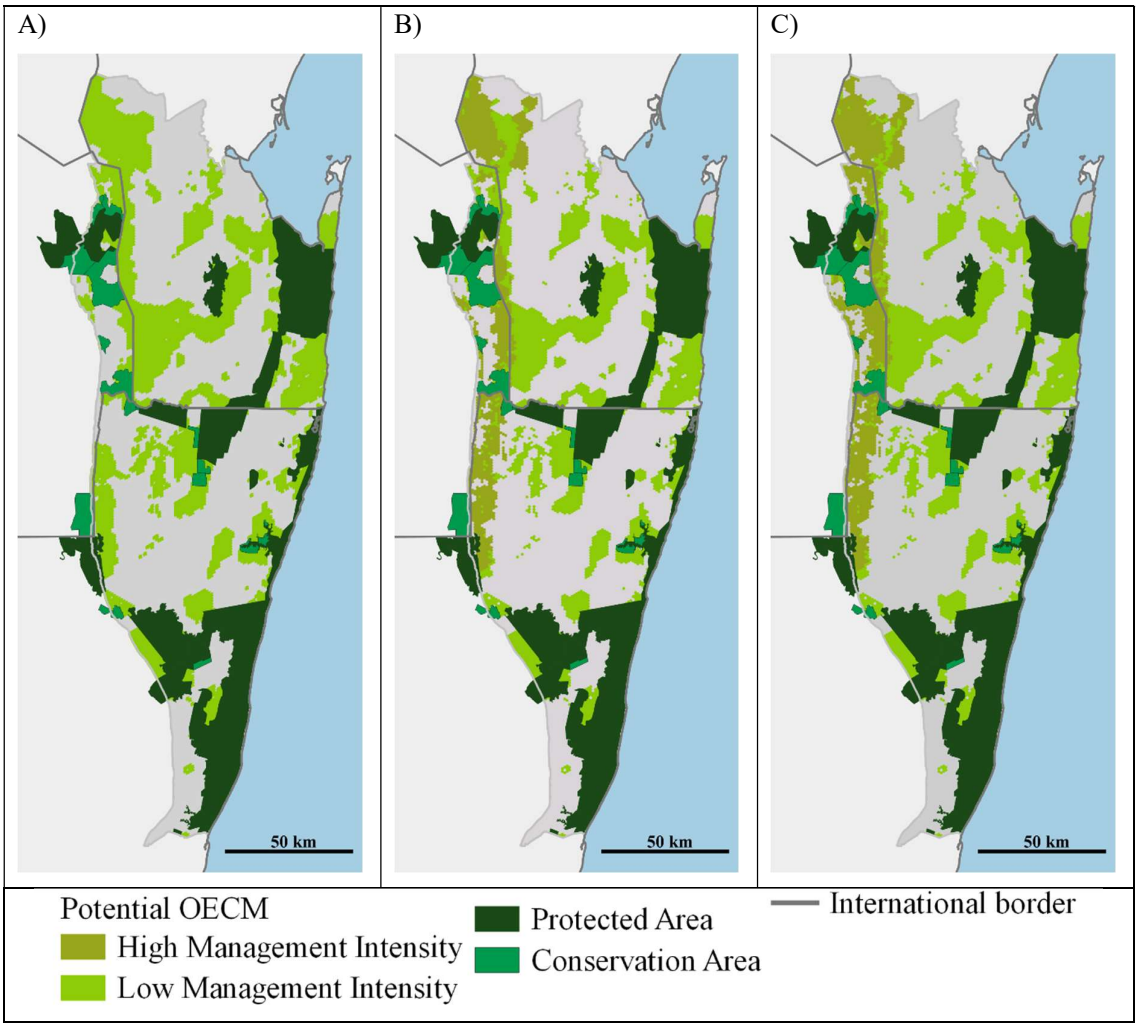



***Figure 3-4: Best portfolios produced by Marxan with Zones for the three scenarios. Scenario A assumed that all species can be found in Low Management Intensity OECMs, Scenario B assumed that species known to be sensitive to over-harvesting and direct persecution in Maputaland are not found in Low Management Intensity OECMs, Scenario C assumed that species known to be globally sensitive are not found in Low Management Intensity OECMs.***

The selection frequency results showed that most planning units were never selected to be part of Zones 1 or 2, i.e. they had a selection frequency of 0, but that the planning units that were selected were generally selected in most of the 1000 runs, i.e. had a selection frequency of 900 or more (Figure 3-5). Thus, overall levels of flexibility were low, with Marxan with Zones generally selecting the same planning units each time, based on the scenario.

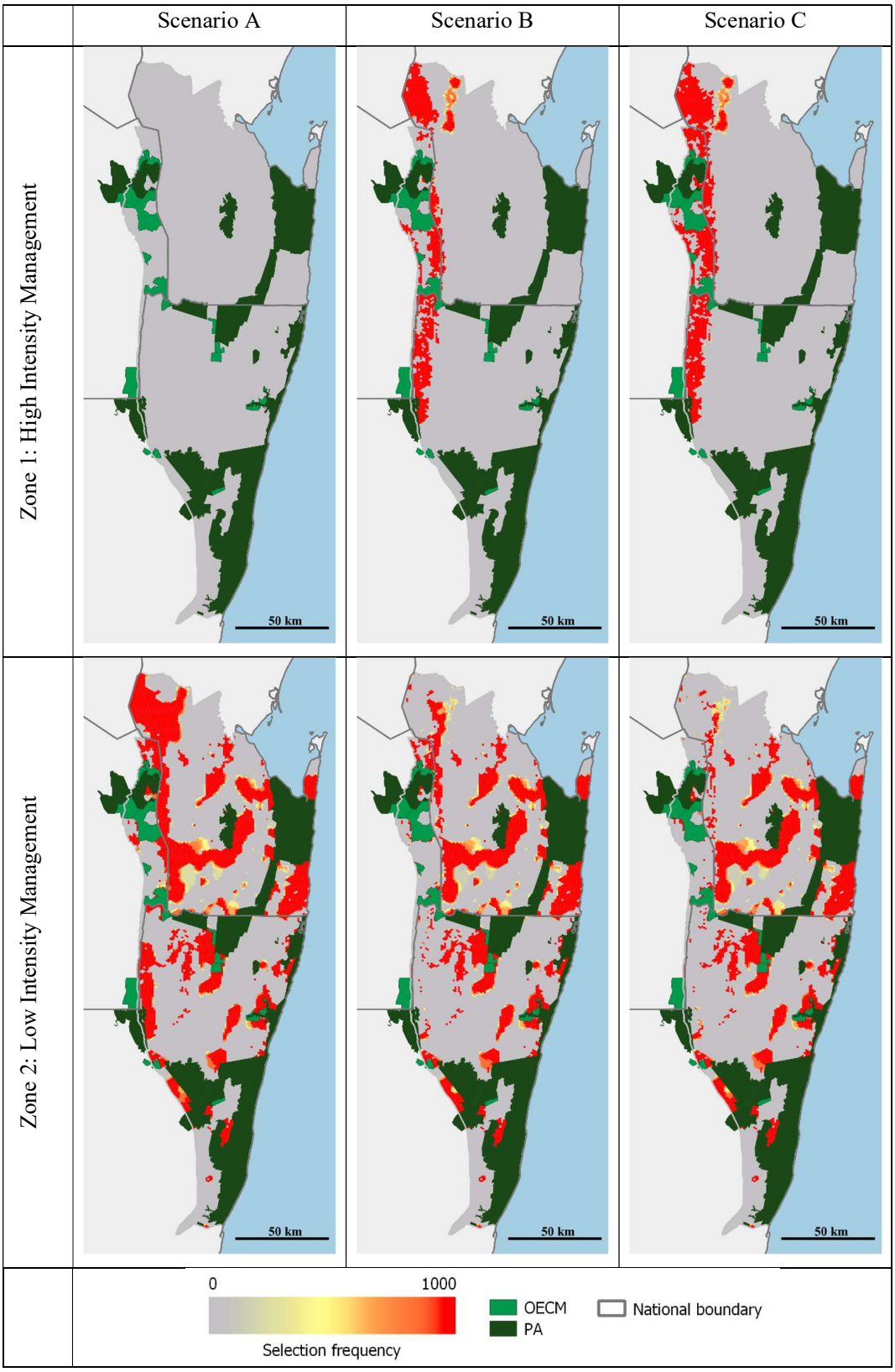



***Figure 3-5: Marxan with Zones zone selection frequency results for the three scenarios, showing the number of times each planning unit was selected in each of the 1000 runs.***

## 3.4 Discussion

### 3.1.8. Spatial prioritisation trends in Maputaland

Systematic conservation planning is designed to be an ongoing process that involves updating the underlying data to account for changes in the planning region and new data sources (Knight et al. 2006). However, few of the spatial prioritisation analyses published in the literature are repeated and so our Maputaland work provides a rare opportunity to revisit previous work. The first thing to note is that the mean rate of habitat loss was 0.39% per year, so that 29% of the region is now under agriculture or urbanised. This level of habitat loss might seem surprising, considering that the region is part of the Maputaland-Pondoland-Albany biodiversity hotspot and these globally important hotspot areas are selected in part based on their high levels of habitat loss (Myers et al. 2000; Sloan et al. 2014). This situation arises because Maputaland has low potential for commercial agriculture because of its poor-quality soils, which also explains why ecotourism is seen by government as the most viable land-use for reducing poverty. Part of the government investment to increase ecotourism has involved infrastructure development, which has dramatically reduced travel times between Mozambique and South Africa. This will boost tourist numbers, but continued benefits will depend on careful conservation planning, as the increase in accessibility has already led to people settling next to an improved road and the resultant spread of agriculture in areas containing important biodiversity.

The other main change measured through the Maputaland conservation planning system has been the increase in land managed for conservation, which went from 21.2% in 2006 (Smith et al. 2008) to 28.6% of the overlapping area in 2020. Part of this increase came from extending the Maputo National Park to the South African border, an initiative that was guided in part by the 2006 conservation planning outputs (Smith et al. 2009). However, most of the additional area has come from the formal recognition of privately- and communally-managed land that was already under conservation. This recognition has been important, especially for the communally-managed CAs, as it has increased access to funding and support from government and NGOs (De Vos et al. 2019). These funding agencies have also focused on sites that are important for biodiversity, again partly informed by the 2006 conservation planning outputs, so that the number of landcover type targets met increased from 12 in 2006 to 19 in 2020 when just focusing on PAs, and 21 when including PAs and CAs. All of this suggests that a model based on recognising and supporting the existing efforts of landowners has the potential to increase the area of land under conservation, although the current slow rate of expansion also suggests this process will need much more resourcing.

The amount of land identified through our spatial prioritisation was similar to that identified by the earlier analysis, with the new priority areas plus the existing PAs and CAs being 52 to 54% of the total land area of Maputaland, compared to 50.5% in the 2006 outputs (Smith et al. 2008). This

percentage appears high, but the region is a biodiversity hotspot and relatively untransformed, which partly explains why it already has high PA and CA coverage. Moreover, in this analysis we identified that at least two thirds of these new priority areas would only require low intensity management, which would involve lower levels of resourcing and have fewer impacts on people living there. The important change between the current and 2006 outputs is the location of the priority areas, which now include large portions of the Lubombo Mountains in the western part of Maputaland. This is chiefly because our analysis included data on four times the number of species, many of which are threatened and/or endemic plant species found in these montane habitats (Darbyshire et al. 2019).

This change has three broader implications. First, it highlights the benefits of a target-based approach that includes broad ecosystem types as conservation features (Pressey et al. 2003). Second, it shows how sensitive spatial conservation planning can be to the inclusion of narrow-range species (Butchart et al. 2015). This is because data on these species reduces the flexibility of the area which can be selected to meet conservation targets. This outcome has important implications for real-world conservation planning, especially as many of the region's plants and invertebrates are still poorly known. Third, it illustrates the benefits of collecting data on these species. For our analysis, we benefited from using a range of new species distribution data that has been collected or collated during the previous 15 years (Darbyshire et al. 2019; Monadjem et al. 2021). But more importantly, we collected new field data for 62 plant species. This fieldwork took places over 6 weeks and was relatively inexpensive, but it provided invaluable information to produce more accurate species distribution maps, which in many cases identified new important areas but also showed that some species are more widespread than previously described.

### 3.1.9. OECMs and management effectiveness

During the 2010 to 2020 period covered by the CBD's Aichi targets, the global extent of terrestrial PA coverage only increased from 15% to 17% (Rodrigues & Cazalis 2020). This reflects the difficulty of creating new PAs, especially state-governed PAs (Watson et al. 2014). With the Global Biodiversity Framework Target 3 calling for 30% of the terrestrial realm to be managed for conservation (CBD 2021), OECMs are expected to play a much larger role as they do not depend on changing existing governance or management systems. Instead, creating new OECMs will largely involve documenting and giving official recognition to ongoing efforts, engaging with landowners and forming wider, more diffuse networks of land managers (MacKinnon 2020). In some cases, it could be argued that recognising an OECM will make no difference on the ground, as by definition these sites have been envisaged to guarantee positive and sustainable long-term *in situ* conservation outcomes (Pressey et al. 2021). However, there are two main ways in which OECM status is likely to improve conservation outcomes. First, they will have more legal recognition and so could be given

better protection from outside threats, such as infrastructure development. Second, they could be the focus of additional resourcing from government and non-governmental groups to maintain or increase their biodiversity value, in terms of support, advice and funding (Alves-Pinto et al. 2021).

Both the processes of legal recognition and providing additional resources have a range of associated costs (Naidoo et al. 2006). This emphasises that the systematic conservation planning approach is just as relevant for identifying where to establish new OECMs as it is for establishing new PAs. However, such planning needs to be explicit about the type of management actions needed, when managing species that are threatened by over-harvesting or direct persecution, as these species are absent from many landscapes unless directly managed (Maxwell et al. 2016). We accounted for this in our analysis by planning for two types of OECM. The low management intensity OECMs would be similar to existing CAs, which are important for conserving landcover types and many species in areas where subsistence and commercial harvesting and poaching have impacted a number of sensitive species. The high intensity management OECMs are explicitly designed to account for this over-harvesting and poaching, but they would need additional resources at similar levels to existing PAs to ensure the conservation of the sensitive species. This could involve management to protect large mammals and birds from hunting for meat, large carnivores from direct persecution and a range of species from medicinal use. Many of these species have potentially wide ranges, so the existing PAs in Maputaland already meet targets for 25 of the 27 animal species that are regionally threatened. However, 13 of the plant species threatened by over-harvesting are under-represented in the existing PA network, which explains why most of the priority areas selected for OECMs occur in the Lubombo Mountains that are rich in endemic plant species. Ensuring the persistence of these species should be a collaboration with relevant stakeholders and could involve protection of vulnerable locations within the OECMs, as well as projects to encourage the propagation of plants.

When analysing the extent to which the existing PAs and CAs meet the species targets, our expectation is that the existing PAs are as effective as the proposed high intensity management OECMs, and the existing CAs are as effective as the proposed low intensity management OECMs. At present, we know that this assumption for PAs is false for at least some species. An obvious example is that while the PAs in Eswatini and Mozambique contain suitable habitat for white rhino and black rhino, these species are only found in Maputaland in the South African PAs (African Rhino Range States 2015). Thus, our spatial prioritisation assumes that every existing PA, as well as proposed new high intensity management OECMs, will be effectively managed so that they contain the actual, rather than potential range of every one of the species included in our analysis. This will not be straightforward and will involve extra investment, but a wealth of evidence highlights the benefits of increasing the management effectiveness of the existing PAs as part of developing ecological networks (Coad et al. 2019).

### 3.1.10. Broader implications

Our work in Maputaland provides a number of insights with broader relevance for systematic conservation planning and developing ecological networks. The first concerns the slow progress, even though the region has been the focus of trans-frontier conservation area initiatives for more than two decades. This is evidenced by the fact that the land under conservation management has only increased from 20% to 27.3%, falling well short of the 52% to 54% needed to meet the targets we identified through our analysis. Moreover, some of the actual increase came from changes in South African legislation to recognise some of the private game reserves as PAs, which highlights the need to account for a wider range of conservation area types when assessing conservation area extent (Sykes 2020). The second insight is the importance of having a mixed set of PAs and CAs in the region, and how this will be further strengthened by the addition of new OECMs. This mix increases the legitimacy of an ecological network by broadening the type and number of groups and organisations involved in governance and management (Donald et al. 2019). It also diversifies funding sources in such a way that the state continues as the main contributor, but some funds will also come from ecotourism programs, sustainable use of natural resources and donors. Lastly, the privately- and communally-managed sites help meet targets for under-represented biodiversity elements and increase the extent and connectivity of ecological networks.

The complexity of guiding the development of such networks highlights the great benefits from using systematic conservation planning, but it also shows how traditional approaches for targeting globally important biodiversity may need to be updated. The traditional approach seeks to create new PAs based on these factors: (i) global analyses to identify priority regions, (ii) regional analyses to identify priority areas within each priority region, and (iii) local scale implementation projects. The latter are based on purchasing land, compensating land-owners or supporting land-owners to designate their own PAs, which then informs the delineation of the boundaries of the resultant conservation areas (Adams et al. 2019). Our Maputaland analysis fulfils this regional role but within the context of developing new OECMs, there is a need for finer scale analyses that account for landowner willingness under different conditions and scenarios. In particular, research is needed to understand, firstly, what sort of conservation interventions the landowners would be willing to implement, secondly, how that willingness might vary with external support and funding, and finally, the impacts that these interventions would have on the different focal species. Co-developing this research with local communities would also provide opportunities to gain from their insights and collect fine-scale species distribution data.

# 4 Identifying a network of Key Biodiversity Areas for Mozambique


## Abstract

The world is facing an ecological crisis resulting from unprecedented rates of biodiversity loss. To reverse this trend, nations signatories to the Convention on Biological Diversity (CBD) have committed to conserve at least 30% of the marine and terrestrial realms by 2030, focusing in part on areas that are critical for biodiversity. One way to do this is to identify Key Biodiversity Areas (KBAs) to inform the strategic development of networks of conservation areas. However, we lack case studies showing how to implement the KBA identification process in high-biodiversity, low-income countries. Here we present an example from Mozambique that delineated KBAs at national scale across seven taxonomic groups. As part of this, we outline a post hoc theory of change that underpinned the project, giving it a clear and adaptive plan for meeting the stipulated targets. We then describe: (1) how the project developed the institutional structures and training to build the necessary capabilities and capacity; and (2) the process of identifying marine and terrestrial KBAs. Through this process, we carried out the Red Listing of species and identified 29 KBAs. We also explain the important role of the Ministry of Land and Environment, which helped ensure the country now has a rigorous standardised process for producing important biodiversity data, and mainstreaming the information into decision-making on biodiversity and environmental matters. The KBA system established in Mozambique demonstrates that this process can be implemented in low-income nations, showing that other countries could follow this lead with appropriate financial support.

## 4.1 Introduction

The world is facing unprecedented rates of biodiversity loss (Butchart et al. 2010; Díaz et al. 2020). In response, the signatories of the UN Convention on Biological Diversity (CBD) have agreed on the Kunming-Montreal Global Biodiversity Framework; a plan with 23 targets designed to improve biodiversity conservation by 2030 (CBD 2022). Under Target 3, parties commit to conserve at least 30% of marine and terrestrial areas by 2030, with a focus on areas that are important for biodiversity and ecosystem services (CBD 2022). This focus is needed as existing protected area (PA) network are generally biased towards areas that are remote and unsuitable for agriculture, leaving many species and ecosystems under-represented (Joppa & Pfaff 2009; Butchart et al. 2015; Venter et al. 2018). Research suggests that placing PAs in more representative locations would increase the number of species under protection by 37 times the current levels (Venter et al. 2014; Wauchope et al. 2022). Moreover, without addressing this problem, increasing the area of land under conservation to meet targets based on percentage coverage will not help the international community address the extinction rate of species or the loss of ecosystems (Visconti et al. 2019). Thus, we need a harmonised approach to guide identification of sites of importance for biodiversity, as part of ensuring that future conservation areas help fill the current gaps (Schaaf & Clamote Rodrigues 2016).

The Key Biodiversity Area (KBA) approach was designed to identify sites that contribute significantly to the global persistence of biodiversity (IUCN 2016). The KBA methodology was developed through strong engagement with taxonomic experts, conservation practitioners and stakeholders, and provides an umbrella approach that captures and harmonises existing approaches for identifying globally important sites for biodiversity. In this way, it is designed to help overcome biases in selecting areas that are important for biodiversity, as well as where monitoring is needed (Nogueira et al. 2021). The KBA methodology is a criteria-based approach, with quantitative thresholds, designed to identify important areas for biodiversity in an objective, repeatable and transparent manner (IUCN 2016). It involves gathering global data on biodiversity elements, then combining it with local data and expertise to identify sites that contain at least a threshold proportion of the global distribution of important species and ecosystems. In this way, identifying KBAs could play a major part in ensuring the scaling up of an effective implementation of the Global Biodiversity Framework (Perino et al. 2022).

Meeting these targets depends on each of the 196 countries that are signatories to the CBD fulfilling their commitments. Thus, KBA identification should ideally be carried out at the national level with government leadership. Such government involvement is important because it also builds pride and ownership, and improves mainstreaming and implementation across a wide range of relevant sectors (Perino et al. 2022). However, most countries lack the capacity and capability to carry out such

analyses, and this is a particular issue for the many tropical countries with high levels of biodiversity but limited species distribution data and conservation resources. Fortunately, the KBA approach was designed with these limitations in mind (Meyer et al. 2015). In particular, each site is assessed independently against global data, with no requirement for information about other sites within the country to have been collected (Brooks et al. 2015). In addition, it only needs one species or ecosystem to meet one sub-criterion and its associated threshold to trigger KBA status, therefore identifying important sites is not dependent on surveying for a range of taxa, and the process entails incorporating stakeholders and local, regional and international experts to engage and fill the knowledge gaps (KBA Standards and Appeals Committee 2022). All of this suggests that the KBA methodology could be applied successfully in any country, but we lack case studies from lower-income countries to test this and understand what is needed to build the required capabilities and capacity. Here we provide such a case study from Mozambique, explaining the steps involved and presenting the results.

Mozambique is a country in south-east Africa (Figure 4-1) that overlaps with three biodiversity hotspots (Mittermeier et al. 2004), partly because it contains 6,280 plant species, 9% are found nowhere else (Darbyshire et al. 2019). In addition, 4,270 animal species have been recorded, and it contains a wide range of terrestrial, marine and coastal ecosystems (MITADER 2019). The country is also one of the world's poorest, with limited resources for identifying and mapping its important species. As part of meeting its international commitments, the Government of Mozambique through its Ministry of Land and Environment recently oversaw the process of identifying KBAs at a national scale across different taxonomic groups. However, achieving this was not just a case of collecting and analysing the available data, as there was also the need to develop the systems needed to underpin the identification and implementation process. Here we: (i) present a theory of change for the KBA process and describe how the Government of Mozambique developed the institutional structures and training to build the necessary capabilities and capacity; (ii) document the process used to identify marine and terrestrial KBAs in Mozambique, and (iii) document the globally important biodiversity that the resultant KBA network supports.

## 4.2 Methods

### 4.2.1 Theory of change

As a signatory to the Convention on Biological Diversity, Mozambique is committed to biodiversity conservation and focused on improving the mechanisms by which biodiversity and environmental matters are incorporated into conservation and development decision-making. As part of this, the country recognised the value of having comprehensive baseline information on the geographical and

conservation status of nationally and globally important species. This is why the Ministry of Land and Environment worked closely with the Wildlife Conservation Society (WCS) on a three-year, quarter of a million dollar project to identify KBAs in Mozambique. The project structure was based on previous successful initiatives, adopted to account for local conditions and it can be summarised using a post-hoc theory of change (e.g. Ivanova et al. 2016). A post-hoc theory of change is one developed after the completion of a project to help measure success and inform future work by explaining the steps involved using a standardised reporting approach (Figure 4-2). Developing a theory of change involves defining the long-term goals for a project or strategy and then mapping backwards to identify the necessary preconditions for success, and they are increasingly used in conservation to help understand and monitor project effectiveness (Rice et al. 2020).




***Figure 4-1: Mozambique's provinces, including Maputo City, which also has provincial status.***

The Mozambique KBA project began in 2019 and recognised that identifying these important sites involved achieving a number of preparatory stages (Figure 4-2). At the institutional level, the first step was to develop a group consisting of three people from WCS (including the lead author of this chapter) and two from the Ministry of Land and Environment who would together coordinate the project. This group then organised the formation of the data management team. This data management team consisted of eight people of recently graduated biologists (hereafter "data management

assistants") who gathered and prepared data. Finally, this entailed establishing a KBA National Coordination Group (IUCN 2016) consisting of 20 national institutions including governmental, academia, research institutes, civil society and private sector, whose role is to facilitate the KBA identification process, help build consensus among the different stakeholders and validate the results of the KBA process at national level.

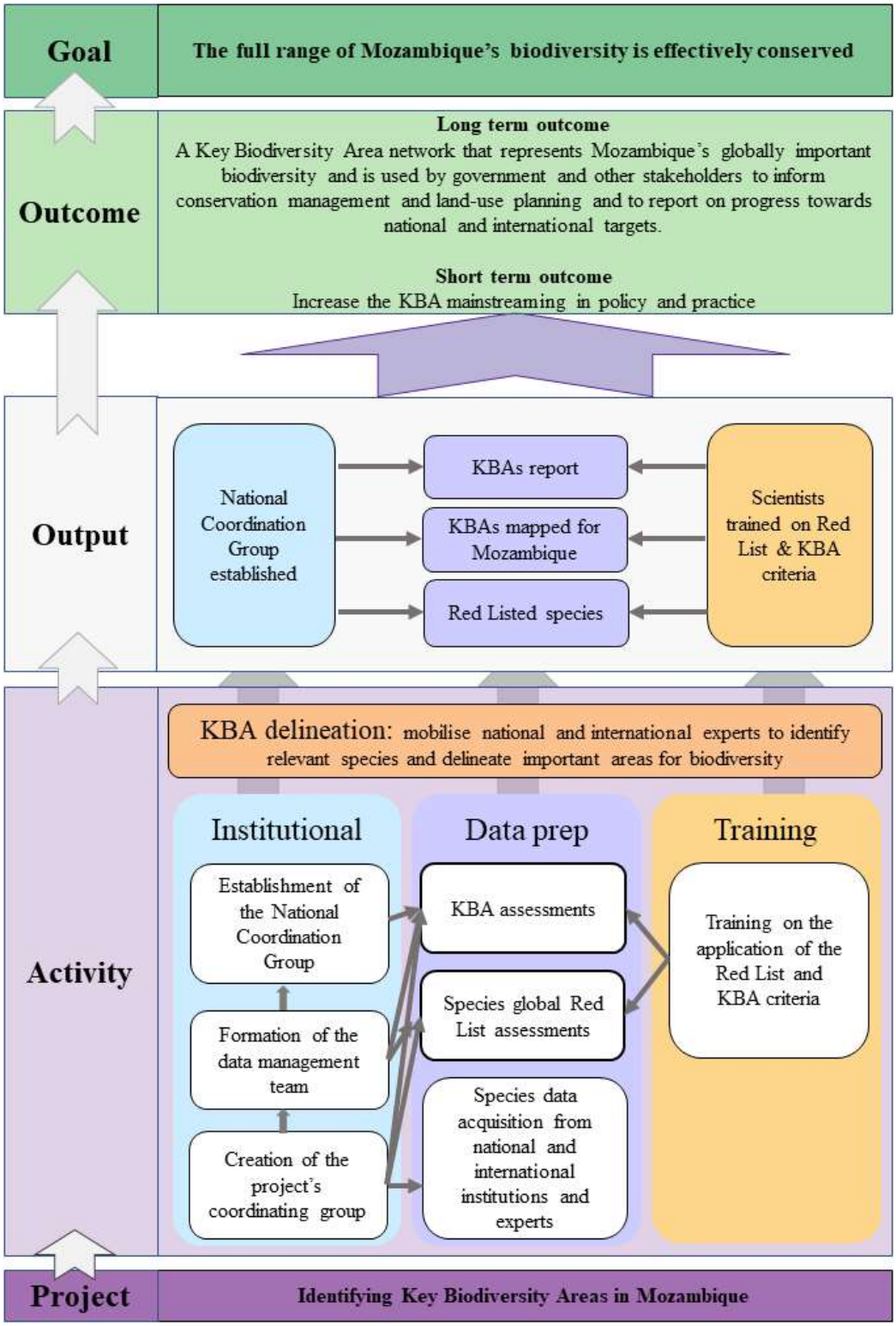



***Figure 4-2: Post-hoc theory of change for the Mozambique's KBAs analysis.***

This KBA National Coordination Group was also responsible for organising IUCN Red List (extinction risk) assessments of key taxonomic groups, mostly based on the data that had been collated by the data management assistants. This is because identifying KBAs depends on knowing the Red List status of the focal species and such data were absent for a number of taxonomic groups. Red List assessments for plants had already been conducted independently by in-country and international botanical expertise, primarily as part of the Tropical Important Plant Areas programme in Mozambique (Darbyshire et al. 2023) and through the IUCN-SSC Southern African Plants Specialist Group. Therefore, the Coordination Group ran one workshop in 2019 to provide training for 41 taxonomic experts from Mozambique, Eswatini and South Africa so they could then apply the Red Listing process to amphibians, freshwater fish, invertebrates and reptiles. As part of the training component of the project (Figure 4-2), the Coordination Group also organised and ran one workshop for training academics, scientists, taxon experts and relevant stakeholders to apply the KBA standard, making sure they became familiar with the KBA criteria and associated thresholds and could apply them correctly. This workshop involved 70% of the experts involved in the Red Listing training and assessment, as well as 29 additional experts, scientists and stakeholders.

### 4.2.2 Species data acquisition

The initial steps for applying the KBA methodology involve compiling a list of species (hereafter trigger species) that are likely to trigger the KBA criteria and thresholds. We developed this list based on both existing global and national priority species lists (Lim et al. 2023), excluding those without a recent confirmed occurrence in the study region (IUCN 2016). We focused on three types of species. The first was species that have been assessed globally under the IUCN Red List and their data was available for download from the IUCN Red List web site (Table S4-1). The datasets available consisted of species locality records, range map, and number of mature individuals (IUCN 2016).

For the second type of species we focused on endemic species, which are those whose global distribution range is entirely restricted to a defined geographic area such as a region, country or specific site (IUCN 2016). For this study we concentrated on what are known as national endemics (Kraus et al. 2023), which in this case are endemic species with their global range wholly restricted to Mozambique and their conservation can only be ensured within the country. Datasets for these species were primarily obtained from experts in each taxonomic group, but we also gathered data from herbaria and museum databases and the literature. The third type was the range restricted species, which are those with a global distribution range less than or equal to 10,000 $km^2$ (IUCN 2016). Almost all the endemic species included here qualify as range restricted species, but we considered this third group as a way of capturing those species with their distribution range not entirely within

Mozambique and not greater than 10,000 km$^2$. Data for these species were again obtained primarily from experts but also from herbaria and museum databases.

### 4.2.3 KBA assessment

Species-based KBA assessments consist of evaluating the species data at a particular site against the KBA criteria and its associated quantitative thresholds to verify whether the site contains a large enough proportion of the trigger species (Table 4-1). Several different types of metric can be used to calculate these proportions, i.e., number of mature individuals, extent of suitable habitat, range and number of localities. Meeting one criterion or sub-criterion is sufficient for a species in a particular site to trigger KBA status. However, KBA proposers are encouraged to apply as many KBA criteria as possible and broaden this to include a wide range of taxonomic groups in the same site, given data availability, in order to strengthen the KBA status attained by this site. Our dataset was adequate to apply Criteria A1: threatened species, B1: individual geographically restricted species, B2: co-occurring geographically restricted species and D1: demographic aggregation (IUCN 2016).

***Table 4-1: Key Biodiversity Area criteria and threshold applied in the Mozambique assessment.***

| **A. Threatened biodiversity** | |
|---|---|
| A1a | ≥0.5% of global population size and ≥5 reproductive units (RU) of a CR/EN species |
| A1b | ≥1.0% of global population size and ≥10 RU of a VU species |
| A1c | ≥0.1% of global population size and ≥5 RU of a species listed as CR/EN due only to past/current decline [= Red List A1, A2, A4 only] |
| A1d | ≥0.2% of global population size and ≥10 RU of a species listed as VU due only to past/current decline [= Red List A1, A2, A4 only] |
| A1e | Effectively the entire population size of a CR/EN species |
| **B. Geographically restricted biodiversity** | |
| B1. Individual geographically restricted species | ≥10% of global population size and ≥10 RU of any species |
| B2. Co-occurring geographically restricted species | ≥1% of global population size of each of a number of restricted range species in a taxonomic group: ≥2 species or 0.02% of the total number of species in the taxonomic group, whichever is larger |
| **D. Biological processes** | |
| D1. | Demographic aggregations |
| D1b | Site is among largest 10 aggregations of the species |

As over 60% of our dataset consisted of GPS records, we mostly based our assessments on the number of localities. A locality is defined as “a discrete population to the extent this can be inferred, given the degree of habitat fragmentation and what is known about the dispersal capabilities of the

species" (IUCN 2022). This assessment parameter was adequate, given that the dataset we worked with consisted of many species with highly restricted distributions. In such cases, it is easy to determine the number of localities, based on the information about the number of locations on the IUCN Red List. A location is a distinct area where a single threatening event can have a rapid negative impact on all the individuals of a taxon present (IUCN 2012). Given the degree of habitat patchiness, each of the locations provided on the IUCN Red List for these species consisted of discrete populations, therefore, representing an adequate proxy to determine locality.

We also used the number of mature individuals and range as other assessment parameters, particularly for bird and mammal species. These taxonomic groups have a much broader range and have been well studied, so the global dataset was obtained from the IUCN Red List. Information about these species in Mozambique was obtained from taxon experts and national population census reports from the National Administration of Conservation Areas. KBA assessment for these widely distributed species consisted of identifying the existing management units throughout Mozambique that contained high enough proportions of the total range or mature individuals to trigger KBA status.

### 4.2.4 KBA delineation and validation

The next step was to delineate geographic boundaries for potential KBA sites. According to the KBA framework, boundaries should be drawn in a way that the proposed KBA site is ecologically relevant and suitable to be managed as one site. We derived the initial boundaries by overlapping the distributional data for each of the KBA trigger species per site. These initial boundaries are known as ecological boundaries (KBA Standards and Appeals Committee 2022). For species with a limited number of records, we estimated the geographic distribution based on taxon experts' knowledge about habitat requirements of the species. We then improved these boundaries by taking into account any existing important sites for biodiversity. This step is necessary because in some cases a newly proposed KBA site might intersect with an existing important area for biodiversity, such as an Important Bird Area or an Alliance for Zero Extinction site. In such cases, the KBA delineation involves building consensus with proposers of the existing site to foster agreement on modifying the boundaries where necessary.

The next step consists of refining the ecological boundaries to produce practical KBA boundaries. This involves accounting for existing management units, such as conservation areas, and considering land ownership and stakeholders' input. KBA proposers are encouraged to consult with various stakeholders, particularly those with knowledge about biodiversity, local tenure and resource management (KBA Standards and Appeals Committee 2022). We brought together a wide range of experts and stakeholders, including civil society, in a workshop where discussions created consensus

on the delineation of the boundaries for each potential KBA site. Where areas overlapped with an existing PA, boundary delineation followed the borders of that PA. When the delineation was completed, we reassessed the sites to check whether they contained enough of each KBA trigger species to meet the thresholds.

Once this process was completed, we submitted the KBA draft assessments to Mozambique's National Coordination Group for evaluation and validation. Because the KBA process accounts for strong involvement of local experts and stakeholders, this Group plays a key role in supporting the whole KBA identification process. After we addressed the follow-up comments and suggestions provided, the National Coordination Group validated the KBA proposals and submitted them to the KBA regional focal point where another evaluation was conducted. This entailed checking whether the KBA criteria had been applied correctly and that all relevant information had been included. Resulting from this process, one site was excluded out of the 30 submitted to the KBA regional focal point. The remaining 29 KBA proposals were submitted to the Global KBA secretariat for final evaluation and publication. On the conclusion of this final stage, the Mozambique government officially acknowledged the successful KBA results and we presented this in a workshop led by the Ministry of Land and Environment.

## 4.3 Results

### 4.3.1 KBA trigger species, endemism and IUCN red list status

The Mozambique KBA analysis comprised 180 species (Table S4-1). These consisted of eight amphibians, 11 birds, eight freshwater and six marine fish, 11 insects, two marine and six terrestrial mammals, 102 plants and 26 reptiles. Of the 180 species, 128 are endemic to Mozambique, most of which are plants (83 species) and reptiles (25 species). The percentage of the assessed species in each taxonomic group that are endemic ranged from 96.2% for reptiles to 12.5% for mammals (Figure 4-3a). For threat status, 163 of these species have been evaluated on the IUCN Red List, leaving 17 species of fish, insect and reptile species not evaluated. Of these assessed species, 75.0% are listed as Critically Endangered, Endangered or Vulnerable, and this ranged between 100.0% for the assessed mammals to 89.2% for plants (Figure 4-3b).

### 4.3.2 KBA criteria and trigger species

The 180 species used in the analysis triggered three high-level KBA criteria, namely A Threatened biodiversity, B Geographically restricted biodiversity and D Biological processes (Table 4-1).

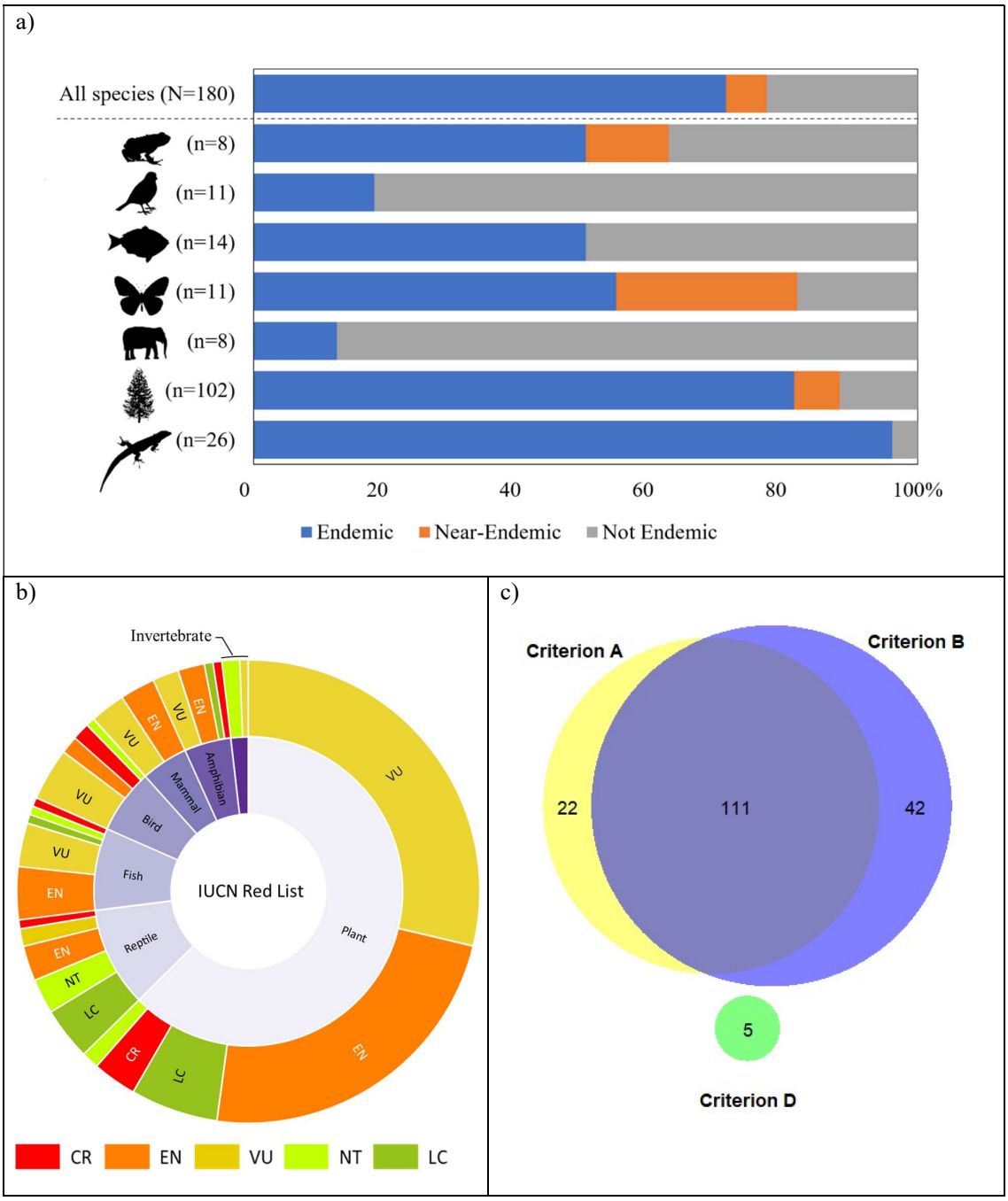



***Figure 4-3: (a) Levels of endemism per taxonomic group; (b) species global conservation status per taxonomic group based on the IUCN red list of threatened species; (c) overall illustration of trigger species under Criterion A, B and D and those triggering both A and B.***

A total of 133 species triggered Criterion A, with 22 species triggering no other criteria (Figure 4-3c). Most of these species (37.2%) triggered Sub-Criterion A1b because they are Vulnerable, followed by 34.4% triggering A1a because they are Critically Endangered or Endangered species. Fewer species triggered the other Criterion A sub-criteria, with 9.4% triggering A1e because they are Critically

Endangered or Endangered species and their global population is only found at one site, 3.3% triggered by A1c because they are Critically Endangered or Endangered and threatened by current or past population reduction and 2.2% triggered by A1d because they are Vulnerable and threatened by to current or past population reduction.

We found 153 triggered Criterion B, with 42 species triggering no other criteria (Figure 4-3c). These species triggered two sub-criteria, with 84.4% triggering B1 because they are geographically restricted to one site across all their life stages, and 15.6% triggering B2 because they were found in sites with multiple co-occurring geographically restricted species (Table 4-1).

Only five species triggered Criterion D, although all of them triggered no other criteria. They all met Sub-Criterion D1b, as they were found at sites where their population of mature individuals is among their 10 largest aggregations worldwide (Table 4-1).

### 4.3.3 Biodiversity sites of global importance

We identified 29 KBAs in Mozambique with a total area of 139,947 $km^2$. This consists of 25 terrestrial sites, which cover 17% of the country's continental territory, and four marine sites, which cover 1% of the Exclusive Economic Zone (Figure 4-4a). We found 24 sites contain species that are triggered by Criterion A and Criterion B, one site that is triggered by Criterion A and Criterion D, two sites contain species that are only triggered by Criterion A and two sites contain species that are only triggered by Criterion D (Table 4-2).

Of these sites, ten hold the entire population size of species that are Critically Endangered or Endangered and trigger Criterion A1e (Figure 4-4b). The percentage of sites containing species that triggered the other different A1 sub-criteria ranged from 10% for A1d (Vulnerable species threatened by past or current decline) to 72% for A1a (Critically Endangered or Endangered species). We also found that 23 sites contain species that trigger Criterion B1 because of their restricted distribution, and three sites trigger Sub-Criterion D1b because they are among the largest 10 aggregations known for the species worldwide.

### 4.3.4 KBAs overlapping with formal network of PAs

We found that 85% of the total extent of Mozambique's KBAs falls partially or totally within an existing conservation area. The proportion varies from 48% (n=14) overlapping with PAs, to 20% (n=6) with forest reserves. Sites designated based on international conventions, such as Ramsar sites, World Heritage Sites and Biosphere Reserves, cover 17% (n=5) of the KBA network.

a) b)

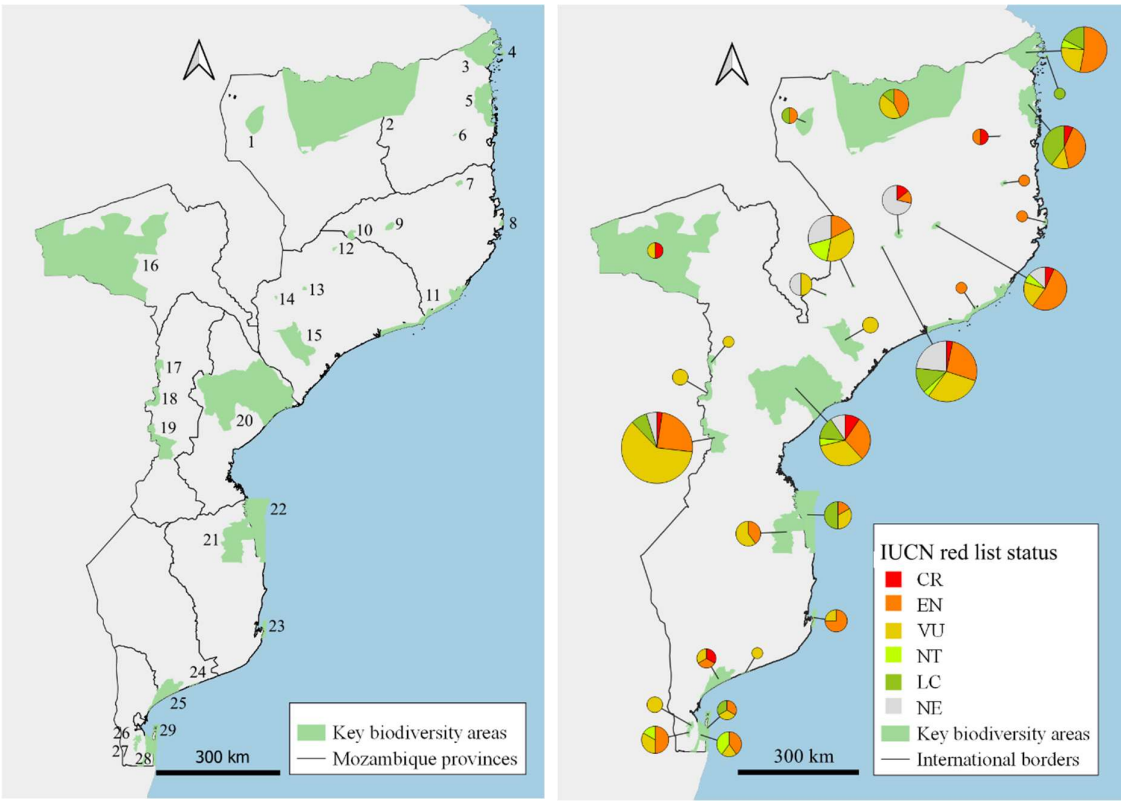



***Figure 4-4: (a) The 29 Key Biodiversity Areas identified in Mozambique. The numbers refer to the sites listed in Table 4-2.; (b) Proportions of species and their global IUCN threat status per KBA.***

***Table 4-2: List of Key Biodiversity Areas of Mozambique, including the province in which they are located, the number of trigger species they contain and the KBA sub-criteria of these trigger species***

| ID | KBA name | KBA area ($km^2$) | Province | Species | KBA sub-criteria of trigger species |
|---|---|---|---|---|---|
| 1 | Njesi Plateau | 1,996 | Niassa | 2 | A1a; A1e, B1 |
| 2 | Niassa Special Reserve | 42,708 | Niassa | 7 | A1a; A1b; A1c; A1d, B1 |
| 3 | Palma | 4,556 | Cabo Delgado | 17 | A1a, A1b, A1e, B1, B2 |
| 4 | Vamizi | 87 | Cabo Delgado | 1 | D1b |
| 5 | Quiterajo | 3,064 | Cabo Delgado | 15 | A1a, A1b, A1e, B1, B2 |
| 6 | Taratibu | 25 | Cabo Delgado | 2 | A1a; A1e, B1 |
| 7 | Eráti | 148 | Nampula | 1 | A1a, B1 |
| 8 | Matibane Forest Reserve | 109 | Nampula | 1 | A1a, B1 |
| 9 | Ribáuè-Mphalwe | 265 | Nampula | 15 | A1a; A1b; A1e; B1; B2 |
| 10 | Mount Inago | 326 | Nampula | 7 | A1a, A1e, B1, B2 |
| 11 | APAIPS | 2,512 | Zambezia | 1 | A1a, B1 |
| 12 | Mount Namuli | 53 | Zambezia | 30 | A1a; A1b; A1e; B1; B2 |
| 13 | Mount Mabu | 61 | Zambezia | 17 | A1a, A1b, B1; B2 |

| | | | | | |
|---|---|---|---|---|---|
| 14 | Mount Chiperone | 36 | Zambezia | 4 | A1b, B1, B2 |
| 15 | Derre Forest Reserve | 3,984 | Zambezia | 2 | A1b, B1 |
| 16 | Tchuma-Tchato Cahora Bassa Lake | 38,180 | Tete | 2 | A1a, A1b, A1c, A1d |
| 17 | Serra Choa | 516 | Manica | 1 | A1b |
| 18 | Machipanda | 756 | Manica | 2 | A1b |
| 19 | Chimanimani National Park | 2,371 | Manica | 41 | A1a; A1b; A1e; B1 |
| 20 | Gorongosa and Marromeu Complex | 23,088 | Sofala | 21 | A1a, A1b, A1c, A1d, A1e, B1 |
| 21 | Inhassoro-Vilankulos | 5,357 | Inhambane | 5 | A1a; A1b; B1 |
| 22 | Great Bazaruto | 5,236 | Inhambane | 7 | A1a, A1b, A1c, A1d, B1 |
| 23 | Tofo | 342 | Inhambane | 4 | D1b |
| 24 | Chongoene | 33 | Gaza | 1 | A1a, B1 |
| 25 | Manhiça-Bilene | 2,070 | Gaza | 3 | A1a, A1b, A1e, B1 |
| 26 | Matutuine | 195 | Maputo | 2 | A1b, B1 |
| 27 | Licuáti Forest Reserve | 141 | Maputo | 6 | A1a, A1b, B1 |
| 28 | Maputo National Park | 1,040 | Maputo | 5 | A1a; A1b; B1 |
| 29 | Ponta do Ouro Partial Marine Reserve | 698 | Maputo | 3 | A1b; A1c, D1b |

## 4.4 Discussion

### 4.4.1 Mozambique's KBAs

Mozambique has high levels of species richness and endemism, as illustrated by the fact that it overlaps with three regional biodiversity hotspots (Marchese 2015) and six centres of plant endemism (Darbyshire et al. 2019). However, as is common for countries in the tropics (Meyer et al. 2015; Stephenson & Stengel 2020), there is relatively limited data on most species across different taxonomic groups within the whole country (Monadjem et al. 2010; Tolley et al. 2016; Neves et al. 2019). Despite this, we were able to successfully complete the KBA assessment process, identifying 29 KBAs throughout the country (WCS et al. 2021a). This built on work that had been going on for more than a decade to bring together the available distribution data for the different taxonomic groups, and the effort needed to produce this information should not be under-estimated. But the KBA assessment itself was largely dependent on the success of the six days of intensive training involving approximately 50 people (WCS et al. 2021b), suggesting that it would be relatively straightforward to repeat the process in other countries with similarly limited resources.

It is, however, worth noting the impacts of the limited biodiversity data on the KBA assessment process (Handley et al. 2023). One important aspect is that most KBAs were triggered by the presence of a small number of important species, with six sites containing only one species and five sites containing two species (Table 4-2). Moreover, most of those species were triggered under Criterion B, which is based on limited geographic range (IUCN 2016). Therefore, it could be that some of these

species only appear to have limited range because they are under-sampled, and they may stop being trigger species once their ranges are better mapped (Lomba et al. 2010). Nevertheless, it has been shown that for some species a small number of records is the true reflection of their rarity (Rivers et al. 2011; Roberts et al. 2016). Additionally, it is also possible that sites may contain other important species for which data are not available. Failing to detect a species when it is actually present can result in false absences (Chen et al. 2013), leading to inadequate estimates of its distribution and population size and resulting in poor management decisions, particularly for rare and threatened species (Moore et al. 2011; Alexander et al. 2012; Zhang et al. 2014). Thus, there is a high likelihood that many more KBAs will be identified in Mozambique once more distribution data on more species are collected. For example, in our study we also shortlisted 15 potential sites that may be identified as KBAs once more data becomes available, and another study identified Important Plant Areas in additional locations based on new data on threatened species (Darbyshire et al. 2023).

Nevertheless, the KBA process has been designed to be applicable in data poor conditions (Meyer et al. 2015). It is applied on a site-by-site basis without needing knowledge about other sites, as long as the global population has been estimated (Brooks et al. 2015). Moreover, the KBA approach has the flexibility to allow assessment of a species against as many criteria as possible (KBA Standards and Appeals Committee 2022). This helps overcome issues with data availability. For example, for this study four marine species only qualified as KBA triggers when assessed under Criterion D1 (demographic aggregations), rather than under Criterion A1 (threatened species), despite their threatened status under the IUCN Red List. These were the Vulnerable Reef Manta Ray (*Mobula alfredi*) and the three Endangered species, Oceanic Manta Ray (*Mobula birostris*), Whale Shark (*Rhincodon typus*) and Zebra Shark (*Stegostoma tigrinum*). Criterion B is also pivotal for identifying important sites for species that are likely to be threatened but have not yet been assessed by the Red List. However, species that trigger a KBA under Criterion B1 must meet the proportion threshold of 10% of the global distribution at a site and also meet the reproductive units threshold of 10 mature individuals (KBA Standards and Appeals Committee 2022). For species that have not been Red Listed, information about reproductive units at a site is not readily available. This can be overcome to a certain extent through engaging with relevant stakeholders and experts to obtain the missing information, including the number of mature individuals. Improving data quality is therefore still an important need.

### 4.4.2 Implementing management of KBAs

A core part of the KBA delineation process is accounting for management units (KBA Standards and Appeals Committee 2022). This is because KBAs have been envisaged as being actual management units or having potential to be managed as a unit, so it is essential to understand the degrees to which

effective management can take place (Oppel et al. 2018). Although delineating KBA boundaries following the borders of an existing management unit often involves including part of the land that does not contain the trigger species (Chapter 1), selecting sites that are already under conservation management has obvious benefits. The most prominent is the guarantee that each site will be formally recognised and its management is then mainstreamed into existing governance systems. In addition, KBAs provide the baseline data that can inform where monitoring should take place. This is because KBA identification entails gathering global data for species under assessment to determine their proportion at a site. Consequently, managers can use this information to monitor species population trends, guided by the KBA ecological boundaries showing exactly where the species are found.

For Mozambique, 12 of the 29 KBAs that were identified in this study are already fully protected and seven are under partial protection. Management in such KBAs could be based on the actual distributional and population size data of the KBA trigger species. This should also account for management effectiveness (Ament & Cumming 2016), which may require much more resourcing to ensure persistence of the trigger elements. In a large park with narrowly distributed species, for example, adequate management may entail delivering direct resources to defined zones that contain extents of the species' suitable habitats. Management for the remaining ones should be determined on the basis of the social and political context (Wauchope et al. 2022). In Mozambique, one option could be co-management, as this model has been applied successfully in the existing PA network. Alternatively, some of the KBAs could become OECMs (other effective area-based conservation measures) (IUCN-WCPA Task Force on OECMs 2019). This approach allows for the areas to be managed in such a way that they achieve many different objectives while also contributing towards biodiversity conservation goals. This model would galvanise support from local communities and wider society to yield desirable outcomes.

It is, however, important to recognise that the KBA methodology is not a prioritisation process (Smith et al. 2019). KBA status does not imply any type of management (Smith et al. 2019; Handley et al. 2023). Moreover, the KBAs in Mozambique have been mostly triggered by range restricted and threatened species, thereby missing extensive areas of ecosystems and habitat types that do not contain such species, such as the country's grasslands, woodlands, and large areas of savanna. These habitats are important in the provision of a varied range of ecosystem services (Liang et al. 2016; Mori et al. 2017; Brockerhoff et al. 2017; Dobson et al. 2022). Systematic conservation planning also has the benefit of accounting for elements that are not identified through the KBA process, such as species that meet the proportion thresholds but do not meet the reproductive-unit threshold due to limited availability of data. There is, therefore, a need to establish synergies between the KBA and systematic conservation planning approaches, as the latter identifies networks of priority areas that meet conservation targets whilst minimising impacts on other sectors (Margules & Pressey 2000).

This will ensure that KBAs are considered when deciding where to prioritise conservation action, producing functional landscape that maintains connectivity (Kaim et al. 2017; Iwamura et al. 2018; Cunningham et al. 2021) and account for the local context and availability of funding from different national governments, donors and agencies (Joseph et al. 2009; Bolam et al. 2019).

### 4.4.3 Incorporating KBAs into environmental decision making

One of the most important aspects of the KBA process is its focus on embedding the process into national decision making. This is in part because the KBA methodology has been designed as an umbrella approach to capture and harmonise the other existing approaches used to identify areas of biodiversity importance. Being an inclusive approach, the KBA process is a working basis for conservation prioritisation (Di Marco et al. 2016), as it covers all taxonomic groups and ecosystems identified and recognised nationally, building pride and ownership to national governments (Perino et al. 2022). In Mozambique, KBAs have been formally recognised by the government through the Ministry of Land and Environment, which is responsible for addressing biodiversity and environmental matters and reporting to international conventions. This involved supporting the establishment of the KBA National Coordination Group, which integrates the IUCN Red List working group with the role of assisting the government to mainstream KBA and Red List matters into decision-making. Also, the KBA National Coordination Group serves as a platform within the government for involving a range of national, regional and international experts. This has the advantage of building the capabilities of Mozambican experts and national institutional capacity to identify important species, and then use the information to determine areas of global importance.

Mozambique now has a rigorous, standardised process for producing important biodiversity data to inform decision making. As a signatory of the CBD, the country is better equipped to update the National Biodiversity Strategic Action Plan to include targets relevant to KBAs and the Red List, and to report on these matters at international level. More specifically, KBAs are now recognised in the national territory development plan and integrated into a number of Ministerial Decrees and Diplomas aimed at improving bird conservation and marine spatial prioritisation. These steps are of particular importance because KBAs help to raise the profile of areas containing less charismatic species, which can then be the focus for monitoring and vigilance, thus helping to safeguard the site even when not protected (KBA Standards and Appeals Committee 2022). Thus, the KBA system in Mozambique shows that this process is possible and other countries are encouraged to follow this lead.

# 5 A spatial prioritisation to inform national-level planning for 30 by 30.

**Abstract**

Achieving Global Biodiversity Framework Target 3 will require countries to create new sites to conserve 30% of the world by 2030, supplementing their existing protected area (PA) systems to produce "effectively and equitably managed, ecologically representative and well-connected" ecological networks. Here we present a spatial conservation prioritisation from Mozambique, which used the Marxan with Zones software package to identify where best to place new OECMs (other effective area-based conservation measures) to meet Target 3. We identified 162 vegetation types and 525 important species to be used in the analysis, mapped their distributions and set targets for how much of each should be included in the ecological network. We then ran two scenarios, both of which automatically selected the PAs, but one of which did not automatically select the existing conservation area ("Without CAs") and one that did ("With CAs"). We found that the existing PAs cover 10.8% of the country and meet targets for 31 of the 162 vegetation classes and 208 of the 525 species, and the CAs cover an additional 14.0% and in combination with the PAs meet targets for an additional 40 vegetation classes and 33 species. The Marxan with Zones analyses found that under the most suitable set of parameters, an additional 161,423 $km^2$ was needed to meet the missing targets in the Without CAs scenario, and 107,081 $km^2$ in the With CAs scenario. Thus, meeting Target 3 will require a large increase in the amount of land managed for conservation in Mozambique, and funding this adequately to ensure effective management will not be straightforward. By focusing on creating new OECMs though, our analysis identified where different types of conservation management would be required and showed that much of this land is compatible with community-based ecotourism and the sustainable harvesting of natural resources.

## 5.1 Introduction

From the local to the global level, biodiversity loss and poverty alleviation are deeply intertwined (Díaz et al. 2020). Therefore, they need to be tackled holistically. In the past, the main policy instrument used by the international conservation community has been establishing state managed protected areas (PAs) (Watson et al. 2014), with each country that is a signatory to the Convention on Biological Diversity (CBD) aiming to protect a minimum of 17% of the terrestrial realm and 10% of the marine realm by 2020 (CBD 2010). Although significant progress has been made towards achieving these targets, the effectiveness of the PAs has been questioned (Geldmann et al. 2019). It is believed that many PAs have been placed in unsuitable areas for species and ecosystems, thus failing to represent biodiversity (Joppa & Pfaff 2009; Butchart et al. 2015). At the same time, the impacts of PAs on poverty alleviation are mixed, as while people living next to PAs have higher levels of well-being (Naidoo et al. 2019), many PAs have impacted people's lives and livelihoods by limiting their access and use of these sites (Oldekop et al. 2016). This is why parties to the Convention on Biological Diversity (CBD) recently developed the Kunming-Montreal Global Biodiversity Framework to account for these different issues (CBD 2022).

The new Global Biodiversity Framework Target 3 focuses on site-based conservation and its most high-profile component is the commitment to conserve at least 30% of the terrestrial and marine realms by 2030 (CBD 2022). However, this target also stresses the need for conserving areas that are important for biodiversity. Two of the widely used approaches for identifying these important sites are Key Biodiversity Areas (KBAs), which identify sites that provide a significant contribution for biodiversity persistence globally on the basis of species and ecosystems (IUCN 2016), and Important Plant Areas (IPAs), which focus on identifying areas that are important on the basis of their plants and fungus species and habitats (Darbyshire et al. 2017). Both are criteria-threshold based approaches and entail gathering worldwide data on the biodiversity elements, then assessing which sites contain sufficient proportions to trigger KBA or IPA status locally. These initiatives are also designed to collect and collate primary biological data to help achieve Target 1 of the Global Biodiversity Framework, which states that by 2030 "all areas are under participatory integrated biodiversity inclusive spatial planning and/or effective management processes" (CBD 2022).

Target 3 also highlights the role of other effective area-based conservation measures (OECMs), as well as PAs, in achieving site-based conservation. OECMs are areas that are primarily managed to achieve a wide range of objectives whilst contributing to *in situ* biodiversity conservation as a secondary objective (MacKinnon et al. 2020; Maxwell et al. 2020). There is increasing recognition of the potential importance of OECMs (Gurney et al. 2021). This is partly because Target 3 emphasises that sites should be equitably governed and OECMs are more likely to be community-owned and

managed (Fidler et al. 2022). It is also because the growth in state-owned PAs has been slow, so it is near impossible for the 30 by 30 component of Target 3 to be met without including a wider range of conservation area types. As a first step, there is a need to identify OECMs by acknowledging and documenting the existing areas where biodiversity conservation is already a secondary management objective. This includes sacred forests, fishing grounds and other communally- and privately-owned and managed conservancies (Gurney et al. 2021). Recognising these OECMs is important because it puts local communities and other relevant stakeholders at the forefront of managing their land and seas, building equitability and strengthening livelihoods. Moreover, it provides new opportunities to conserve under-represented biodiversity whilst placing minimal impact on communities' livelihoods, so there is also a need to identify where new OECMs could be created and supported.

This process of identifying where to create new OECMs that complement the existing PAs and OECMs is not straightforward though, as it depends on accounting for a whole range of ecological, social and socio-economic data. A widely used approach to inform this process is systematic conservation planning (Margules & Pressey 2000), which is an approach based on working with relevant stakeholders to define the conservation context and translate this into transparent objectives that feed into a spatial prioritisation for guiding action on the ground (Groves & Game 2016). Systematic conservation planning is widely used (Sinclair et al. 2018) and has informed national spatial plans in several countries (Bicknell et al. 2017; Metcalfe et al. 2022). However, this previous work has not explicitly accounted for different types of conservation management and ownership, so we lack research on how to identify PAs and OECMs to inform national plans. Here we present an example from Mozambique that is designed to account for these factors, building on a nationwide assessment of KBAs (WCS et al. 2021) and IPAs (Darbyshire et al. 2023), and providing guidance on how a country can fulfil its CBD Target 3 commitments.

Mozambique is a country in southern Africa with a number of existing PAs and sites that could potentially meet the OECM definition. Despite much of the country already being under some type of conservation management, recent work has shown that some important biodiversity remains unprotected (e.g. Chapter 4) and (WCS et al. 2021; Darbyshire et al. 2023). Moreover, Mozambique has committed to the Global Biodiversity Framework Target 3, with is emphasis on effectiveness and equitability. This is why we used Marxan with Zones to identify priority areas for conservation in Mozambique, as it is designed to account for the fact that different management zones can have different conservation outcomes (Watts et al. 2009). For example, some PAs and similar conservation areas (CAs) are more intensively managed to conserve certain sensitive species, but as part of that require more funding and have more restrictions on how the sites are used (Craigie et al. 2010). In contrast, other CAs that include some potential OECMs are designed as multiple use zones, and so are

only effective at conserving ecosystems and less sensitive species but are less expensive to manage and have fewer negative impacts on local people (Alves-Pinto et al. 2021).

Here we use Marxan with Zones to investigate the impacts of planning for different management zones at a national level using Mozambique as a case study. Our analyses consisted of: (i) selecting and mapping the distributions of a set of conservation features, namely vegetation types and important species, and determining which species depend on high intensity management; (ii) setting targets for how much of each conservation feature should be managed for conservation; (iii) assessing the extent to which the current network of PAs and CAs meets these conservation targets; (iv) running Marxan with Zones to identify where best to create new high management intensity and low management intensity zones to meet any target gaps, based on two scenarios that excluded or included existing CAs from the current network.

## 5.2 Methods

### 5.2.1 Study area and conservation context

Mozambique is a country in south-east Africa, with an area of about 800,000 $km^2$ and a coastline of approximately 2,500 km along the Indian Ocean (Figure 5-1). The country is home to a wide range of habitat types, including miombo woodland, savannas, grasslands, mangroves, coastal forest to the east and montane forests to the west. Mozambique's conservation model consists of PAs and CAs. The PAs include national parks, national reserves and special reserves that are highly enforced and managed by the National Administration of Conservation Areas, which is the responsible governmental agency under the Ministry of Land and Environment. The CAs include a wide range of types such as forest reserves, buffer zones, community conservation areas and hunting reserves. The forest reserves are managed by the Forestry Department, under the Ministry of Land and Environment, with very limited enforcement and low funding level. Management of the other CA types is the responsibility of a wide range of stakeholders.

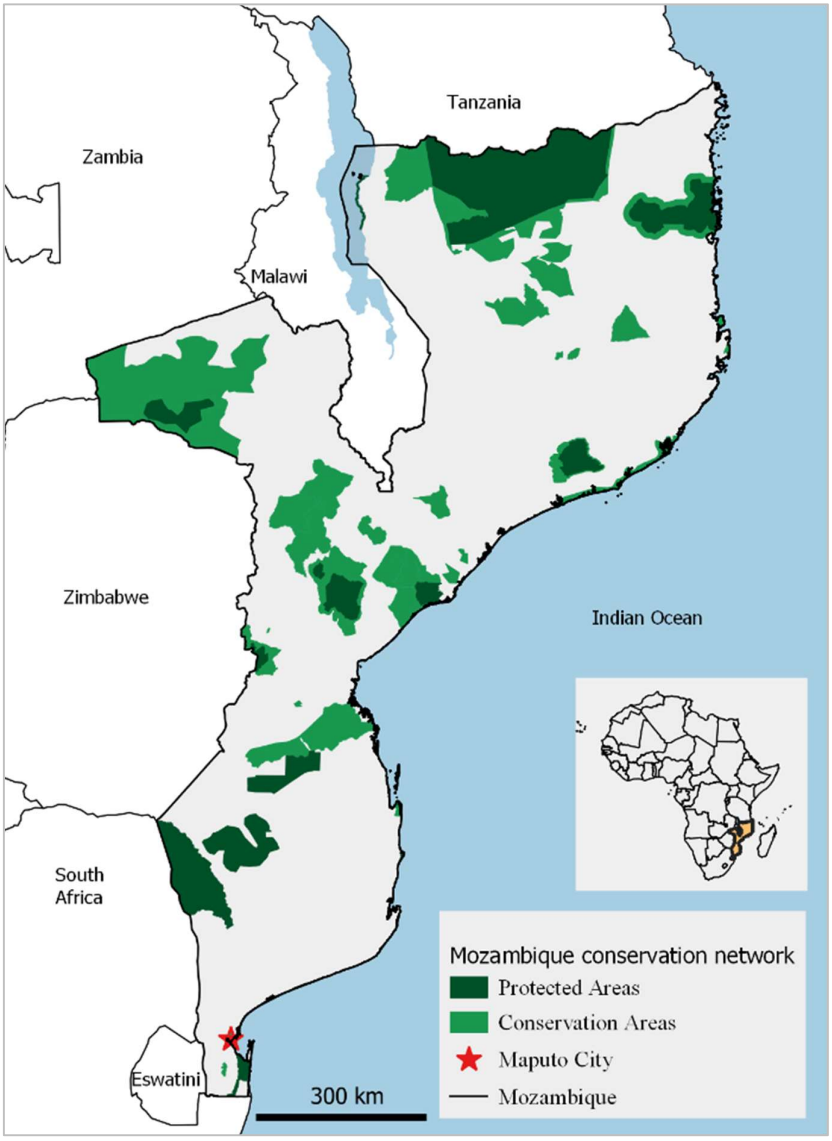



***Figure 5-1: Mozambique's network of protected areas and conservation areas***

### 5.2.2 Defining the management zones

Our analysis was based on identifying a set of potential OECMs to complement the existing PAs and CAs. We decided to identify two types of potential OECM based on their levels of management intensity. One type was based on the high management intensity of the PAs and CAs that are often state- or privately-owned and designed to protect high value species, such as large mammals impacted by poaching or important timber tree species. The other type was based on community-managed CAs, which generally have lower densities of high value species and lower management costs and impacts on people. As a general rule, we sought to identify low management intensity sites as much as possible, given their lower costs and impacts on people, only selecting high management intensity sites when they were needed to meet targets for sensitive species.

Spatial conservation prioritisations generally involve dividing the planning region into a number of planning units and then selecting the sets of planning units that should be managed for conservation. Our analysis was based on Marxan with Zones (Watts et al. 2009), which lets the user define different management zones and then allocates planning units to these zones based on meeting targets for a number of conservation features, minimising planning unit costs and ensuring connectivity. With this approach, each planning unit has a zone-specific cost, and each conservation feature can have a zone-specific contribution. Our analysis consisted of defining three types of management zone. Zone 1 was the High Management Intensity Zone, which contributed to targets for all ecosystem types and species, including those that are sensitive to over-harvesting or direct persecution. Conserving these sensitive species involves additional law enforcement and/or species-specific management, so we assigned a higher planning unit cost because of this extra management effort. Zone 2 was the Low Management Intensity Zone, which we defined as not contributing to targets for species sensitive to over-harvesting or direct persecution, i.e., the habitat for a sensitive species would not count towards meeting the target for that species if it was found in a planning unit allocated to this zone. A planning unit assigned to this zone would also have a lower planning unit cost, given the lower levels of management activity. Zone 3 was the Other Zone, which represented land that was not managed for conservation and so planning units assigned to this zone did not contribute to meeting any of the conservation feature targets and had a very low management cost.

### 5.2.3 Selecting and mapping the conservation features

When selecting the conservation features to be used in our analyses, we used vegetation types to represent Mozambique's broader biodiversity (Figure 5-2) and a set of threatened and endemic species to ensure they were also represented (Noss 1990). The vegetation types were based on (Lötter et al. 2021) and we produced a map of each type by removing areas impacted by anthropogenic actions, as shown in Mozambique's land use map (FNDS 2020). This process consisted of extracting the transformed landscape from the land use map and clipping it to the vegetation map. To do this we used the libraries "sf", "raster" and "rmapshaper" in R. Finally, we used QGIS to merge the outputs to produce a 30 m resolution map.

We compiled an initial list of terrestrial species based on those identified in previous KBA (WCS et al. 2021) and IPA (Darbyshire et al. 2023) analyses in Mozambique. This was followed by consultation with relevant experts to establish a final list that would account for species that were not part of the KBA and IPA assessments. Then, using data from the IUCN Red List of Threatened Species (IUCN 2023), we identified which species are threatened by over-harvesting or direct persecution globally.

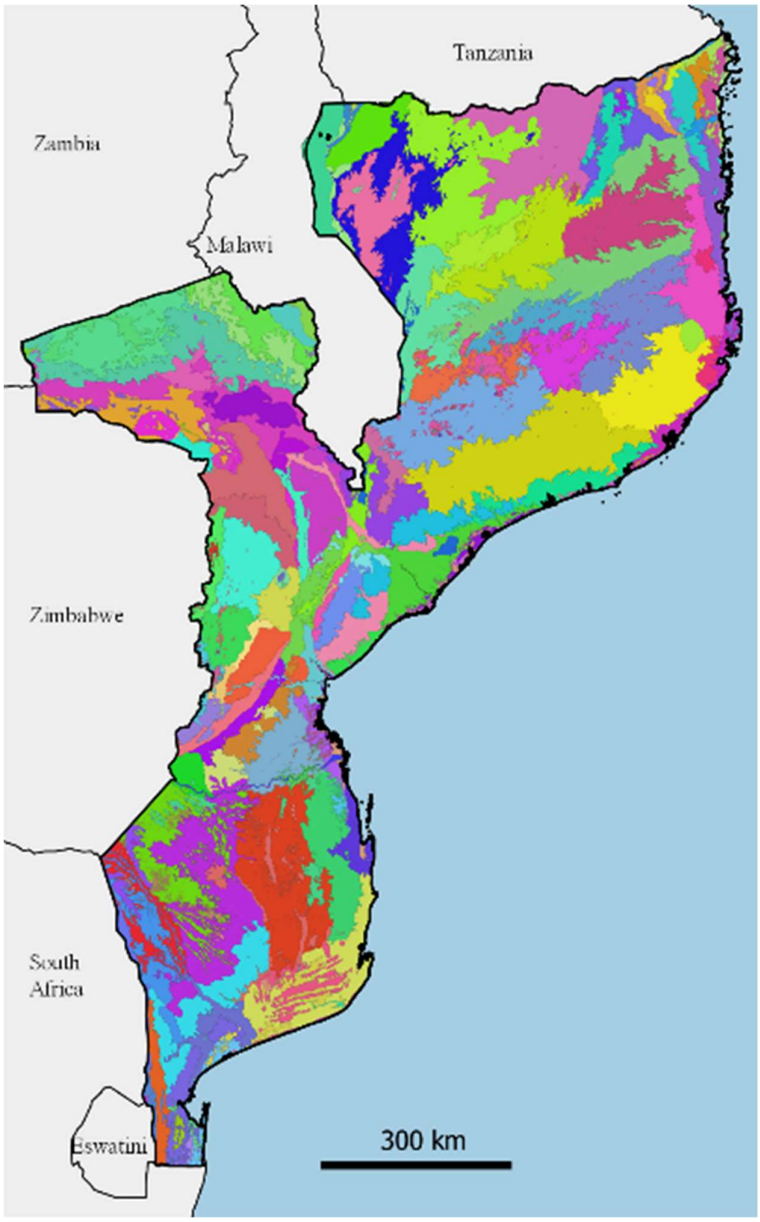



***Figure 5-2: Vegetation types in Mozambique based on Lötter et al (2021). The names of the vegetation types are given in Figure S5-1.***

Mapping the distribution of species was primarily based on the KBA and IPA datasets. This was combined with data from national, regional and international natural history collections and field surveys for some plant species. For the first steps in producing the maps, we compiled species range maps from the IUCN Red List. This was straightforward for all birds, freshwater fish and mammals included in the analyses. For most amphibians, invertebrates, plants and some reptiles we used the Maximum Entropy model (Phillips et al. 2006) to produce a distribution model map for every species as, long as sufficient data was available. Modelling was not adequate for species with fewer than 10

records (Lomba et al. 2010), therefore mapping for such species was based on their distribution point data. We then converted these range maps into Extent of Suitable Habitat maps. We did this by first using the literature, together with the IUCN Red List and experts' input, to determine which landcover types were not suitable for each species. We then used QGIS to remove the area of modelled distribution/data points overlapping with unsuitable landcover types for each species.

### 5.2.4 Setting the conservation features targets

We used four approaches to set the representation targets for the conservation features. For each of the 162 natural vegetation types, we set target as 20% of their historical extent. We chose this 20% target because it is used by the South African government (SANBI 2018) and accounted for the fact that, together with the existing PAs and CAs, this should identify approximately 30% of the country in line with the CBD target. In situations where the available extent of a vegetation type was already less than 20% of its original coverage, we used the total range remaining as the conservation target.

For birds and mammals, we set targets based on the minimum proportion of each species range that is required to guarantee its persistence. This was calculated using a widely adopted approach based on the species' global range (Rodrigues et al. 2004; Butchart et al. 2015). For ranges less than 1,000 $km^2$ the proportion target is 100%. Those with a range ≥ 250,000 $km^2$, we set 10% of the country's range. Finally, for ranges with values in between, targets were based on a log-linear scale.

We used a different approach for amphibians, freshwater fish, invertebrates, plants and reptiles, as most of these species are endemic to Mozambique and had much smaller ranges (Pfab et al. 2011). Based on the IUCN Red List, for threatened species assessed under Criterion B we set their targets at 100% of their distribution in Mozambique. For threatened species evaluated under Criterion A, as well as Near Threatened (NT) and Least Concern (LC) species, determining their targets entailed calculating their global target set as 10,000 mature individuals and 11 localities (Pfab et al. 2011). It is worth noting that we used locality instead of location (IUCN Standards and Petitions Committee 2022) given the scarcity of information about threats. For each locality, we set a minimum threshold of 4 $km^2$ in light of the 2x2 grid cell for AOO of the IUCN Red List. We produced the final target for each species by multiplying their global target by the estimated proportion of their global range in Mozambique. For species with point data maps and without the IUCN Red List status, the conservation target was set to conserve all known populations in Mozambique.

### 5.2.5 Producing the planning units and cost layer

Producing the Mozambique planning system involved dividing the country's land surface into a number of planning units. Following Nhancale and Smith (2011), the planning units were based on a

series of hexagons with an area of 25 $km^2$. Then we combined the hexagon with the existing network of PAs and CAs. This was to ensure an accurate representation of the boundaries of these PAs and CAs in the planning system. This produced 36,599 planning units, which was sufficiently small to produce near optimal results in Marxan with Zones, especially as 4,318 were in the PAs and so automatically selected at the beginning of each analysis.

We then imported the planning units into the CLUZ conservation planning plug-in for QGIS (Smith 2019). Using CLUZ, we imported the spatial data and then calculated the amount of each conservation feature found in each planning unit. The planning unit cost for each zone was based on the agricultural opportunity cost and its management cost. The agricultural cost layer was derived from the 30 m resolution African agriculture potential map (Hengl et al. 2021), which scores areas from 0 to 5 depending on the number of constraints the soil has for farming (Figure 5-3). For each planning unit we summed the number of constraints per pixel, thus accounting for the fact that the planning unit area varied, and did a reverse linear transformation, so the highest value of 100 was given to the planning unit with the lowest summed number of constraints and the lowest value of 0.001 was given to the planning unit with the highest summed number of constraints. These values were then set as the planning unit costs for Zone 2. For Zone 1 we added 100 to their Zone 2 value to ensure that the cost of managing these sites in Zone 1 was always higher than managing a site in Zone 2. This addition of 100 was arbitrary but did not influence the results because it only impacted prioritisation within zones, rather than between zones. For Zone 3 we set the planning unit cost for every planning unit as 0.001, so the cost of assigning a planning unit to that zone was never higher than assigning it to a zone under conservation management.

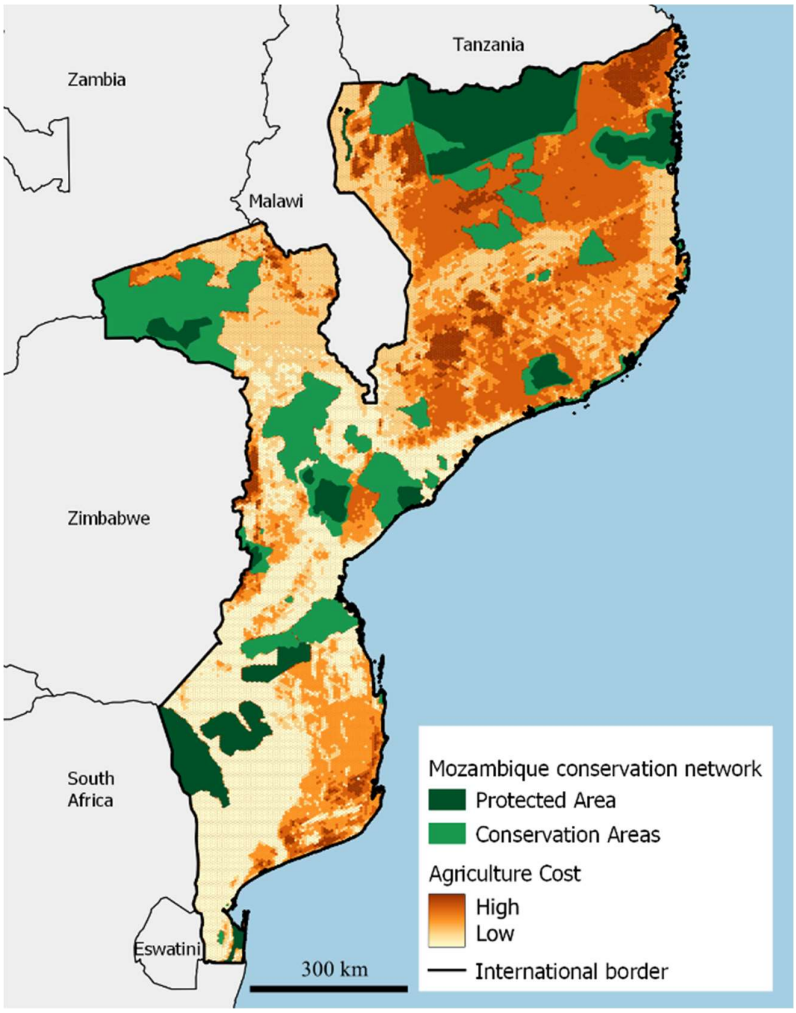



***Figure 5-3: Agricultural opportunity cost layer that was basis of the different planning unit costs per zone used in the spatial prioritisation analysis for Mozambique.***

### 5.2.6 Spatial prioritisation analysis

We carried out two Marxan with Zones analyses: (1) the "Without CAs scenario" that involved giving locked-in status in Zone 1 for the PAs; (2) the "With CAs scenario" that involved giving locked-in status in Zone 1 for the PAs and locked-in status in Zone 2 for the CAs. Locked-in status ensures that the planning units are automatically assigned to the intended zone when running Marxan with Zones. In both cases, we specified that species that are sensitive to over-harvesting or direct persecution would only meet their conservation target in Zone 1. We then used CLUZ to carry out a gap analysis, working out the extent to which the current network of PAs (Without CAs scenarios) and the current network of PAs plus other CA types (With CAs scenarios) meet the conservation features targets.

We were then ready to run Marxan with Zones. Marxan uses a simulated annealing algorithm to identify near-optimal solutions (Watts et al. 2009). This involves running the software a number of

times, with each run involving a number of iterations. Each run generally produces a slightly different result, so Marxan with Zones identifies the 'best' output based on the one with the lowest cost and the 'selection frequency' output, which counts the number of times each planning unit is assigned to each zone in each of the run results. Planning units that are selected in every output can be seen as 'irreplaceable', whereas planning units with lower selection frequency scores could be swapped with similar, unselected planning units to produce similarly efficient outputs (Baisero et al. 2022). As a first step, we worked out the Feature Penalty Factor (FPF) values (Serra-Sogas et al. 2020), which is the penalty for not meeting the conservation target in a particular run. We used CLUZ to carry out a sensitivity analysis and found that a FPF of 1.5 was sufficient to ensure every run output from Marxan with Zones met all the targets. We also carried out sensitivity analyses to select the number of runs and the number of iterations in each run needed to produce near-optimal results, finding that 1,000 runs of 100 million iterations produced efficient results.

We were then ready to run the main analyses for the two scenarios. For each scenario we ran 15 analyses, varying the Boundary Length Modifier (BLM) and Zone Boundary Cost Modifier values. The BLM value influences how fragmented the Marxan with Zones outputs are, so setting a high cost identifies larger patches of planning units assigned to the same zone (Watts et al. 2009). The Zone Boundary Cost Modifier influences the extent to which Marxan with Zones preferentially selects patches of zones that neighbour each other. In these analyses, we wanted patches of Zone 1 and Zone 2 to be more likely to share borders than with patches of Zone 3, so set a Zone Boundary Cost Modifier value of 1 between Zone 1 and Zone 3, and Zone 2 and Zone 3, and set values between Zone 1 and Zone 2 as lower than 1. Therefore, for both the Without CAs and With CAs, we investigated the impact of using BLM values of 0.01, 0.02, 0.03, 0.04 and 0.05 and Zone Boundary Cost Modifier values between Zone 1 and Zone 2 of 0.25, 0.5 and 0.75. We then used the best output from each analysis and calculated for Zones 1 and 2 the total area, the number of patches and the mean patch size. From this, we identified the parameters to produce results that balanced the need for patches of planning units to be large enough to be ecologically viable but not so large as to produce inefficient results (Smith et al, 2010). We then compared these outputs to the current distribution of CAs, KBAs and IPAs.

## 5.3 Results

Mozambique contains 12 PAs and 47 CAs (Figure 5-1). The area of PAs is 85,027 $km^2$ and the area of the CAs is 110,789 $km^2$, giving a total of 195,816 $km^2$ (24.8%) of the country's terrestrial and inland water. The conservation features used for our analysis consisted of 162 natural vegetation types and 525 species across seven taxonomic groups including 11 amphibians, 37 birds, 12 freshwater fish, 14 invertebrates, 13 mammals, 410 plants and 28 reptiles (Table S5-1). Of these species, 45.1% are

globally threatened and 55.2% are endemic or near-endemic to Mozambique. The PAs in Mozambique meet targets for 31 of the 162 vegetation types and 208 of the 525 species. The PAs and the CAs combined meet targets for an additional 40 vegetation types and 33 species (Figure 5-4).

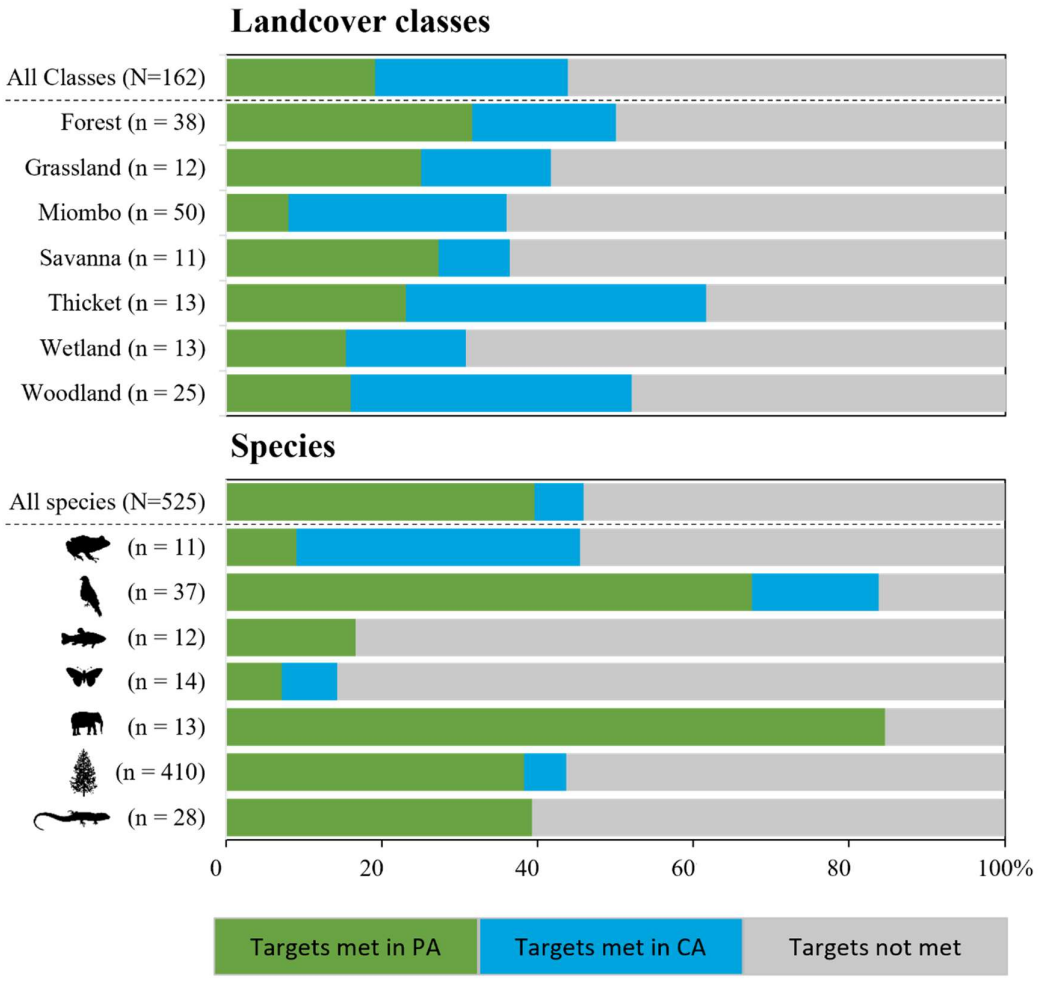



***Figure 5-4: The proportion of conservation features (species and vegetation types) meeting targets within the existing network of protected areas (PAs) and conservation areas (CAs) in Mozambique.***

For the Without CAs analysis, the extent of the high management intensity areas ranged from 93,658 $km^2$ for the scenario with a BLM of 0.01 and a Zone Boundary Cost Modifier of 0.5 to 96,409 $km^2$ for the scenario with a BLM of 0.05 and a Zone Boundary Cost Modifier of 0.75 (Figure 5-5). The extent of the low management intensity areas ranged from 140,483 $km^2$ for the scenario with a BLM of 0.01 and a Zone Boundary Cost Modifier of 0.75 to 174,416 $km^2$ for the scenario with a BLM of 0.05 and a Zone Boundary Cost Modifier of 0.5 (Figure 5-5). All the Without CAs scenarios show a relatively consistent spatial pattern, with small patches of high management intensity areas in the north, south-east and south. Larger low management intensity areas are also mostly found in the north, south-east and south, but they are more widespread, occurring throughout the country (Figure 5-5).

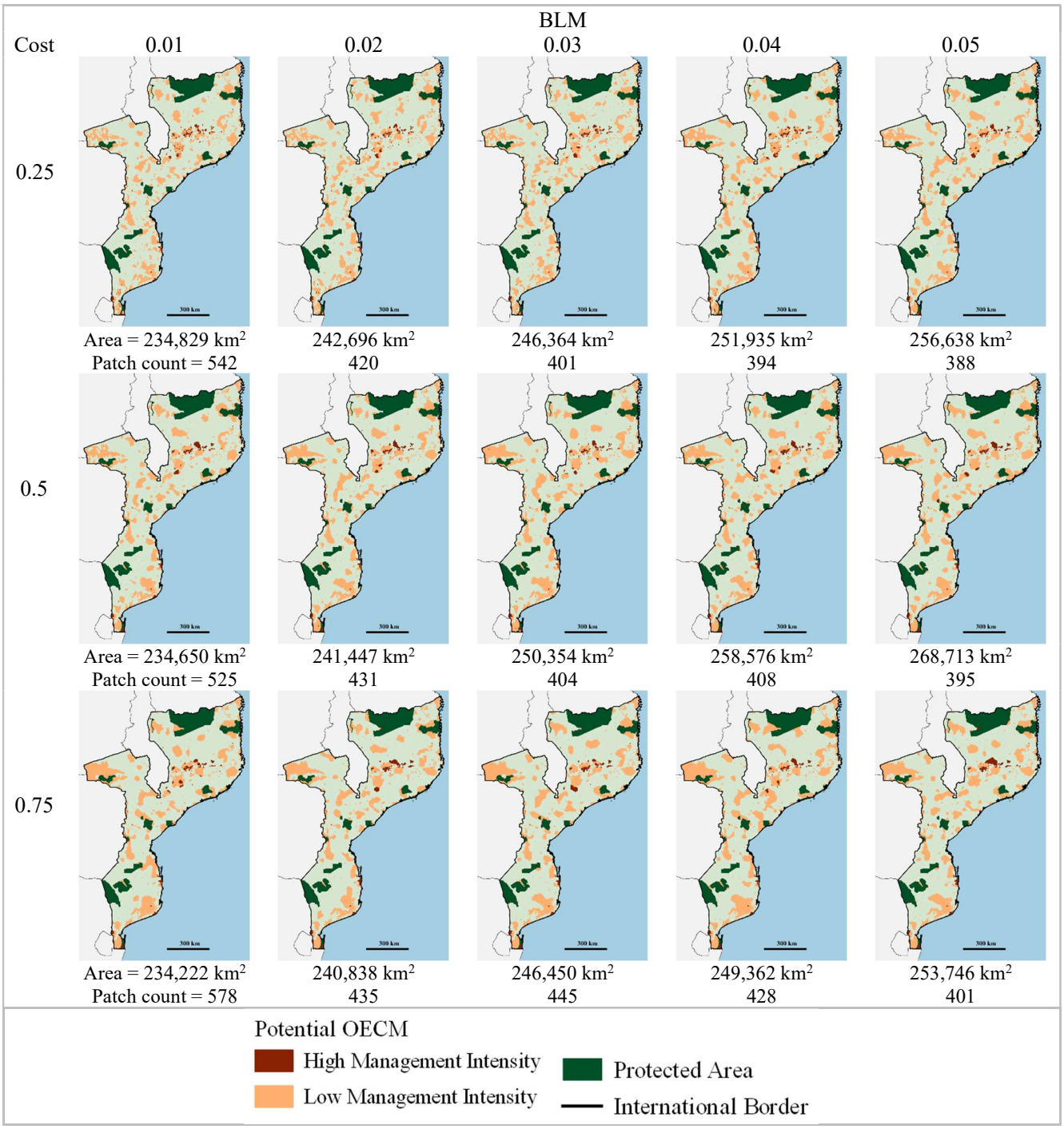



***Figure 5-5: Marxan with Zone best outputs for the Without CAs scenarios, showing the total area selected for Zones 1 and 2 and the patch count of planning units assigned to either of these zones.***

For the With CAs Scenarios: the extent of High Management Intensity areas ranged from 93,919 km² for the scenario with a BLM of 0.01 and a Zone Boundary Cost Modifier of 0.5 to 96,649 km² for the scenario with a BLM of 0.05 and a Zone Boundary Cost Modifier of 0.75 (Figure 5-6). The extent of

Low Management Intensity areas ranged from 200,523 km$^2$ for the scenario with a BLM of 0.01 and Zone Boundary Cost Modifier of 0.75 to 235,416 km2 for the scenario with a BLM of 0.05 and Zone Boundary Cost Modifier of 0.25 (Figure 5-6). The results from the 15 With CAs scenarios show a similar pattern to the Without CAs scenarios, with most of the High Management Intensity and Low Management Intensity areas in the north, south-east and south (Figure 5-6).

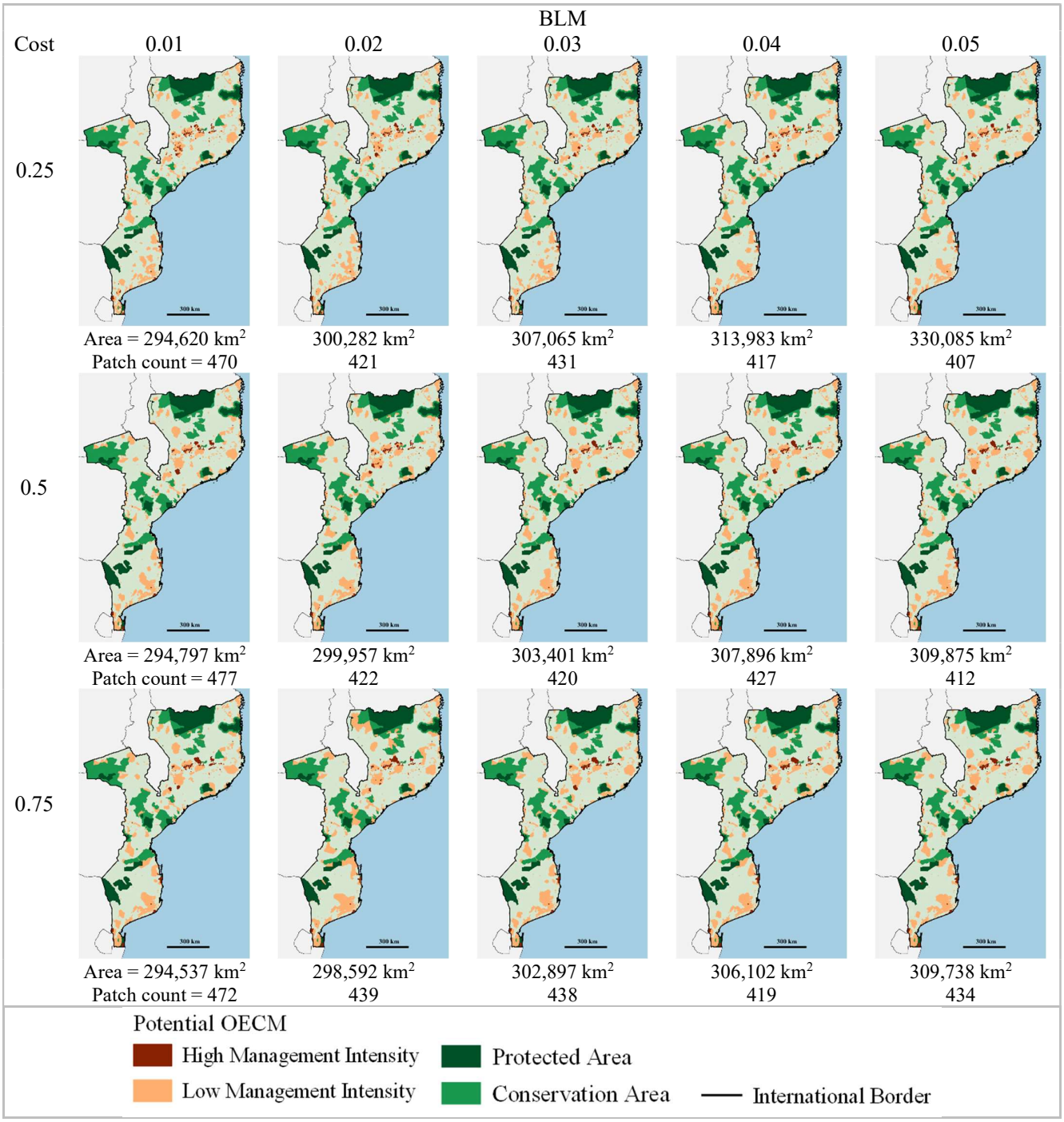


***Figure 5-6: Marxan with Zone best outputs for the With CAs scenarios, showing the total area selected for Zones 1 and 2 and the patch count of planning units assigned to either of these zones.***

The spatial characteristics of the priority areas identified by Marxan with Zones were strongly influenced by the BLM and Zone Boundary Cost Modifier values (Figure 5-7). For the Without CAs analysis, the number of patches ranged from 388 patches and a mean patch size of 659 $km^2$ for the BLM = 0.05 and Zone Boundary Cost Modifier = 0.25 scenario to 578 patches and a mean patch size of 403 $km^2$ for the BLM = 0.01 and Zone Boundary Cost Modifier = 0.75 scenario (Figure 5-5). When the CAs are included, the number of patches ranged from 407 patches and a mean patch size of 809 $km^2$ for the BLM = 0.05 and Zone Boundary Cost Modifier = 0.25 scenario to 477 patches and a mean patch size of 616 $km^2$ for the BLM = 0.01 and Zone Boundary Cost Modifier = 0.5 (Figure 5-6). Based on all this, we selected the BLM of 0.03 and a Zone Boundary Cost Modifier of 0.75 scenario as the most suitable for informing future discussions, based on balancing efficiency and reducing fragmentation.

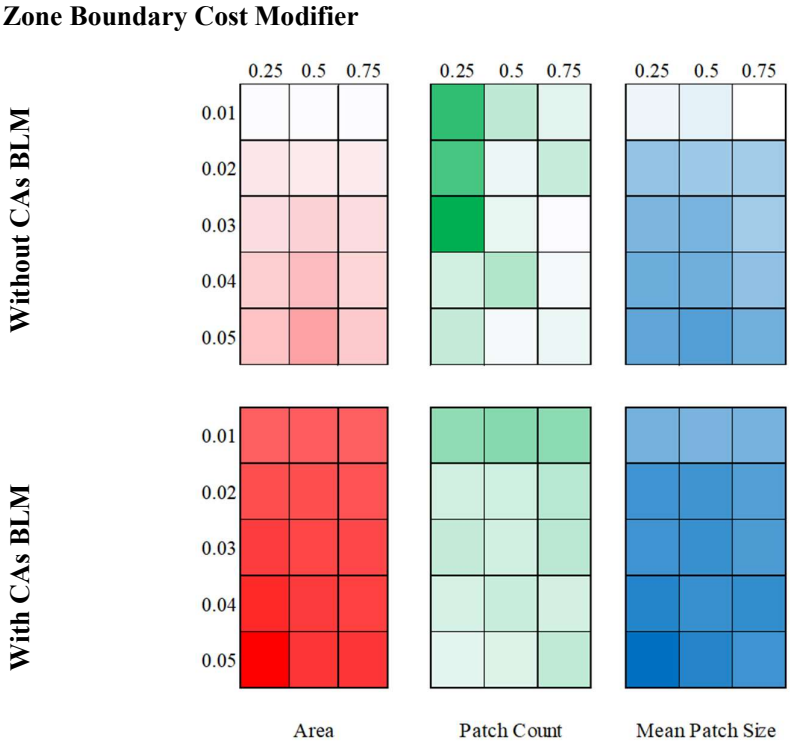



***Figure 5-7: Schematic on how the Marxan with Zones best results for the With CAs and Without CAs scenario were influenced by the BLM and Zone Boundary Cost Modifier values between Zone 1 and Zone 2. Results with a larger combined area of Zone 1 and Zone 2, a larger number of patches of Zone 1 and Zone 2 planning units and larger patch sizes are in darker colours.***

The selection frequency scores for these two most suitable analyses are fairly similar. In both cases, most of the Zone 1 planning units were never selected but a small number were repeatedly selected, whereas for Zone 2 there were also some patches of medium selection frequency scores (Figure 5-8).

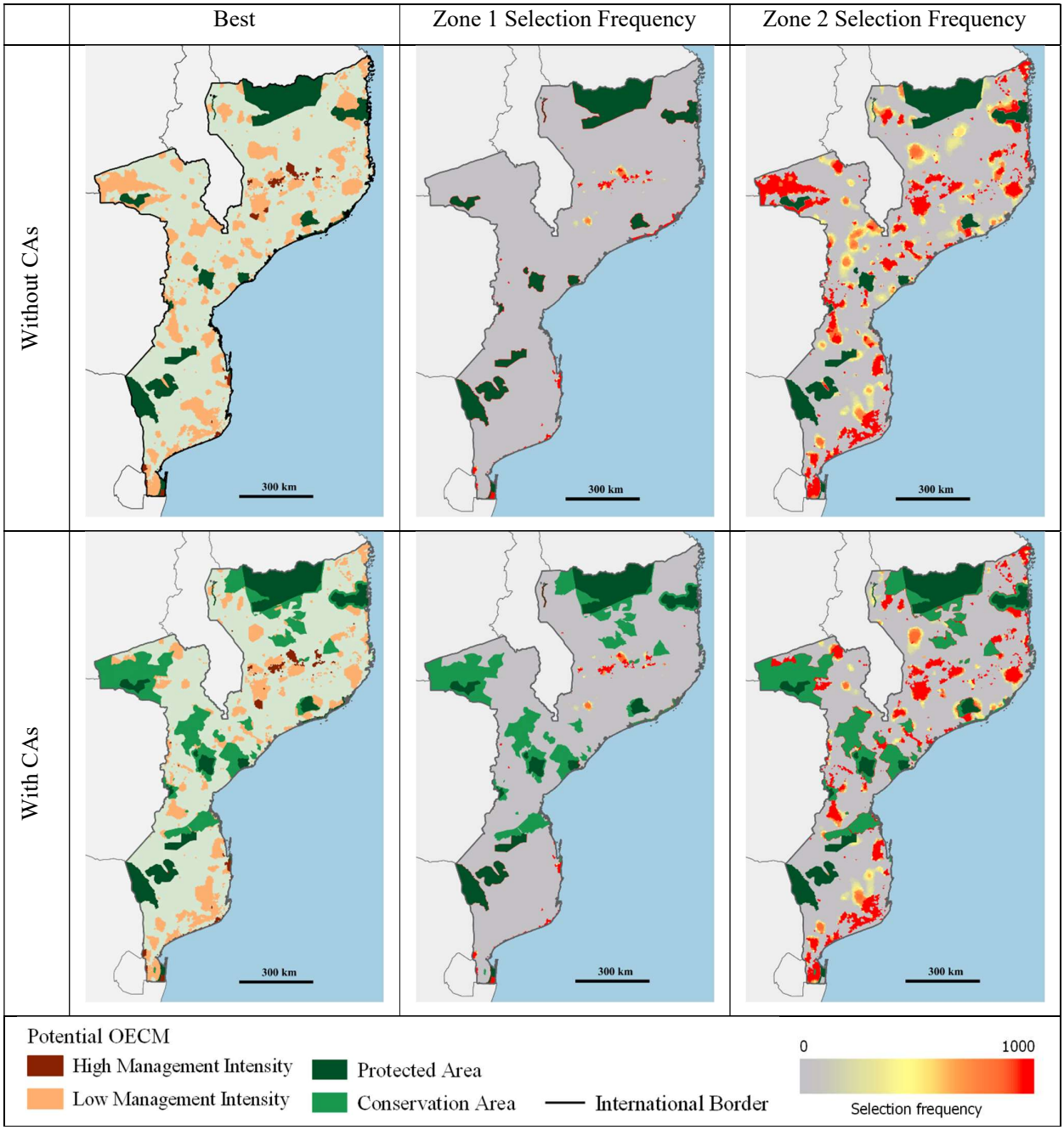



***Figure 5-8: The best output, Zone 1 selection frequency scores and Zone 2 selection frequency scores for the most suitable Marxan with Zones from the Without Conservation Areas (CAs) and With CA scenarios.***

Comparing the selection frequency scores from the most suitable Without CAs scenario with the boundaries of the CAs overlaid shows that 25.6% of the land found in CAs had selection frequency values of 0. There are some patches of planning units with medium selection frequency scores found in CAs in the centre and north, and some patches of high selection frequency scores in central and north-west Mozambique (Figure 5-9).

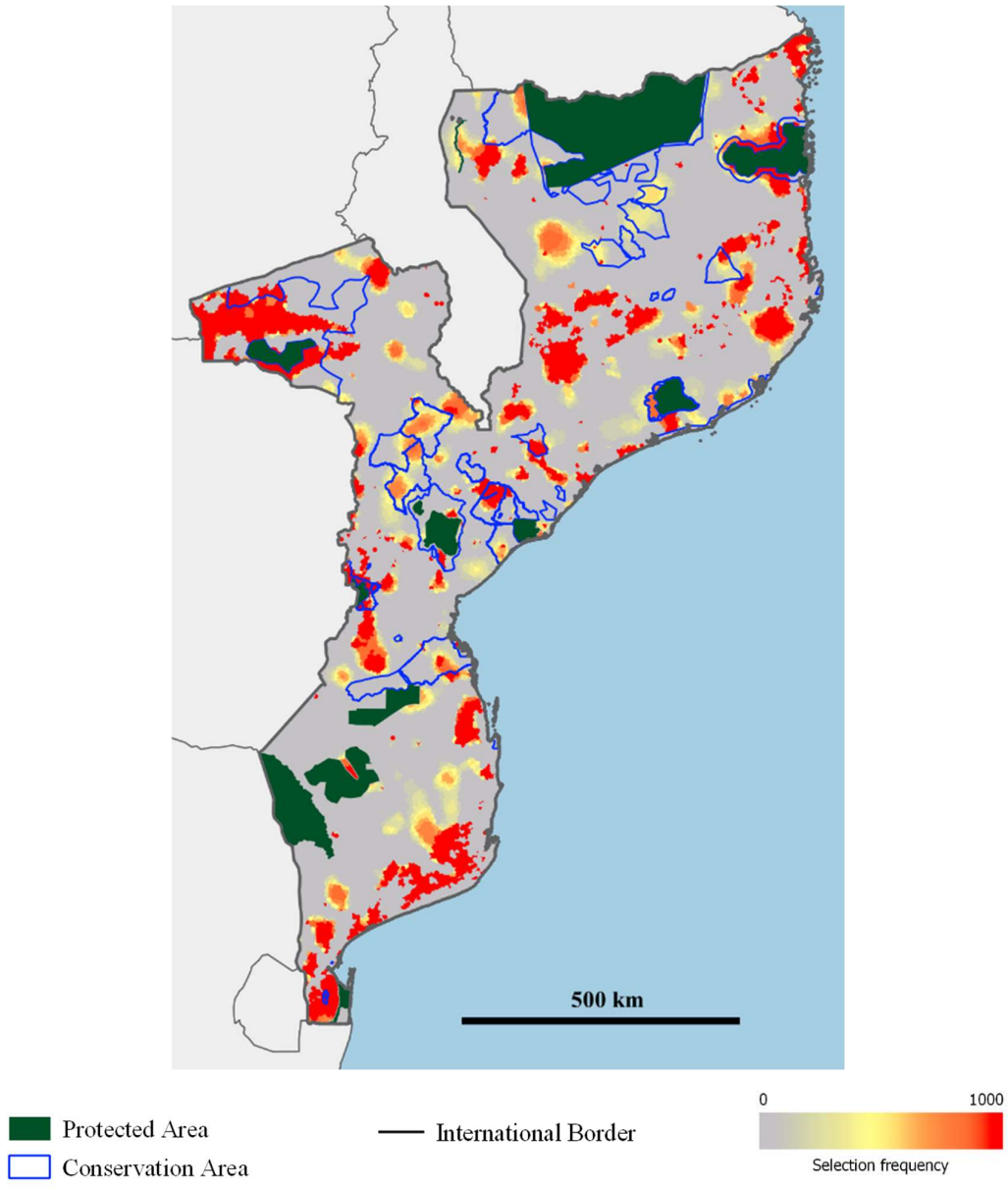



***Figure 5-9: Selection frequency for the most suitable Without CAs analysis with Conservation Areas overlaid.***

Comparing the selection frequency scores from the most suitable With CAs scenario with the boundaries of the KBAs and IPAs produced less distinctive results, partly because most of the KBAs

are also PAs or CAs and most of the IPAs are very small. We did find that 39.2% of the currently unprotected land found in KBAs and 30.5% of the currently unprotected land found in IPAs had selection frequency values of 0 (Figure 5-10).

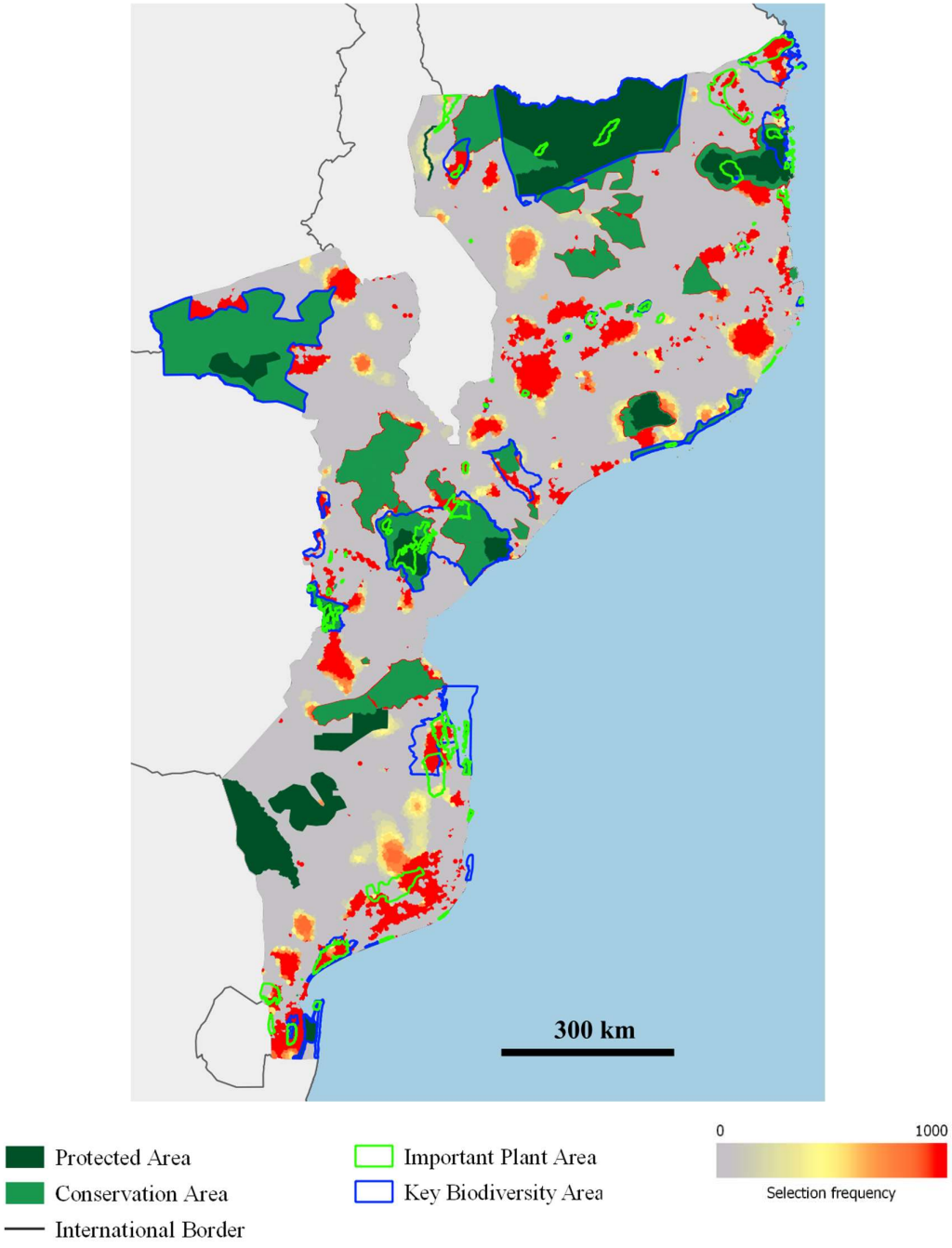



***Figure 5-10: Selection frequency for the most suitable Without CAs analysis with Key Biodiversity Area and Important Plant Area boundaries overlaid.***

## 5.4 Discussion

To meet the Kunming-Montreal Global Biodiversity Framework targets it is imperative that signatory nations deliver on their commitments (Perino et al. 2022). The first step in scaling this up involves establishing theories of change, drawing a road map and articulating clearly how different stakeholders can contribute to achieving the targets effectively (Addison et al. 2020; Burgass et al. 2021). This will require balancing biodiversity conservation with other relevant social, economic, environmental and climate goals (Milner-Gulland et al. 2021; Watson et al. 2021). Using a systematic conservation planning approach provides an effective way of informing such decisions (Margules & Pressey 2000). It is a defensible and transparent model to assist decision-making on spatial prioritisation. This study is the first national-scale spatial prioritisation analysis aimed at evaluating various zoning scenarios for conservation land-use. Here we discuss the role of Mozambique's current network of PAs and CAs in meeting conservation targets. We also discuss how the spatial prioritisation results could help identify new OECMs to achieve positive biodiversity outcomes, and then discuss the broader implications for implementation and management.

### 5.4.1 Current conservation area network

Mozambique's network of PAs meets the targets for 19% of the vegetation types and the PAs and CAs combined meet targets for 44% of them. Moreover, about a half of Mozambique's extent of forest types, thickets and woodlands are found within the PAs and CAs network. This shows that the existing network of PAs and CAs in Mozambique provides an important contribution to conserving broader biodiversity and the provision of ecosystem services (Watson et al. 2016). However, these PAs and CAs do not meet targets for approximately two thirds of wetland, miombo and savanna types and half of grassland and forest types. Thus, many vegetation types are under-represented, although the CAs play an important role in conserving some types that are missing from the PAs. This is similar to studies from other countries, where other types of conservation area are important for filling gaps in the PA system (Schleicher et al. 2017; Clements et al. 2019; Palfrey et al. 2022). This pattern is less strong for the species results though, where the PAs meet 40% of the targets and adding the CAs to the analysis only increases this to 46%. Moreover, the CAs do not help meet any targets for the freshwater fish, mammals and reptiles that were part of this analysis, meet their conservation targets (Figure 5-4).

The fact that Mozambique's network of PAs and CAs are unrepresentative of the country's biodiversity is not surprising as national analyses from other countries have found similar results. This pattern is often because PAs are generally located in remote, upland areas (Joppa & Pfaff 2009) and other areas with low economic value for agriculture (Venter et al. 2018), and this is also likely the case in Mozambique. This occurred because the PAs established in the colonial era were designed to

protect important populations of large mammal populations, which had generally been extirpated on land that was valuable for agriculture. The reasons for creating the CAs depend on their type. The Forest Reserves were mostly designated for the sustainable harvesting of commercial timber species, but they also include natural forest types and are important for some endemic species. Buffer zones and community reserves are often located next to existing PAs and generally protect more of already well-represented vegetation types, as they are mostly designed to provide local communities with the right to access natural resources for alternative livelihoods in remoter areas (Hoffmann 2022). Hunting reserves also neighbour PAs and so also often protect more of the same vegetation types, but some are not contiguous and so help meet targets by conserving new ecosystems. This is why CAs as a whole play an important role in filling the target gaps, particularly for vegetation types, showing the benefit of enabling a wide range of conservation types in the country (Hanson et al. 2020).

This explanation of the role of the PAs and different types of CA does not explain why these sites are much more effective at representing vegetation types than species, or why CAs add little in meeting species targets. For example, the PAs and CAs fail to meet targets for 86% of invertebrates, three quarters of fish, two thirds of reptiles, and more than half of amphibians and plants (Figure 5-4), and the CAs only help meet targets for an additional 33 species. One possible reason is that broad biodiversity elements, such as vegetation types, may not be effective at representing important species (Brooks et al. 2004). It is also possible that important species are better sampled in the existing PAs (Boakes et al. 2010), and while the distribution modelling we used helps account for that bias, it is still possible that the CA network contains more habitat for these species than we accounted for in our analyses. However, the key reason is probably more fundamental, as historically, the allocation of PAs and CAs was not based on conserving threatened, endemic or range restricted species and ecosystems. This is why we need a systematic conservation planning approach to identify how to address these gaps.

### 5.4.2 Spatial prioritisation scenarios

Systematic conservation planning is an operational model for identifying and implementing the conservation of priority areas (Knight et al. 2006). A key part of this involves framing and developing broad objectives, which then informs the subsequent spatial prioritisation stage to identify where to focus conservation resources (Groves & Game 2016). Our spatial prioritisation analysis presented here should be seen as a first step in guiding conservation action in Mozambique, as while it was underpinned by national targets and commitments related to the Global Biodiversity Framework, it did not involve the major stakeholders. Instead, it was designed to investigate approximately how much land would need to be managed for conservation under different scenarios, and to identify the best Marxan with Zones parameter values for producing efficient and ecologically viable results.

Thus, there is still a need for future work that includes relevant stakeholders to establish a conservation plan framework and account for additional relevant socio-economic data (Obura et al. 2021; Niemiec et al. 2021; Cunningham et al. 2023), based on the targets and Marxan with Zones parameters we developed.

We investigated two scenarios, each of which consisted of 15 analyses. Based on these analyses we selected one set of parameters that produced the most suitable results and so the rest of this discussion will be based on that set of parameters. For the most suitable analysis from the Without CAs scenarios, 10.6% of the country is already in PAs and we identified the need for an additional 1.4% to be assigned to the High Management Intensity zone and 19.7% for the Low Management Intensity zone. Thus, the total area under conservation would be 31.7%, just exceeding the 30% target outlined in the Global Biodiversity Framework Target 3. This near tripling of the area of land under conservation would obviously have a huge impact on the people of Mozambique, although most of it would be in low management intensity areas and so would still be suitable for livelihoods that did not lead to landcover change, such as some types of pastoralism and ecotourism (Gurney et al. 2021). These results are obviously strongly driven by the targets that were used, which were based on a number of quantitative approaches commonly used in systematic conservation planning but still underpinned by the values of the people setting them (Smith et al, 2019). This further emphasises the need for future analyses to include a wider range of stakeholders.

For the most suitable analysis from the With CAs scenarios, 10.6% of the country is already in PAs and CAs, and we identified the need for an additional 1.3% to be assigned to the High Management Intensity zone and 12.1% for the Low Management Intensity zone. Thus, the total area under conservation would be 41.2%, producing a much less efficient result than when the CAs were not automatically included in the analyses. This is further illustrated by the finding that 25.6% of the planning units within the existing CAs were never selected by Marxan with Zones in the most suitable of the Without CAs analyses (Figure 5-9). There are some exceptions, as the CAs in the west neighbouring Zimbabwe are selected because they meet targets for some important vegetation types. There are also a few sites in the north and far south of Mozambique that contain the distributions of some endemic species. However, most CAs do not contain the distributions of endemic species, which is further illustrated by the patchy overlap between the selection frequency scores and the currently unprotected KBAs and IPAs (Figure 5-10). Thus, while a number of studies have shown that different types of conservation area are important in filling representation gaps in PA networks (Lindsey et al. 2017; Clements et al. 2019), our results show they may do so in a largely inefficient manner.

This might suggest that it would be more efficient to degazette the existing CAs that are not needed to meet the targets and to replace them with areas highlighted in our prioritisation (Fuller et al. 2010).

However, extreme caution is needed before investigating this idea. This is because the CAs were created for a number of reasons, not just for meeting biodiversity targets, and little of this was accounted for in our analysis. In particular, many of these sites were established to support people's livelihoods and so degazetting them could have severe impacts on equitability. Besides, it could be argued that the results from the With CAs scenario reflects both the reliance of Mozambique's rural population on natural resources and the country's high levels biological diversity and endemism (Butchart et al. 2015). Moreover, our analysis was based in part on identifying low management intensity zones that would be highly suited for community-managed OECMs, based on maintaining ecosystem services, and ecotourism where appropriate, all of which could be at the heart of sustainable rural economies. Thus, the current CBD target of 30% could be seen as a reference minimum threshold (Xu et al 2021) to inform actions towards the ambitious 2050 vision of "living in harmony with nature" (CBD 2022).

### 5.4.3 Management effectiveness and implementation

Our analysis was based on zoning for conservation management, rather than for different types of conservation area or land ownership. This is because we were more interested in accounting for conservation outcomes and reflects the fact that environmental legislation in Mozambique supports a broad range of conservation area types (MITADER 2019), so there is scope for tailoring the type of conservation area created at a site to local conditions. The zones were based on management intensity, with the High Management Intensity zone designed for species that are sensitive to over-harvesting or direct persecution. In this zone effective management requires anti-poaching enforcement (Hilborn et al. 2006) and sometimes implementing species-specific projects, such as plant propagation for valuable medicinal plants (Senkoro et al. 2019). Thus, our analyses were highly dependent on whether the endemic and range restricted amphibians, invertebrates, plant and reptile species are listed as sensitive to over-harvesting or direct persecution based on the IUCN Red List. Such data are likely to be robust but we know from previous work that this sensitivity can be different at a regional and global level (Chapter 3) and that over-harvesting pressure on some species can be much lower in remote areas with lower human population density (Ahrends et al. 2010; Di Marco et al. 2018).

Our analysis also assumed that all of the existing PAs currently have high management intensity and all existing CAs have low management intensity, and that these sites are effectively managed and so fully contribute to meeting the relevant targets. But we know that is not the case. Conservation in Mozambique is still impacted by the 16 year long civil war that disrupted conservation governance, management and infrastructure, leading to increased levels of exploitation (Stalmans et al. 2019). The PAs have relatively high levels of resources and law enforcement, but funding disproportionately benefits a small number of them. For CAs funding and management can be low and habitat

degradation is common through the expansion of subsistence farming and charcoal production (Tokura et al. 2020). This is important because our scenarios show that even more land is required, with up to an additional 20% of Mozambique prioritised for low management intensity conservation. Thus, it is vital that any expansion of the conservation estate does not lead to resources being spread more thinly, especially as previous research has shown that it is often more effective to use resources to improve the effectiveness of existing PAs, rather than create new ones (Adams et al. 2019).

To achieve the targets used in our analyses, the Government of Mozambique will need to diversify conservation funding and management, with a focus on long-term financial sustainability (Murphy et al. 2021). This will not be easy, as the country still faces serious challenges in providing essential services for its citizens, and increasing climate change impacts makes this even harder. Such a situation means that funding for conservation is always likely to be highly constrained, but this makes the systematic conservation planning approach even more important. Our work in Mozambique shows it is relatively straightforward to plan at this national level for different types of management zone, even in low-income countries where primary biological data is not readily available. So, the next step is for the government to lead on framing a conservation planning project to inform action on the ground directly. This should involve defining objectives and targets through partnerships with relevant stakeholders, such as non-governmental organisations, conservation donors and community representatives. Building on the subsequent insights from systematic conservation planning, Mozambique can take further steps to developing networks of PAs and OECMs that conserve the country's globally important biodiversity and support people's livelihoods and wellbeing.

# 6 Discussion

## 6.1 Overview

In response to global biodiversity declines, signatory nations to the Convention on Biological Diversity (CBD) have set the ambitious Kunming-Montreal Global Biodiversity Framework. As part of this, they have committed to scaling up conservation areas to encompass a third of the globe by 2030, with a vision of "living in harmony with nature by 2050" (CBD 2022). This involves balancing the need to provide adequate standards of living for people, including access to food and clean water (Díaz et al. 2019), whilst minimising actions causing environmental degradation and halting biodiversity loss. In order to be successful, conservation efforts must focus on areas that are important for biodiversity (Butchart et al. 2012). One framework for establishing such pathways is based on systematic conservation planning (Margules & Pressey 2000). This is a context-specific approach that involves (a) framing the scope of the initiative, (b) setting measurable objectives, defining the minimum amount of biological features that need to be represented, and (c) identifying and implementing networks of priority areas for conservation (Kukkala & Moilanen 2013).

The Global Biodiversity Framework targets are a set of commitments by national governments, so this approach needs to be implemented at the national level, both through transfrontier conservation projects and national level planning (Bicknell et al. 2017). Doing so ensures that multiple values of biodiversity are recognised and inform steps through which to promote engagement with a wide range of stakeholders (Armitage et al. 2020). This puts local stakeholders at the centre of leading context-specific transformative actions and mainstreaming targets that can foster accountable positive biodiversity outcomes (Perino et al. 2022). This thesis looked at how to do this for Mozambique, both through a specific transfrontier project in Maputaland and through national-level work, providing insights that can inform similar work around the world. In this chapter, I begin by outlining how this thesis contributes to the literature and then identify three broad themes that link much of this work. Next, I discuss limitations with this research and suggest ideas for future work and finish by making recommendations for policymakers.

## 6.2 Contributions to the literature

### 6.2.1 Chapter 2

In a wide range of fields including, for example, ecology, conservation biology and climate change, predictive species distribution models (SDMs) are largely used to work out potential species habitat distribution in a defined region (Phillips et al. 2006; Elith et al. 2011; Renner et al. 2015). Species

distribution models are used as a proxy to inform conservation prioritisation (Buchanan & Leidner 2018; Domisch et al. 2019; Macdonald et al. 2019; Allen et al. 2023). I contribute to the literature by modelling the species extent of suitable habitat to inform the delineation of Important Plant Areas (IPAs) and Key Biodiversity Areas (KBAs).

In the process of identifying IPAs and KBAs one of the most challenging steps is the process of delineating boundaries. The initial boundaries are known as ecological boundaries, as they are primarily based on assembling distributional data of trigger elements (KBA Standards and Appeals Committee 2022). However, this is not straightforward (e.g. Chapter 3; (Darbyshire et al. 2023). As a robust and repeatable method, segmentation has been used in a number of studies to map vegetation communities (Massada et al. 2012; Munyati et al. 2013; Munyati 2018). Through analysis carried out as part of **Chapter 2**, I show that segmentation is a valuable proxy to inform initial delineation of boundaries for IPAs and KBAs. This produces outputs that could form the basis of stakeholder engagement to produce the final site boundaries.

To ensure that important areas for biodiversity are mainstreamed into decision-making, the process of identification is led at the national level. For that reason, KBA guidelines do not recommend delineating an area beyond the political borders of a country, other than for sites that are existing or near-completed transfrontier conservation areas (KBA Standards and Appeals Committee 2022). In **Chapter 2**, I show that in a transfrontier centre of endemism where habitats of trigger elements straddle international borders, it is better to highlight the exact areas prior to any cross-border conservation planning. This could then inform initiatives aimed at developing cross-border or regional plans in light of Target 1 of the Global Biodiversity Framework, which aims to target "areas of high biodiversity importance" (CBD 2022). I use the IPA and KBA delineation to show that political borders divide important sites for biodiversity into smaller units that may not be viable separately.

### 6.2.2 Chapter 3

**Chapter 3** builds on an initial Maputaland conservation plan system developed about 15 years ago, which aimed at establishing transfrontier conservation areas across Eswatini, Mozambique and South Africa (Smith et al. 2008). Systematic conservation planning is designed to be an ongoing process (Groves & Game 2016), and I carry out the first analysis to update a systematic conservation planning analysis designed for implementation, over a decade after the initial analysis, adding more data and widening the stakeholder base. The pattern of the findings highlights the impact of including more data on endemic species, as the results show the importance of the Lubombo Mountains in contributing to meeting the conservation targets.

Worldwide, systematic conservation planning has been extensively used to identify where new conservation areas should be located (Sinclair et al. 2018), but largely without considering the type of management that could be implemented. Recent developments in this field are shifting towards taking management into account. In general, this involves using Marxan with Zones, as this allows the user to allocate planning units into specific zones based on meeting conservation targets (Watts et al. 2009). Much of this work has focused on marine conservation planning (Klein et al. 2010), so my contribution to the literature consisted of identifying terrestrial community-led conservation areas that accounted for ecotourism potential.

Management effectiveness of protected areas is regarded pivotal for a successful conservation outcome (Negret et al. 2020; Zafra-Calvo & Geldmann 2020; Powlen et al. 2021). In Marxan with Zones, different management types can be considered, including what actions should be implemented to accomplish each zone's objective (Adams et al. 2016; Law et al. 2017). I contribute to the literature by investigating scenarios that accounting for management effectiveness in terms of conservation outcomes for species sensitive to over-harvesting or direct persecution, identifying where potential OECMs would need more resources to guarantee management effectiveness.

### 6.2.3 Chapter 4

**Chapter 4** focused on mobilising comprehensive global distributional data across a wide range of taxonomic groups to then apply the KBA criteria at a national level. As part of this I developed a post-hoc theory of change, showing how biodiversity knowledge could be incorporated into decision-making in Mozambique. As part of this, I showed how a low-income nation established a mechanism to: (i) mainstream biodiversity matters; (ii) inform development decisions; (iii) work towards meeting global biodiversity targets; and (iv) report back at the international level about biodiversity.

The process of identifying KBAs is based on strong engagement with relevant stakeholders across a range of disciplines, with a focus on local expertise (Plumptre et al. 2019). I contributed to the literature by establishing a novel way of implementing KBA identification in a low-resourced country. This process involved mobilising a wide range of expertise at national, regional and international levels to identify important species and delineate important areas for biodiversity.

The KBA approach is designed to account for the limited availability of data (Meyer et al. 2015). Each site is evaluated independently in the study region, provided that relevant global data for trigger elements has been gathered (Brooks et al. 2015). Moreover, trigger elements are assessed against all criteria based on adequate data, but meeting one of the criteria is sufficient to qualify (KBA Standards and Appeals Committee 2022). I contribute to the literature by demonstrating the benefits of that

approach. In particular, I show that a number of marine species could not trigger KBA status when evaluated against Sub-criterion A1 aimed at threatened species, despite being Vulnerable and Endangered under the IUCN Red List, but did qualify when assessed under Sub-criterion D1 on demographic aggregations.

### 6.2.4 Chapter 5

Governments need to step up at the national level to develop plans that will help draw pathways towards achieving internationally agreed commitments (Bicknell et al. 2017). In **Chapter 5**, I contribute to the literature by applying Marxan with Zones to identify a nationwide potential terrestrial network of OECMs. This is in a country that has never carried out spatial prioritisation analyses at a national scale before.

In many parts of the world, there is growing recognition that OECMs provide significant contributions to the current network of PAs (Geldmann et al. 2021; Rodríguez-Rodríguez et al. 2021; Shiono et al. 2021). They complement the PAs helping to meet national and international conservation targets (Dudley et al. 2018) because some OECMs contain under-represented landcover types and species. Management of these areas could be the responsibility of a wide range of stakeholders or groups of people including local communities. This could galvanise support to achieve positive biodiversity outcomes efficiently with less resources. I contribute to the literature through illustrating, based on a gap analysis for Mozambique, that OECMs provide a significant contribution to the current network of PAs. These OECMs are managed cost-effectively, but improvements are necessary to ensure management effectiveness and yield positive biodiversity outcomes.

## 6.3 Synthesis

### 6.3.1 Benefits of more biodiversity data

Systematic conservation planning depends on the availability of high-quality biodiversity to make informed decisions about where to focus conservation actions. However, there is always a trade-off in whether to spend resources on collecting more data or using existing datasets and focusing on conservation implementation (Grantham et al. 2008). Our work from Maputaland shows that having more data on more species highlighted areas that were missed in a previous spatial prioritisation (Smith et al. 2008). In particular, this is apparent in the Lubombo Mountains to the west, which are geomorphologically different from the eastern area that consists of a low-lying and flat coastal plane and contains more PAs. The geomorphological differences between the east and west (du Randt 2018) leads to significant differences with respect to species composition, particularly for plants creating

ecological communities that could be regarded as sub-centres of endemism within the wider biodiversity hotspot (Darbyshire et al. 2019).

At the national level, where additional data has been gathered for the last 15 years (Monadjem et al. 2010; Timberlake et al. 2011; Conradie et al. 2016; Tolley et al. 2019), our findings show that there are priority areas that are currently outside of the existing PAs. This is particularly because of the wide range of endemic species across different taxonomic groups, which were the focus of our national scale KBA identification (Chapter 4). Thus, I found large benefits in collating the existing data at a national scale, and focusing resources on surveying poorly-documented taxa in a biodiversity hotspot. Collecting more data will also enable us to learn more about level of species' rarity and endemism to better inform future analyses (Rivers et al. 2011; Roberts et al. 2016).

In addition to the species data, our analysis also benefited from a new Mozambique vegetation map (Lötter et al. 2021). This was because most of the work over the last decade collating species data has focused on endemic, near-endemic and range restricted species, so this dataset is biased towards areas with high concentrations of such species. By including distribution data on different vegetation types, we could ensure that our national-scale systematic conservation planning could account for broader biodiversity (Noss, 1990).

### 6.3.2 Comparison and synergies between the IPA, KBA and systematic conservation planning approaches

There are many different ways to identify important sites for biodiversity (Eken et al. 2004; Ricketts et al. 2005; BirdLife International 2014; Plantlife International 2018) and to prioritise conservation action (Watts et al. 2009; Moilanen 2012; Di Minin et al. 2013). This has also led to discussions in the literature about which is the best approach. In this study, I focused on the IPA (Darbyshire et al. 2017) and KBA approaches (IUCN 2016) and how they relate to systematic conservation planning.

My research revealed four important differences between the IPA and KBA approaches during this study, which stem from the former being developed for plants and fungi and the latter for all taxonomic groups and ecosystems. First, they differ in the proportion thresholds used and the species KBA criteria consist of both proportion threshold and reproductive unit threshold. I found that this is a significant limitation when evaluating plant species against the KBA criteria because population census data for this group is not readily available. This is not a requirement for the IPA approach. Second, IPAs account for species, subspecies and varieties, whereas the KBAs focuses on species level. Third, the KBA approach requires that the data being used for assessment is recent enough to ensure that a KBA will be identified in a place where the species still extant. This is not a requirement

for the IPA, although the approach stresses that it is always advisable to double check. Fourth, KBAs needs to be delineated to overlap with existing management units or with the potential to be a management unit. This is to ensure that the KBAs are mainstreamed into decision making and stand a high chance of being implemented. This is less of a consideration in the IPA approach.

Despite these differences, I found that the two approaches are complementary. In particular, in **Chapter 2** I found that all the important plant species identified in the KBA analysis are also important for triggering IPAs and that the process for identifying sites for these species was near identical. The only key difference is that the IPA process identified important sub-species and varieties that were absent from the KBA list. Comparing the IPA and KBA and systematic conservation planning outputs in **Chapter 5**, also showed high levels of overlap, although the IPAs are generally much smaller than the KBAs and include more land that is currently not protected.

The best approach for achieving a more holistic plan for guiding conservation action is systematic conservation planning, which helps meet the identified targets (Woodley et al. 2019). Because Mozambique has conducted both IPA and KBA identification at national scale, which involved identifying priority species, I used the data from these initiatives to inform systematic conservation planning at the national scale. Through establishing synergies between systematic conservation planning, KBAs (Smith et al. 2019) and IPAs, I produced a network of potential OECMs that account for areas that have been identified as important for biodiversity in Mozambique. By overlapping IPAs, KBAs and the best systematic conservation planning output (Figure 5-10), I found that an area of 5,513 $km^2$ covers all three and remains outside the current network of protected areas. Because the Kunming-Montreal Global Biodiversity Framework stresses the need for conserving areas that are important for biodiversity (CBD 2022), inclusion of this land into Mozambique's network of PAs would help the country to meet national and international conservation targets.

The processes of identifying IPAs and KBAs in Mozambique also involved building capabilities particularly on Red Listing and institutional capacity. Moreover, both are based on strong engagement with experts and other relevant stakeholders, therefore creating the recognition that biodiversity data should be used in decision making.

### 6.3.3 The value of systematic conservation planning for informing OECMs

Identifying new OECMs is complex, so the systematic conservation planning approach helps define objectives and measure the contribution of different types of conservation area (Kukkala & Moilanen 2013). In **Chapter 3** and **Chapter 5**, I follow this approach to first carry out a gap analysis then identify where it would be most appropriate to allocate new OECMs to fill the gaps in the existing

network of PAs in a cost-effective way (McIntosh et al. 2016). This was possible in both chapters because the core part of the systematic conservation planning is based on accounting for complementarity whilst ensuring representativeness of the conservation elements (Geldmann et al. 2021). Moreover, it seeks to establish a viable network of areas through connectivity to guarantee movement of species within the planning region.

It is worth noting that, until about a decade ago, the application of the systematic conservation planning did not incorporate aspects on management (Klein et al. 2010). However, there is a need to account for different management zones, so Marxan with Zones can play a large role (Watts et al. 2009). In **Chapter 3** and **Chapter 5**, I apply Marxan with Zones to identify OECMs based on two management zones, first in a transfrontier biodiversity hotspot then at a national level. It was possible to carry out such analysis because Marxan with Zones allows the user to place planning units into specific zones, accounting for different management types by defining how they would meet different targets. Inspired by the need to develop an inclusive conservation system, in both chapters I defined zones based on their required management intensity. Although implementation of these conservation systems will require further analyses that account for locally relevant data, both plans give an opportunity for government to enlarge their base of stakeholder involvement in conservation (Rodríguez-Rodríguez et al. 2021).

## 6.4 Limitations and future work

### 6.4.1 Data

Data is the key component informing the analysis of spatial prioritisation for biodiversity conservation (Heinicke et al. 2019; Xuereb et al. 2021). I found that data gathered over the last decade in Mozambique and the transfrontier region of Maputaland has played a major role in informing the identification of important areas for biodiversity, which then formed the basis to identify priority areas for conservation. However, despite all efforts to mobilise data, extensive areas of Mozambique remain under-sampled. In addition, our data set consisted of endemic, near-endemic and range restricted species, with plants species being disproportionately represented. It is therefore necessary to target less surveyed geographical areas and taxonomic groups in the future. This should then be followed by the Red Listing process, building on experience from the IUCN Species Survival Commission Southern Africa Plant Specialist Group.

Spatial conservation prioritisation also use socio-economic data (Gilby et al. 2021) to inform where action should be focused. For the Mozambique analysis, I used agricultural opportunity costs as the socio-economic cost metric, as agriculture is the major land use (FNDS 2020). Building from this, future work should produce opportunity cost data for a wider range of industries, such as mining. In

Chapter 3, I show that ecotourism data can play a major role in informing the identification of conservation areas in Maputaland. Mozambique is a prime tourism destination, so future research should also identify areas with high ecotourism potential at a national scale.

In the national KBA analysis (Chapter 4), I show that our knowledge about species distribution in the country is limited. As our knowledge about species endemism improves, the current pattern of important sites is likely to change, especially for sites identified based on one or two species. Therefore, future research is needed, first targeting these areas and associated species and then expanding into other areas. This is particularly relevant because KBAs are being used to inform where new development initiatives should be avoided and the implementation of offsetting policies.

Setting conservation targets is a core part of systematic conservation planning, so analyses are sensitive to the methods used (Levin et al. 2015; Green et al. 2018). In Chapters 3 and 5 I used the approach developed by Pfab et al. (2011) to set conservation targets for some species because this method has been successfully used in South Africa. This approach is based on the IUCN Red List global assessments, so the targets represent the minimum amount of area or populations that the planning region can contribute to ensuring biodiversity persistence. In some cases, this led to very low targets being set, so there is a need for more data and tools to assist with the process of setting species targets in line with Red List assessments. In some cases the stakeholders felt the targets were too low for some taxonomic groups, so I was able to overcome this limitation by adopting different approaches (Rodrigues et al. 2004; Smith et al. 2008). Another relevant aspect to consider in conservation is climate change. The impact of climate change in Mozambique is noticeable, particularly with more frequent and intense natural hazards occurring. In order to yield positive biodiversity outcomes, future work on conservation planning needs to account for climate change and land-use change in a systematic way (Stralberg et al. 2020).

### 6.4.2 Management effectiveness

Site-based approaches, mainly in the form of PAs, have been a major focus of efforts to address biodiversity loss over the decades (Maxwell et al. 2018; Bhola et al. 2021). The intended objectives of these areas together with OECMs will only be achieved if they are managed effectively (Geldmann et al. 2019, 2021; Wauchope et al. 2022). I assumed that management of the current network of protected areas (PAs) and conservation areas (CAs) throughout the planning region is effective. Thus, these areas contribute to meeting the conservation targets for the biodiversity features.

However, it is important to note that this does not reflect reality on the ground. Thus, more research is needed on the actual impacts of management effectiveness on the species used in our analyses. In the

PAs management is largely adequate, so research is particularly needed on how effective CAs are in terms of conservation outcomes (Allan et al. 2022). This research should account for funding levels in CAs and how best to address the current shortfalls (Miller et al. 2013). Through my thesis, I show that there is high potential for ecotourism in the region, so there is also a need to investigate whether this could be an adequate source of funding for areas being managed with low intensity.

### 6.4.3 More stakeholder engagement

Conservation is a cross-disciplinary field, therefore decisions about what, where, and how to implement such initiatives require engaging with a wide range of stakeholders (Wheeler & Root-Bernstein 2020; Hamelin et al. 2023). As part of Chapter 3, I established a group of taxon experts and ecologists working in the region on conservation management in public and private areas, the governmental and academic sectors. However, I did not have the opportunity to engage with landowners to gather their contributions. Therefore, more research is needed on the context of engaging with landowners and other social groups, and how to incorporate such data into planning. For Chapter 5, I focused on accounting for national policies and international commitments with limited stakeholder engagement. In Mozambique, future work needs to account for stakeholder engagement with an emphasis on the local level.

## 6.5 Recommendations for policymakers and practitioners

The process of identifying areas that are important for biodiversity, including IPAs and KBAs, is pivotal in our effort to reverse the trend of biodiversity loss (Jones et al. 2020; Mateo-Ramírez et al. 2021). This is especially relevant because parties to the CBD have committed to focusing conservation on areas that are important for biodiversity (CBD 2022). Throughout my thesis, I show the benefits of applying global approaches to inform national plans and systematic conservation planning analyses. Governments, together with a broader range of stakeholders, including the private sector, should encourage more of these initiatives. This should then be translated into policy to guide sustainable development ensuring biodiversity persistence. Moreover, data sets from these analyses provide information about species' IUCN Red List status, distributions and levels of endemism. Therefore, effort should be made to refer to these data sets when making decisions about matters related to biodiversity and socio-economic development. There is also a need to develop theories of change to clearly articulate how national and international commitments should be translated into action (Rice et al. 2020).

Mozambique has made significant improvements in conservation management since the end of the civil war two decades ago, but several issues remain to be addressed, particularly in the CAs. Despite this, I show through the identification of KBAs that some CAs, especially forest reserves, contain

important species that are found nowhere else. A similar pattern was found in another study aimed at identifying important plant areas in Mozambique (Darbyshire et al. 2023). This means that those CAs are the only places worldwide where this species can be conserved. I would therefore encourage the government and conservation practitioners to consider addressing the management of such areas in new ways, especially as research shows that focusing on existing network of conservation areas is often the most cost-effective way of improving conservation (Adams et al. 2016).

Although the management of CAs needs to be addressed urgently, considerations should be made on a site-by-site basis and ideally be informed by clear and targeted studies. This is also relevant to initiatives aimed at expanding existing PAs. Moreover, future studies need to account for global changes in order to inform policy and practice. Working closely with the government can lead to effective changes to policy. For example, I have shown that the government can take the lead in the KBA process. Carrying out KBA assessment in this way helps the government understand the relevance of the process and speed up mainstreaming steps into policy. In addition, it broadens government's knowledge about the available expertise, helps identify gaps provides evidence on how to develop solutions.

## 6.6 Conclusions

The 196 countries signatories to the CBD recently committed to bold targets on reverse the trend of biodiversity loss (CBD 2022). This is an ideal moment for completing analyses that will inform changes as we proceed towards 2030. Over time, Mozambique shifted from species-oriented conservation to landscape-focused approaches, including transfrontier initiatives. In this thesis I have applied global methodologies to identify key areas for biodiversity, identified priority areas for conservation and provided insights on how those areas could be managed. I also showed how to identify community-based conservation areas (Armitage et al. 2020) that can be funded cost-effectively through ecotourism in a transfrontier biodiversity hotspot. Various lessons can be drawn from these results and can be applied across many parts of the world. Through this I have shown that applying well-targeted spatial prioritisation approaches, with a focus on management effectiveness and putting the local communities and stakeholders at the forefront of conceptualising and implementing initiatives, is the best approach for yielding positive biodiversity outcomes.

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# Supplementary material

## Chapter 2 - Supplementary material

***Table S2-1: Details of KBA and IPA criteria applied in the Maputaland analysis***

| Criteria | Description | Thresholds |
|---|---|---|
| **A – KBA** | **Threatened biodiversity** | **Site regularly contains one or more of:** |
| **A1 (KBA)** | **Threatened species**<br>A1 sites hold a significant proportion of the global threatened species. | a) ≥0.5% + ≥5 RUs of a CR or EN.<br>b) ≥1% + ≥10 RUs of a VU.<br>c) ≥0.1% + ≥5 RUs of a CR or EN due only to past and present population reduction.<br>d) ≥0.2% + ≥10 RUs of a VU due only to past and present population reduction.<br>e) 100% global population size of a CR or EN. |
| **B – KBA** | **Geographically restricted biodiversity** | **Site regularly contains:** |
| **B1 (KBA)** | **Individual geographically restricted species**<br>B1 sites hold a significant proportion of the global geographically restricted species. | ≥10% of the global population size AND ≥10 RUs |
| **A – IPA** | **Threatened species** | **Site known, thought or inferred to contain:** |
| **A(i) IPA** | **Globally threatened species**<br>A(i) sites hold one or more **globally threatened** species and meet the associated thresholds. | ≥1% of the global population AND/OR<br>≥5% of the national population OR<br>5 best sites for the species nationally |
| **A(ii) IPA** | **Regionally threatened species**<br>A(ii) sites contain one or more **regionally threatened** species and meet the associated thresholds. | ≥5% of the national population OR<br>5 best sites for the species nationally |
| **A(iv) IPA** | **Range restricted endemic species**<br>A(iv) sites hold one or more **range restricted endemic** species that are potentially threatened. | ≥1% of the global population AND/OR<br>≥5% of the national population OR<br>5 best sites for the species nationally |

***Table S2-2: List of trigger species for the KBA and IPA assessment. The list includes both fauna and flota taxonomic groups. The KBA approach is applicable for the two taxonomic groups whereas, the IPA methodology is relevant for plants only.***

| Taxon Group | Taxon Name | IUCN status Global | IUCN Status Regional | KBA trigger criteria | IPA trigger criteria |
|---|---|---|---|---|---|
| Amphibian | *Breviceps carruthersi* | NE | NE | B1 | – |
| Amphibian | *Breviceps passmorei* | NE | NE | B1 | – |
| Freshwater Fish | *Marcusenius sp. nov. kosi* | EN | EN | A1a; B1 | – |
| Freshwater Fish | *Silhouettea sibayi* | EN | EN | A1a; B1 | – |
| Invertebrate | *Afrarchaea ansieae* | NE | NE | B1 | – |
| Invertebrate | *Agriocnemis ruberrima ruberrima* | EN | EN | A1a | – |
| Invertebrate | *Amblyphymus adspersus* | NT | NT | B1 | – |
| Invertebrate | *Anomalipus decosteri* | NE | NE | B1 | – |
| Invertebrate | *Anomalipus parallelus zulu* | NE | NE | B1 | – |
| Invertebrate | *Anomalipus savii savii* | NE | NE | B1 | – |
| Invertebrate | *Arytropteris basalis* | VU | VU | A1b | – |
| Invertebrate | *Chaleponcus delagoanus* | NE | NE | B1 | – |
| Invertebrate | *Cheiramiona haddadi* | NE | NE | B1 | – |
| Invertebrate | *Cheiramiona tembensis* | NE | NE | B1 | – |
| Invertebrate | *Dendryphantes matumi* | NE | NE | B1 | – |
| Invertebrate | *Doratogonus major* | NT | NE | B1 | – |
| Invertebrate | *Evarcha striolata* | NE | NE | B1 | – |
| Invertebrate | *Gamasomorpha longisetosa* | NE | NE | B1 | – |
| Invertebrate | *Gulella appletoni* | NE | NE | B1 | – |
| Invertebrate | *Henicovalva lebomboensis* | NE | NE | B1 | – |
| Invertebrate | *Hortipes aelurisiepae* | NE | NE | B1 | – |
| Invertebrate | *Iolaus lulua* | VU | NE | A1b | – |
| Invertebrate | *Latrodectus umbukwane* | NE | NE | B1 | – |
| Invertebrate | *Parepistaurus inhaca* | NE | NE | B1 | – |
| Invertebrate | *Phlegra arborea* | NE | NE | B1 | – |

| | | | | | |
|---|---|---|---|---|---|
| Invertebrate | *Phoenicocampa terinata* | NE | NE | B1 | – |
| Invertebrate | *Phymeurus illepidus* | VU | VU | A1b | – |
| Invertebrate | *Planochelas haddadi* | NE | NE | B1 | – |
| Invertebrate | *Poachelas refugus* | NE | NE | B1 | – |
| Invertebrate | *Pomatonota dregii* | VU | VU | A1b; B1 | – |
| Invertebrate | *Proandricus hlatikulu* | NE | NE | B1 | – |
| Invertebrate | *Scarabaeus bornemizzai* | NE | NT | B1 | – |
| Invertebrate | *Scarabaeus clericus* | NE | NE | B1 | – |
| Invertebrate | *Scarabaeus clericus* | NE | NE | B1 | – |
| Invertebrate | *Spinotarsus ingwavuma* | NE | NE | B1 | – |
| Invertebrate | *Teriomima zuluana* | VU | VU | A1b | – |
| Invertebrate | *Tugelana butleri* | NE | NE | B1 | – |
| Invertebrate | *Zerenopsis meraca* | NE | NE | B1 | – |
| Invertebrate | *Zinophora brevilobata* | NT | NE | B1 | – |
| Invertebrate | *Natalina wesseliana* | VU | NE | A1b | – |
| Plant | *Acridocarpus natalitius var. linearifolius* | VU | NE | – | A(i) |
| Plant | *Asparagus radiatus* | NE | NE | B1 | A(iv); B(ii) |
| Plant | *Australluma ubomboensis* | NE | LC | B1 | B(ii) |
| Plant | *Barleria lebomboensis* | NE | NE | B1 | A(iv); B(ii) |
| Plant | *Barleria oxyphylla* | VU | NE | A1b | A(i); B(ii) |
| Plant | *Blepharis sericea* | NE | Critically Rare | B1 | A(iii); A(iv); B(ii) |
| Plant | *Blepharis swaziensis* | NE | LC | B1 | B(ii) |
| Plant | *Brachystelma tenue* | NE | EN | B1 | A(iv); B(ii) |
| Plant | *Brachystelma vahrmeijeri* | NE | EN | B1 | A(iv); B(ii) |
| Plant | *Ceropegia aloicola* | EN | NE | A1a; B1 | A(i); B(ii) |
| Plant | *Ceropegia arenaria* | NE | EN | B1 | A(i); B(ii) |
| Plant | *Cola dorrii* | EN | NE | A1a | A(i) |
| Plant | *Combretum eugeneanum* | NE | NE | B1 | A(iv); B(ii) |
| Plant | *Delosperma lebomboense* | NE | LC | B1 | B(ii) |
| Plant | *Dicerocaryum forbesii* | LC | NE | – | B(ii) |

| | | | | | |
|---|---|---|---|---|---|
| Plant | *Dicliptera quintasii* | VU | EN | A1b; B1 | A(i); B(ii) |
| Plant | *Dioscorea sylvatica* | VU | VU | A1b | A(i) |
| Plant | *Empogona maputensis* | EN | LC | A1a; B1 | A(i); B(ii) |
| Plant | *Encephalartos aplanatus* | VU | NE | A1b; B1 | A(i); B(ii) |
| Plant | *Encephalartos lebomboensis* | EN | EN | A1a; B1 | A(i) |
| Plant | *Encephalartos ngoyanus* | EN | EN | A1a; B1 | A(i) |
| Plant | *Encephalartos senticosus* | EN | EN | A1a; B1 | A(i); B(ii) |
| Plant | *Encephalartos umbeluziensis* | EN | NE | A1a; B1 | A(i); B(ii) |
| Plant | *Euphorbia baylissii* | VU | NE | A1b; B1 | A(i); B(ii) |
| Plant | *Euphorbia keithii* | NE | NE | B1 | A(i); B(ii) |
| Plant | *Gladiolus brachyphyllus* | NE | LC | B1 | B(ii) |
| Plant | *Helichrysum moggii* | LC | NE | B1 | B(ii) |
| Plant | *Indigofera gobensis* | CR | NE | A1a; B1 | A(i); B(ii) |
| Plant | *Jatropha latifolia var. subeglandulosa* | NE | NE | – | A(iv); B(ii) |
| Plant | *Metathelypteris burrowsiorum* | NE | NE | B1 | A(iv); B(ii) |
| Plant | *Mondia whitei* | NE | EN | B1 | A(iv); B(ii) |
| Plant | *Pachycarpus lebomboensis* | NE | Rare | B1 | A(iv); B(ii) |
| Plant | *Panicum sancta-luciense* | NE | Rare | B1 | A(iv); B(ii) |
| Plant | *Pavetta vanwykiana* | LC | NE | B1 | B(ii) |
| Plant | *Phyllanthus sp. nov.* | NE | NE | B1 | A(iv); B(ii) |
| Plant | *Polygala francisci* | VU | NE | A1b; B1 | A(i); B(ii) |
| Plant | *Prionostemma delagoensis* | NT | NE | B1 | B(ii) |
| Plant | *Prunus africana* | VU | VU | A1b | A(i); B(ii) |
| Plant | *Psychotria amboniana subsp. mosambicensis* | VU | NE | – | A(i); B(ii) |
| Plant | *Psydrax fragrantissima* | NT | VU | B1 | A(ii); B(ii) |
| Plant | *Raphia australis* | VU | VU | A1b; B1 | A(i); B(ii) |
| Plant | *Rytigynia celastroides var. australis* | VU | NE | – | A(i) |
| Plant | *Sclerochiton apiculatus* | VU | VU | A1b; B1 | A(i); B(ii) |
| Plant | *Searsia kwazuluana* | NE | Rare | B1 | A(iv); B(ii) |
| Plant | *Solanum litoraneum* | EN | NE | A1a; B1 | A(i); B(ii) |

| | | | | | |
|---|---|---|---|---|---|
| Plant | *Tephrosia forbesii subsp. Forbesii* | VU | NE | – | A(i) |
| Plant | *Tephrosia forbesii subsp. Inhacensis* | VU | NE | – | A(i); B(ii) |
| Plant | *Tephrosia gobensis* | NE | NE | B1 | A(iv); B(ii) |
| Plant | *Thesium jeanae* | NE | Rare | B1 | A(iv); B(ii) |
| Plant | *Thesium polygaloides* | VU | VU | A1b; B1 | A(i); B(ii) |
| Plant | *Tragia glabrata var. hispida* | NE | NE | – | B(ii) |
| Plant | *Warburgia salutaris* | EN | EN | A1a | A(i) |
| Plant | *Warneckea parvifolia* | EN | CR | A1a; B1 | A(i); B(ii) |
| Plant | *Xylopia torrei* | EN | NE | A1a; B1 | A(i); B(ii) |
| Plant | *Zanthoxylum delagoense* | LC | NE | – | B(ii) |
| Reptile | *Acontias parietalis* | LC | LC | B1 | – |
| Reptile | *Afroedura marleyi* | LC | LC | B1 | – |
| Reptile | *Bradypodion setaroi* | LC | LC | B1 | – |
| Reptile | *Kinixys natalensis* | VU | VU | A1b | – |
| Reptile | *Lycophidion pygmaeum* | LC | LC | B1 | – |
| Reptile | *Platysaurus lebomboensis* | LC | LC | B1 | – |
| Reptile | *Scelotes fitzsimonsi* | LC | LC | B1 | – |
| Reptile | *Scelotes vestigifer* | LC | LC | B1 | – |
| Reptile | *Smaug warreni* | LC | LC | B1 | – |

# Chapter 3 - Supplementary material

***Table S3-1: List of conservation features used in the Maputaland conservation planning system, their targets and their total area/population size within the planning region. Columns Scen A to Scen C represent the scenarios of whether the feature would contribute towards targets if found in the Low Management Intensity Zone.***

| Id | Name | Target | Total | Scen A | Scen B | Scen C |
|---|---|---|---|---|---|---|
| 1 | Lubombo aquatic N | 864.6 | 3333.1 | 1 | 1 | 1 |
| 2 | Lubombo aquatic C | 731.1 | 2184.0 | 1 | 1 | 1 |
| 3 | Lubombo aquatic S | 411.2 | 1440.6 | 1 | 1 | 1 |
| 4 | Rock-faces N | 1879.5 | 2850.2 | 1 | 1 | 1 |
| 5 | Rock-faces C | 1294.9 | 1752.7 | 1 | 1 | 1 |
| 6 | Rock-faces S | 661.7 | 947.3 | 1 | 1 | 1 |
| 7 | Lubombo grassland N | 6395.9 | 24135.0 | 1 | 1 | 1 |
| 8 | Lubombo grassland C | 363.4 | 1107.6 | 1 | 1 | 1 |
| 9 | Lubombo grassland S | 342.2 | 1167.2 | 1 | 1 | 1 |
| 10 | Lubombo woodland N | 41143.3 | 158248.3 | 1 | 1 | 1 |
| 11 | Lubombo woodland C | 12128.6 | 37178.1 | 1 | 1 | 1 |
| 12 | Lubombo woodland S | 7349.0 | 27169.3 | 1 | 1 | 1 |
| 13 | Lubombo thicket N | 7500.7 | 29311.4 | 1 | 1 | 1 |
| 14 | Lubombo thicket C | 7251.1 | 21649.4 | 1 | 1 | 1 |
| 15 | Lubombo thicket S | 1310.5 | 4487.2 | 1 | 1 | 1 |
| 16 | Lubombo forest N | 5855.1 | 8533.1 | 1 | 1 | 1 |
| 17 | Lubombo forest C | 3505.7 | 4261.5 | 1 | 1 | 1 |
| 18 | Lubombo forest S | 117.4 | 154.2 | 1 | 1 | 1 |
| 19 | Acacia woodland N | 64721.5 | 125916.7 | 1 | 1 | 1 |
| 20 | Acacia woodland S | 13266.6 | 25623.2 | 1 | 1 | 1 |
| 21 | Acacia thicket N | 4898.4 | 20455.7 | 1 | 1 | 1 |
| 22 | Acacia thicket S | 6977.1 | 30292.4 | 1 | 1 | 1 |
| 23 | Floodplain grassland N | 7620.0 | 14054.8 | 1 | 1 | 1 |
| 24 | Floodplain grassland S | 1060.3 | 3326.2 | 1 | 1 | 1 |
| 25 | Reed beds | 4132.2 | 13929.9 | 1 | 1 | 1 |
| 26 | Riverine thicket | 4788.5 | 13848.8 | 1 | 1 | 1 |
| 27 | Riverine forest | 2265.7 | 3258.3 | 1 | 1 | 1 |
| 28 | Sedge and grass swamp | 4687.4 | 12812.8 | 1 | 1 | 1 |
| 29 | Hygrophilous grasslands | 19318.7 | 55729.5 | 1 | 1 | 1 |
| 30 | Woody grassland | 44536.5 | 164839.7 | 1 | 1 | 1 |
| 31 | Terminalia woodland | 70157.9 | 289955.7 | 1 | 1 | 1 |
| 32 | Woodland on red sands | 1990.2 | 8501.1 | 1 | 1 | 1 |
| 33 | Sand thicket | 34951.2 | 45176.7 | 1 | 1 | 1 |
| 34 | Sand forest | 50656.0 | 65775.1 | 1 | 1 | 1 |
| 35 | Inland evergreen forest | 9975.4 | 14981.1 | 1 | 1 | 1 |
| 36 | Swamp forest | 0.0 | 4584.9 | 1 | 1 | 1 |
| 37 | Mangroves N | 2884.9 | 4639.9 | 1 | 1 | 1 |

| | | | | | | |
|---|---|---|---|---|---|---|
| 38 | Mangroves S | 143.4 | 143.4 | 1 | 1 | 1 |
| 39 | Beach | 917.1 | 3779.3 | 1 | 1 | 1 |
| 40 | Dune thicket | 1574.8 | 2252.8 | 1 | 1 | 1 |
| 41 | Dune forest N | 3065.0 | 4272.1 | 1 | 1 | 1 |
| 42 | Dune forest S | 8128.7 | 11732.4 | 1 | 1 | 1 |
| 43 | Towns | 0.0 | 32758.5 | 1 | 1 | 1 |
| 44 | Subsistence agriculture | 0.0 | 396476.3 | 1 | 1 | 1 |
| 45 | Commercial agriculture | 0.0 | 45327.7 | 1 | 1 | 1 |
| 46 | Plantations | 0.0 | 73658.9 | 1 | 1 | 1 |
| 47 | Quarry | 0.0 | 389.9 | 1 | 1 | 1 |
| 48 | Open Water | 17001.6 | 70840.0 | 1 | 1 | 1 |
| 49 | Mud Flats | 3065.7 | 12773.7 | 1 | 1 | 1 |
| 50 | Salt marsh | 3528.3 | 14701.1 | 1 | 1 | 1 |
| 1001 | *Breviceps carruthersi* | 0.0 | 10835.7 | 1 | 1 | 1 |
| 1002 | *Breviceps passmorei* | 2099.6 | 2346.0 | 1 | 1 | 1 |
| 1003 | *Breviceps sopranus* | 13200.0 | 64269.3 | 1 | 1 | 1 |
| 1004 | *Hyperolius pickersgilli* | 0.0 | 433.2 | 1 | 1 | 1 |
| 2001 | *Anthus brachyurus* | 200.0 | 163725.6 | 1 | 1 | 1 |
| 2002 | *Aquila rapax* | 88900.0 | 840323.1 | 1 | 0 | 0 |
| 2003 | *Bucorvus leadbeateri* | 63000.0 | 713912.3 | 1 | 0 | 0 |
| 2004 | *Caprimulgus natalensis* | 90600.0 | 248432.4 | 1 | 1 | 1 |
| 2005 | *Ciconia nigra* | 27300.0 | 986187.7 | 1 | 0 | 0 |
| 2006 | *Circaetus fasciolatus* | 63000.0 | 71222.3 | 1 | 1 | 1 |
| 2007 | *Circus ranivorus* | 2400.0 | 106603.0 | 1 | 1 | 1 |
| 2008 | *Ephippiorhynchus senegalensis* | 25000.0 | 110038.4 | 1 | 1 | 1 |
| 2009 | *Falco biarmicus* | 27000.0 | 957999.9 | 1 | 0 | 0 |
| 2010 | *Falco concolor* | 30000.0 | 250121.3 | 1 | 1 | 1 |
| 2011 | *Glaucidium capense* | 4200.0 | 123968.0 | 1 | 1 | 1 |
| 2012 | *Gorsachius leuconotus* | 80200.0 | 265345.3 | 1 | 1 | 1 |
| 2013 | *Gyps africanus* | 0.0 | 891350.7 | 1 | 0 | 0 |
| 2014 | *Gyps coprotheres* | 0.0 | 886400.0 | 1 | 0 | 0 |
| 2015 | *Halcyon senegaloides* | 12500.0 | 308575.4 | 1 | 1 | 1 |
| 2016 | *Macheiramphus alcinus* | 30000.0 | 276934.4 | 1 | 1 | 1 |
| 2017 | *Microparra capensis* | 300.0 | 99645.5 | 1 | 1 | 1 |
| 2018 | *Mycteria ibis* | 200.0 | 123706.6 | 1 | 1 | 1 |
| 2019 | *Necrosyrtes monachus* | 9100.0 | 330061.6 | 1 | 1 | 0 |
| 2020 | *Neotis denhami* | 900.0 | 168687.4 | 1 | 1 | 1 |
| 2021 | *Nettapus auritus* | 40000.0 | 106603.0 | 1 | 1 | 1 |
| 2022 | *Pelecanus rufescens* | 300.0 | 276530.1 | 1 | 1 | 1 |
| 2023 | *Podica senegalensis* | 1200.0 | 34222.4 | 1 | 1 | 1 |
| 2024 | *Poicephalus cryptoxanthus* | 57900.0 | 731027.7 | 1 | 0 | 0 |
| 2025 | *Polemaetus bellicosus* | 77800.0 | 866728.9 | 1 | 0 | 0 |
| 2026 | *Sagittarius serpentarius* | 100000.0 | 276863.7 | 1 | 0 | 0 |
| 2027 | *Scotopelia peli* | 3000.0 | 16997.1 | 1 | 1 | 1 |
| 2028 | *Smithornis capensis* | 100.0 | 196425.3 | 1 | 1 | 1 |
| 2029 | *Terathopius ecaudatus* | 40300.0 | 866728.9 | 1 | 0 | 0 |

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|---|---|---|---|---|---|---|
| 2030 | *Torgos tracheliotos* | 50000.0 | 865755.7 | 1 | 0 | 0 |
| 2031 | *Trigonoceps occipitalis* | 166700.0 | 866016.5 | 1 | 0 | 0 |
| 2032 | *Turnix nanus* | 100.0 | 250456.1 | 1 | 1 | 1 |
| 2033 | *Tyto capensis* | 7600.0 | 19297.3 | 1 | 1 | 1 |
| 3001 | *Chetia brevis* | 9756.2 | 11175.8 | 1 | 1 | 1 |
| 3002 | *Chiloglanis emarginatus* | 9736.9 | 11637.4 | 1 | 1 | 1 |
| 3003 | *Enteromius gurneyi* | 0.0 | 559.3 | 1 | 1 | 1 |
| 3004 | *Marcusenius sp nov kosi* | 0.0 | 4568.4 | 1 | 1 | 1 |
| 3005 | *Oreochromis mossambicus* | 13200.0 | 82860.6 | 1 | 1 | 1 |
| 3006 | *Serranochromis meridianus* | 13200.0 | 30311.9 | 1 | 1 | 1 |
| 3007 | *Silhouettea sibayi* | 13200.0 | 30311.9 | 1 | 1 | 1 |
| 3008 | *Streptocephalus dendrophorus* | 1.0 | 1.0 | 1 | 1 | 1 |
| 4001 | *Agriocnemis ruberrima ruberrima* | 76800.0 | 95367.9 | 1 | 1 | 1 |
| 4002 | *Amblyphymus adspersus* | 69300.0 | 461590.7 | 1 | 1 | 1 |
| 4003 | *Anomalipus parallelus zulu* | 1.0 | 1.0 | 1 | 1 | 1 |
| 4004 | *Anomalipus savii savii* | 3.0 | 3.0 | 1 | 1 | 1 |
| 4005 | *Arytropteris basalis* | 2.0 | 2.0 | 1 | 1 | 1 |
| 4006 | *Copris inhalatus inhalatus* | 29400.0 | 146865.3 | 1 | 1 | 1 |
| 4007 | *Copris urus* | 14200.0 | 71032.5 | 1 | 1 | 1 |
| 4008 | *Digrammacris bifIdus* | 4.0 | 4.0 | 1 | 1 | 1 |
| 4009 | *Exoprosopa mozambica* | 13200.0 | 20785.1 | 1 | 1 | 1 |
| 4010 | *Iolaus lulua* | 13200.0 | 15434.3 | 1 | 1 | 1 |
| 4011 | *Onthophagus giuseppecarpanetoi* | 48600.0 | 161935.6 | 1 | 1 | 1 |
| 4012 | *Parepistaurus inhaca* | 2.0 | 2.0 | 1 | 1 | 1 |
| 4013 | *Phoenicocampa terinata* | 1.0 | 1.0 | 1 | 1 | 1 |
| 4014 | *Phymeurus illepidus* | 4.0 | 4.0 | 1 | 1 | 1 |
| 4015 | *Pomatonota dregii* | 2862.2 | 3298.5 | 1 | 1 | 1 |
| 4016 | *Scarabaeus bornemizzai* | 46400.0 | 242140.9 | 1 | 1 | 1 |
| 4017 | *Scarabaeus clericus* | 1.0 | 1.0 | 1 | 1 | 1 |
| 4018 | *Teriomima zuluana* | 56900.0 | 85706.8 | 1 | 1 | 1 |
| 4019 | *Tugelana butleri* | 4.0 | 4.0 | 1 | 1 | 1 |
| 4020 | *Zerenopsis meraca* | 3.0 | 4.0 | 1 | 1 | 1 |
| 4021 | *Henicovalva lebomboensis* | 1.0 | 1.0 | 1 | 1 | 1 |
| 4022 | *Zinophora brevilobata* | 9.0 | 9.0 | 1 | 1 | 1 |
| 5001 | *Afrarchaea ansieae* | 1.0 | 1.0 | 1 | 1 | 1 |
| 5002 | *Chaleponcus delagoanus* | 1.0 | 1.0 | 1 | 1 | 1 |
| 5003 | *Cheiramiona haddadi* | 1.0 | 1.0 | 1 | 1 | 1 |
| 5004 | *Cheiramiona tembensis* | 1.0 | 1.0 | 1 | 1 | 1 |
| 5005 | *Dendryphantes matumi* | 1.0 | 1.0 | 1 | 1 | 1 |
| 5006 | *Doratogonus major* | 0.0 | 1663.1 | 1 | 1 | 1 |
| 5007 | *Evarcha striolata* | 4.0 | 4.0 | 1 | 1 | 1 |
| 5008 | *Gamasomorpha longisetosa* | 3.0 | 3.0 | 1 | 1 | 1 |
| 5009 | *Hortipes aelurisiepae* | 9.0 | 9.0 | 1 | 1 | 1 |
| 5010 | *Latrodectus umbukwane* | 4.0 | 4.0 | 1 | 1 | 1 |
| 5011 | *Natalina wesseliana* | 13200.0 | 23602.4 | 1 | 1 | 1 |
| 5012 | *Phlegra arborea* | 4.0 | 4.0 | 1 | 1 | 1 |

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|---|---|---|---|---|---|---|
| 5013 | *Planochelas haddadi* | 2.0 | 2.0 | 1 | 1 | 1 |
| 5014 | *Poachelas refugus* | 2.0 | 2.0 | 1 | 1 | 1 |
| 5015 | *Proandricus hlatikulu* | 1.0 | 1.0 | 1 | 1 | 1 |
| 5016 | *Spinotarsus ingwavuma* | 4.0 | 4.0 | 1 | 1 | 1 |
| 5017 | *Zinophora brevilobata* | 36400.0 | 352747.1 | 1 | 1 | 1 |
| 6001 | *Amblysomus marleyi* | 14400.0 | 48141.0 | 1 | 1 | 1 |
| 6002 | *Ceratotherium simum* | 23.0 | 3807.4 | 1 | 0 | 0 |
| 6003 | *Diceros bicornis* | 290.0 | 2380.5 | 1 | 0 | 0 |
| 6004 | *Giraffa camelopardalis giraffa* | 48.0 | 3433.1 | 1 | 1 | 0 |
| 6005 | *Hippopotamus amphibius* | 650.0 | 5647.5 | 1 | 1 | 0 |
| 6006 | *Loxodonta africana* | 600.0 | 2543.1 | 1 | 0 | 0 |
| 6007 | *Lycaon pictus* | 18.0 | 109.2 | 1 | 0 | 0 |
| 6008 | *Myosorex sclateri* | 5400.0 | 17340.3 | 1 | 1 | 1 |
| 6009 | *Nesotragus moschatus zuluensis* | 16.0 | 12819.7 | 1 | 0 | 0 |
| 6010 | *Ourebia ourebi ourebi* | 1000.0 | 10472.9 | 1 | 0 | 0 |
| 6011 | *Panthera leo* | 300.0 | 1042.0 | 1 | 0 | 0 |
| 6012 | *Panthera pardus* | 13.0 | 1080.9 | 1 | 0 | 0 |
| 6013 | *Redunca fulvorufula* | 16.0 | 1140.7 | 1 | 0 | 0 |
| 6014 | *Smutsia temminckii* | 5000.0 | 873983.2 | 1 | 0 | 0 |
| 6015 | *White rhino North* | 0.0 | 2378.3 | 1 | 0 | 0 |
| 6016 | *White rhino South* | 0.0 | 1429.0 | 1 | 0 | 0 |
| 6017 | *Black rhino North* | 90.0 | 1469.6 | 1 | 0 | 0 |
| 6018 | *Black rhino South* | 200.0 | 910.9 | 1 | 0 | 0 |
| 7001 | *Acridocarpus natalitius var linearifolius* | 61400.0 | 102253.1 | 1 | 1 | 1 |
| 7002 | *Adenium swazicum* | 13200.0 | 23449.3 | 1 | 0 | 0 |
| 7003 | *Adenopodia schlechteri* | 0.0 | 4890.1 | 1 | 1 | 1 |
| 7004 | *Albizia suluensis* | 5694.1 | 6301.0 | 1 | 1 | 1 |
| 7005 | *Aloe keithii* | 21000.0 | 37974.4 | 1 | 0 | 0 |
| 7006 | *Asclepias gordon-grayae* | 2.0 | 2.0 | 1 | 1 | 1 |
| 7007 | *Aspalathus gerrardii* | 13200.0 | 18639.5 | 1 | 1 | 1 |
| 7008 | *Asparagus minutiflorus* | 5.0 | 5.0 | 1 | 1 | 1 |
| 7009 | *Asparagus radiatus* | 0.0 | 7660.2 | 1 | 1 | 1 |
| 7010 | *Australluma ubomboensis* | 0.0 | 10289.0 | 1 | 0 | 0 |
| 7011 | *Barleria lebomboensis* | 2.0 | 2.0 | 1 | 1 | 1 |
| 7012 | *Barleria oxyphylla* | 0.0 | 11992.7 | 1 | 1 | 1 |
| 7013 | *Begonia homonyma* | 13200.0 | 23616.1 | 1 | 1 | 1 |
| 7014 | *Berchemia sp nov* | 6.0 | 6.0 | 1 | 0 | 0 |
| 7015 | *Blepharis sericea* | 0.0 | 14439.3 | 1 | 1 | 1 |
| 7016 | *Blepharis swaziensis* | 29600.0 | 82014.3 | 1 | 1 | 1 |
| 7017 | *Bonatea lamprophylla* | 31400.0 | 83259.0 | 1 | 1 | 1 |
| 7018 | *Brachystelma sandersonii* | 61400.0 | 291759.3 | 1 | 1 | 1 |
| 7019 | *Brachystelma tenue* | 35800.0 | 136642.6 | 1 | 1 | 1 |
| 7020 | *Brachystelma vahrmeijeri* | 3.0 | 4.0 | 1 | 1 | 1 |
| 7021 | *Bulbostylis parvinux* | 57200.0 | 580017.2 | 1 | 1 | 1 |
| 7022 | *Callilepis corymbosa* | 3.0 | 3.0 | 1 | 1 | 1 |
| 7023 | *Caputia medleywoodii* | 1.0 | 1.0 | 1 | 1 | 1 |

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|---|---|---|---|---|---|---|
| 7024 | *Celosia nervosa* | 1.0 | 1.0 | 1 | 1 | 1 |
| 7025 | *Ceropegia arenaria* | 2.0 | 2.0 | 1 | 1 | 1 |
| 7026 | *Ceropegia cimiciodora* | 2.0 | 2.0 | 1 | 1 | 1 |
| 7027 | *Cola dorrii* | 95700.0 | 407685.0 | 1 | 1 | 1 |
| 7028 | *Combretum eugeneanum* | 11.0 | 19.0 | 1 | 1 | 1 |
| 7029 | *Crassula lanceolata subsp lebomboensis* | 0.0 | 14029.5 | 1 | 1 | 1 |
| 7030 | *Crassula maputensis* | 55700.0 | 332444.3 | 1 | 1 | 1 |
| 7031 | *Cussonia arenicola* | 2.0 | 2.0 | 1 | 1 | 1 |
| 7032 | *Cyathocoma bachmannii* | 0.0 | 1.0 | 1 | 1 | 1 |
| 7033 | *Delosperma lebomboense* | 13200.0 | 40965.0 | 1 | 1 | 1 |
| 7034 | *Dicerocaryum forbesii* | 59300.0 | 332444.3 | 1 | 1 | 1 |
| 7035 | *Dicliptera quintasii* | 30400.0 | 452718.3 | 1 | 1 | 1 |
| 7036 | *Dioscorea sylvatica* | 11747.2 | 14393.0 | 1 | 1 | 0 |
| 7037 | *Disperis woodii* | 2.0 | 2.0 | 1 | 1 | 1 |
| 7038 | *Distephanus inhacensis* | 40100.0 | 92324.9 | 1 | 1 | 1 |
| 7040 | *Empogona maputensis* | 47200.0 | 106759.2 | 1 | 1 | 1 |
| 7041 | *Encephalartos aplanatus* | 20820.0 | 55141.3 | 1 | 0 | 0 |
| 7042 | *Encephalartos ferox subsp ferox* | 57200.0 | 146771.5 | 1 | 0 | 0 |
| 7043 | *Encephalartos lebomboensis* | 33000.0 | 74039.4 | 1 | 0 | 0 |
| 7044 | *Encephalartos ngoyanus* | 28000.0 | 74039.4 | 1 | 0 | 0 |
| 7045 | *Encephalartos senticosus* | 4454.2 | 5286.5 | 1 | 0 | 0 |
| 7046 | *Encephalartos umbeluziensis* | 23400.0 | 83141.2 | 1 | 0 | 0 |
| 7047 | *Englerina schlechteri* | 21100.0 | 83141.2 | 1 | 1 | 1 |
| 7048 | *Euphorbia baylissii* | 1.0 | 1.0 | 1 | 1 | 1 |
| 7049 | *Excoecaria madagascariensis* | 3.0 | 3.0 | 1 | 1 | 1 |
| 7050 | *Fimbristylis aphylla* | 2.0 | 3.0 | 1 | 1 | 1 |
| 7051 | *Freesia laxa subsp azurea* | 27800.0 | 35470.9 | 1 | 1 | 1 |
| 7052 | *Gardenia thunbergia Thunb* | 4.0 | 4.0 | 1 | 1 | 1 |
| 7053 | *Habenaria woodii* | 2.0 | 2.0 | 1 | 1 | 1 |
| 7054 | *Haworthiopsis limifolia var limifolia* | 5.0 | 5.0 | 1 | 1 | 1 |
| 7055 | *Haworthiopsis limifolia var ubomboensis* | 31900.0 | 151209.4 | 1 | 1 | 1 |
| 7056 | *Helichrysum moggii* | 2.0 | 2.0 | 1 | 1 | 1 |
| 7057 | *Hermannia micropetala* | 37400.0 | 158610.5 | 1 | 1 | 1 |
| 7058 | *Heywoodia lucens Sim* | 13200.0 | 16332.3 | 1 | 1 | 1 |
| 7059 | *Indigofera gobensis* | 2.0 | 4.0 | 1 | 1 | 1 |
| 7060 | *Kniphofia leucocephala* | 13200.0 | 16909.9 | 1 | 1 | 1 |
| 7061 | *Kniphofia tysonii subsp lebomboensis* | 34000.0 | 191531.7 | 1 | 1 | 1 |
| 7062 | *Krauseola mosambicina* | 0.0 | 1.0 | 1 | 1 | 1 |
| 7063 | *Lumnitzera racemosa var racemosa* | 11.0 | 16.0 | 1 | 1 | 1 |
| 7064 | *Metathelypteris burrowsiorum* | 1.0 | 1.0 | 1 | 0 | 0 |
| 7065 | *Mondia whitei* | 13200.0 | 19720.8 | 1 | 0 | 0 |
| 7066 | *Mystacidium aliceae* | 13200.0 | 19720.8 | 1 | 0 | 0 |
| 7067 | *Nesaea wardii* | 3.0 | 3.0 | 1 | 1 | 1 |
| 7068 | *Nidorella tongensis* | 2.0 | 2.0 | 1 | 1 | 1 |
| 7069 | *Ocimum reclinatum* | 23200.0 | 55755.4 | 1 | 1 | 1 |
| 7070 | *Orbea gerstneri subsp gerstneri* | 1.0 | 1.0 | 1 | 0 | 0 |

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|---|---|---|---|---|---|---|
| 7071 | *Oxygonum dregeanum var strictum* | 55200.0 | 452996.0 | 1 | 1 | 1 |
| 7072 | *Pachycarpus concolor subsp arenicola* | 97800.0 | 452996.0 | 1 | 1 | 1 |
| 7073 | *Pachycarpus lebomboensis* | 20500.0 | 71823.5 | 1 | 1 | 1 |
| 7074 | *Panicum sanctaluciense* | 11.0 | 12.0 | 1 | 1 | 1 |
| 7075 | *Pavetta vanwykiana* | 57000.0 | 362846.3 | 1 | 1 | 1 |
| 7076 | *Pavonia dregei* | 72200.0 | 362846.3 | 1 | 1 | 1 |
| 7077 | *Pelargonium tongaense* | 34000.0 | 136200.6 | 1 | 1 | 1 |
| 7078 | *Plectranthus psammophilus* | 1.0 | 3.0 | 1 | 1 | 1 |
| 7079 | *Polygala francisci* | 4.0 | 4.0 | 1 | 1 | 1 |
| 7080 | *Prionostemma delagoensis* | 26300.0 | 70943.4 | 1 | 1 | 1 |
| 7081 | *Prunus africana* | 32900.0 | 93154.5 | 1 | 0 | 0 |
| 7082 | *Psychotria amboniana subsp mosambicensis* | 0.0 | 3588.6 | 1 | 1 | 1 |
| 7083 | *Psydrax fragrantissima* | 65100.0 | 325748.0 | 1 | 1 | 1 |
| 7084 | *Psydrax moggii* | 46000.0 | 325748.0 | 1 | 1 | 1 |
| 7085 | *Putterlickia neglecta* | 68000.0 | 300973.8 | 1 | 1 | 1 |
| 7086 | *Raphia australis* | 0.0 | 3102.0 | 1 | 1 | 1 |
| 7087 | *Raphionacme elsana* | 57000.0 | 173455.7 | 1 | 1 | 1 |
| 7088 | *Restio zuluensis* | 38600.0 | 154596.5 | 1 | 1 | 1 |
| 7089 | *Rytigynia celastroides var australis* | 5.0 | 5.0 | 1 | 1 | 1 |
| 7090 | *Schlechterina mitostemmatoides* | 13200.0 | 50170.9 | 1 | 1 | 1 |
| 7091 | *Sclerochiton apiculatus* | 61200.0 | 378918.2 | 1 | 1 | 1 |
| 7092 | *Searsia kwazuluana* | 55100.0 | 220506.4 | 1 | 1 | 1 |
| 7093 | *Selaginella culverwellii* | 6.0 | 6.0 | 1 | 1 | 1 |
| 7094 | *Stangeria eriopus* | 22000.0 | 98277.8 | 1 | 0 | 0 |
| 7095 | *Streptocarpus confusus subsp lebomboensis* | 30100.0 | 194383.8 | 1 | 1 | 0 |
| 7096 | *Syncolostemon ramosus* | 3.0 | 3.0 | 1 | 1 | 1 |
| 7097 | *Tephrosia gobensis* | 3.0 | 3.0 | 1 | 1 | 1 |
| 7098 | *Thesium jeanae* | 5.0 | 5.0 | 1 | 1 | 1 |
| 7099 | *Thesium polygaloides* | 44100.0 | 123867.4 | 1 | 1 | 1 |
| 7100 | *Vangueria monteiroi* | 52000.0 | 261860.6 | 1 | 1 | 1 |
| 7101 | *Warburgia salutaris* | 62200.0 | 240439.1 | 1 | 0 | 0 |
| 7102 | *Warneckea parvifolia* | 24000.0 | 258887.7 | 1 | 1 | 1 |
| 7103 | *Xylopia torrei* | 13200.0 | 61936.7 | 1 | 1 | 1 |
| 7104 | *Zanthoxylum delagoense* | 24000.0 | 201692.1 | 1 | 1 | 1 |
| 8001 | *Acontias parietalis* | 92100.0 | 410365.2 | 1 | 1 | 1 |
| 8002 | *Afroedura marleyi* | 49800.0 | 149298.7 | 1 | 1 | 1 |
| 8003 | *Bradypodion setaroi* | 27200.0 | 92816.8 | 1 | 1 | 1 |
| 8004 | *Kinixys natalensis* | 59400.0 | 295434.0 | 1 | 1 | 1 |
| 8005 | *Lycophidion pygmaeum* | 47300.0 | 140941.4 | 1 | 1 | 1 |
| 8006 | *Platysaurus lebomboensis* | 47200.0 | 183284.9 | 1 | 1 | 1 |
| 8007 | *Scelotes arenicolus* | 24300.0 | 101617.3 | 1 | 1 | 1 |
| 8008 | *Scelotes fitzsimonsi* | 13200.0 | 26225.5 | 1 | 1 | 1 |
| 8009 | *Scelotes vestigifer* | 18400.0 | 53662.4 | 1 | 1 | 1 |
| 8010 | *Smaug warreni* | 89400.0 | 397039.9 | 1 | 0 | 0 |

# Chapter 4 - Supplementary material

***Table S4-1: List of trigger species with the KBA criteria and sub-criteria summarised for each species.***

| Taxon Group | Species | IUCN Red List | KBA criteria triggered |
|---|---|---|---|
| Amphibian | *Arthroleptis francei* | VU | A1b(iv) |
| Amphibian | *Hyperolius spinigularis* | VU | A1b(iv) |
| Amphibian | *Hyperolius swynnertoni* | LC | B1(iv) |
| Amphibian | *Nothophryne baylissi* | EN | A1a(iv) |
| Amphibian | *Nothophryne inagoensis* | EN | A1a(iv); B1(iv) |
| Amphibian | *Nothophryne ribauensis* | EN | A1a(iv); A1e(iv); B1(iv) |
| Amphibian | *Nothophryne unilurio* | CR | A1a(iv); A1e(iv); B1(iv) |
| Amphibian | *Strongylopus rhodesianus* | VU | A1b(iv) |
| Bird | *Apalis lynesi* | NT | B1(i) |
| Bird | *Arcanator orostruthus* | VU | A1b(i) |
| Bird | *Artisornis sousae* | EN | A1a(i); A1e(i); B1(i) |
| Bird | *Balearica regulorum* | EN | A1a(i); A1c(i) |
| Bird | *Bugeranus carunculatus* | VU | A1b(i); A1d(i) |
| Bird | *Chamaetylas choloensis* | VU | A1b(i); B1(i); B2(i) |
| Bird | *Gyps africanus* | CR | A1a(i); A1c(i) |
| Bird | *Hirundo atrocaerulea* | VU | A1b(i) |
| Bird | *Sternula balaenarum* | VU | A1b(i); B1(i) |
| Bird | *Swynnertonia swynnertoni* | VU | A1b(i); B2(i) |
| Bird | *Trigonoceps occipitalis* | CR | A1a(i); A1c(i) |
| Fish (freshwater) | *Chetia brevis* | EN | A1a(iv) |
| Fish (freshwater) | *Nothobranchius hengstleri* | EN | A1a(iv); B1(iv) |
| Fish (freshwater) | *Nothobranchius kadleci* | NT | B1(iv) |
| Fish (freshwater) | *Nothobranchius krammeri* | VU | A1b(iv); B1(iv) |
| Fish (freshwater) | *Nothobranchius krysanovi* | VU | A1b(iv); B1(iv) |
| Fish (freshwater) | *Nothobranchius niassa* | VU | A1b(iv); B1(iv) |
| Fish (freshwater) | *Oreochromis mortimeri* | CR | A1a(iv); A1c(iv) |
| Fish (freshwater) | *Silhouettea sibayi* | EN | A1a(iv) |
| Fish (marine) | *Mobula alfredi* | VU | D1b(i) |
| Fish (marine) | *Mobula birostris* | EN | D1b(i) |
| Fish (marine) | *Parablennius lodosus* | VU | A1b(iv) |
| Fish (marine) | *Rhincodon typus* | EN | D1b(i) |
| Fish (marine) | *Stegostoma tigrinum* | EN | D1b(i) |
| Fish (marine) | *Caranx Ignobilis* | LC | D1b(i) |
| Insect | *Acraea dondoensis* | | B1(v) |
| Insect | *Alaena lamborni* | | B1(v) |

| | | | |
|---|---|---|---|
| Insect | *Amblyphymus adspersus* | NT | B1(iv) |
| Insect | *Baliochila woodi* | | B1(v) |
| Insect | *Cymothoe baylissi* | | B1(v) |
| Insect | *Graphium junodi* | | B1(v) |
| Insect | *Iolaus (Epamera) malaikae* | | B1(v) |
| Insect | *Iolaus lulua* | VU | A1b(v); B1(v) |
| Insect | *Neocoenyra bioculata* | | B1(v) |
| Insect | *Pseudonympha cyclops* | | B1(iv) |
| Insect | *Scarabaeus bornemizzai* | NT | B1(iv) |
| Mammal | *Carpitalpa arendsi* | VU | A1b(iv); B1(iv) |
| Mammal (marine) | *Dugong dugon* | VU | A1d(i) |
| Mammal | *Hippopotamus amphibius* | VU | A1b(i); A1d(i) |
| Mammal | *Loxodonta africana* | EN | A1a(i); A1c(i) |
| Mammal | *Lycaon pictus* | EN | A1a(i) |
| Mammal | *Panthera leo* | VU | A1b(i); A1d(i) |
| Mammal | *Paraxerus vincenti* | EN | A1a(iv) |
| Mammal (marine) | *Sousa plumbea* | EN | A1a(iv); A1c(iv) |
| Plant | *Acacia latispina* | VU | A1b(iv); B1(iv); B2(iv) |
| Plant | *Acacia quiterajoensis* | LC | B1(v) |
| Plant | *Acacia torrei* | LC | B1(iv) |
| Plant | *Aeschynomene aphylla* | VU | A1b(iv); B1(iv) |
| Plant | *Agelanthus patelii* | EN | A1a(v); A1e(v); B1(v) |
| Plant | *Alloeochaete namuliensis* | VU | A1b(iv); B1(iv) |
| Plant | *Allophylus torrei* | EN | A1a(v); B1(v) |
| Plant | *Aloe plowesii* | VU | A1b(v); B1(v) |
| Plant | *Aloe ribauensis* | EN | A1a(v); B1(v) |
| Plant | *Aloe rulkensii* | CR | A1a(i); A1e(i); B1(i) |
| Plant | *Aspidoglossum glabellum* | EN | A1a(v); B1(v) |
| Plant | *Brachystelma nutans* | VU | A1b(v); B1(v) |
| Plant | *Buchnera subglabra* | VU | A1b(v); B1(v) |
| Plant | *Centella obtriangularis* | VU | A1b(v); B1(v) |
| Plant | *Chassalia colorata* | EN | A1a(v); B1(v) |
| Plant | *Cissus aristolochiifolia* | VU | A1b(v); B1(v) |
| Plant | *Clerodendrum lutambense* | VU | A1b(iv); B1(iv) |
| Plant | *Coffea schliebenii* | VU | A1b(iv); B1(iv) |
| Plant | *Coleus namuliensis* | LC | B1(v) |
| Plant | *Cordia stuhlmannii* | VU | A1b(v); B1(v) |
| Plant | *Crossopetalum mossambicense* | EN | A1e(v) |
| Plant | *Crotalaria insignis* | VU | A1b(v); B1(v) |
| Plant | *Crotalaria namuliensis* | LC | B1(v) |

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|---|---|---|---|
| Plant | *Croton aceroides* | EN | A1a(v); B1(v) |
| Plant | *Croton Inhambanensis* | VU | A1b(v); B1(v) |
| Plant | *Cynanchum oresbium* | VU | A1b(v); B1(v) |
| Plant | *Danthoniopsis chimanimaniensis* | EN | A1a(v); B1(v) |
| Plant | *Didymosalpinx callianthus* | EN | A1a(iv); B1(iv); B2(iv) |
| Plant | *Dierama inyangense* | EN | A1a(v); B1(v) |
| Plant | *Disa zimbabweensis* | VU | A1b(v); B1(v) |
| Plant | *Dissotis pulchra* | VU | A1b(v); B1(v) |
| Plant | *Dissotis swynnertonii* | VU | A1b(v); B1(v) |
| Plant | *Dombeya leachii* | EN | A1a(v); A1e(v); B1(v) |
| Plant | *Dorstenia zambesiaca* | VU | A1b(iv); B1(iv) |
| Plant | *Ecbolium hastatum* | EN | A1a(v); B1(v) |
| Plant | *Empogona jenniferae* | EN | A1a(v); B1(v) |
| Plant | *Empogona maputensis* | EN | A1a(v); B1(v) |
| Plant | *Erica lanceolifera* | VU | A1b(v); B1(v) |
| Plant | *Faurea racemosa* | EN | A1a(v); B1(v) |
| Plant | *Ficus muelleriana* | EN | A1a(v); B1(v) |
| Plant | *Gladiolus zimbabweensis* | VU | A1b(v); B1(v) |
| Plant | *Grewia limae* | EN | A1a(iv); B1(iv) |
| Plant | *Gutenbergia westii* | VU | A1b(v); B1(v) |
| Plant | *Helixanthera schizocalyx* | EN | A1a(v); B1(v) |
| Plant | *Icuria dunensis* | EN | A1a(v); B1(v) |
| Plant | *Impatiens psychadelphoides* | VU | A1b(v); B1(v) |
| Plant | *Impatiens wuerstenii* | VU | A1b(v); B1(v) |
| Plant | *Isoglossa namuliensis* | CR | A1a(v); A1e(v); B1(v) |
| Plant | *Jatropha subaequiloba* | VU | A1b(v); B1(v) |
| Plant | *Justicia gorongozana* | LC | B1(v) |
| Plant | *Maerua andradae* | LC | B1(iv) |
| Plant | *Memecylon incisilobum* | CR | A1a(i); A1e(i); B1(i) |
| Plant | *Memecylon nubigenum* | EN | A1a(v); B1(v) |
| Plant | *Micklethwaitia carvalhoi* | VU | A1b(iv); B2(iv) |
| Plant | *Mimosa busseana* | LC | B1(iv); B2(iv) |
| Plant | *Morella chimanimaniana* | EN | A1a(v); B1(v) |
| Plant | *Neobolusia ciliata* | EN | A1a(v); B1(v) |
| Plant | *Ochna dolicharthros* | VU | A1b(v); A1e(v); B1(v) |
| Plant | *Olinia chimanimani* | EN | A1a(v); B1(v) |
| Plant | *Ormocarpum schliebenii* | NT | B1(iv) |
| Plant | *Otiophora lanceolata* | VU | A1b(v); B1(v) |
| Plant | *Oxyanthus biflorus* | EN | A1a(iv); B1(iv); B2(iv) |

| | | | |
|---|---|---|---|
| Plant | *Oxyanthus strigosus* | EN | A1a(iv); A1e(iv); B1(iv); B2(iv) |
| Plant | *Ozoroa gomesiana* | VU | A1b(v); B1(v) |
| Plant | *Pavetta gurueensis* | VU | A1b(v); B1(v) |
| Plant | *Pavetta lindina* | EN | A1a(iv); B1(iv); B2(iv) |
| Plant | *Plectranthus cucullatus* | VU | A1b(v); B1(v) |
| Plant | *Plectranthus mandalensis* | VU | A1b(v); B1(v) |
| Plant | *Polygala francisci* | VU | A1b(v); B1(v) |
| Plant | *Polygala zambesiaca* | VU | A1b(v); B1(v) |
| Plant | *Polysphaeria ribauensis* | EN | A1a(v); A1e(v); B1(v) |
| Plant | *Protea enervis* | VU | A1b(v); B1(v) |
| Plant | *Pseudomussaenda mozambicensis* | EN | A1a(iv); B1(iv) |
| Plant | *Psydrax fragrantissima* | NT | B1(v) |
| Plant | *Pterocephalus centennii* | CR | A1a(v); A1e(v); B1(v) |
| Plant | *Raphia australis* | VU | A1b(i); B1(i) |
| Plant | *Rhynchosia chimanimaniensis* | EN | A1a(v); B1(v) |
| Plant | *Rytigynia torrei* | EN | A1a(v); B1(v) |
| Plant | *Schistostephium oxylobum* | VU | A1b(v); B1(v) |
| Plant | *Schizochilus lepidus* | VU | A1b(v); B1(v) |
| Plant | *Sclerochiton apiculatus* | VU | A1b(v); B1(v) |
| Plant | *Sclerochiton hirsutus* | VU | A1b(v); B1(v) |
| Plant | *Sericanthe chimanimaniensis* | VU | A1b(v); B1(v) |
| Plant | *Streptocarpus brachynema* | EN | A1a(v); A1e(v); B1(v) |
| Plant | *Streptocarpus hirticapsa* | VU | A1b(v); B1(v) |
| Plant | *Streptocarpus myoporoides* | EN | A1a(i); B1(i) |
| Plant | *Streptocarpus umtaliensis* | LC | B1(v) |
| Plant | *Stylochaeton tortispathus* | VU | A1b(iv); B1(iv) |
| Plant | *Syncolostemon oritrephes* | VU | A1b(v); B1(v) |
| Plant | *Synsepalum sp. nov.* | EN | A1a(v); B1(v) |
| Plant | *Tarenna longipedicellata* | VU | A1b(v); B1(v) |
| Plant | *Tarenna pembensis* | EN | A1a(iv)B1(iv) |
| Plant | *Vepris allenii* | EN | A1a(v); B1(v) |
| Plant | *Vepris drummondii* | VU | A1b(v); B1(v) |
| Plant | *Vepris macedoi* | EN | A1a(v); B1(v) |
| Plant | *Warneckea cordiformis* | CR | A1a(iv); A1e(iv); B1(iv) |
| Plant | *Warneckea parvifolia* | EN | A1a(v); B1(v) |
| Plant | *Xerophyta kirkii* | LC | B1(v) |
| Plant | *Xerophyta pseudopinifolia* | LC | B1(v) |
| Plant | *Xylia mendoncae* | VU | A1b(v); B1(v) |
| Plant | *Xylopia torrei* | EN | A1a(v); B1(v) |

| | | | |
|---|---|---|---|
| Plant | *Xyris asterotricha* | VU | A1b(v); B1(v) |
| Reptile | *Afroedura gorongosa* | LC | B1(iv) |
| Reptile | *Atheris mabuensis* | EN | A1a(iv); B1(iv); B2(iv) |
| Reptile | *Cordylus meculae* | LC | B1(iv) |
| Reptile | *Cycloderma frenatum* | EN | A1a(iv) |
| Reptile | *Dipsadoboa montisilva* | NT | B1(iv); B2(iv) |
| Reptile | *Lygodactylus chiperone* | | B1(iv); B2(iv) |
| Reptile | *Lygodactylus inago* | | B1(iv); B2(iv) |
| Reptile | *Lygodactylus mabu* | | B1(iv); B2(iv) |
| Reptile | *Lygodactylus ribaue* | | B1(iv); B2(iv) |
| Reptile | *Mochlus lanceolatus* | LC | B1(iv) |
| Reptile | *Nadzikambia baylissi* | NT | B1(iv); B2(iv) |
| Reptile | *Nadzikambia chiperone* | | B1(iv); B2(iv) |
| Reptile | *Nadzikambia inago* | | B1(iv); B2(iv) |
| Reptile | *Nadzikambia namuli* | | B1(iv); B2(iv) |
| Reptile | *Nadzikambia ribaue* | | B1(iv); B2(iv) |
| Reptile | *Rhampholeon bruessoworum* | CR | A1a(iv); B1(iv); B2(iv) |
| Reptile | *Rhampholeon gorongosae* | EN | A1a(iv); A1e(iv); B1(iv) |
| Reptile | *Rhampholeon marshalli* | VU | A1b(iv); B1(iv) |
| Reptile | *Rhampholeon maspictus* | NT | B1(iv); B2(iv) |
| Reptile | *Rhampholeon namuli* | | B1(iv); B2(iv) |
| Reptile | *Rhampholeon nebulauctor* | VU | A1b(iv); B1(iv); B2(iv) |
| Reptile | *Rhampholeon tilburyi* | EN | A1a(iv); B1(iv); B2(iv) |
| Reptile | *Scelotes duttoni* | LC | B1(iv) |
| Reptile | *Scelotes insularis* | LC | B1(iv) |
| Reptile | *Scolecoseps broadleyi* | NT | B1(iv) |
| Reptile | *Smaug mossambicus* | LC | B1(iv) |

***Table S4-2: List of the 29 KBAs identified in Mozambique and the species that triggered KBA status per site. Of the 29 KBAs, 25 are terrestrial and four are marine namely: Vamizi, Great Bazaruto, Tofo and Ponta do Ouro Partial Marine Reserve.***

| ID | KBA name | Province | KBA area km2 | KBA criteria | KBA trigger species | Taxonomic Group | IUCN status |
|---|---|---|---|---|---|---|---|
| 1 | Njesi Plateau | Niassa | 1996 | A1a(i); A1e(i); B1(i) | *Artisornis sousae* | Bird | EN |
| | | | | B1(iv) | *Cordylus meculae* | Reptile | LC |
| 2 | Niassa Special Reserve | Niassa | 42708 | A1a(i); A1c(i) | *Loxodonta africana* | Mammal | EN |
| | | | | A1b(i); A1d(i) | *Hippopotamus amphibius* | Mammal | VU |
| | | | | A1a(i) | *Lycaon pictus* | Mammal | EN |
| | | | | A1b(i); A1d(i) | *Panthera leo* | Mammal | VU |
| | | | | A1a(iv) | *Cycloderma frenatum* | Reptile | EN |
| | | | | B1(iv) | *Cordylus meculae* | Reptile | LC |
| | | | | A1b(iv); B1(iv) | *Nothobranchius niassa* | Freshwater Fish | VU |
| 3 | Palma | Cabo Delgado | 4556 | B1(iv) | *Scolecoseps broadleyi* | Reptile | NT |
| | | | | A1b(iv); B1(iv) | *Nothobranchius krammeri* | Freshwater Fish | VU |
| | | | | A1a(iv); B1(iv) | *Nothobranchius hengstleri* | Freshwater Fish | EN |
| | | | | A1a(v); B1(v) | *Chassalia colorata* | Plant | EN |
| | | | | A1a(iv); B1(iv) | *Grewia limae* | Plant | EN |
| | | | | A1a(iv); B1(iv); B2(iv) | *Pavetta lindina* | Plant | EN |
| | | | | A1e(v) | *Crossopetalum mossambicense* | Plant | EN |
| | | | | A1a(v); B1(v) | *Vepris allenii* | Plant | EN |
| | | | | A1b(iv); B1(iv) | *Clerodendrum lutambense* | Plant | VU |
| | | | | A1b(iv); B1(iv) | *Coffea schliebenii* | Plant | VU |
| | | | | A1a(iv); A1e(iv); B1(iv); B2(iv) | *Oxyanthus strigosus* | Plant | EN |
| | | | | B1(iv) | *Ormocarpum schliebenii* | Plant | NT |
| | | | | A1a(iv); B1(iv); B2(iv) | *Oxyanthus biflorus* | Plant | EN |
| | | | | A1b(v); A1e(v); B1(v) | *Ochna dolicharthros* | Plant | VU |
| | | | | B1(v) | *Acacia quiterajoensis* | Plant | LC |

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|---|---|---|---|---|---|---|---|
| | | | | B1(iv); B2(iv) | *Mimosa busseana* | Plant | LC |
| | | | | A1a(iv); B1(iv); B2(iv) | *Didymosalpinx callianthus* | Plant | EN |
| 4 | Vamizi | Cabo Delgado | 87 | D1b(i) | *Caranx Ignobilis* | Marine Fish | LC |
| 5 | Quiterajo | Cabo Delgado | 3064 | A1a(iv); A1e(iv); B1(iv) | *Warneckea cordiformis* | Plant | CR |
| | | | | A1a(iv); B1(iv) | *Grewia limae* | Plant | EN |
| | | | | A1a(iv); A1e(iv); B1(iv); B2(iv) | *Oxyanthus strigosus* | Plant | EN |
| | | | | A1a(iv); B1(iv) | *Pseudomussaenda mozambicensis* | Plant | EN |
| | | | | A1a(iv); B1(iv); B2(iv) | *Pavetta lindina* | Plant | EN |
| | | | | A1a(v); B1(v) | *Vepris allenii* | Plant | EN |
| | | | | A1a(iv)B1(iv) | *Tarenna pembensis* | Plant | EN |
| | | | | A1b(iv); B2(iv) | *Micklethwaitia carvalhoi* | Plant | VU |
| | | | | A1b(iv); B1(iv) | *Stylochaeton tortispathus* | Plant | VU |
| | | | | A1b(iv); B1(iv); B2(iv) | *Acacia latispina* | Plant | VU |
| | | | | B1(v) | *Acacia quiterajoensis* | Plant | LC |
| | | | | B1(v) | *Justicia gorongozana* | Plant | LC |
| | | | | B1(iv) | *Maerua andradae* | Plant | LC |
| | | | | B1(iv); B2(iv) | *Mimosa busseana* | Plant | LC |
| | | | | B1(iv) | *Ormocarpum schliebenii* | Plant | NT |
| 6 | Taratibu | Cabo Delgado | 25 | A1a(iv); A1e(iv); B1(iv) | *Nothophryne unilurio* | Amphibian | CR |
| | | | | A1a(v); B1(v) | *Rytigynia torrei* | Plant | EN |
| 7 | Eráti | Nampula | 148 | A1a(v); B1(v) | *Allophylus torrei* | Plant | EN |
| 8 | Matibane Forest Reserve | Nampula | 109 | A1a(v); B1(v) | *Icuria dunensis* | Plant | EN |
| 9 | Ribáuè-Mphalwe | Nampula | 265 | B1(iv); B2(iv) | *Dipsadoboa montisilva* | Reptile | NT |
| | | | | A1a(iv); B1(iv); B2(iv) | *Rhampholeon tilburyi* | Reptile | EN |
| | | | | B1(iv); B2(iv) | *Nadzikambia ribaue* | Reptile | |
| | | | | B1(iv); B2(iv) | *Lygodactylus ribaue* | Reptile | |
| | | | | A1a(iv); A1e(iv); B1(iv) | *Nothophryne ribauensis* | Amphibian | EN |
| | | | | A1a(i); A1e(i); B1(i) | *Aloe rulkensii* | Plant | CR |
| | | | | A1a(v); B1(v) | *Aloe ribauensis* | Plant | EN |

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|---|---|---|---|---|---|---|---|
| | | | | A1a(i); B1(i) | *Streptocarpus myoporoides* | Plant | EN |
| | | | | A1a(v); A1e(v); B1(v) | *Dombeya leachii* | Plant | EN |
| | | | | A1a(v); A1e(v); B1(v) | *Polysphaeria ribauensis* | Plant | EN |
| | | | | A1a(v); B1(v) | *Memecylon nubigenum* | Plant | EN |
| | | | | A1b(v); B1(v) | *Cynanchum oresbium* | Plant | VU |
| | | | | A1b(v); B1(v) | *Cissus aristolochiifolia* | Plant | VU |
| | | | | A1a(v); B1(v) | *Vepris macedoi* | Plant | EN |
| | | | | A1b(v); B1(v) | *Plectranthus cucullatus* | Plant | VU |
| 10 | Mount Inago | Nampula | 326 | A1a(iv); B1(iv); B2(iv) | *Rhampholeon bruessoworum* | Reptile | CR |
| | | | | B1(iv); B2(iv) | *Nadzikambia inago* | Reptile | |
| | | | | B1(iv); B2(iv) | *Lygodactylus inago* | Reptile | |
| | | | | A1a(iv); B1(iv) | *Nothophryne inagoensis* | Amphibian | EN |
| | | | | B1(v) | *Cymothoe baylissi* | Insect | |
| | | | | B1(v) | *Alaena lamborni* | Insect | |
| | | | | B1(v) | *Neocoenyra bioculata* | Insect | |
| 11 | APAIPS | Zambezia | 2512 | A1a(v); B1(v) | *Icuria dunensis* | Plant | EN |
| 12 | Mount Namuli | Zambezia | 53 | A1a(iv) | *Paraxerus vincenti* | Mammal | EN |
| | | | | B1(i) | *Apalis lynesi* | Bird | NT |
| | | | | A1b(i) | *Arcanator orostruthus* | Bird | VU |
| | | | | A1b(i); B1(i); B2(i) | *Chamaetylas choloensis* | Bird | VU |
| | | | | A1a(iv); B1(iv); B2(iv) | *Atheris mabuensis* | Reptile | EN |
| | | | | B1(iv); B2(iv) | *Nadzikambia namuli* | Reptile | |
| | | | | B1(iv); B2(iv) | *Rhampholeon namuli* | Reptile | |
| | | | | A1a(iv); B1(iv); B2(iv) | *Rhampholeon tilburyi* | Reptile | EN |
| | | | | A1b(iv) | *Arthroleptis francei* | Amphibian | VU |
| | | | | A1b(iv) | *Hyperolius spinigularis* | Amphibian | VU |
| | | | | A1a(iv) | *Nothophryne baylissi* | Amphibian | EN |
| | | | | B1(v) | *Baliochila woodi* | Insect | |
| | | | | B1(v) | *Cymothoe baylissi* | Insect | |
| | | | | B1(v) | *Iolaus (Epamera) malaikae* | Insect | |

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|---|---|---|---|---|---|---|---|
| | | | | B1(v) | *Alaena lamborni* | Insect | |
| | | | | B1(v) | *Neocoenyra bioculata* | Insect | |
| | | | | A1a(v); A1e(v); B1(v) | *Agelanthus patelii* | Plant | EN |
| | | | | A1b(iv); B1(iv) | *Alloeochaete namuliensis* | Plant | VU |
| | | | | A1b(v); B1(v) | *Brachystelma nutans* | Plant | VU |
| | | | | B1(v) | *Coleus namuliensis* | Plant | LC |
| | | | | A1a(v); B1(v) | *Faurea racemosa* | Plant | EN |
| | | | | A1a(v); B1(v) | *Helixanthera schizocalyx* | Plant | EN |
| | | | | A1a(v); A1e(v); B1(v) | *Isoglossa namuliensis* | Plant | CR |
| | | | | A1a(v); B1(v) | *Memecylon nubigenum* | Plant | EN |
| | | | | A1b(v); B1(v) | *Pavetta gurueensis* | Plant | VU |
| | | | | A1b(v); B1(v) | *Plectranthus mandalensis* | Plant | VU |
| | | | | A1b(v); B1(v) | *Sclerochiton hirsutus* | Plant | VU |
| | | | | B1(v) | *Xerophyta kirkii* | Plant | LC |
| | | | | B1(v) | *Xerophyta pseudopinifolia* | Plant | LC |
| | | | | B1(v) | *Crotalaria namuliensis* | Plant | LC |
| 13 | Mount Mabu | Zambezia | 61 | A1b(i) | *Arcanator orostruthus* | Bird | VU |
| | | | | A1b(i); B1(i); B2(i) | *Chamaetylas choloensis* | Bird | VU |
| | | | | A1b(i); B2(i) | *Swynnertonia swynnertoni* | Bird | VU |
| | | | | A1a(iv); B1(iv); B2(iv) | *Atheris mabuensis* | Reptile | EN |
| | | | | B1(iv); B2(iv) | *Dipsadoboa montisilva* | Reptile | NT |
| | | | | B1(iv); B2(iv) | *Lygodactylus mabu* | Reptile | |
| | | | | B1(iv); B2(iv) | *Nadzikambia baylissi* | Reptile | NT |
| | | | | B1(iv); B2(iv) | *Rhampholeon maspictus* | Reptile | NT |
| | | | | A1b(iv) | *Arthroleptis francei* | Amphibian | VU |
| | | | | B1(v) | *Cymothoe baylissi* | Insect | |
| | | | | B1(v) | *Baliochila woodi* | Insect | |
| | | | | B1(v) | *Iolaus (Epamera) malaikae* | Insect | |
| | | | | B1(v) | *Neocoenyra bioculata* | Insect | |
| | | | | A1a(v); B1(v) | *Faurea racemosa* | Plant | EN |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | | | | A1a(v); B1(v) | *Helixanthera schizocalyx* | Plant | EN |
| | | | | A1b(v); B1(v) | *Pavetta gurueensis* | Plant | VU |
| | | | | A1b(v); B1(v) | *Sclerochiton hirsutus* | Plant | VU |
| 14 | Mount Chiperone | Zambezia | 36 | A1b(i); B1(i); B2(i) | *Chamaetylas choloensis* | Bird | VU |
| | | | | A1b(iv); B1(iv); B2(iv) | *Rhampholeon nebulauctor* | Reptile | VU |
| | | | | B1(iv); B2(iv) | *Nadzikambia chiperone* | Reptile | |
| | | | | B1(iv); B2(iv) | *Lygodactylus chiperone* | Reptile | |
| 15 | Derre Forest Reserve | Zambezia | 3984 | A1b(iv); B1(iv) | *Nothobranchius krysanovi* | Freshwater Fish | VU |
| | | | | A1b(v); B1(v) | *Tarenna longipedicellata* | Plant | VU |
| 16 | Tchuma-Tchato Cahora Bassa Lake | Tete | 38180 | A1b(i); A1d(i) | *Hippopotamus amphibius* | Mammal | VU |
| | | | | A1a(iv); A1c(iv) | *Oreochromis mortimeri* | Freshwater Fish | CR |
| 17 | Serra Choa | Manica | 516 | A1b(i) | *Hirundo atrocaerulea* | Bird | VU |
| 18 | Machipanda | Manica | 756 | A1b(iv); B1(iv) | *Carpitalpa arendsi* | Mammal | VU |
| | | | | A1b(i) | *Hirundo atrocaerulea* | Bird | VU |
| 19 | Chimanimani National Park | Manica | 2371 | A1b(iv); B1(iv) | *Carpitalpa arendsi* | Mammal | VU |
| | | | | A1b(iv); B1(iv) | *Rhampholeon marshalli* | Reptile | VU |
| | | | | B1(iv) | *Smaug mossambicus* | Reptile | LC |
| | | | | B1(iv) | *Hyperolius swynnertoni* | Amphibian | LC |
| | | | | A1b(iv) | *Strongylopus rhodesianus* | Amphibian | VU |
| | | | | B1(iv) | *Pseudonympha cyclops* | Insect | |
| | | | | B1(v) | *Graphium junodi* | Insect | |
| | | | | A1a(v); A1e(v); B1(v) | *Pterocephalus centennii* | Plant | CR |
| | | | | A1a(v); B1(v) | *Danthoniopsis chimanimaniensis* | Plant | EN |
| | | | | A1a(v); B1(v) | *Morella chimanimaniana* | Plant | EN |
| | | | | A1a(v); B1(v) | *Neobolusia ciliata* | Plant | EN |
| | | | | A1a(v); B1(v) | *Olinia chimanimani* | Plant | EN |
| | | | | A1a(v); B1(v) | *Empogona jenniferae* | Plant | EN |
| | | | | A1a(v); B1(v) | *Ficus muelleriana* | Plant | EN |
| | | | | A1a(v); B1(v) | *Rhynchosia chimanimaniensis* | Plant | EN |

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|---|---|---|---|---|---|---|---|
| | | | | A1a(v); B1(v) | *Aspidoglossum glabellum* | Plant | EN |
| | | | | A1a(v); B1(v) | *Dierama inyangense* | Plant | EN |
| | | | | A1a(v); B1(v) | *Synsepalum sp. nov.* | Plant | EN |
| | | | | A1b(iv); B1(iv) | *Aeschynomene aphylla* | Plant | VU |
| | | | | A1b(v); B1(v) | *Aloe plowesii* | Plant | VU |
| | | | | A1b(v); B1(v) | *Buchnera subglabra* | Plant | VU |
| | | | | A1b(v); B1(v) | *Centella obtriangularis* | Plant | VU |
| | | | | A1b(v); B1(v) | *Crotalaria insignis* | Plant | VU |
| | | | | A1b(v); B1(v) | *Dissotis pulchra* | Plant | VU |
| | | | | A1b(v); B1(v) | *Dissotis swynnertonii* | Plant | VU |
| | | | | A1b(v); B1(v) | *Erica lanceolifera* | Plant | VU |
| | | | | A1b(v); B1(v) | *Gutenbergia westii* | Plant | VU |
| | | | | A1b(v); B1(v) | *Otiophora lanceolata* | Plant | VU |
| | | | | A1b(v); B1(v) | *Protea enervis* | Plant | VU |
| | | | | A1b(v); B1(v) | *Schistostephium oxylobum* | Plant | VU |
| | | | | A1b(v); B1(v) | *Schizochilus lepidus* | Plant | VU |
| | | | | A1b(v); B1(v) | *Sericanthe chimanimaniensis* | Plant | VU |
| | | | | A1b(v); B1(v) | *Streptocarpus hirticapsa* | Plant | VU |
| | | | | A1b(v); B1(v) | *Syncolostemon oritrephes* | Plant | VU |
| | | | | A1b(v); B1(v) | *Vepris drummondii* | Plant | VU |
| | | | | A1b(v); B1(v) | *Xyris asterotricha* | Plant | VU |
| | | | | B1(v) | *Streptocarpus umtaliensis* | Plant | LC |
| | | | | A1b(v); B1(v) | *Polygala zambesiaca* | Plant | VU |
| | | | | A1b(v); B1(v) | *Impatiens psychadelphoides* | Plant | VU |
| | | | | A1b(v); B1(v) | *Gladiolus zimbabweensis* | Plant | VU |
| | | | | A1b(v); B1(v) | *Disa zimbabweensis* | Plant | VU |
| 20 | Gorongosa and Marromeu Complex | Sofala | 23088 | A1b(i); A1d(i) | *Hippopotamus amphibius* | Mammal | VU |
| | | | | A1a(i); A1c(i) | *Gyps africanus* | Bird | CR |
| | | | | A1a(i) | *Lycaon pictus* | Mammal | EN |
| | | | | A1a(i); A1c(i) | *Loxodonta africana* | Mammal | EN |

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|---|---|---|---|---|---|---|---|
| | | | | A1b(i); A1d(i) | *Panthera leo* | Mammal | VU |
| | | | | A1a(i); A1c(i) | *Trigonoceps occipitalis* | Bird | CR |
| | | | | A1a(i); A1c(i) | *Balearica regulorum* | Bird | EN |
| | | | | A1b(i); A1d(i) | *Bugeranus carunculatus* | Bird | VU |
| | | | | B1(iv) | *Afroedura gorongosa* | Reptile | LC |
| | | | | A1a(iv) | *Cycloderma frenatum* | Reptile | EN |
| | | | | A1a(iv); A1e(iv); B1(iv) | *Rhampholeon gorongosae* | Reptile | EN |
| | | | | B1(iv) | *Nothobranchius kadleci* | Freshwater Fish | NT |
| | | | | B1(v) | *Acraea dondoensis* | Insect | |
| | | | | B1(v) | *Graphium junodi* | Insect | |
| | | | | B1(iv) | *Acacia torrei* | Plant | LC |
| | | | | A1b(v); B1(v) | *Impatiens wuerstenii* | Plant | VU |
| | | | | B1(v) | *Justicia gorongozana* | Plant | LC |
| | | | | A1a(v); A1e(v); B1(v) | *Streptocarpus brachynema* | Plant | EN |
| | | | | A1b(v); B1(v) | *Cordia stuhlmannii* | Plant | VU |
| | | | | A1b(iv); B1(iv) | *Dorstenia zambesiaca* | Plant | VU |
| | | | | A1b(v); B1(v) | *Tarenna longipedicellata* | Plant | VU |
| 21 | Inhassoro-Vilankulos | Inhambane | 5357 | A1b(v); B1(v) | *Croton Inhambanensis* | Plant | VU |
| | | | | A1a(v); B1(v) | *Ecbolium hastatum* | Plant | EN |
| | | | | A1a(v); B1(v) | *Croton aceroides* | Plant | EN |
| | | | | A1b(v); B1(v) | *Ozoroa gomesiana* | Plant | VU |
| | | | | A1b(v); B1(v) | *Xylia mendoncae* | Plant | VU |
| 22 | Great Bazaruto | Inhambane | 5236 | A1d(i) | *Dugong dugon* | Mammal | VU |
| | | | | A1a(iv); A1c(iv) | *Sousa plumbea* | Mammal | EN |
| | | | | B1(iv) | *Mochlus lanceolatus* | Reptile | LC |
| | | | | B1(iv) | *Scelotes insularis* | Reptile | LC |
| | | | | B1(iv) | *Scelotes duttoni* | Reptile | LC |
| | | | | A1b(v); B1(v) | *Jatropha subaequiloba* | Plant | VU |
| 23 | Tofo | Inhambane | 342 | D1b(i) | *Mobula alfredi* | Fish | VU |
| | | | | D1b(i) | *Mobula birostris* | Fish | EN |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | | | | D1b(i) | *Rhincodon typus* | Fish | EN |
| | | | | D1b(i) | *Stegostoma tigrinum* | Fish | EN |
| 24 | Chongoene | Gaza | 33 | A1b(v); B1(v) | *Iolaus lulua* | Insect | VU |
| 25 | Manhiça-Bilene | Gaza | 2070 | A1a(iv) | *Chetia brevis* | Freshwater Fish | EN |
| | | | | A1a(i); A1e(i); B1(i) | *Memecylon incisilobum* | Plant | CR |
| | | | | A1b(i); B1(i) | *Raphia australis* | Plant | VU |
| 26 | Matutuine | Maputo | 195 | A1b(v); B1(v) | *Sclerochiton apiculatus* | Plant | VU |
| | | | | A1b(v); B1(v) | *Polygala francisci* | Plant | VU |
| 27 | Licuáti Forest Reserve | Maputo | 141 | A1b(v); B1(v) | *Sclerochiton apiculatus* | Plant | VU |
| | | | | A1b(v); B1(v) | *Polygala francisci* | Plant | VU |
| | | | | A1a(v); B1(v) | *Empogona maputensis* | Plant | EN |
| | | | | B1(v) | *Psydrax fragrantissima* | Plant | NT |
| | | | | A1a(v); B1(v) | *Warneckea parvifolia* | Plant | EN |
| | | | | A1a(v); B1(v) | *Xylopia torrei* | Plant | EN |
| 28 | Maputo National Park | Maputo | 1040 | A1b(v); B1(v) | *Sclerochiton apiculatus* | Plant | VU |
| | | | | B1(iv) | *Scarabaeus bornemizzai* | Insect | NT |
| | | | | A1a(iv) | *Silhouettea sibayi* | Freshwater Fish | EN |
| | | | | A1a(i); A1c(i) | *Loxodonta africana* | Mammal | EN |
| | | | | B1(iv) | *Amblyphymus adspersus* | Insect | NT |
| 29 | Ponta do Ouro Partial Marine Reserve | Maputo | 698 | D1b(i) | *Caranx Ignobilis* | Marine Fish | LC |
| | | | | A1b(iv) | *Parablennius lodosus* | Fish | VU |
| | | | | A1a(iv); A1c(iv) | *Sousa plumbea* | Mammal | EN |

# Chapter 5 - Supplementary material

***Table S5-1: List of conservation features used in the Mozambique conservation planning system, their targets and their total area/population size within the planning region. Columns Scen All and Scen over Harvested, represents whether the species would contribute towards the targets if found in the Low Management Intensity Zone.***

| Id | Name | Target | Scen All | Scen over Harvested |
|---|---|---|---|---|
| 1 | Amaramba Moist Miombo | 2822.462 | 1 | 1 |
| 2 | Amatonga Lowland Semideciduous Forest | 110.2995 | 1 | 1 |
| 3 | Angonia Escarpment Miombo | 1446.761 | 1 | 1 |
| 4 | Angonia Gneiss Montane Miombo | 8.5815 | 1 | 1 |
| 5 | Angonia Montane Moist Miombo | 39.98 | 1 | 1 |
| 6 | Bangomatete Rhyolite Dry Woodland | 118.58 | 1 | 1 |
| 7 | Banhine Inland Salt Pans | 827.491 | 1 | 1 |
| 8 | Barue Escarpment Miombo | 1066.91 | 1 | 1 |
| 9 | Barue Plateau Moist Miombo | 2136.234 | 1 | 1 |
| 10 | Bilene Coastal Forest | 617.02 | 1 | 1 |
| 11 | Buzi-Pungwe Alluvial Vegetation | 666.6772 | 1 | 1 |
| 12 | Canxixe Lowland Dry Woodland | 1845.548 | 1 | 1 |
| 13 | Central Lowland Moist Forest | 246.4216 | 1 | 1 |
| 14 | Central Mid-elevation Moist Forest | 64.48742 | 1 | 1 |
| 15 | Central Montane Forest | 8.562252 | 1 | 1 |
| 16 | Central Submontane Forest | 29.45908 | 1 | 1 |
| 17 | Cheringoma Coastal Moist Miombo | 1260.326 | 1 | 1 |
| 18 | Cheringoma Coastal Palm Savanna | 336.3022 | 1 | 1 |
| 19 | Cheringoma Escarpment Moist Miombo | 336.019 | 1 | 1 |
| 20 | Cheringoma Limestone Gorge Forest | 36.62858 | 1 | 1 |
| 21 | Cheringoma Plateau Moist Miombo | 841.5368 | 1 | 1 |
| 22 | Chimanimani Montane Grassland | 59.07006 | 1 | 1 |
| 23 | Chimanimani Montane Miombo | 239.3726 | 1 | 1 |
| 24 | Chimoio Moist Miombo | 648.0171 | 1 | 1 |
| 25 | Chitonga Montane Wooded Grassland | 27.95778 | 1 | 1 |
| 26 | Choa Escarpment Moist Miombo | 629.8674 | 1 | 1 |
| 27 | Coastal Berlinia Miombo | 490.2558 | 1 | 1 |
| 28 | Coastal Lagoons | 26.1897 | 1 | 1 |
| 29 | Dombe Basalt Dry Woodland | 753.7172 | 1 | 1 |
| 30 | Dombe Escarpment Miombo | 1187.631 | 1 | 1 |

| | | | | |
|---|---|---|---|---|
| 31 | Freshwater Lake | 1495.355 | 1 | 1 |
| 32 | Gaza Sandy Guibourtia Woodland | 2961.24 | 1 | 1 |
| 33 | Gorongosa Escarpment Moist Miombo | 89.22178 | 1 | 1 |
| 34 | Gorongosa Foothills Moist Miombo | 802.3614 | 1 | 1 |
| 35 | Gorongosa Montane Grassland | 8.002442 | 1 | 1 |
| 36 | Guro Dry Woodland | 2108.008 | 1 | 1 |
| 37 | Gurue Plateau Moist Miombo | 1279.088 | 1 | 1 |
| 38 | Icuria Coastal Forest | 42.8424 | 1 | 1 |
| 39 | Inhambane Coastal Miombo | 2463.734 | 1 | 1 |
| 40 | Inhambane Dune Thicket | 100.1765 | 1 | 1 |
| 41 | Inhamitanga Sand Forest | 401.9132 | 1 | 1 |
| 42 | Inharrime Coastal Palm Savanna | 1053.158 | 1 | 1 |
| 43 | Ironwood Dry Forest | 834.6702 | 1 | 1 |
| 44 | Lake Niassa Lowland Miombo | 345.3034 | 1 | 1 |
| 45 | Lebombo-KwaZulu Natal Scarp Forest | 7.260916 | 1 | 1 |
| 46 | Lebombo Summit Sourveld | 12.63314 | 1 | 1 |
| 47 | Lichinga Escarpment Moist Miombo | 3359.86 | 1 | 1 |
| 48 | Lichinga Montane Moist Miombo | 2213.134 | 1 | 1 |
| 49 | Lichinga Wooded Grassland | 36.40024 | 1 | 1 |
| 50 | Licuati Sand Thicket | 114.9423 | 1 | 1 |
| 51 | Limpopo-Olifants Riverine Forest | 20.63918 | 1 | 1 |
| 52 | Limpopo Lowland Woodland | 2467.724 | 1 | 1 |
| 53 | Limpopo Ridge Mopane Woodland | 1179.29 | 1 | 1 |
| 54 | Lugenda Lowland Dry Miombo | 4725.806 | 1 | 1 |
| 55 | Luia Basalt Mopane Woodland | 723.5008 | 1 | 1 |
| 56 | Lupata Plateau Dry Woodland | 359.9876 | 1 | 1 |
| 57 | Lupilichi Escarpment Miombo | 183.0961 | 1 | 1 |
| 58 | Lurio Riverine Forest | 0.580671 | 1 | 1 |
| 59 | Lurio Valley Dry Miombo | 5285.312 | 1 | 1 |
| 60 | Mabu Moist Miombo | 4639.516 | 1 | 1 |
| 61 | Macanga Montane Moist Miombo | 1322.321 | 1 | 1 |
| 62 | Macomia Lowland Deciduous Forest | 854.3122 | 1 | 1 |
| 63 | Madanda Rubber Sand Thicket | 944.5338 | 1 | 1 |
| 64 | Madanda Sand Forest | 663.0754 | 1 | 1 |
| 65 | Madanda Sandstone Dry Woodland | 756.4264 | 1 | 1 |
| 66 | Sandstone Mopane Woodland | 1707.119 | 1 | 1 |
| 67 | Makonde Bamboo Thicket | 1097.98 | 1 | 1 |
| 68 | Malema Granite Escarpment Miombo | 1541.221 | 1 | 1 |
| 69 | Manda Moist Miombo | 1412.596 | 1 | 1 |
| 70 | Manica Montane Grassland | 91.8238 | 1 | 1 |

| | | | | |
|---|---|---|---|---|
| 71 | Maputaland Coastal Forest | 49.8582 | 1 | 1 |
| 72 | Maputaland Coastal Wooded Grassland | 390.095 | 1 | 1 |
| 73 | Maputaland Dune Forest | 32.63476 | 1 | 1 |
| 74 | Maputaland Sand Forest | 82.68608 | 1 | 1 |
| 75 | Maputo Alluvial Vegetation | 1412.285 | 1 | 1 |
| 76 | Maputo Riverine Forest | 1.63543 | 1 | 1 |
| 77 | Maravia Plateau Miombo | 2700.566 | 1 | 1 |
| 78 | Maringue Sandstone Dry Woodland | 540.6684 | 1 | 1 |
| 79 | Marrupa Plateau Moist Miombo | 1433.856 | 1 | 1 |
| 80 | Matondonvela Moist Miombo | 4889.258 | 1 | 1 |
| 81 | Mazoe Gneiss Dry Thicket | 350.0296 | 1 | 1 |
| 82 | Mecufi Sandstone Dry Woodland | 31.2642 | 1 | 1 |
| 83 | Mecula Summit Grassland | 4.663622 | 1 | 1 |
| 84 | Memba Dry Deciduous Lowland Forest | 484.139 | 1 | 1 |
| 85 | Memba Dry Miombo | 2088.506 | 1 | 1 |
| 86 | Mocuba Moist Miombo | 6520.308 | 1 | 1 |
| 87 | Monapo Klippe Dry Woodland | 83.4048 | 1 | 1 |
| 88 | Montepuez Plateau Moist Miombo | 3343.762 | 1 | 1 |
| 89 | Morrumbala Lowland Moist Miombo | 1544.324 | 1 | 1 |
| 90 | Morrumbala Plateau Moist Miombo | 630.9024 | 1 | 1 |
| 91 | Mossurize Escarpment Miombo | 494.195 | 1 | 1 |
| 92 | Mueda Dry Sand Thicket | 142.1919 | 1 | 1 |
| 93 | Mueda Escarpment Miombo | 433.4432 | 1 | 1 |
| 94 | Mueda Midslope Deciduous Forest | 247.4208 | 1 | 1 |
| 95 | Mueda Mixed Dry Miombo | 1100.571 | 1 | 1 |
| 96 | Mueda Plateau Moist Forest | 32.0841 | 1 | 1 |
| 97 | Nametil Moist Miombo | 3444.12 | 1 | 1 |
| 98 | Nampula Coastal Palm Savanna | 810.9996 | 1 | 1 |
| 99 | Nampula Granite Escarpment Miombo | 3478.252 | 1 | 1 |
| 100 | Nampula Ironwood Forest | 4.824314 | 1 | 1 |
| 101 | Namuli Montane Grassland | 6.748694 | 1 | 1 |
| 102 | Nangade Deciduous Newtonia Forest | 102.0867 | 1 | 1 |
| 103 | Northern Coastal Dry Woodland | 1234.802 | 1 | 1 |
| 104 | Northern Inselberg Forest | 0.491036 | 1 | 1 |
| 105 | Northern Inselberg Woodland | 1232.572 | 1 | 1 |
| 106 | Northern Lebombo Bushveld | 336.7342 | 1 | 1 |
| 107 | Northern Lowland Moist Forest | 18.87809 | 1 | 1 |
| 108 | Northern Mid-elevation Moist Forest | 33.44604 | 1 | 1 |
| 109 | Northern Montane Forest | 4.038298 | 1 | 1 |
| 110 | Northern Submontane Forest | 14.08833 | 1 | 1 |

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|---|---|---|---|---|
| 111 | Nungo Moist Miombo | 3209.18 | 1 | 1 |
| 112 | Nwambiya-Pumbe Sandy Bushveld | 1475.965 | 1 | 1 |
| 113 | Nwambiya Sand Thicket | 442.5706 | 1 | 1 |
| 114 | Pande Sand Thicket | 146.506 | 1 | 1 |
| 115 | Pangue Dry Miombo | 4969.654 | 1 | 1 |
| 116 | Pebane Sandy Shrub Miombo | 1516.379 | 1 | 1 |
| 117 | Ribaue Granite Escarpment Miombo | 1865.775 | 1 | 1 |
| 118 | Rift Valley Floodplain Wooded Grassland | 777.4312 | 1 | 1 |
| 119 | Rift Valley Lowland Woodland | 1222.766 | 1 | 1 |
| 120 | Rovuma Alluvial Vegetation | 263.5438 | 1 | 1 |
| 121 | Rovuma Basin Coastal Forest | 341.8082 | 1 | 1 |
| 122 | Rovuma Coastal Moist Miombo | 324.057 | 1 | 1 |
| 123 | Rovuma Coastal Wooded Grassland | 195.9668 | 1 | 1 |
| 124 | Rovuma Coral Rag Thicket | 35.1263 | 1 | 1 |
| 125 | Rovuma Dune Thicket | 36.15432 | 1 | 1 |
| 126 | Rovuma Riverine Forest | 3.198174 | 1 | 1 |
| 127 | Save Alluvial Vegetation | 140.0203 | 1 | 1 |
| 128 | Save Coastal Miombo | 97.76642 | 1 | 1 |
| 129 | Save Coastal Palm Savanna | 531.8548 | 1 | 1 |
| 130 | Save Lowland Dry Woodland | 1940.214 | 1 | 1 |
| 131 | Save Riverine Forest | 17.9128 | 1 | 1 |
| 132 | Save Sand Forest | 622.9204 | 1 | 1 |
| 133 | Save Valley Chalk Thicket | 123.6512 | 1 | 1 |
| 134 | Songo Granite Dry Woodland | 657.1374 | 1 | 1 |
| 135 | Southern Inselberg Woodland | 74.59858 | 1 | 1 |
| 136 | Southern Lebombo Bushveld | 252.233 | 1 | 1 |
| 137 | Southern Mopane Woodland | 4891.254 | 1 | 1 |
| 138 | Stormberg Dry Woodland | 590.8086 | 1 | 1 |
| 139 | Subtropical Coastal Salt Marshes | 56.0096 | 1 | 1 |
| 140 | Subtropical Seashore Vegetation | 17.63559 | 1 | 1 |
| 141 | Subtropical Swamp Forest | 2.694752 | 1 | 1 |
| 142 | Tembe Sandy Bushveld | 1001.3 | 1 | 1 |
| 143 | Tete Gabbro Dry Woodland | 1142.87 | 1 | 1 |
| 144 | Tete Mixed Dry Miombo | 3694.982 | 1 | 1 |
| 145 | Tropical Coastal Salt Marshes | 577.5632 | 1 | 1 |
| 146 | Tropical Indian Ocean African Mangrove | 695.7524 | 1 | 1 |
| 147 | Tropical Seashore Vegetation | 52.0888 | 1 | 1 |
| 148 | Tropical Swamp Forest | 0.120926 | 1 | 1 |
| 149 | Ulongue Plateau Grassland | 9.0648 | 1 | 1 |
| 150 | Urronga Lowland Dry Woodland | 2420.428 | 1 | 1 |

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|---|---|---|---|---|
| 151 | Vilanculos Coastal Miombo | 706.1138 | 1 | 1 |
| 152 | Western Maputaland Clay Bushveld | 764.6792 | 1 | 1 |
| 153 | Zambezi Alluvial Vegetation | 434.3538 | 1 | 1 |
| 154 | Zambezi Chenier Dune Thicket | 344.9852 | 1 | 1 |
| 155 | Zambezi Delta Floodplain Grassland | 2113.8 | 1 | 1 |
| 156 | Zambezi Delta Lowland Forest | 1645.791 | 1 | 1 |
| 157 | Zambezi Riverine Forest | 71.75014 | 1 | 1 |
| 158 | Zambezi Valley Mopane Woodland | 3011.182 | 1 | 1 |
| 159 | Zambezi Valley Sand Forest | 84.9954 | 1 | 1 |
| 160 | Zambezian Papyrus Wetland | 349.6342 | 1 | 1 |
| 161 | Zambezian Sand Thicket | 241.1444 | 1 | 1 |
| 162 | Zumbo Montane Miombo | 762.8146 | 1 | 1 |
| 211 | Cultivos arboreos | 0 | 1 | 1 |
| 212 | Cultivos nao arboreo | 0 | 1 | 1 |
| 221 | Plantacao florestal | 0 | 1 | 1 |
| 251 | Areas habitacionais | 0 | 1 | 1 |
| 1001 | Arthroleptis francei | 39.79 | 1 | 1 |
| 1002 | Hyperolius spinigularis | 20.645 | 1 | 1 |
| 1003 | Hyperolius stictus | 44 | 1 | 1 |
| 1004 | Hyperolius swynnertoni | 212 | 1 | 1 |
| 1005 | Mertensophryne anotis | 726 | 1 | 1 |
| 1006 | Nothophryne baylissi | 370.786 | 1 | 1 |
| 1007 | Nothophryne inagoensis | 92.786 | 1 | 1 |
| 1008 | Nothophryne ribauensis | 0.18 | 1 | 1 |
| 1009 | Nothophryne unilurio | 7.18 | 1 | 1 |
| 1010 | Poyntonophrynus beiranus | 826 | 1 | 1 |
| 1011 | Strongylopus rhodesianus | 392.515 | 1 | 1 |
| 2001 | Anthus brachyurus | 602.79 | 1 | 1 |
| 2002 | Apalis lynesi | 516.669 | 1 | 1 |
| 2003 | Aquila rapax | 17221.25 | 1 | 0 |
| 2004 | Arcanator orostruthus | 302.81 | 1 | 1 |
| 2005 | Artisornis sousae | 517.431 | 1 | 1 |
| 2006 | Balearica regulorum | 3575.75 | 1 | 0 |
| 2007 | Bucorvus leadbeateri | 13464.14 | 1 | 0 |
| 2008 | Bugeranus carunculatus | 1428.36 | 1 | 0 |
| 2009 | Calherodius leuconotus | 1543.62 | 1 | 1 |
| 2010 | Caprimulgus natalensis | 591.69 | 1 | 1 |
| 2011 | Chamaetylas choloensis | 106.56 | 1 | 1 |
| 2012 | Ciconia nigra | 2839.01 | 1 | 0 |
| 2013 | Circaetus fasciolatus | 4024.77 | 1 | 1 |

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|---|---|---|---|---|
| 2014 | Circus ranivorus | 1987.52 | 1 | 1 |
| 2015 | Ephippiorhynchus senegalensis | 2602.95 | 1 | 1 |
| 2016 | Falco biarmicus | 15182.54 | 1 | 0 |
| 2017 | Falco concolor | 2391.96 | 1 | 1 |
| 2018 | Glaucidium capense | 3577.15 | 1 | 1 |
| 2019 | Gyps africanus | 6851.97 | 1 | 0 |
| 2020 | Gyps coprotheres | 5159.71 | 1 | 0 |
| 2021 | Halcyon senegaloides | 998.88 | 1 | 1 |
| 2022 | Hirundo atrocaerulea | 193.89 | 1 | 1 |
| 2023 | Macheiramphus alcinus | 2481.04 | 1 | 1 |
| 2024 | Microparra capensis | 1845.82 | 1 | 1 |
| 2025 | Mycteria ibis | 1363.31 | 1 | 1 |
| 2026 | Necrosyrtes monachus | 4962 | 1 | 1 |
| 2027 | Neotis denhami | 186.9 | 1 | 0 |
| 2028 | Nettapus auritus | 1302.29 | 1 | 1 |
| 2029 | Pelecanus rufescens | 2191.77 | 1 | 1 |
| 2030 | Podica senegalensis | 2015.77 | 1 | 1 |
| 2031 | Poicephalus cryptoxanthus | 4165.17 | 1 | 0 |
| 2032 | Polemaetus bellicosus | 4996.31 | 1 | 0 |
| 2036 | Swynnertonia swynnertoni | 206.782 | 1 | 1 |
| 2037 | Terathopius ecaudatus | 2076.74 | 1 | 0 |
| 2039 | Trigonoceps occipitalis | 6557.2 | 1 | 0 |
| 2040 | Turnix nanus | 2053.89 | 1 | 1 |
| 2041 | Tyto capensis | 874.03 | 1 | 1 |
| 3001 | Chetia brevis | 220 | 1 | 1 |
| 3002 | Chiloglanis emarginatus | 160.089 | 1 | 1 |
| 3003 | Marcusenius sp nov kosi | 0.2 | 1 | 1 |
| 3004 | Nothobranchius hengstleri | 8 | 1 | 1 |
| 3005 | Nothobranchius kadleci | 2.741 | 1 | 1 |
| 3006 | Nothobranchius krammeri | 13.64 | 1 | 1 |
| 3007 | Nothobranchius krysanovi | 80.718 | 1 | 1 |
| 3008 | Nothobranchius niassa | 5.55 | 1 | 1 |
| 3009 | Oreochromis mortimeri | 64.798 | 1 | 1 |
| 3010 | Oreochromis mossambicus | 605 | 1 | 1 |
| 3011 | Serranochromis meridianus | 220 | 1 | 1 |
| 3012 | Silhouettea sibayi | 454.101 | 1 | 1 |
| 4001 | Agriocnemis ruberrima ssp. ruberrima | 61 | 1 | 1 |
| 4002 | Acraea dondoensis | 1 | 1 | 1 |
| 4003 | Alaena lamborni | 2 | 1 | 1 |
| 4004 | Amblyphymus adspersus | 529.12 | 1 | 1 |

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|---|---|---|---|---|
| 4005 | Baliochila woodi | 1 | 1 | 1 |
| 4006 | Copris inhalatus inhalatus | 396.95 | 1 | 1 |
| 4007 | Cymothoe baylissi | 2 | 1 | 1 |
| 4008 | Graphium junodi | 5 | 1 | 1 |
| 4009 | Iolaus malaikae | 1 | 1 | 1 |
| 4010 | Lepidochrysops intermedia | 1 | 1 | 1 |
| 4011 | Lepidocrysops delicata | 1 | 1 | 1 |
| 4012 | Neocoenyra bioculata | 2 | 1 | 1 |
| 4013 | Physophorina livingstonii | 1090.29 | 1 | 1 |
| 4014 | Scarabaeus bornemisszai | 259 | 1 | 1 |
| 5001 | Carpitalpa arendsi | 256.12 | 1 | 1 |
| 5002 | Ceratotherium simum | 3800.86 | 1 | 0 |
| 5003 | Diceros bicornis | 4940.71 | 1 | 0 |
| 5004 | Giraffa camelopardalis giraffa | 1549.05 | 1 | 1 |
| 5005 | Hippopotamus amphibius | 1070.45 | 1 | 1 |
| 5006 | Loxodonta africana | 24887.46 | 1 | 0 |
| 5007 | Lycaon pictus | 2024.55 | 1 | 0 |
| 5008 | Nesotragus moschatus | 3306.68 | 1 | 0 |
| 5009 | Ourebia ourebi ourebi | 7083.26 | 1 | 0 |
| 5010 | Panthera leo | 7300.29 | 1 | 0 |
| 5011 | Panthera pardus | 4289.5 | 1 | 0 |
| 5012 | Paraxerus vincenti | 92.823 | 1 | 0 |
| 5013 | Smutsia temminckii | 1988.19 | 1 | 0 |
| 6001 | Acacia latispina | 705.647 | 1 | 1 |
| 6002 | Acacia quiterajoensis | 1641.78 | 1 | 1 |
| 6003 | Acacia torrei | 887.77 | 1 | 1 |
| 6004 | Acridocarpus natalitius linearifolius | 1080.5 | 1 | 0 |
| 6005 | Adenium swazicum | 342 | 1 | 0 |
| 6006 | Adenopodia schlechteri | 279.731 | 1 | 1 |
| 6007 | Aeollanthus viscosus | 5 | 1 | 1 |
| 6008 | Aeschynomene aphylla | 250.171 | 1 | 1 |
| 6009 | Aeschynomene chimanimaniensis | 3 | 1 | 1 |
| 6010 | Aeschynomene grandistipulata | 88 | 1 | 1 |
| 6011 | Aeschynomene inyangensis | 2 | 1 | 1 |
| 6012 | Aeschynomene minutiflora grandiflora | 859.41 | 1 | 1 |
| 6013 | Aeschynomene mossambicensis mossambicensis | 474.92 | 1 | 1 |
| 6014 | Aeschynomene pawekiae | 0 | 1 | 1 |
| 6015 | Afrocanthium ngonii | 2 | 1 | 1 |
| 6016 | Afrosciadium rhodesicum | 1 | 1 | 1 |
| 6017 | Agelanthus deltae | 2 | 1 | 1 |

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|---|---|---|---|---|
| 6018 | Agelanthus patelii | 1 | 1 | 1 |
| 6019 | Alloeochaete namuliensis | 1.93 | 1 | 1 |
| 6020 | Allophylus mossambicensis | 124.685 | 1 | 1 |
| 6021 | Allophylus torrei | 412.816 | 1 | 1 |
| 6022 | Aloe argentifolia | 1 | 1 | 0 |
| 6023 | Aloe ballii makurupiniensis | 4 | 1 | 0 |
| 6024 | Aloe cannellii | 1 | 1 | 0 |
| 6025 | Aloe excelsa breviflora | 0 | 1 | 0 |
| 6026 | Aloe hazeliana hazeliana | 152.39 | 1 | 0 |
| 6027 | Aloe keithii | 348.17 | 1 | 0 |
| 6028 | Aloe menyharthii ensifolia | 147.785 | 1 | 0 |
| 6029 | Aloe munchii Christian | 88.34 | 1 | 0 |
| 6030 | Aloe plowesii Reynolds | 7 | 1 | 0 |
| 6031 | Aloe ribauensis | 1 | 1 | 0 |
| 6032 | Aloe rulkensii | 2 | 1 | 0 |
| 6033 | Aloe wildii | 1 | 1 | 0 |
| 6034 | Ammannia elata | 1 | 1 | 1 |
| 6035 | Ammannia fernandesiana | 2 | 1 | 1 |
| 6036 | Ammannia moggii | 1 | 1 | 1 |
| 6037 | Ammannia pedroi | 1 | 1 | 1 |
| 6038 | Aneilema arenicola | 2 | 1 | 1 |
| 6039 | Anisopappus paucidentatus | 88 | 1 | 1 |
| 6040 | Antherotoma angustifolia | 2 | 1 | 1 |
| 6041 | Anthospermum zimbabwense | 1 | 1 | 1 |
| 6042 | Asclepias cucullata scabrifolia | 132 | 1 | 1 |
| 6043 | Asclepias graminifolia | 2 | 1 | 1 |
| 6044 | Asparagus chimanimanensis | 6 | 1 | 1 |
| 6045 | Asparagus radiatus | 132 | 1 | 1 |
| 6046 | Aspidoglossum | 1 | 1 | 1 |
| 6047 | Australluma ubombonesis | 88.25 | 1 | 1 |
| 6048 | Baphia massaiensis | 1032.44 | 1 | 1 |
| 6049 | Baphia ovata | 270.61 | 1 | 1 |
| 6050 | Baphia punctulata palmensis | 4 | 1 | 1 |
| 6051 | Barleria laceratiflora | 1 | 1 | 1 |
| 6052 | Barleria oxyphylla | 220 | 1 | 1 |
| 6053 | Barleria setosa | 4 | 1 | 1 |
| 6054 | Barleria torrei | 1478.31 | 1 | 1 |
| 6055 | Barleria vollesenii | 1 | 1 | 1 |
| 6056 | Bauhinia burrowsii | 2020.43 | 1 | 1 |
| 6057 | Bersama swynnertonii | 88 | 1 | 1 |

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|---|---|---|---|---|
| 6058 | Blepharis dunensis | 4 | 1 | 1 |
| 6059 | Blepharis gazensis | 3 | 1 | 1 |
| 6060 | Blepharis swaziensis | 276.89 | 1 | 1 |
| 6061 | Bothriocline moramballae | 423.28 | 1 | 1 |
| 6062 | Bothriocline steetziana | 127.65 | 1 | 1 |
| 6063 | Brachychloa fragilis | 0 | 1 | 1 |
| 6064 | Brachystegia oblonga | 48.274 | 1 | 1 |
| 6065 | Buchnera chimanimaniensis | 88 | 1 | 1 |
| 6066 | Buchnera subglabra | 5 | 1 | 1 |
| 6067 | Buchnera wildii | 132 | 1 | 1 |
| 6068 | Bussea xylocarpa | 9 | 1 | 1 |
| 6069 | Caroxylon littoralis | 1 | 1 | 1 |
| 6070 | Carpolobia suaveolens | 6 | 1 | 1 |
| 6071 | Casearia celastroides | 6 | 1 | 1 |
| 6072 | Celosia pandurata | 4 | 1 | 1 |
| 6073 | Centella obtriangularis | 8 | 1 | 1 |
| 6074 | Ceropegia aloicola | 2 | 1 | 1 |
| 6075 | Ceropegia chimanimaniensis | 4 | 1 | 1 |
| 6076 | Ceropegia cyperifolia | 4 | 1 | 1 |
| 6077 | Ceropegia nutans | 2 | 1 | 1 |
| 6078 | Chamaecrista paralias | 220 | 1 | 1 |
| 6079 | Chascanum angolense zambesiacum | 3 | 1 | 1 |
| 6080 | Chascanum schlechteri torrei | 1 | 1 | 1 |
| 6081 | Chassalia colorata | 753.05 | 1 | 1 |
| 6082 | Chlorophytum pygmaeum rhodesianum | 3 | 1 | 1 |
| 6083 | Cineraria pulchra | 88 | 1 | 1 |
| 6084 | Clerodendrum robustum macrocalyx | 1 | 1 | 1 |
| 6085 | Clutia sessilifolia | 3 | 1 | 1 |
| 6086 | Coffea schliebenii | 2 | 1 | 1 |
| 6087 | Cola cheringoma | 3 | 1 | 1 |
| 6088 | Cola clavata | 2 | 1 | 1 |
| 6089 | Cola dorrii | 132 | 1 | 1 |
| 6090 | Coleus caudatus | 88 | 1 | 1 |
| 6091 | Coleus namuliensis | 2 | 1 | 1 |
| 6092 | Combretum caudatisepalum | 1 | 1 | 1 |
| 6093 | Combretum lindense | 1 | 1 | 1 |
| 6094 | Combretum stocksii | 733 | 1 | 1 |
| 6095 | Cordia megiae | 4 | 1 | 1 |
| 6096 | Cordia stuhlmannii | 1319.36 | 1 | 1 |
| 6097 | Crassula lanceolata | 88 | 1 | 1 |

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|---|---|---|---|---|
| 6098 | Crassula morrumbalensis | 1 | 1 | 1 |
| 6099 | Crepidorhopalon flavus | 544.699 | 1 | 1 |
| 6100 | Crepidorhopalon namuliense | 2 | 1 | 1 |
| 6101 | Crossopetalum mossambicense | 3 | 1 | 1 |
| 6102 | Crotalaria mocubensis | 2.603 | 1 | 1 |
| 6103 | Crotalaria namuliensis | 1 | 1 | 1 |
| 6104 | Crotalaria paraspartea | 0 | 1 | 1 |
| 6105 | Crotalaria phylicoides | 132 | 1 | 1 |
| 6106 | Crotalaria torrei | 2 | 1 | 1 |
| 6107 | Croton aceroides | 2 | 1 | 1 |
| 6108 | Croton inhambanensis | 1316.32 | 1 | 1 |
| 6109 | Croton leuconeurus mossambicensis | 1 | 1 | 1 |
| 6110 | Crotonogynopsis australis | 1 | 1 | 1 |
| 6111 | Cyanotis chimanimaniensis | 88 | 1 | 1 |
| 6112 | Cynanchum oresbium | 1 | 1 | 1 |
| 6113 | Cynorkis anisoloba | 3 | 1 | 1 |
| 6114 | Cyrtorchis glaucifolia | 0 | 1 | 1 |
| 6115 | Danthoniopsis chimanimaniensis | 724.97 | 1 | 1 |
| 6116 | Dianthus chimanimaniensis | 1 | 1 | 1 |
| 6117 | Dicerocayum forbesii | 220 | 1 | 1 |
| 6118 | Dicliptera sp. b | 2 | 1 | 1 |
| 6119 | Didymosalpinx callianthus | 2 | 1 | 1 |
| 6120 | Dierama plowesii | 2 | 1 | 1 |
| 6121 | Digitaria megasthenes | 1 | 1 | 1 |
| 6122 | Dioscorea sylvatica | 20.804 | 1 | 1 |
| 6123 | Disa chimanimaniensis | 88 | 1 | 1 |
| 6124 | Disa zimbabweensis | 1 | 1 | 1 |
| 6125 | Disperis mozambicensis | 0 | 1 | 1 |
| 6126 | Dissotis johnstoniana johnstoniana | 3 | 1 | 1 |
| 6127 | Dissotis pulchra | 514.456 | 1 | 1 |
| 6128 | Dissotis swynnertonii | 280.33 | 1 | 1 |
| 6129 | Distephanus inhacensis | 638.52 | 1 | 1 |
| 6130 | Dolichandrone alba | 703.17 | 1 | 1 |
| 6131 | Dombeya lastii | 0 | 1 | 1 |
| 6132 | Dombeya leachii | 4 | 1 | 1 |
| 6133 | Dorstenia zambesiaca | 3 | 1 | 1 |
| 6134 | Dovyalis sp. a | 1 | 1 | 1 |
| 6135 | Duosperma dichotomum | 1 | 1 | 1 |
| 6136 | Ecbolium hastatum | 2 | 1 | 1 |
| 6137 | Elaeodendron fruticosum | 625 | 1 | 1 |

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|---|---|---|---|---|
| 6138 | Empogona jenniferae | 1 | 1 | 1 |
| 6139 | Empogona maputensis | 220 | 1 | 1 |
| 6140 | Encephalartos aplantus | 1 | 1 | 0 |
| 6142 | Encephalartos chimanimaniensis | 31.678 | 1 | 0 |
| 6143 | Encephalartos ferox emersus | 434.872 | 1 | 0 |
| 6144 | Encephalartos lebomboensis | 135.52 | 1 | 0 |
| 6145 | Encephalartos munchii | 0 | 1 | 0 |
| 6146 | Encephalartos pterogonus | 1 | 1 | 0 |
| 6147 | Encephalartos turneri | 520 | 1 | 0 |
| 6148 | Encephalartos umbeluziensis | 234 | 1 | 0 |
| 6149 | Englerina oedostemon | 3 | 1 | 1 |
| 6150 | Englerina schlechteri | 220 | 1 | 1 |
| 6151 | Entada mossambicensis | 0 | 1 | 1 |
| 6152 | Eragrostis desolata | 3 | 1 | 1 |
| 6153 | Eragrostis sericata | 1 | 1 | 1 |
| 6154 | Erica lanceolifera | 1 | 1 | 1 |
| 6155 | Erica pleiotrichaild | 88 | 1 | 1 |
| 6156 | Erica pleiotricha pleiotricha | 2 | 1 | 1 |
| 6157 | Erica wildii | 88 | 1 | 1 |
| 6158 | Eriocaulon infaustum | 1 | 1 | 1 |
| 6159 | Eriolaena rulkensii | 1 | 1 | 1 |
| 6160 | Eriospermum mackenii phippsii | 3 | 1 | 1 |
| 6161 | Erythrococca zambesiaca | 2 | 1 | 1 |
| 6162 | Eugenia sp. a | 132 | 1 | 1 |
| 6163 | Euphorbia ambroseae ambroseae | 5 | 1 | 1 |
| 6164 | Euphorbia angularis | 1 | 1 | 1 |
| 6165 | Euphorbia baylissii | 5 | 1 | 1 |
| 6166 | Euphorbia bougheyi | 742 | 1 | 1 |
| 6167 | Euphorbia citrina | 1 | 1 | 1 |
| 6168 | Euphorbia contorta | 2 | 1 | 1 |
| 6169 | Euphorbia corniculata | 683.45 | 1 | 1 |
| 6170 | Euphorbia crebrifolia | 88.28 | 1 | 1 |
| 6171 | Euphorbia decliviticola | 2 | 1 | 1 |
| 6172 | Euphorbia depauperata tsetserrensis | 1 | 1 | 1 |
| 6173 | Euphorbia grandicornis sejuncta | 1 | 1 | 1 |
| 6174 | Euphorbia graniticola | 50.937 | 1 | 1 |
| 6175 | Euphorbia keithii | 1 | 1 | 1 |
| 6176 | Euphorbia knuthii johnsonii | 2 | 1 | 1 |
| 6177 | Euphorbia knuthii Knuthii | 132 | 1 | 1 |
| 6178 | Euphorbia marrupana | 3 | 1 | 1 |

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| 6179 | Euphorbia namuliensis | 1 | 1 | 1 |
| 6180 | Euphorbia neohalipedicola | 1 | 1 | 1 |
| 6181 | Euphorbia ramulosa | 3 | 1 | 1 |
| 6182 | Euphorbia schlechteri | 2 | 1 | 1 |
| 6183 | Euphorbia unicornis | 2 | 1 | 1 |
| 6184 | Faurea rubriflora | 320 | 1 | 1 |
| 6185 | Ficus muelleriana | 2 | 1 | 1 |
| 6186 | Gelrebia rostrata | 2 | 1 | 1 |
| 6187 | Geranium exellii | 1 | 1 | 1 |
| 6188 | Gladiolus brachyphyllus | 0 | 1 | 1 |
| 6189 | Gladiolus zimbabweensis | 2 | 1 | 1 |
| 6190 | Gnidia chapmanii | 3 | 1 | 1 |
| 6191 | Grewia filipes | 2 | 1 | 1 |
| 6192 | Grewia limae | 220 | 1 | 1 |
| 6193 | Grewia occidentalis littoralis | 220.82 | 1 | 1 |
| 6194 | Grewia transzambesica | 800.35 | 1 | 1 |
| 6195 | Guibourtia schliebenii | 132 | 1 | 1 |
| 6196 | Gutenbergia westii | 220.35 | 1 | 1 |
| 6197 | Gyrodoma hispida | 532 | 1 | 1 |
| 6198 | Habenaria hirsutissima | 2 | 1 | 1 |
| 6199 | Halophila ovalis linearis | 1 | 1 | 1 |
| 6200 | Heinsia mozambicensis | 1 | 1 | 1 |
| 6201 | Helichrysum africanum | 93.15 | 1 | 1 |
| 6203 | Helichrysum moorei | 7 | 1 | 1 |
| 6204 | Helichrysum rhodellum | 1 | 1 | 1 |
| 6205 | Helichrysum silvaticum | 132 | 1 | 1 |
| 6206 | Helixanthera schizocalyx | 3 | 1 | 1 |
| 6207 | Hermannia micropetala | 320 | 1 | 1 |
| 6208 | Hermannia torrei Wild | 0 | 1 | 1 |
| 6209 | Hexalobus mossambicensis | 368.58 | 1 | 1 |
| 6210 | Hibiscus torrei | 1 | 1 | 1 |
| 6211 | Huernia erectiloba | 320 | 1 | 1 |
| 6212 | Huernia leachii | 1 | 1 | 1 |
| 6213 | Huernia verekeri pauciflora | 2 | 1 | 1 |
| 6214 | Hugonia elliptica | 1 | 1 | 1 |
| 6215 | Hymenodictyon austro-africanum | 1 | 1 | 1 |
| 6216 | Icuria dunensis | 130.09 | 1 | 0 |
| 6217 | Impatiens salpinx | 153.2 | 1 | 1 |
| 6218 | Impatiens wuerstenii | 8 | 1 | 1 |
| 6219 | Indigofera gobensis | 1 | 1 | 1 |

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|---|---|---|---|---|
| 6220 | Indigofera pseudomoniliformis | 478.935 | 1 | 1 |
| 6221 | Indigofera torrei | 2 | 1 | 1 |
| 6222 | Ipomoea sp. nov. | 1 | 1 | 1 |
| 6223 | Isoglossa namuliensis | 1 | 1 | 1 |
| 6224 | Jamesbrittenia carvalhoi | 220 | 1 | 1 |
| 6225 | Jatropha latifolia subeglandulosa | 1 | 1 | 1 |
| 6226 | Jatropha scaposa | 822 | 1 | 1 |
| 6227 | Jatropha subaequiloba | 1 | 1 | 1 |
| 6228 | Justicia gorongozana | 220 | 1 | 1 |
| 6229 | Justicia niassensis | 0 | 1 | 1 |
| 6230 | Justicia sp. a | 2 | 1 | 1 |
| 6231 | Justicia sp. b | 1 | 1 | 1 |
| 6232 | Justicia subcordatifolia | 1 | 1 | 1 |
| 6233 | Kalanchoe hametiorum | 220 | 1 | 1 |
| 6234 | Kalanchoe velutina chimanimanensis | 3 | 1 | 1 |
| 6235 | Kleinia chimanimaniensis | 2 | 1 | 1 |
| 6236 | Leucas nyassae velutina | 1 | 1 | 1 |
| 6237 | Liparis hemipilioides | 0 | 1 | 1 |
| 6238 | Lobelia blantyrensis | 1 | 1 | 1 |
| 6239 | Lobelia cobaltica | 88 | 1 | 1 |
| 6240 | Lopholaena brickellioides | 4 | 1 | 1 |
| 6241 | Lotus wildii | 44 | 1 | 1 |
| 6242 | Lysimachia gracilipes | 1 | 1 | 1 |
| 6243 | Maerua andradae | 220 | 1 | 1 |
| 6244 | Memecylon aenigmaticum | 1 | 1 | 1 |
| 6245 | Memecylon incisilobum | 1 | 1 | 1 |
| 6246 | Memecylon insulare | 1 | 1 | 1 |
| 6247 | Memecylon nubigenum | 1 | 1 | 1 |
| 6248 | Memecylon rovumense | 2 | 1 | 1 |
| 6249 | Memecylon torrei | 3 | 1 | 1 |
| 6250 | Mesanthemum africanum | 132 | 1 | 1 |
| 6251 | Micklethwaitia carvalhoi | 305.24 | 1 | 1 |
| 6252 | Millettia ebenifera | 265.795 | 1 | 1 |
| 6253 | Millettia mossambicensis | 775.29 | 1 | 1 |
| 6254 | Momordica mossambica | 0 | 1 | 1 |
| 6255 | Monodora carolinae | 1 | 1 | 1 |
| 6256 | Moraea niassensis | 1 | 1 | 1 |
| 6257 | Morella chimanimaniana | 178.226 | 1 | 1 |
| 6258 | Nectaropetalum carvalhoi | 3 | 1 | 1 |
| 6259 | Neobolusia ciliata | 88 | 1 | 1 |

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| 6260 | Ochna angustata | 220 | 1 | 1 |
| 6261 | Ochna beirensis | 1 | 1 | 1 |
| 6262 | Ochna dolicharthros | 5 | 1 | 1 |
| 6263 | Ocimum natalense | 3 | 1 | 1 |
| 6264 | Ocimum reclinatum | 2 | 1 | 1 |
| 6265 | Oldenlandia cana | 2 | 1 | 1 |
| 6266 | Oldenlandia verrucitesta | 1 | 1 | 1 |
| 6267 | Olea chimanimani | 88 | 1 | 1 |
| 6268 | Olinia chimanimani | 145.61 | 1 | 1 |
| 6269 | Orbea halipedicola | 379 | 1 | 1 |
| 6270 | Orthosiphon scedastophyllus | 1 | 1 | 1 |
| 6271 | Otiophora inyangana inyangana | 132 | 1 | 1 |
| 6272 | Otiophora inyangana | 8 | 1 | 1 |
| 6273 | Otiophora lanceolata | 5 | 1 | 1 |
| 6274 | Oxyanthus biflorus | 3 | 1 | 1 |
| 6275 | Oxyanthus strigosus | 475 | 1 | 1 |
| 6276 | Ozoroa gomesiana | 569 | 1 | 1 |
| 6277 | Pachycarpus concolor arenicola | 5 | 1 | 1 |
| 6278 | Pachystigma sp. a of f.z | 1 | 1 | 1 |
| 6279 | Pavetta chapmanii | 3 | 1 | 1 |
| 6280 | Pavetta comostyla inyangensis | 4 | 1 | 1 |
| 6281 | Pavetta curalicola | 1 | 1 | 1 |
| 6282 | Pavetta dianeae | 2 | 1 | 1 |
| 6283 | Pavetta gardeniifolia appendiculata | 2 | 1 | 1 |
| 6284 | Pavetta gurueensis | 6 | 1 | 1 |
| 6285 | Pavetta lindina | 3 | 1 | 1 |
| 6286 | Pavetta mocambicensis | 4 | 1 | 1 |
| 6287 | Pavetta pumila | 532 | 1 | 1 |
| 6288 | Pavetta umtalensis | 4 | 1 | 1 |
| 6289 | Pavetta vanwykiana | 320 | 1 | 1 |
| 6290 | Pearsonia mesopontica | 2 | 1 | 1 |
| 6291 | Pelargonium mossambicense | 4 | 1 | 1 |
| 6292 | Phyllanthus bernerianus glaber | 220 | 1 | 1 |
| 6293 | Phyllanthus manicaensis | 1 | 1 | 1 |
| 6294 | Phyllanthus tsetserrae | 1 | 1 | 1 |
| 6295 | Pimpinella mulanjensis | 2 | 1 | 1 |
| 6296 | Platycaulos quartziticola | 88 | 1 | 1 |
| 6297 | Plectranthus caudatus | 1 | 1 | 1 |
| 6298 | Plectranthus cucullatus | 0 | 1 | 1 |
| 6299 | Plectranthus guruensis | 0 | 1 | 1 |

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|---|---|---|---|---|
| 6300 | Plectranthus mandalensis | 1 | 1 | 1 |
| 6301 | Plectranthus sessilifolius | 44 | 1 | 1 |
| 6302 | Polygala adamsonii | 5 | 1 | 1 |
| 6303 | Polygala francisci | 220 | 1 | 1 |
| 6304 | Polygala zambesiaca | 220 | 1 | 1 |
| 6305 | Polysphaeria harrisii | 4 | 1 | 1 |
| 6306 | Polystachya subumbellata | 21.029 | 1 | 1 |
| 6307 | Premna hans-joachimii | 532 | 1 | 1 |
| 6308 | Protea caffra gazensis | 88 | 1 | 1 |
| 6309 | Protea enervis | 2 | 1 | 1 |
| 6310 | Prunus africana | 578 | 1 | 0 |
| 6311 | Psychotria amboniana mosambicensis | 275 | 1 | 1 |
| 6312 | Psydrax fragrantissima | 272.61 | 1 | 1 |
| 6313 | Psydrax moggii | 104.17 | 1 | 1 |
| 6314 | Pyrostria chapmanii a of f.z | 2 | 1 | 1 |
| 6315 | Pyrostria sp. D | 3 | 1 | 1 |
| 6316 | Raphia australis | 294.986 | 1 | 1 |
| 6317 | Rhynchosia chimanimaniensis | 132 | 1 | 1 |
| 6318 | Rhynchosia genistoides | 1 | 1 | 1 |
| 6319 | Rhynchosia stipata Meikle | 4 | 1 | 1 |
| 6320 | Rhynchosia swynnertonii | 320 | 1 | 1 |
| 6321 | Rotheca luembensis niassensis | 1 | 1 | 1 |
| 6322 | Rotheca verdcourtii | 3 | 1 | 1 |
| 6323 | Rytigynia sp. c of f.z | 1 | 1 | 1 |
| 6324 | Rytigynia torrei | 2 | 1 | 1 |
| 6325 | Sansevieria subspicata | 334.28 | 1 | 1 |
| 6326 | Schistostephium oxylobum | 220.62 | 1 | 1 |
| 6327 | Schizochilus lepidus | 1 | 1 | 1 |
| 6328 | Schlechterina mitostemmatoides | 88 | 1 | 1 |
| 6329 | Sclerochiton apiculatus | 220 | 1 | 1 |
| 6330 | Sclerochiton hirsutus | 1 | 1 | 1 |
| 6331 | Scorodophloeus torrei | 140.883 | 1 | 1 |
| 6332 | Selago swynnertonii leiophylla | 1 | 1 | 1 |
| 6333 | Senecio aetfatensis | 1 | 1 | 1 |
| 6334 | Senecio peltophorus | 1 | 1 | 1 |
| 6335 | Senegalia latistipulata | 152.43 | 1 | 1 |
| 6336 | Sericanthe chimanimaniensis | 202.79 | 1 | 1 |
| 6337 | Siphonochilus kilimanensis | 373.68 | 1 | 1 |
| 6338 | Solanum litoraneum | 5 | 1 | 1 |
| 6339 | Spermacoce kirkii | 533 | 1 | 1 |

| | | | | |
|---|---|---|---|---|
| 6340 | Spermacoce schlechteri | 1 | 1 | 1 |
| 6341 | Stachys didymantha | 2 | 1 | 1 |
| 6342 | Stapelia unicornis | 1 | 1 | 1 |
| 6343 | Stomatostemma pendulina | 320 | 1 | 1 |
| 6344 | Streptocarpus acicularis | 1 | 1 | 0 |
| 6345 | Streptocarpus brachynema | 4 | 1 | 0 |
| 6346 | Streptocarpus erubescens | 1 | 1 | 0 |
| 6347 | Streptocarpus grandis septentrionalis | 1 | 1 | 0 |
| 6348 | Streptocarpus hirticapsa | 300 | 1 | 0 |
| 6349 | Streptocarpus leptopus | 1 | 1 | 0 |
| 6350 | Streptocarpus michelmorei | 2 | 1 | 0 |
| 6351 | Streptocarpus myoporoides | 2 | 1 | 0 |
| 6352 | Streptocarpus umtaliensis | 3 | 1 | 0 |
| 6353 | Stylochaeton tortispathus | 1 | 1 | 1 |
| 6354 | Syncolostemon flabellifolius | 88 | 1 | 1 |
| 6355 | Syncolostemon oritrephes | 2 | 1 | 1 |
| 6356 | Synsepalum chimanimani | 6 | 1 | 1 |
| 6357 | Tarenna longipedicellata | 1320 | 1 | 1 |
| 6358 | Tarenna pembensis | 4 | 1 | 1 |
| 6359 | Tarenna sp | 1 | 1 | 1 |
| 6360 | Tephrosia chimanimaniana | 88 | 1 | 1 |
| 6361 | Tephrosia chimanimaniana | 1 | 1 | 1 |
| 6362 | Tephrosia faulknerae | 0 | 1 | 1 |
| 6363 | Tephrosia forbesii | 2 | 1 | 1 |
| 6364 | Tephrosia gobensis | 3 | 1 | 1 |
| 6365 | Tephrosia longipes | 1 | 1 | 1 |
| 6366 | Tephrosia longipes | 1 | 1 | 1 |
| 6367 | Tephrosia montana | 4 | 1 | 1 |
| 6368 | Tephrosia praecana | 220 | 1 | 1 |
| 6369 | Thesium chimanimaniense | 2 | 1 | 1 |
| 6370 | Thesium dolichomeres | 4 | 1 | 1 |
| 6371 | Thesium pygmaeum | 3 | 1 | 1 |
| 6372 | Thespesia mossambicensis | 88 | 1 | 1 |
| 6373 | Triainolepis sancta | 44 | 1 | 1 |
| 6374 | Triaspis hypericoides canescens | 1 | 1 | 1 |
| 6375 | Triaspis suffulta | 0 | 1 | 1 |
| 6376 | Tricalysia jasminiflora hypotephros | 1 | 1 | 1 |
| 6377 | Tricliceras elatum | 1 | 1 | 1 |
| 6378 | Tricliceras lanceolatum | 1 | 1 | 1 |
| 6379 | Tricliceras longepedunculatum | 2 | 1 | 1 |

| | | | | |
|---|---|---|---|---|
| 6380 | Tritonia moggii | 321 | 1 | 1 |
| 6381 | Tulbaghia friesii | 2 | 1 | 1 |
| 6382 | Turbina longiflora | 3 | 1 | 1 |
| 6383 | Utricularia podadena | 1 | 1 | 1 |
| 6384 | Uvaria rovumae | 0 | 1 | 1 |
| 6385 | Vahlia capensis macrantha | 2 | 1 | 1 |
| 6386 | Vangueria domatiosa | 2 | 1 | 1 |
| 6387 | Vangueria monteiroi | 520 | 1 | 1 |
| 6388 | Vepris allenii | 3 | 1 | 1 |
| 6389 | Vepris drummondii | 220 | 1 | 1 |
| 6390 | Vepris macedoi | 2 | 1 | 1 |
| 6391 | Vepris sp. Nov | 4 | 1 | 1 |
| 6392 | Vernonia calvoana meridionalis | 44 | 1 | 1 |
| 6393 | Vernonia nepetifolia | 2 | 1 | 1 |
| 6394 | Viscum littorum | 1 | 1 | 1 |
| 6395 | Vismianthus punctatus | 132 | 1 | 1 |
| 6396 | Vitex francesiana | 88.79 | 1 | 1 |
| 6397 | Wahlenbergia subaphylla | 88.17 | 1 | 1 |
| 6398 | Warneckea albiflora | 2 | 1 | 1 |
| 6399 | Warneckea cordiformis | 2 | 1 | 1 |
| 6400 | Warneckea parvifolia | 312.31 | 1 | 1 |
| 6401 | Xerophyta argentea | 88.74 | 1 | 1 |
| 6402 | Xylia mendoncae | 320.56 | 1 | 1 |
| 6403 | Xylopia lukei | 132.22 | 1 | 1 |
| 6404 | Xylopia tenuipetala | 132.11 | 1 | 1 |
| 6405 | Xylopia torrei | 926.43 | 1 | 1 |
| 6406 | Xyris asterotricha | 5 | 1 | 1 |
| 6407 | Xyris makuensis | 3 | 1 | 1 |
| 6408 | Zanthoxylum delagoense | 132 | 1 | 1 |
| 6409 | Zanthoxylum tenuipedicellatum | 2 | 1 | 1 |
| 6410 | Warburgia salutaris | 5 | 1 | 1 |
| 6411 | Polysphaeria ribauensis | 0 | 1 | 1 |
| 6412 | Polystachya songaniensis | 0 | 1 | 1 |
| 7001 | Acontias parietalis | 220 | 1 | 1 |
| 7002 | Afroedura gorongosa | 132 | 1 | 1 |
| 7003 | Afroedura marleyi | 220 | 1 | 1 |
| 7004 | Atheris mabuensis | 58.991 | 1 | 1 |
| 7005 | Bradypodion setaroi | 22.75 | 1 | 1 |
| 7006 | Cordylus meculae | 132 | 1 | 1 |
| 7007 | Cycloderma frenatum | 264 | 1 | 1 |

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|---|---|---|---|---|
| 7008 | Dipsadoboa montisilva | 67.03 | 1 | 1 |
| 7009 | Kinixys natalensis | 220 | 1 | 1 |
| 7010 | Leptotyphlops telloi | 132 | 1 | 1 |
| 7011 | Lycophidion pygmaeum | 220 | 1 | 1 |
| 7012 | Mochlus lanceolatus | 44 | 1 | 1 |
| 7013 | Nadzikambia baylissi | 53.75 | 1 | 1 |
| 7014 | Platysaurus imperator | 264 | 1 | 1 |
| 7015 | Platysaurus lebomboensis | 132 | 1 | 1 |
| 7016 | Platysaurus torquatus | 264 | 1 | 1 |
| 7017 | Rhampholeon bruessoworum | 2.394 | 1 | 1 |
| 7018 | Rhampholeon gorongosae | 45.56 | 1 | 1 |
| 7019 | Rhampholeon marshalli | 38.298 | 1 | 1 |
| 7020 | Rhampholeon maspictus | 52.61 | 1 | 1 |
| 7021 | Rhampholeon nebulauctor | 11.51 | 1 | 1 |
| 7022 | Rhampholeon tilburyi | 10.81 | 1 | 1 |
| 7023 | Scelotes arenicolus | 44 | 1 | 1 |
| 7024 | Scelotes duttoni | 5.85 | 1 | 1 |
| 7025 | Scelotes insularis | 73.93 | 1 | 1 |
| 7026 | Scolecoseps broadleyi | 220 | 1 | 1 |
| 7027 | Smaug mossambicus | 132 | 1 | 1 |
| 7028 | Smaug warreni | 44 | 1 | 0 |

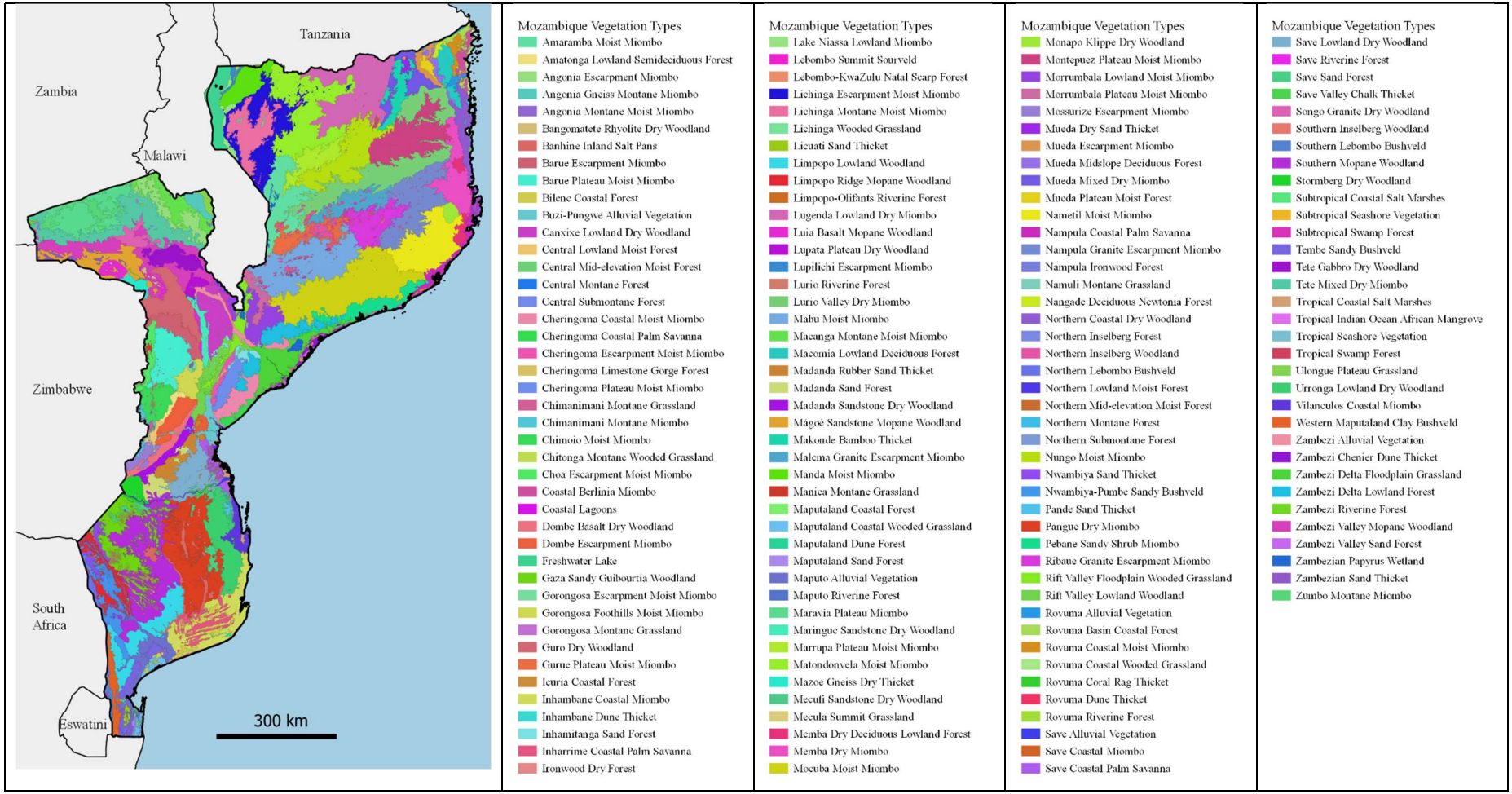



***Figure S5-1: Vegetation types of Mozambique by Lötter et al (2021).***