Identifying Priority Protection Zones within Protected Areas A Biodiversity Sensitivity-Value Analysis of the Kruger National Park, South Africa

DISSERTATION

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ABSTRACT

Approaches to conservation and natural resource management are maturing rapidly in response to changing perceptions of biodiversity and ecological systems. In response, traditional spatial planning techniques, previously restricted to the initial identification of new protected areas, are used here as a decision-support tool for co-ordinating activity zoning and management planning. Different land use practises (wilderness zonation) and varied degrees of elephant impact (elephant impact zonation) combine to form a disturbance template against which local prioritization of tourism, management and research actions are stratified for the Kruger National Park, South Africa. Areas of higher priority protection status are identified by zoning graduated levels of biodiversity sensitivity-value and classifying them according to the influence of different land use practices and varying levels of elephant impact. Compositional, structural and functional components of biodiversity, represented as grid layers in a GIS, are combined to form collective sensitivity and value grids and ultimately an overall biodiversity sensitivity-value index. As a result, 11% of KNP is classified as highly sensitive and valuable, 32% is moderately sensitive and valuable, and the remaining 57% has relatively low values of biodiversity sensitivity and value. This combination of high biodiversity sensitivity and value will facilitate spatial evaluations of human and/or elephant impact risk and the prioritisation of management actions.

A park zonation plan is an important conflict management template for the development of spatial activity planning and co-ordination of conservation, tourism and visitor experience initiatives in and around protected areas. Zones are identified along a spectrum of environmental modification, from pristine wilderness to high intensity leisure. The results indicate that 45% of KNP is composed of wilderness (n=62; 22 areas < 10 000 ha; 29 areas between 10 000 – 20 000 ha; 8 areas between 20 000 – 30 000 ha; and 3 areas of > 30 000 ha of intact wilderness). Wilderness areas are combined with biodiversity sensitivity and value, resulting in the identification of 17% of KNP as priority protection zones, consisting of valuable and sensitive wilderness areas. Density-driven elephant impact is another major environmental concern for most southern African protected areas. Although, this subject is fraught with the complexities of ecosystem theory and contentious ethical issues, the influence of naturally occurring high and low elephant densities on biodiversity components should underpin the identification of zones for potential management intervention. KNP is classified into six zones of persistent elephant concentrations over 25 years of dry-season scenarios. Consequently, 43% of KNP has experienced consistently high concentrations of elephants over the last 25 years. When combined with BIOSEVA, 19% of KNP is classified as highly valuable and sensitive and subjected to persistently high concentrations of elephants.

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DISCLAIMER

The results presented in this thesis are based on my own research at the Faculty of Earth and Life Sciences of the Vrije Universiteit Amsterdam and South African National Parks, Scientific Services. All assistance received from other individuals and organisations has been acknowledged and full reference is made to all published and unpublished sources. This thesis has not been submitted previously for a degree at any institution.

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~ **CHAPTER 1** ~

GENERAL INTRODUCTION

1.1. SETTING THE SCENE

Approaches to conservation and natural resource management are maturing rapidly in response to changing perceptions of biodiversity and ecological systems (Poiani *et al.* 2000). In past decades, biodiversity was viewed in terms of species richness in isolation of the supporting ecosystems, which were considered static and predictable (Fiedler *et al.* 1997). As a result, conservation activities are often focused in areas rich in species numbers and/or which contain rare species. However, in doing so conservation initiatives often ignore the different levels and the complex spatial patterns formed by the diversity of life on earth (Levin 2000) *viz.* composition, structure and function (Noss 1990; Figure 1). In response to this concern, systematic approaches to reserve identification and design were developed to ensure a representative sample of global biodiversity is sustainably protected (Margules & Pressey 2000).

These approaches often include, i) the assessment of conservation value, vulnerability and irreplaceability in the selection of priority conservation areas (Conservation or Biodiversity Assessment), and ii) the development of a framework of implementation to expand the current network of protected areas in response to such conservation or biodiversity assessments (Conservation or Biodiversity Planning) (Jackelman *et al.* 2007; Knight *et al.* 2007). Although these techniques are well represented in the literature (Margules *et al.* 1988, Rebelo & Siegfried 1992; Pressey *et al.* 1993, Lombard *et al.* 1997; Cowling & Heijnis 2001; Ball 2000; Carwardine *et al.* 2007), an implementation gap between conservation planners and practitioners prevent many plans from being realised (Copeland *et al.* 2007). As a result, current conservation plans are placing more emphasis on delivering tangible conservation goals (Balmform & Cowling 2006; Copeland *et al.* 2007).

Nevertheless, spatial planning is still predominantly restricted to the initial identification and design of new protected areas. Once an area has been proclaimed, the assumption is that biodiversity is completely protected (Zeng *et al.* 2005), albeit in a quasi fossilized state (Alexander 2008). The reality is, even within a formally protected area, disturbance (change) exists as both desirable natural disturbance and undesirable anthropogenic disturbance. The challenge for conservation researchers and managers is to assess what changes are beneficial and what are detrimental to ecosystem integrity. Complicating matters further, biodiversity is not evenly distributed across space or time as it varies in magnitude, significance and vulnerability both geographically and temporally. In other words, biodiversity is heterogeneous both spatially and temporally (Pickett *et al.* 2003). This makes conservation research and management particularly difficult, especially in a country like South Africa (SA), which has been described as a nation of megadiversity (Mittermeier *et al.* 1997).

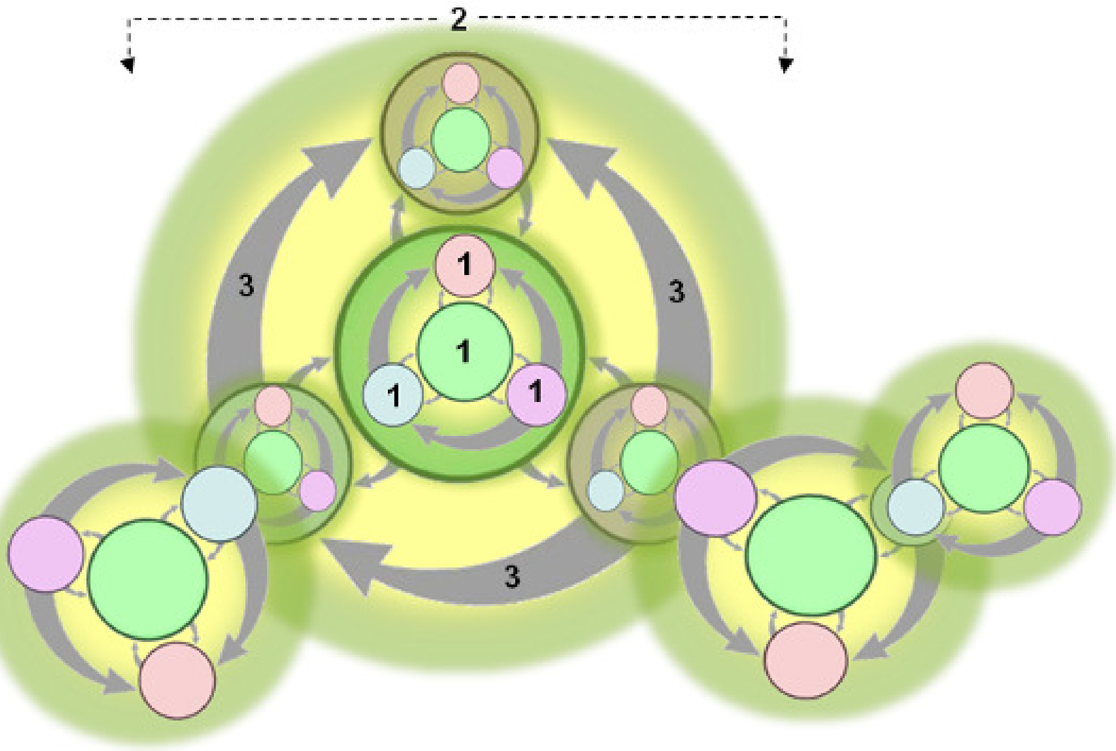


Figure 1: Framework of ecosystem components: composition[1] (type); structure[2] (pattern) and function[3] (process) (Mac Fadyen 2009). The clearly hierarchical and interrelated structure of natural systems is composed of a complex arrangement of elements[1] (genes, species, populations, communities, ecosystems, landscape types); controlled by relationships[2] and interactions[3] among elements (Noss 1990; Amiri & Nakane 2009).

South Africa is ranked as one of the top 25 most bio-diverse countries in the world (WCMC 1992; Reyers *et al.* 2002), comprising almost 10% of the planets' plant species (the fifth highest number in the world) and 7% of its reptile, bird and mammal species (DEAT 2006). SA's global biodiversity estate is disproportionately large considering it only occupies 2% of the world's land surface (King *et al.* 2005). It is therefore essential for key regions to be set aside as protected areas, for the conservation of not only these species but also the ecosystems, landscapes and ecological and evolutionary processes defining the variable states of biodiversity over time (Driver *et al.* 2005). Protected area management should therefore be focused on maintaining ecosystem integrity as opposed to merely "farming for threatened species" (a species-centric approach). Although, protected areas should aim to identify significant species, it is the ecosystems and/or landscapes in which they live that must be protected if conservation is be sustainable (an ecosystem approach). This is particularly important considering natural habitat loss or degradation is regarded as the single biggest threat to biodiversity (Driver *et al.* 2005). Bearing in mind that, *"it is not the strongest species that survive, nor the most intelligent, but the ones most responsive to change"* (Darrow 1987, p.1), conservation strategies must also be adaptive.

The worldwide and local (SA) networks of protected areas are exposed to ever changing and intensifying global population pressure (Balmford *et al*. 2001). As a result, adaptive conservation goals are aimed at moving targets as a paradigm shift from the 'balance of nature' to the 'flux of nature' has occurred in conservation science theory (Wu & Loucks 1991; Kalamandeen & Gillson 2007). Nevertheless, even in the face of this complexity, research and management decisions need to be taken and biodiversity must be conserved. However, since biodiversity is clearly not static, in what state should these ecosystems be conserved, and what are the thresholds of acceptable ecosystem change for the future?

The recent State of the Environment Report (DEAT 2006) resulting from the National Spatial Biodiversity Assessment (Driver *et al*. 2005), provides a detailed assessment of the state of SA's ecosystems. As with any systematic conservation assessment, the objectives were to identify priority areas for biodiversity conservation (Berliner *et al.* 2007). This is particularly important for the wealth of biodiversity lying outside of SA's formally protected areas (Freitag *et al.* 1998; Reyers *et al.* 2002). However, what happens once the area has been proclaimed and formally protected? Land use planners classify the area as a homogenous and benign land use, and local conservation practitioners begin the processes of managing their biodiversity islands, often in isolation of national (and global) irreplaceability knowledge. However, a gradient of land use still occurs within these protected areas in association with local research, management and tourism activities, albeit at a finer scale.

Therefore, I propose that traditional biodiversity assessment and conservation planning techniques be applied within already formally proclaimed protected areas like the Kruger National Park (KNP), to prioritise and focus localised protection efforts in the context of National, and where feasible global, spatial biodiversity sensitivities and values. Once this gradient of biodiversity sensitivity and value has been classified into zones (biodiversity sensitivity-value zonation), it will help guide and co-ordinate conservation, tourism and visitor activity zoning in conjunction with a park-specific Conservation Development Framework (CDF) (KNP 2009). In this way conflicts between different land uses and their inherent biodiversity sensitivity and value may be minimised (KNP 2009). Similarly, results may be applied to Park specific management concerns like the influence of elephants in the KNP, on the landscape and, with neighbouring rural communities. In other words, in this study different tourism land use practises (wilderness zonation) and varied degrees of elephant impact (elephant impact zonation) will comprise a disturbance template against which priority protection areas may be identified and management actions may later be stratified.

1.2. RESEARCH GOALS AND QUESTIONS

This research is based on a systematic conservation planning approach (Margules & Pressey 2000) adapted from traditional spatial biodiversity assessment techniques (Driver *et al.* 2005). The overall aim being to design a Biodiversity Sensitivity-Value Analysis (BIOSEVA) method to determine priority conservation areas spatially in terms of their vulnerability (sensitivity) to disturbance in the context of their National biodiversity significance (value). As a result, the KNP BIOSEVA is applied here as a decision support tool for the integration of systematic conservation planning principles into a spatial planning framework for human activities (land use) and elephant management within the Kruger National Park.

The main research question addressed here is, "**How can priority protection zones be determined within established protected areas, in the face of competing tourism needs and elephant impact, and the influence these may have on biodiversity**"? This is further broken down into the following sub-questions:

- How can priority protection areas be identified from zones of spatially varying levels of biodiversity sensitivity and value of KNP's biophysical template?

- How can this biodiversity sensitivity-value stratification be applied to assess impending impact of different protected area land use practises and varied degrees of elephant disturbance in and around protected areas?

In answering these questions, best available knowledge of local biodiversity may be integrated into a spatial conservation planning process aimed at local disturbance research, monitoring and management. While it is recognised that fire, in combination with a complex web of many other interacting and interrelated elements also exists as an ecosystem modifier, within the context of this study "disturbance" will be restricted to human and elephant related impacts.

1.3. RESEARCH OUTLINE

A schematic outline of this thesis (Figure 2) provides a description of the key concepts of the research I have introduced, and an assessment of the goals and questions. Chapter two further sets the scene by describing the study area and associated past and present management regimes. Chapter three describes the methodology behind the delineation of priority protection zones (see Figure 3) and chapter four demonstrates their application for management. Chapter five provides an overall discussion and conclusion on prioritising protection efforts within established protected areas.

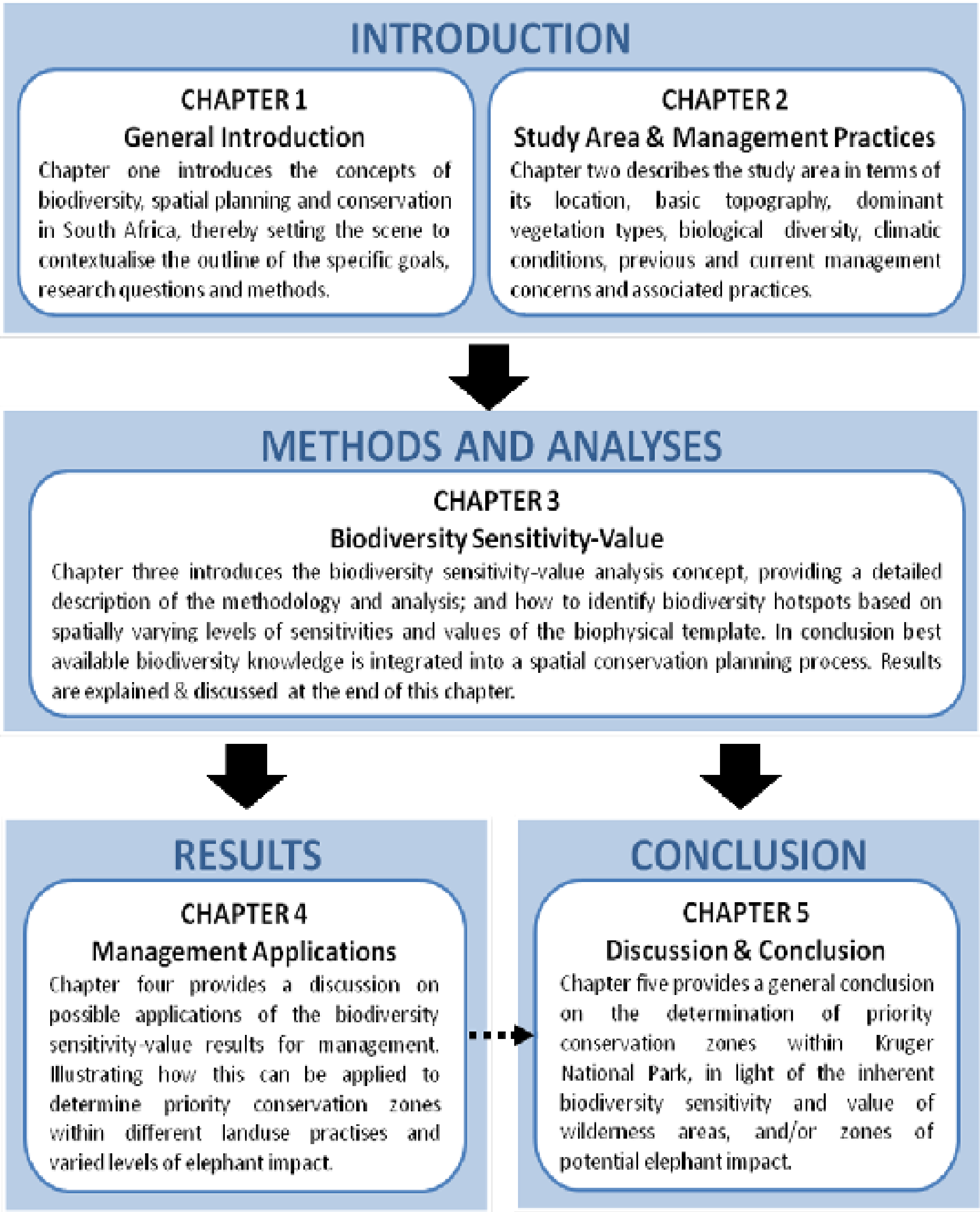


Figure 2: A graphical overview of the research process outlined in the five chapters of this thesis.

In order to determine those zones of higher priority protection status, the KNP must first be stratified into zones graduated by different levels of biodiversity sensitivity and value (Figure 3). Each zone will also fall within a different land use neighbourhood, delineated by internal human pressure (an existing human footprint), which in turn is subjected to different levels of elephant impact, approximated by high to low elephant distribution-density patterns (Figure 3).

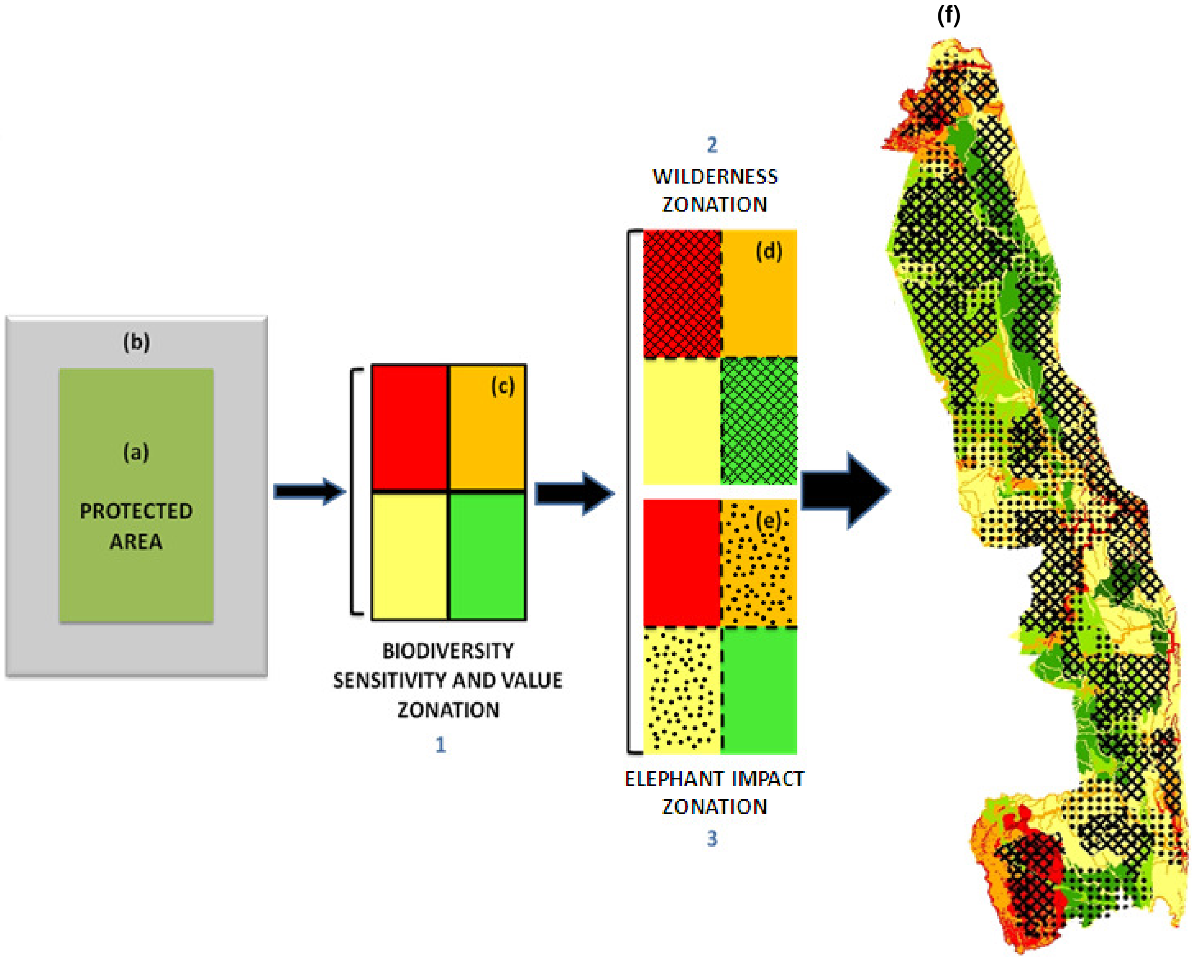


Figure 3: A graphical representation of the research methodology, where Biodiversity Sensitivity-Value stratification (1) is applied to Wilderness (or use zones; (2) and Elephant Impact zones (3). The study area (KNP) is a protected area (a), surrounded by socio-economic human pressures (b). KNP is stratified into zones graduated by different levels of biodiversity sensitivity-values (c), which in turn are compared with different land use zones delineated by internal human pressure (KNP minus existing human impact "footprint" equals remaining wilderness; (d) and elephant impact (approximated by distribution-density patterns; (e) to delineate high priority protection areas (f).

~ **CHAPTER 2** ~

STUDY AREA AND ASSOCIATED MANAGEMENT PRACTICES

South African National Parks (SANParks) is the conservation body responsible for the implementation and management of nationally protected areas in SA. In accordance with the National Environmental Management: Protected Areas (NEM:PA; Act No. 57 of 2003) and Biodiversity Acts (NEM:BA; Act No. 10 of 2004), SANParks is charged with *"the development and management of a system of National Parks that represent the biodiversity, landscapes, and associated heritage assets of South Africa for the sustainable use and benefit of all"* (SANParks 2009, p. 1). SANParks currently manages 22 National Parks, proclaimed under the NEM:PA Act, and is responsible for 53% of SA's protected areas (Hall-Martin & Carruthers 2003) totalling 3.2% (~ 3 923 261 ha) of SA's land mass. The Kruger National Park makes up the bulk of this area (~ 2 million ha) and as such it is the country's premier National Park entrusted with the protection of 50 fish; 505 bird; 35 amphibian; 119 reptile; 1990 plant, 148 mammal and thousands of invertebrate species (Mabunda *et al.* 2003). KNP aims to "*maintain biodiversity in all its natural facets and fluxes, to provide human benefits and build a strong constituency and to preserve as far as possible the wilderness qualities and cultural resources associated with the Park*" (KNP 2009, p. 15). In accepting the spatially and temporally dynamic nature of biodiversity (Figure 1), KNP has adopted an adaptive management approach towards the application of biodiversity conservation. This style of management exists under the realisation that, *"nothing endures but change"* (Heraclitus 540 BC - 480 BC) and therefore if scientists and managers do not adapt and learn, management will fail, to the detriment of the species and ecosystems under its protection.

KNP is situated in the north-eastern corner of South Africa, bordered by Mozambique in the East and Zimbabwe in the North (Figure 4). It is one of the largest protected areas in the world, covering an area of almost two million hectares between latitudes 22˚19'40" S - 25˚31'44" S and longitudes 30˚53'18" E - 32˚01'59" E (Foxcroft *et al*. 2009). Occurring within SA's dominant savanna biome (Low & Rebelo 1996), the KNP consists of 20 board-scale vegetation types (Mucina & Rutherford 2006), classified into 35 landscapes (Gertenbach 1983) or 56 landtypes (Venter 1990). KNP has a gentle undulating topography with the exception of an east-west altitudinal gradient starting from basalt plains at 200m above sea level and ending with granite hills at 700m above sea level (Mabunda *et al.* 2003). With a variable mean annual rainfall of between 350 mm in the north to 950 mm in the south-west, the KNP experiences hot, wet summers between December and March followed by mild, dry winters from June to August (Wessels *et al.* 2006).

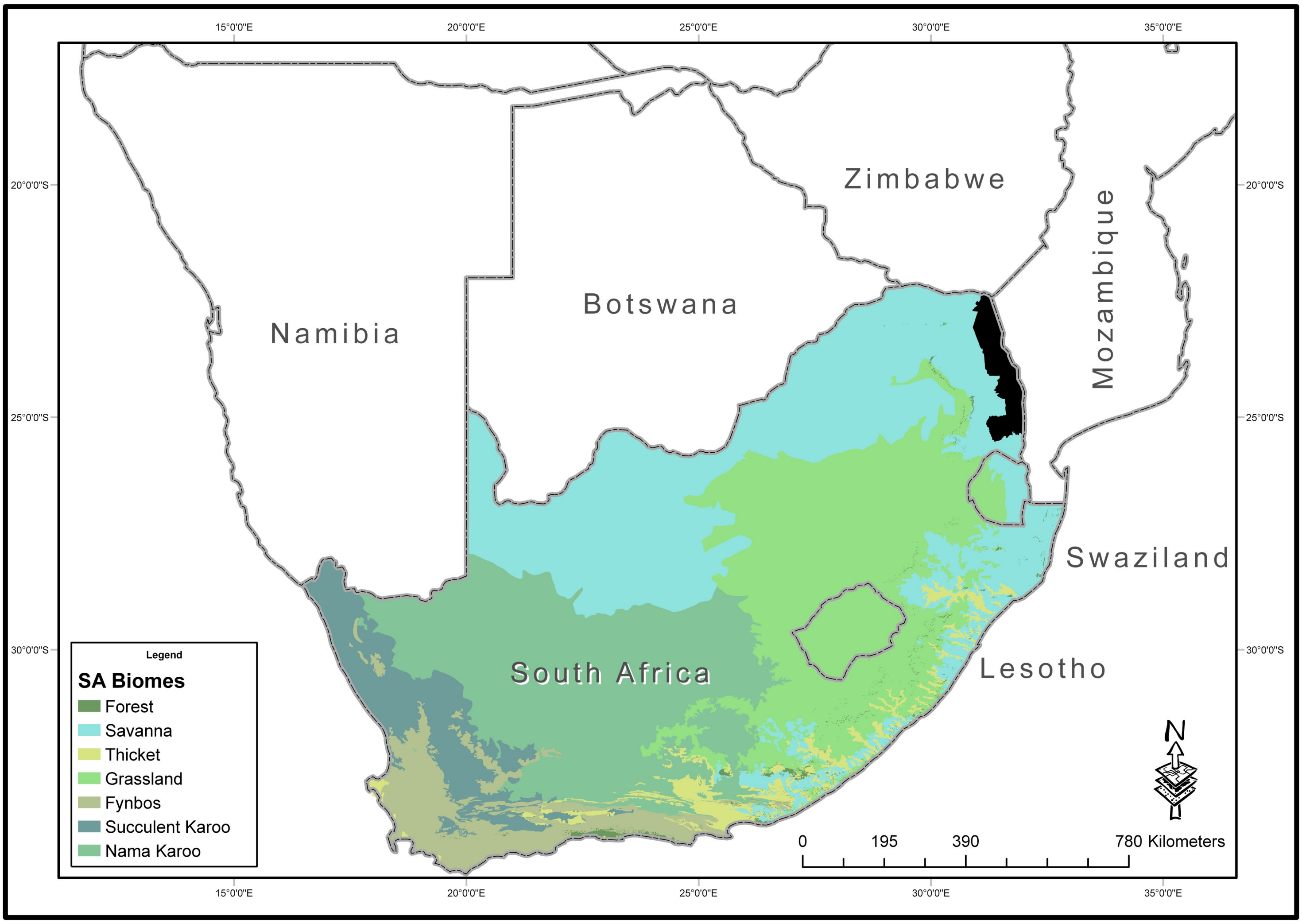


Figure 4: The Kruger National Park is situated in the north-eastern corner of South Africa, within the mesic savanna biome (Mucina & Rutherford 2006).

Since the official proclamation of the KNP in 1926, the protectionist philosophy responsible for its birth has ebbed in response to political, socio-economic, intellectual and ethical changes in society (Carruthers 1995). The science underpinning its nature conservation strategies has evolved iteratively through changing ecological paradigms. From eras of hunter-gatherers (Pre 200 AD); to Iron Age farmers, metalworkers and traders (200-1836 AD); pioneers and hunters (1836-1902); game preservationists (1902-1925); the creation of a National Park (1926-1946); management intervention (1946-1990) and political changes (1990-2002), humans have long been engineers of natural systems (Mabunda *et al.* 2003). The extent of mans' influence on these natural systems is linked to issues of human population density and the evolution of natural science theory and knowledge. In response, associated management regimes have ranged from simple naturalist interest, basic observation of natural phenomenon, *a priori* management intervention, systematic monitoring; *a posteriori* management intervention and finally, researched based adaptive management. Today complex adaptive systems and heterogeneity theories steer research, monitoring and management on a strategic adaptive course (Biggs & Rogers 2003).

These theories stem from the belief that ecological systems are dynamic across space and time, incorporating multiple physical and biological features, processes and scales (Pickett *et al.* 1997). Biodiversity plays an important role in ecosystem function, resilience and the provision of ecosystem services (Figure 1; MEA 2005). As a result, a loss of biodiversity will have a negative effect on human well-being by compromising food security, resilience to natural disasters, energy security, access to clean water and raw materials, human health, social relations, and freedom of choice (MEA 2005). The question may arise as to why management is needed at all if the removal or control of anthropogenic threats, through the proclamation of protected areas, is said to protect natural processes responsible for the self-maintenance of ecosystems (Alexander 2008). The reality is today's protected areas are constrained ecosystems, no longer capable of self-regulation. For example, since the erection of the western boundary fence in 1961 the natural movement of large mammals in the KNP has been disconnected. This single event and the resulting management actions have had far reaching consequences for KNP's biodiversity (Mabunda *et al.* 2003).

In addition, biodiversity is not spatially homogenous and as such management actions must be spatially dynamic. In response to this concern, the KNP was classified into 35 landscapes of similar geomorphology, climate, soil and vegetation patterns and associated fauna Gertenbach (1983). The objective was that these landscapes would represent basic functional units on which all future management decisions would be based Gertenbach (1983). The classification was later refined into 56 landtypes on the basis of similar soil and vegetation patterns, and geological, geomorphological and climatic characteristics for management and research planning (Venter 1990). However, in order to advance conservation efficacy, research efforts should be prioritized on the basis of biodiversity value and management strategies should be formed in light of biodiversity sensitivity at a finer scale than landscape or landtypes (Sarkar *et al.* 2002). As a result, the spatial pattern of biodiversity, in terms of sensitivity and value, forms the basis of a new stratification for management and research planning. Planning that may for example reduce conflicts between different land uses or levels of elephant densities and the intrinsic biodiversity of a protected area.

Efforts have been made in the past to reduce these conflicts through zoning Recreational Opportunities (ROZ) (Figure 5a; Venter *et al.* 1997; Freitag-Ronaldson *et al.* 2003) and Elephant Management Zoning (Figure 5b; Whyte *et al.* 1999). Unfortunately due to the prevailing beliefs of the time and a lack of spatially explicit environmental data, biodiversity maintenance was not the driving principle behind either of these zonations (Freitag-Ronaldson *et al.* 2003). Furthermore, the accurate boundaries of the respective ROZ and elephant management zones only roughly conformed to known boundaries of human influence and elephant distribution (Whyte *et al.* 1999). Needless to say, if the management of human and elephant activities is to be meaningful, boundaries should be drawn using actual human and elephant distribution and density data in a Geographic Information System (GIS) (ESRI 2000). Chapters three and four discuss the methodology behind new approaches to mapping biodiversity (sensitivity and value), human influence (wilderness gradient) and elephant impact (elephant hotspots).

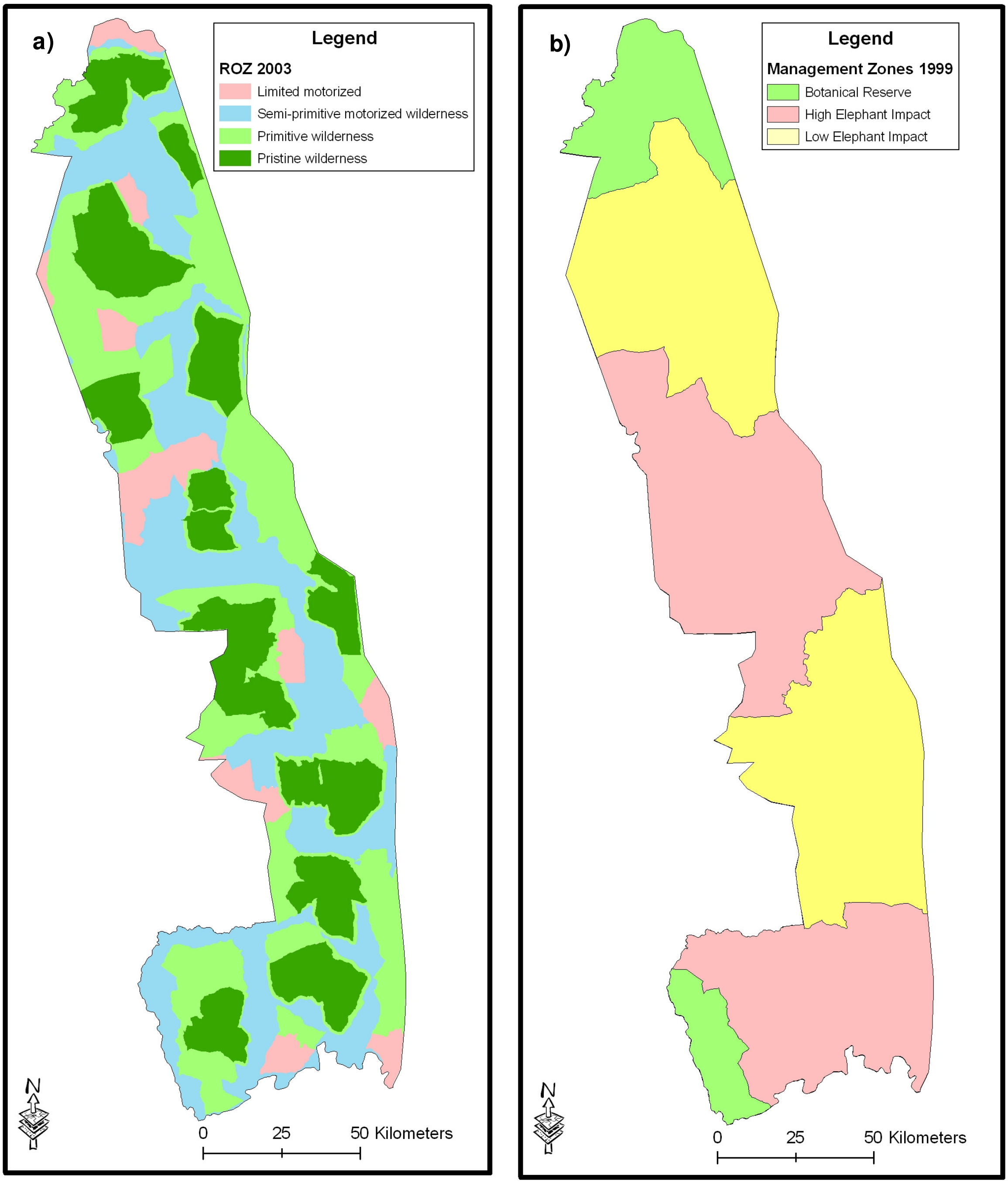


Figure 5: Past land use (wilderness) and elephant impact zonation plans used in KNP: **a)** Recreational Opportunity Zones: Limited motorized; semi-primitive motorized wilderness; primitive wilderness; pristine wilderness (Freitag-Ronaldson *et al.* 2003) and **b)** Elephant Management Zones: High impact zones - elephant population will be allowed to increased until there are indications of biodiversity disturbance; low impact zones - elephant population will be decreased until there are indications that disturbance has been reduced - botanical reserves, medium densities are to be maintained (Whyte *et al.* 1999).

~ **CHAPTER 3** ~

BIODIVERSITY SENSITIVITY-VALUE ANALYSIS

3.1 INTRODUCTION

Conservation scientists need to be proactive in their research, monitoring and management strategies if they wish to answer the perplexing question of how to prioritize protection efforts by focussing resources (Cowling & Heijnis 2001). As a result, the early BIOSEVA process was developed within SANParks, in close consultation with DEAT, as a decision support tool for local spatial biodiversity and conservation planning (Holness & Skowno 2008). Similar in application to a monitoring stratification of environmental variables that best predict ecological patterns (Margules *et al.* 2003), BIOSEVA will help stratify or prioritize protection efforts on the basis of the conservation value of an area (its contribution to the national conservation estate) and its sensitivity (its vulnerability to a specific disturbance type) (Holness & Skowno 2008). The concept can be broken down into three components as follows:

Biodiversity (BIO): Compositional, structural and functional spheres of ecosystems, biological assemblages, species and populations (Figure 1; Noss 1990).

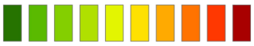
Sensitivity (SE): An element or areas' level of vulnerability to a specific disturbance factor e.g. human use or elephant impact. This component of the model investigates the variation of biodiversity with respect to the variation in tolerance to a specific disturbance (Apte 2002). Disturbance is restricted here to physical disturbance and the sensitivity of steep slopes (topography); soils; drainage systems (hydrology); threatened species (plants and mammals) and vegetation.

Value (VA): An element or areas' total contribution to the National and/or local conservation estate. This component of the model affixes a measure of importance or worth to an element or area with regard to its biodiversity estate. Values are attributed to special habitats (topographic variability); national and local habitat conservation status and species richness (plants and mammals).

Sensitivity and value are described spatially using different environmental layers (Figure 6) in a GIS, representing the different levels of biodiversity complexity in KNP. Different mathematical overlay operations (Saraf 2002) are iteratively tested using ESRI's Spatial Analyst extension and Raster Calculator tool (ESRI 2000) to ensure results are robust in their combination of layers and classification of value ranges (low-high). Similarly, the resulting sensitivity and value layers are spatially combined to form an overall BIOSEVA layer. In quantifying the national value and sensitivity of local biodiversity features, decision makers may be able to mitigate some of the local risks causing a decrease in biodiversity estates (Gaston *et al.* 2002). Examples of local risks are discussed in chapter four.

3.2. METHODOLOGY

Some areas have higher levels of biodiversity than others, as they may contain larger numbers of species or ecosystems, threatened species, or more intact ecological processes (Ferrar & Lötter 2007). In order to measure the sensitivity and value of these areas, all components of biodiversity (Figure 1) must first be translated from real world features into database entities (points, lines and areas) in a GIS (Longley *et al.* 2005). As a result, the KNP BIOSEVA process is divided into three levels of spatial analysis as follows (Figure 6):

1. Data preparation, compilation and scoring (i.e. 1-10; 1 being least sensitive or valuable [green] and 10 being most sensitive or valuable [red]; )

2. Combination of various data layers into meaningful biodiversity sensitivity and value layers

3. Combination of biodiversity sensitivity and value layers into the final BIOSEVA index.

Realistically, data limitations will almost always shape a modelling framework. Not surprisingly, surrogates are therefore often needed to model complex systems like the environment (Ferrier 2002). For example, Cowling and Heijnis (2001) used the intersection of geology (as a surrogate for substratum), topography (as a surrogate for temperature) and climate to *identify broad habitat units called biodiversity entities for systematic conservation planning in the Cape Floristic Region*. Similarly, Knight *et al.* (2007) used the presence of rare and endangered species, range-restricted and biome-restricted species, roosting or breeding sites, and large populations of species to *improve the key biodiversity areas approach for effective conservation planning*. Rouget *et al.* (2004), identified priority areas for habitat, species and process protection by scoring different layers of habitat conservation and protection status, and irreplaceability; plant and animal species irreplaceability and vulnerability; and ecological processes (using surrogates for water production, carbon sequestration, biogeographic nodes, escarpment and climate change resilience) in the first *National Spatial Biodiversity Assessment (NSBA)* for SA. Later Ferrar and Lötter (2007) used a similar approach to map vegetation types, threatened taxa distribution, special landscape features and process surrogates in a finer scale terrestrial analysis for the *Mpumalanga Biodiversity Conservation Plan (MBCP).*

Clearly, there is no single method for identifying biodiversity priority areas across multiple layers, numerous biodiversity features and various spatial scales (Rouget *et al.* 2004). As a result, data limitations are assessed and appropriate data layers selected and/or generated to represent components of KNP's biodiversity status. These layers are combined to form an overall account of KNP's biodiversity sensitivity and value as follows (Figure 6).

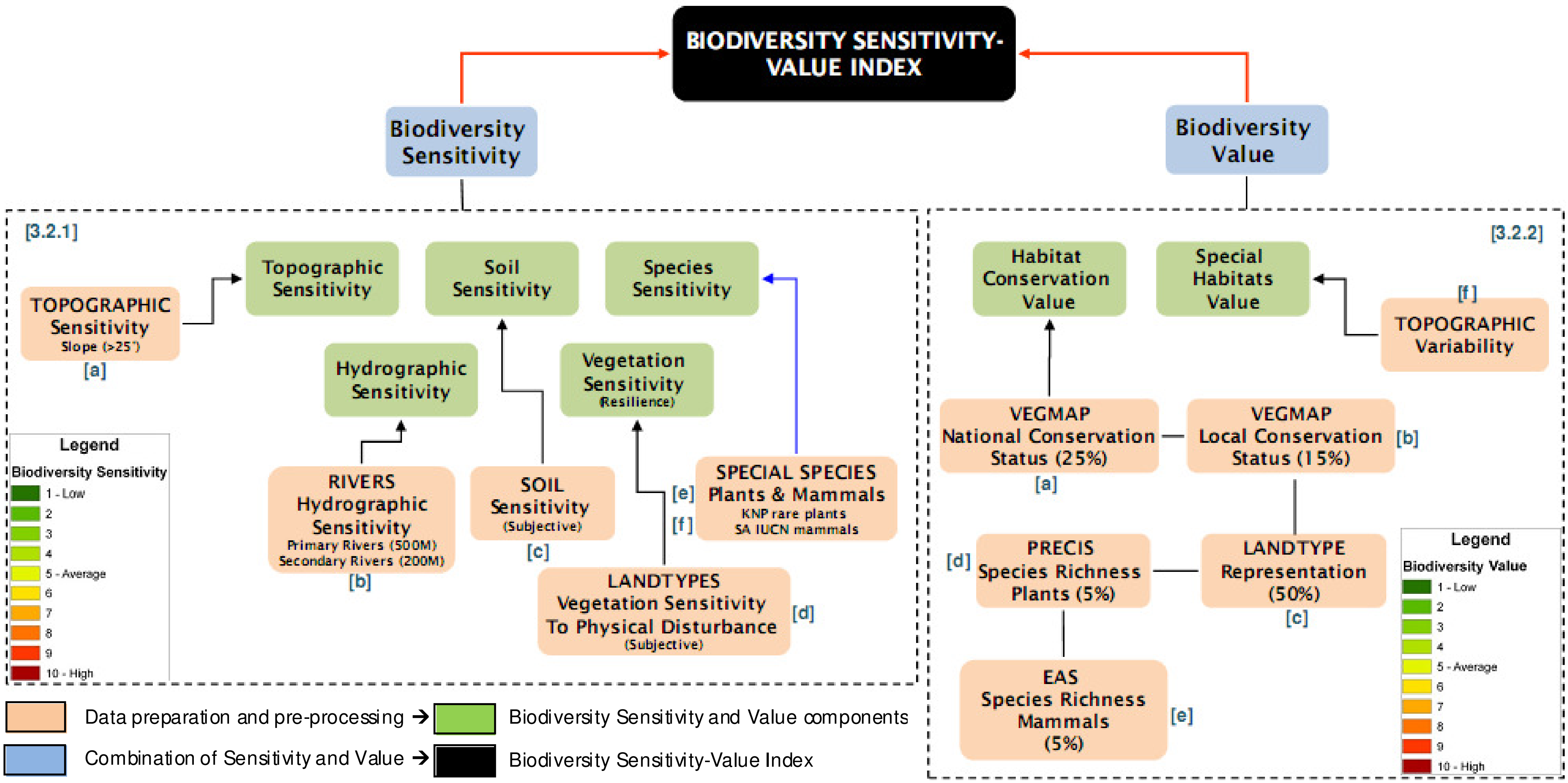


Figure 6: layers and basic processing steps in the Kruger National Park's Biodiversity Sensitivity-Value Analysis (BIOSEVA)

3.2.1 BIODIVERSITY SENSITIVITY

The following section provides a detailed description of each of the biodiversity sensitivity components, the underlying data layers, and their use.

a)	Topographic Sensitivity	d)	Vegetation Sensitivity
b)	Hydrological Sensitivity	e)	Special Species – Plants
c)	Soil Sensitivity	f)	Special Species – Mammals

a) Topographic Sensitivity

A 20 m Digital Elevation Model (DEM), consisting of a sampled array of elevations for a number of ground positions at regularly spaced intervals, was used to derive slope across the KNP. By identifying the maximum rate of change, from each cell to its neighbours, the degree of slope for each cell location is output as a raster grid (Applegate 1995). Apart from the health, safety, environmental and aesthetic considerations, the disturbance of steep slopes may increase soil erosion, sedimentation and ultimately decrease water quality (Williams *et al.* 2008). Since the KNP is generally flat, slope sensitivity is classified as follows: **0-5 (0)**; **5-10 (2)**; **10-15 (6)**; **15-25 (8)** and **>25 (10)** (Figure 8a; Holness, pers comm.).

b) Hydrological Sensitivity

Freshwater ecosystems are in general highly impacted systems (Maree *et al.* 2006). However impacts can be minimized if riparian buffers are protected (Maree *et al.* 2006). Suggested buffer distances in the literature range from 32 m to 100 m (Berliner *et al.* 2007) and 50 m to 200 m (Holness & Skowno 2008; Lagabrielle *et al.* 2009). Erring on the side of caution, the KNP Rivers and drainage lines are buffered from 200 m to 500 m according to river class, thereby ensuring a complete representation of associated riparian areas *viz.* **Primary 0-1 (500 m)** and **Secondary 2-3 (200 m)** rivers. Considering the sensitive nature of riparian areas, all river-buffered areas are scored as **10** with the remaining upland areas scored as **0** (Figure 8b).

c) Soil Sensitivity

Soil disturbance can lead to top soil loss (erosion), the establishment of undesirable (weedy or ruderal) plant species (Hobbs & Huenneke 1992), a reduction in microbial biomass and micro-arthropod populations and ultimately compromise decomposition (Migge-Kleian *et al.* 2007). As a result, 56 land types, which are described on the basis of soil and vegetation patterns and landform characteristics, are classified according to their sensitivity as follows (Venter 1990): An expert estimate of soil sensitivity was calculated per land type, 1 being low sensitivity and 10 being high sensitivity. Accordingly, seven land types are classified as **not sensitive (1-2)**; 28 have **low sensitivity (3-5)**; 17 have **medium sensitivity (6-8)** and four are **highly sensitive (9-10)** (Figure 8c; Venter, *pers comm.*).

d) Vegetation Sensitivity (or resilience to physical disturbance)

Some vegetation types are more resilient to disturbance than others. As a result, 56 land types which are described on the basis of soil and vegetation patterns and landform characteristics, are classified according to their sensitivity as follows (Venter 1990): An expert estimate of vegetation sensitivity/vulnerability was calculated per land type with scores of 2 representing a low sensitivity and 10 high sensitivity. According to these criteria, four land types are classified as **not sensitive (2)**; 18 have **low sensitivity (3-5)**; 31 have **medium sensitivity (6-8)** and three are **highly sensitive (9-10)** (Figure 8d; Eckhardt & Venter, *pers comm*.).

e) Special Species – Plants

The presence of threatened plant species is a key driver of protected area design and management (Ball 2000). As a result all red-data species of the International Union for Conservation of Nature (IUCN) categories, Critically Endangered (CR), Endangered (EN) and Vulnerable (VU) were extracted from the KNP herbarium database (Table 1) and summarised by land type. Although a density interpolation of point data (plant localities) would be ideal, since herbarium records are fraught with collection bias these data are instead summarised by landtype units to ensure all potential habitats are represented. As a result, three landtypes contained **CR** plant species while two contained **E** and another three contained **VU** species (Figure 7). These CR, EN and VU records were then multiplied by 5, 2 and 1 respectively in order to calculate a maximum status score per landtype, thereby also taking occurrence into account (Figure 8e; Table 1).

Table 1: IUCN red-data listed plant species occurring in KNP

SCIENTIFIC NAME	STATUS	OCCURRENCE RANKING
Adenium swazicum	CR	2
Barleria oxyphylla	E	1
Orbea paradoxa	E	1
Rhynchosia vendae	VU	5
Siphonochilus aethiopicus	CR	1
Warburgia salutaris	E	1
Woodia singularis	VU	1

f) Special Species – Mammals

The presence of threatened mammal species is a key driver of protected area design and management (Ball 2000). The University of Pretoria's South African Terrestrial Mammal Distribution project includes distributional data, which was based on information obtained from various natural history museums in South Africa (Keith 2004). Additional data were also obtained from work done by researchers from the Department of Zoology and Entomology, at the university.

The dataset consists of compiled museum records and personal observations translated into GIS shapefiles containing the extent of occurrence for most of the assessed terrestrial mammals (Friedmann & Daly 2004). These Quarter Degree Square (QDS) resolution data, representing 241 mammal species, was spatially combined with the KNP landtypes and all the red-data species of IUCN categories CR, EN and VU extracted (Figure 7; Table 2). The CR, EN and VU records were then multiplied by 5, 2 and 1 respectively to recreate a maximum status score per landtype, thereby taking occurrence into account. As a result, 48 landtypes contained both EN and VU species while 8 contained all three categories **CR**, **EN** and **VU** species (Figure 8).

Figure 7: A representation of the relationships between IUCN Red List Categories (IUCN 2001, and adapted from Rouget *et al.* 2004)

Table 2: IUCN red-data listed mammals from the University of Pretoria mammal distribution database

COMMON NAME	SCIENTIFIC NAME	IUCN
African Wild Dog	*Lycaon pictus*	EN C1
Cheetah	*Acinonyx jubatus*	VU C2a(1)
Greater Musk Shrew	*Crocidura flavescens*	VU B1+2c
Juliana's Golden Mole	*Neamblysomus julianae*	CR B1+2c
Lion	*Panthera leo*	VU C2a (i)
Rough-haired Golden Mole	*Chrysospalax villosus*	VU B1+2c
Spotted-necked Otter	*Lutra maculicollis*	VU A1c
Springhare	*Pedetes capensis*	VU A1cd

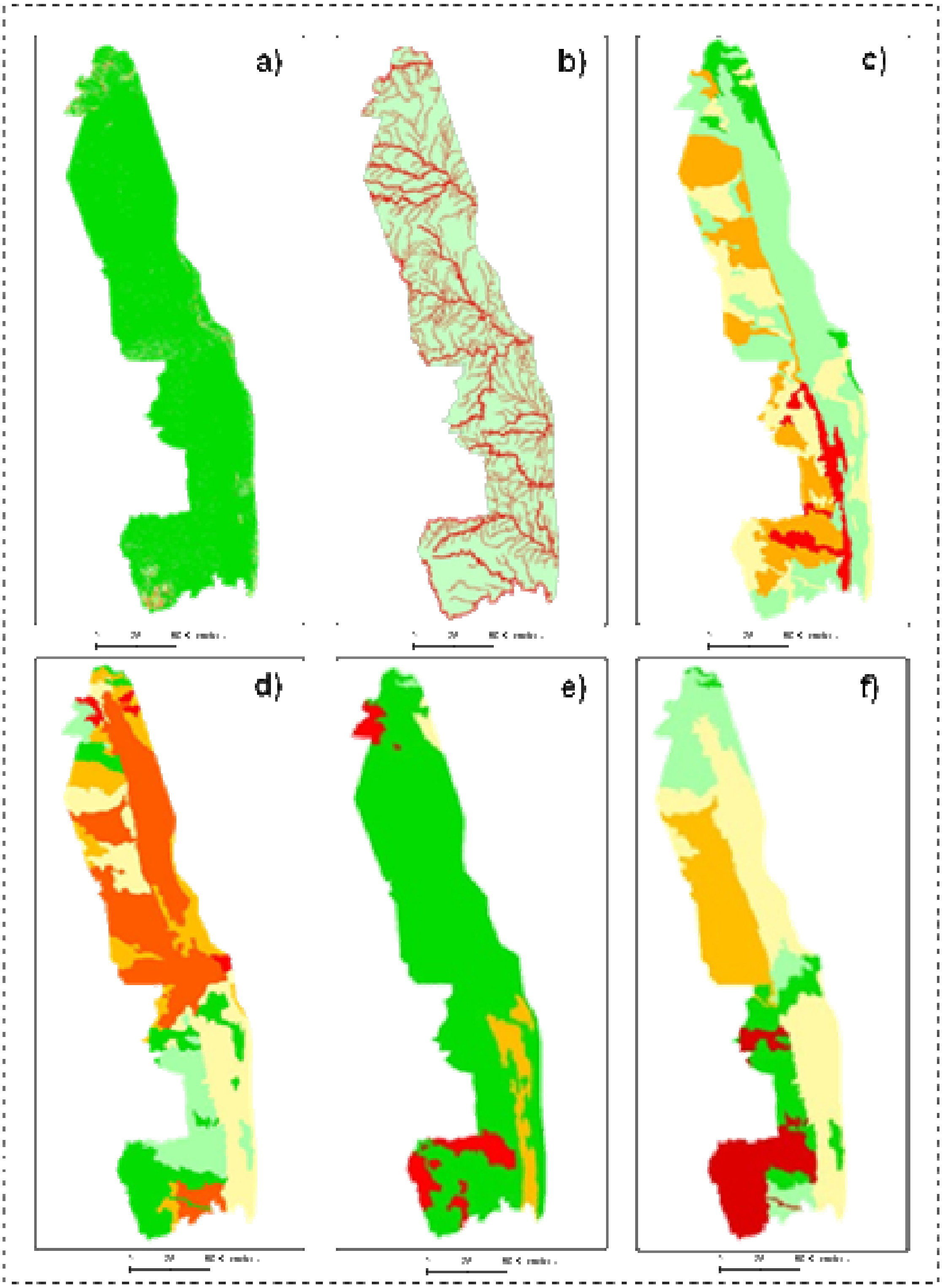


Figure 8: Individual layers used for the Biodiversity Sensitivity analysis. a) Topographic Sensitivity; b) Hydrological Sensitivity; c) Soil Sensitivity; d) Vegetation Sensitivity (or resilience to physical disturbance); e) Special Plant Species; f) Special Mammal Species. Scores range from green-red: green being least sensitive [e.g. score 1] and red being most sensitive [e.g. score 10].

3.2.2 BIODIVERSITY VALUE

The following section provides a detailed description of the biodiversity value components, the underlying data layers, their use and % weighting.

a) National Conservation Status
b) Local Conservation Status
c) Landtype Representation
d) Species Richness - Plants
e) Species Richness - Mammals
f) Topographic Variability

The Vegetation map of South Africa, Lesotho and Swaziland (VEGMAP; Figure 9) forms the basis of the National and local Conservation Status components of Biodiversity Value described in more detail below. VEGMAP was a collaborative initiative funded by the Department of Environmental Affairs and Tourism (DEAT) and is managed by the South African National Biodiversity Institute (SANBI). The aims of this project included (i) an analysis and synthesis of SA vegetation data, and (ii) the derivation of a revised vegetation map (Mucina & Rutherford 2006). As a result, the map accurately reflects the distribution and variation in SA vegetation and indicates the conservation status of each vegetation type Nationally (Rouget *et al.* 2004).

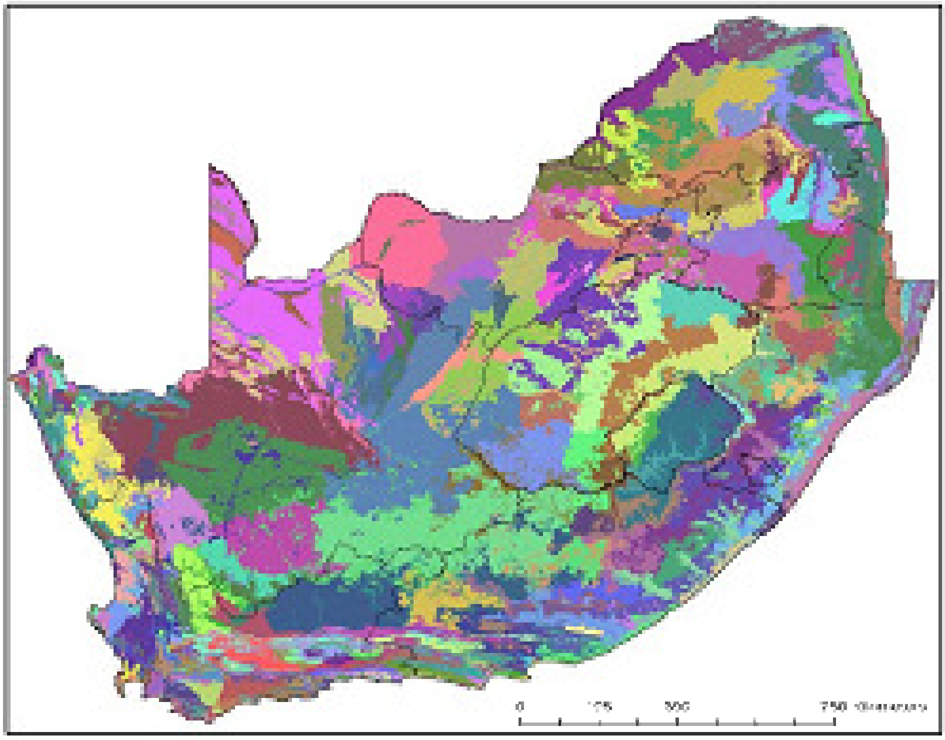


Figure 9: The 441 vegetation types of South Africa, Lesotho and Swaziland (Mucina & Rutherford 2006).

a) National Conservation Status (weighting 25%)

The individual vegetation type conservation status rating of VEGMAP is used in the identification of vegetation types at high risk of degradation. The IUCN Red List Categories and Criteria are used in this classification with the general aim of providing an explicit, objective framework for the classification of the broadest range of vegetation types according to their extinction risk (IUCN 2001). Within KNP, 20 vegetation types are represented of which three are CE (CR); five are VU and 12 are LT (LC) (Figure 7a). The three red list categories were subsequently scored according to their level of extinction risk *viz.* **CR (10)**; **VU (8)** and **LC (6)** (Figure 10).

b) Local Conservation Status (weighting 15%)

The VEGMAP Gap analysis provides a systematic approach for evaluating the protection afforded biodiversity in given areas. Using GIS, current "gaps" have been identified in biodiversity protection areas. These gaps may then be filled by the establishment of new protected areas or changes in land use practices (Scott *et al.* 1993). However, since the KNP is already a conservation area and predominantly within the well protected Savanna Biome, the VEGMAP Gap analysis (the range of values being between 80-500%) is not representative of local conservation targets (management guideline for placement of future developments within KNP). Therefore, the total area of each vegetation type was divided by the area it represents in KNP to calculate the percentage for which KNP is responsible for conserving nationally. The resulting scores are as follows: **80-100% (10)**; **50-80% (8)**; **20-50% (6)** and **0-20% (2)** (Table 3; Figure 10b).

c) Landtype Representation (weighting 50%)

Venter (1990) subdivided the KNP into 56 land types on the basis of similar geology, geomorphology, broad climatic attributes, soil type, vegetation type and landform features (Solomon *et al.* 2003). Each landtype was quantitatively described in terms of the dominant soil series and woody plant species associated with the different hillslope units, as well as landform characteristics such as relief, slope, slope length and stream frequency (Venter 1990). All 56 landtypes were allocated a score ranging from 1-10 to signify the level of representation within KNP, viz. each landtype's area was calculated and divided by the total area of the KNP, expressed as a percentage. The resultant scores are as follows: **0-1% (10)**; **1-2% (8)**; **2-4% (4)** and **>4% (1)** (Figure 10c).

d) Species Richness – Plants (weighting 5%)

SANBI's PRECIS database is an electronic system on Southern African plants for the provision of an efficient customer-driven information service and for producing computer-generated electronic and publishable products. The system contains species richness records from 736 424 specimens across 24 500 taxa available at a QDS resolution (PRECIS 2005). These data were spatially combined with the KNP landtypes, averaged, divided by the respective landtype area (ha) and divided by 0.01 to generate the number of species per km^2 per landtype. The values were then scored as follows: **0.04 – 0.5 (1)**; **1 (4); 2 (6)**; **3 (8)** and **4 - 8 (10)** (Figure 10d).

Table 3: Percentage of each vegetation type protected within KNP

VEGETATION TYPE	KNP (ha)	SA (ha)	% KNP
Sand Forest	1470	24259	6
Subtropical Salt Pans	215	3265	7
Limpopo Ridge Bushveld	41770	278637	15
Granite Lowveld Bushveld	319521	1980715	16
Delagoa Lowveld	48580	272091	18
Lowveld Riverine Forest	3333	15831	21
Phalaborwa Sandy Mopaneveld	39894	139457	29
Makuleke Sandy Bushveld	65288	207615	32
Lowveld Rugged Mopaneveld	105577	314976	34
Subtropical Alluvial Vegetation	23366	66513	35
Malelane Mountain Bushveld	45357	116696	39
Pretoriuskop Sour Bushveld	37244	94292	40
Tsende Mopaneveld	375432	614614	61
Basalt Sweet Arid Lowveld	225393	356898	63
Ironwood Dry Forest	4604	5913	78
Gabbro Grassy Bushveld	103410	107031	97
Nwambyia-Pumbe Sandy Bushveld	16768	16976	99
Northern Lebombo Bushveld	133234	133766	100
Cathedral Mopane Bushveld	27563	27563	100
Mopane Shrubveld	280429	280259	100

e) Species Richness – Mammals (weighting 5%)

In the absence of a complete set of spatial taxa data (missing birds, reptiles, amphibians, invertebrates), large herbivores are used as umbrella species (Solomon *et al*. 2003) in combination with unique habitat units (Landtypes). The Ecological Aerial Census (EAS) data from 1981-1993 was summarized by landtype to represent species richness. This standardised census method, aimed at repeatability, achieved total area coverage by flying parallel strips 800 m apart between 08h00 - 12h00 over $3^1/_2$ months (Whyte & Joubert 1988). Population numbers as well as the distribution of each species were recorded with a latitude and longitude position. Preliminary results indicated that a simple species-richness calculation was not descriptive enough at a landtype scale (since there are only 22 species counted during the census, many of which are wide-ranging). An abundance and distribution ranked species richness index was therefore developed as follows:

$$\mathbf{S} = \frac{\mathbf{A[\sum(B(56\text{-}C)+1))]}}{\mathbf{D/0.01}}$$

A: Number of **species** per landtype

B: **Abundance** of species per landtype

C: **Distribution** range of species *viz.* number of landtypes each species is represented /present in.

D: Area (ha) of landtype

In other words, the number of species per landtype are multiplied by the sum of the number of individuals per species, multiplied by the appropriate distribution weighting per species, divided by the area of each landtype, divided by 0.01 to convert values to species richness per km^2. Resulting values are scored as follows: **0.1-0.5 (2); 0.5-1 (4); 1-3 (6); 3-5(8)** and **4-5 (10)** (Figure 10e).

f) Topographic Variability

The Topographic Position Index (TPI) forms the basis of this layer as it represents the difference between the elevation value of a cell and the average elevation of the neighbourhood around that cell (Jenness 2005). The degree to which cells are higher or lower than their respective slope was used in a slope position classification, while scale was determined by an input neighbourhood of 2 km. TPI values which are significantly higher than the surrounding neighbourhood, are likely to be at or near the top of a hill or ridge. Low values suggest the cell is at or near the bottom of a valley, while values near zero indicate either a flat area or a mid-slope area. A quantitative method is required to compare different landtypes and relate landscape patterns to ecological function with the use of indexes of landscape richness, evenness and patchiness (Romme 1982). In this case the topographic relative richness of each landtype was calculated using the following formula adapted from Romme (1982).

$$\mathbf{R} = \frac{\mathbf{T}}{\mathbf{TMAX}} \times \mathbf{100}$$

R: Topographic Relative Richness
T: Number of different topographic position types present
TMAX: Maximum number of topographic position types possible

Resulting values are scored as follows: **1-5 (2); 5-10 (4); 10-15 (6); 15-20 (8)** and **20-27 (10)** (Figure 10f).

3.2.3 BIODIVERSITY SENSITIVITY-VALUE

In light of the fact that no single formula exists on how to combine multiple biodiversity layers to identify biodiversity priority areas (Rouget *et al.* 2004), a number of mathematic overlay operations were tested using the sensitivity and value datasets. A meaningful formula will present a realistic overall representation of areas with both high sensitivity and value once multiple layers are combined (Figure 11).

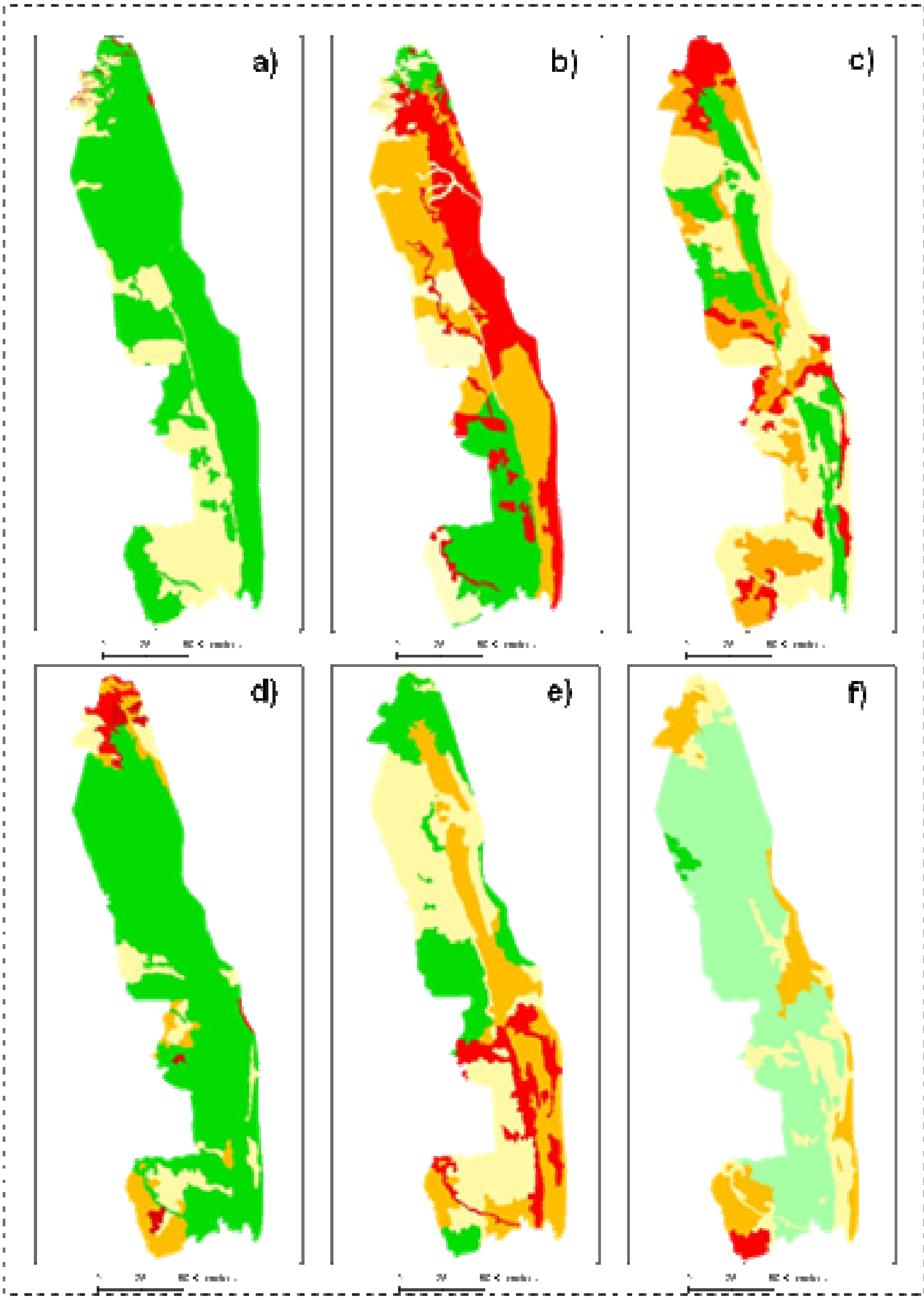


Figure 10: Individual layers of the Biodiversity Value analysis. a) National Conservation Status; b) Local Conservation Status; c) Landtype Representation; d) Plants Species Richness; e) Mammal Species Richness; f) Topographic Variability. Scores range from green-red: green being least valuable (e.g. score 1) and red being most valuable (e.g. score 10).

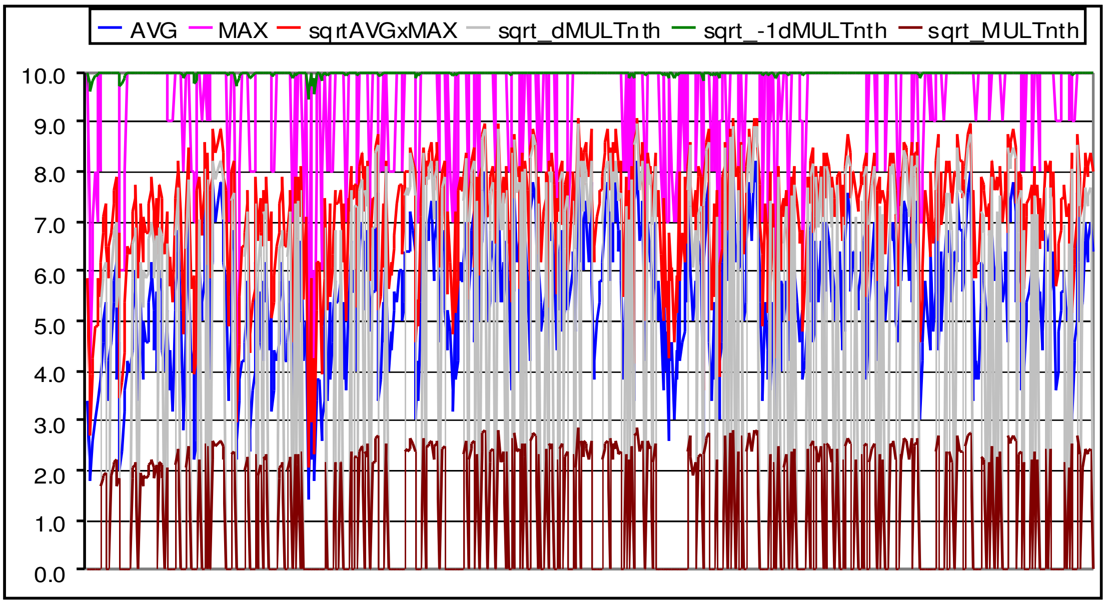


Figure 11: Test results of the different mathematical overlay operations used to combine layers. AVG: Average –> Max values are "diluted"; MAX: Maximum –> Max values –> accumulated; sqrtAVGxMAX: Average adjusted by the Max –> appears to compensates for AVG max dilution; Others: different multiplicative arching overlays –> high / low / variable

☒ **Average (AVG)**: A simple average [=avg()] calculation causes the maximum values to become "diluted". Since the research is aimed at biodiversity conservation we cannot ignore these highly sensitive/valuable features or areas.

☒ **Maximum (MAX)**: Using the MAX statistic however, inflates these scores through the process of geographical accumulation of higher values.

☒ **Others**: I experimented with other formulas used in the field of forestry e.g. variation of multiplicative arching (Biggs 2005 *pers. comm.*), however these results also appeared too high, low or variable across space.

☑ **[sqrt (AVG x MAX)]**: A combination of the above formulae, [sqrt (AVG x MAX)] compensates for AVG dilution and the MAX inflation by adjusting the average statistic by the maximum.

As a result, in an effort to highlight highly sensitive and/or valuable areas, without unrealistically inflating their geographic representation, a function of the square-root of the product of the mean and maximum (**[sqrt (AVG x MAX)]**) is used to combine all biodiversity sensitivity and biodiversity value layers.

3.3 **RESULTS**

Using the formula, [sqrt (AVG x MAX)] to combine biodiversity sensitivity (Figure 8) and value (Figure 10) layers, priority protection areas are defined according to degrees of conservation value and sensitivity. Individual formulae are presented below using math algebra and simple mathematical syntax:

Biodiversity Sensitivity (BIOSE): Is measured by combining the sensitivity scores of steep slopes (topography); drainage systems (hydrology); soils; threatened species (plants and mammals) and vegetation resilience to physical disturbance as follows:

Formula = {(sqrt(avg[topos] x max[topos])) + (sqrt(avg[hydro] x max[hydro])) + (sqrt(avg[soil] x max[soil])) + (sqrt(avg[veg] x max[veg])) + (sqrt(avg[spp] x max[spp]))}

$$\textbf{BIOSE} = \sum_{i=1}^{n} \mathbf{X_i} \{\sqrt{a} \times \max(a); \sqrt{b} \times \max(b); \sqrt{c} \times \max(c); \sqrt{d} \times \max(d); \sqrt{g} \times \max(g)\}$$

a) Topographic Sensitivity

b) Hydrological Sensitivity

c) Soil Sensitivity

d) Vegetation Sensitivity

g) Special Species – Plants and Mammals

Resulting tolerance or vulnerability levels are classified from 1-10 (1 being least sensitive [green] 10 being most sensitive [red]). As a result, 45% of KNP's biodiversity estate has low sensitivity (**1-3**); 35% has medium (**4-5**) and 20% high (**6-8**) (Figure 12).

Biodiversity Value (BIOVA): Is measured by combining the value scores of habitat (national and local conservation status, and local landtype representivity; species richness) and special habitats (topographic variability) of KNP's biodiversity estate:

Formula = {(sqrt(avg[habitat^note] x max[habitat*])) + (sqrt(avg[topov] x max[topov]))} **Note: Habitat = (([Status] * 0.25) + ([Gap] * 0.15) + ([Knp_area] * 0.50) + ([Precis] * 0.05) + ([Eas] * 0.05))**

$$\textbf{BIOVA} = \sum_{i=1}^{n} \mathbf{X_i} \{\sqrt{g} \times \max(g); \sqrt{f} \times \max(f)\}$$

f) Topographic Variability

g) Habitat Value (National Conservation Status; Local Conservation Status; Landtype Representation; Species Richness – Plants and Mammals; Species Richness – Mammals)

$$\text{HABITAT VALUE (g)} = \sum_{i=1}^{n} x_i \{(a \times 0.25); (b \times 0.15); (c \times 0.50); (d \times 0.05); (e \times 0.05)\}$$

a) National Conservation Status
b) Local Conservation Status
c) Landtype Representation
d) Species Richness - Plants
e) Species Richness – Mammals

Resulting levels of importance or worth are classified from 1-10 (1 being least valuable [green] 10 being most valuable [red]). As a result, 51% of KNP has a low biodiversity value (**3-4**); 39% has medium value (**5-6**) and 10% high value (**7-9**) relative to National and local biodiversity estates (Figure 13).

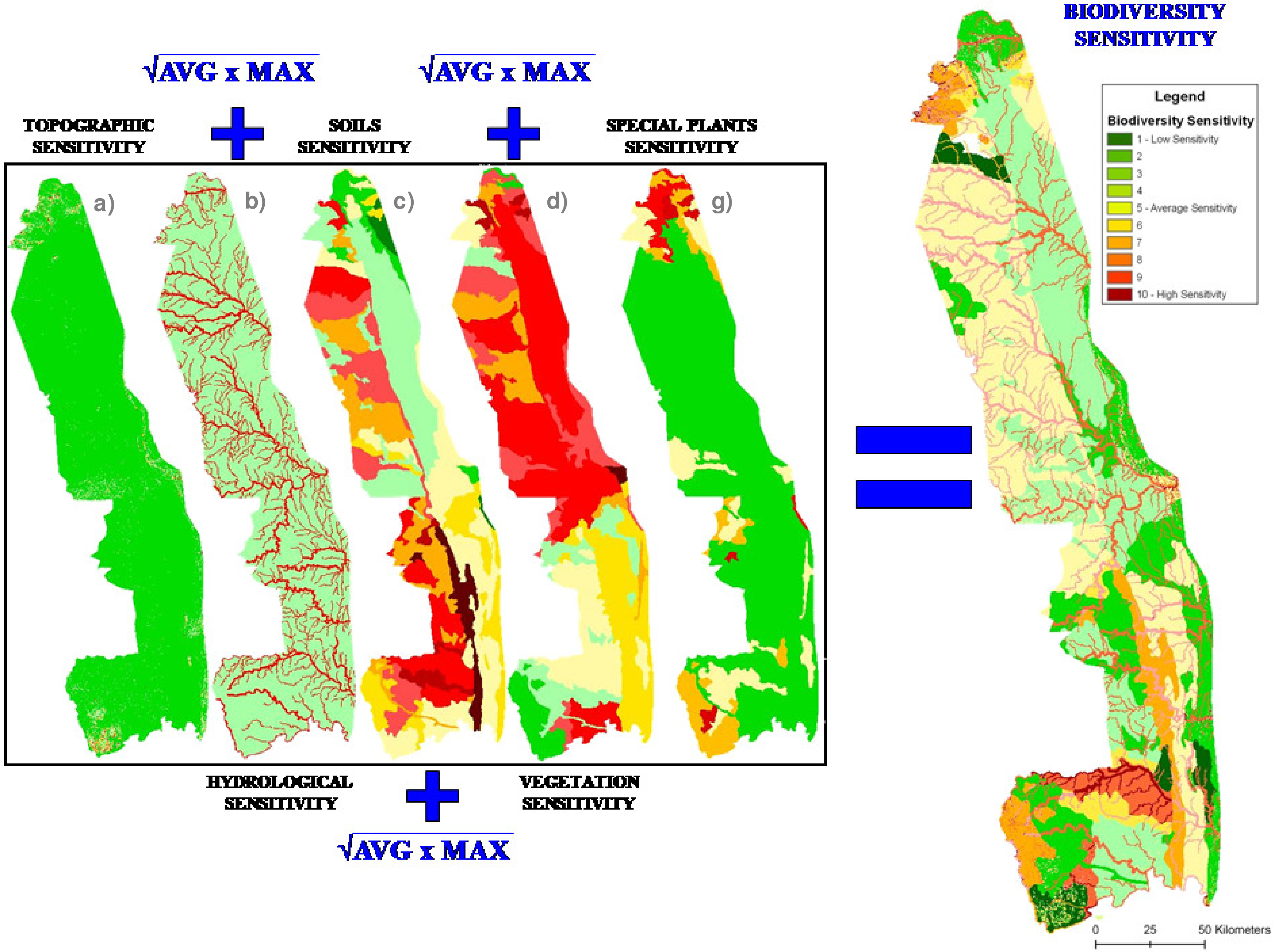
Biodiversity Sensitivity-Value (BIOSEVA): Is calculated by combining the collective biodiversity sensitivity (Figure 12) and value (Figure 13) scores of the KNP as follows (Figure 14):

Formula = {(sqrt(avg[biose] x max[biose])) + (sqrt(avg[biova] x max[biova]))}

$$\text{BIOSEVA} = \sum_{i=1}^{n} x_i \{\sqrt{h} \times \max(h); \sqrt{k} \times \max(k)\}$$

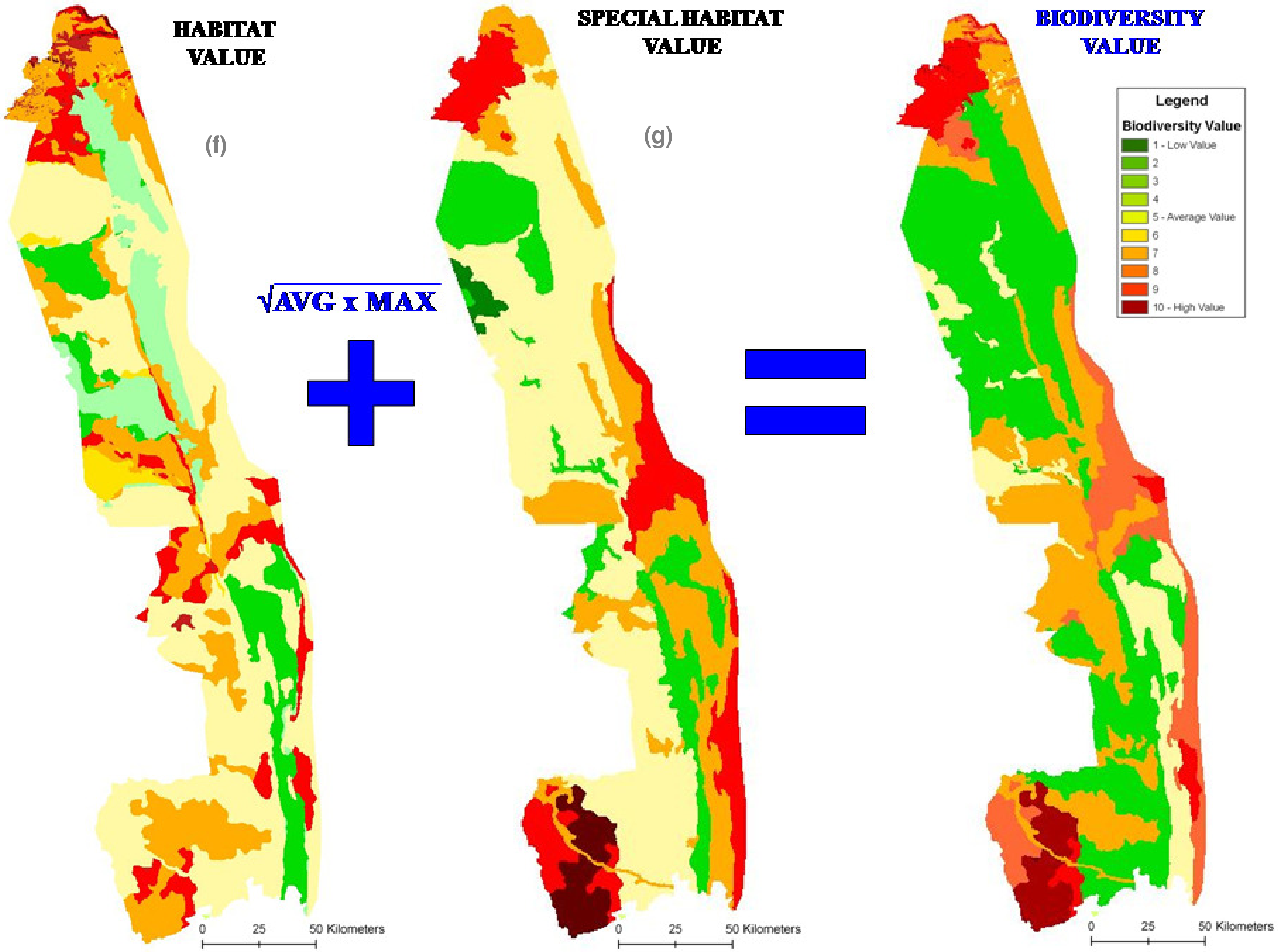
h) Biodiversity Sensitivity
k) Biodiversity Value

Resulting levels of biodiversity sensitivity and value are classified from 1-10 (1 being least valuable [green] 10 being most valuable [red]). As a result, 57% of KNP has a low biodiversity sensitivity-value (**3-4**); 32% has medium sensitivity-value (**5-6**) and 11% high sensitivity-value (**7-9**) (Figure 15). Protection efforts must therefore be prioritised within these areas of high biodiversity sensitivity-value (11%) due to their vulnerability to disturbance and National biodiversity significance.



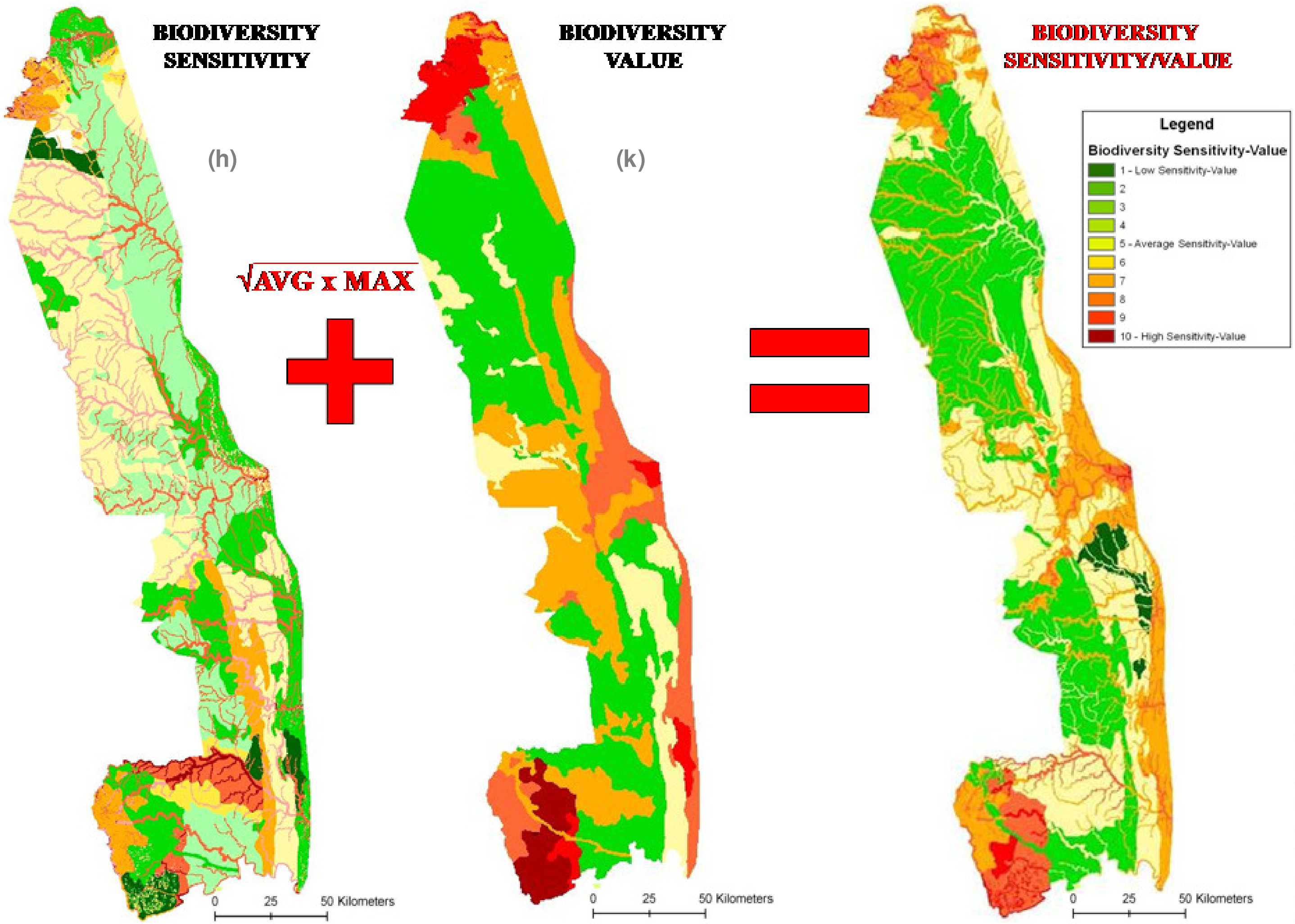
BIOVAL	AREA (HA)	%KNP	JENKS-BIN %
3.46	291999.56	15.40	51
4.00	463756.94	24.46	
4.47	10036.74	0.53	
4.90	164468.71	8.67	
5.00	33312.00	1.76	
5.48	371532.95	19.59	39
5.92	14617.56	0.77	
6.00	74443.00	3.93	
6.32	2.68	0.00	
6.48	45684.28	2.41	
6.93	181165.18	9.55	
7.00	54314.37	2.86	
7.48	47424.64	2.50	10
8.00	61121.53	3.22	
8.49	3243.27	0.17	
8.94	67671.71	3.57	
9.00	1577.23	0.08	
9.49	9869.37	0.52	

Figure 12: The combination of individual Biodiversity Sensitivity layers to form an overall Biodiversity Sensitivity layer.



BIOVAL	AREA (HA)	%	JENKS%
3.46	291999.56	15.40	
4.00	463756.94	24.46	40.38
4.47	10036.74	0.53	
4.90	164468.71	8.67	
5.00	33312.00	1.76	30.02
5.48	371532.95	19.59	
5.92	14617.56	0.77	
6.00	74443.00	3.93	4.70
6.32	2.68	0.00	
6.48	45684.28	2.41	
6.93	181165.18	9.55	14.83
7.00	54314.37	2.86	
7.48	47424.64	2.50	5.72
8.00	61121.53	3.22	
8.49	3243.27	0.17	
8.94	67671.71	3.57	4.34
9.00	1577.23	0.08	
9.49	9869.37	0.52	

Figure 13: The combination of Biodiversity Value layers to form an overall Biodiversity Value layer.



BIOSEVA	AREA (HA)	%KNP	JENKS-BIN %
3.00	32844.91	1.73	
4.00	250151.69	13.19	
4.47	428916.62	22.62	57
4.90	9271.79	0.49	
5.00	367467.97	19.38	
5.29	5647.24	0.30	
5.48	161681.41	8.53	
5.92	178364.62	9.41	
6.00	85991.90	4.53	32
6.32	11686.74	0.62	
6.48	151822.87	8.01	
6.71	12087.88	0.64	
6.93	31510.64	1.66	
7.00	61209.74	3.23	
7.35	42331.31	2.23	
7.48	32312.08	1.70	11
7.94	18059.86	0.95	
8.00	8234.55	0.43	
8.49	6575.41	0.35	
9.00	72.49	0.00	

Figure 14: The combination of (h) Biodiversity Sensitivity and (k) Value layers to form an overall Biodiversity Sensitivity-Value layer.

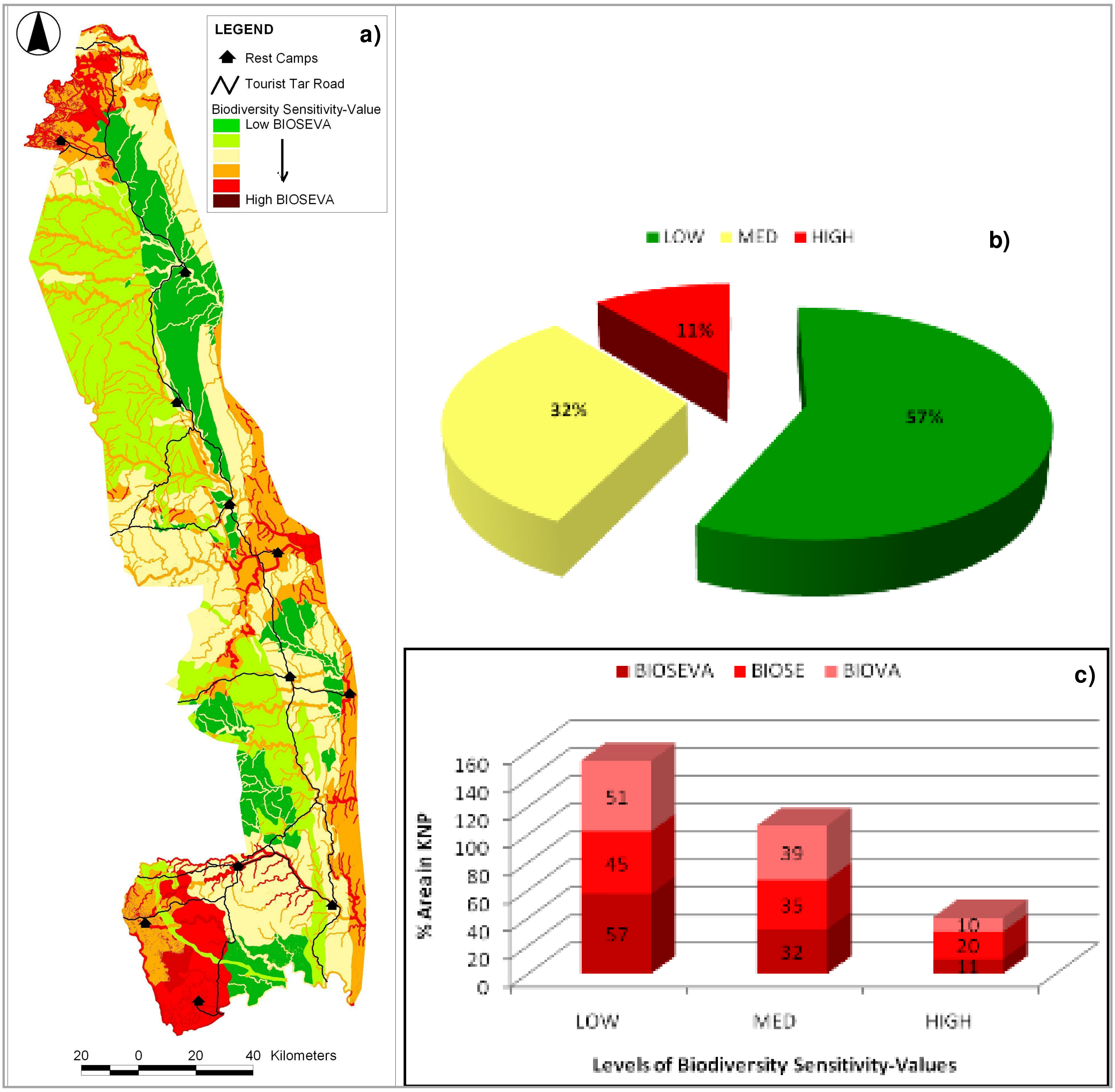


Figure 15: The final combination of all layers, presented as six, natural-break intervals of Biodiversity Sensitivity-Value in a map (a); a pie chart indicating from low biodiversity sensitivity values (57%) to high biodiversity and sensitivity values (11%; (b); and a bar chart indicating the individual sensitivity and value scores which form the overall Biodiversity Sensitivity-Value grid.

In response, the resulting biodiversity sensitivity-value classification will help stratify spatial conservation planning for local tourism, management and research strategies. This is especially important for KNP in light of competing tourism needs and elephant impacts. In other words, highly sensitive and valuable areas subjected to intense land use and/or high levels of elephant impact, may be under threat of degradation. Chapter four will discuss how the biodiversity sensitivity-value stratification (Figure 15) may be applied to assess impending impact of different land use practises and varied degrees of elephant disturbance.

~ CHAPTER 4 ~

MANAGEMENT APPLICATIONS

4.1. INTRODUCTION

Having prioritised protection zones within KNP according to inherent landscape biodiversity sensitivity and value, it is important that these results be used to inform management decisions. Knowing what to conserve and where, is the first step towards effective protected area management (Tucker *et al.* 2005). How to conserve on the other hand is complicated by the ubiquitous nature of disturbance. Management decisions may either mitigate or aggravate biodiversity consequences. If well managed, protected areas should aim to simulate, where necessary, those natural processes which are lost to these isolated environments as a result of fencing, external anthropogenic pressures, local impacts, or others. However, the challenge lies in i) predicting state change; ii) separating desirable (natural) and undesirable (unnatural) change iii) knowing when management intervention is necessary to avoid an unnatural state change; iv) identifying the mechanisms responsible for this unnatural state change; v) identifying the absent or damaged ecosystem process; vi) deciding what management intervention will simulate this specific ecosystem process; vii) and knowing how to adapt management practices appropriately (Figure 16).

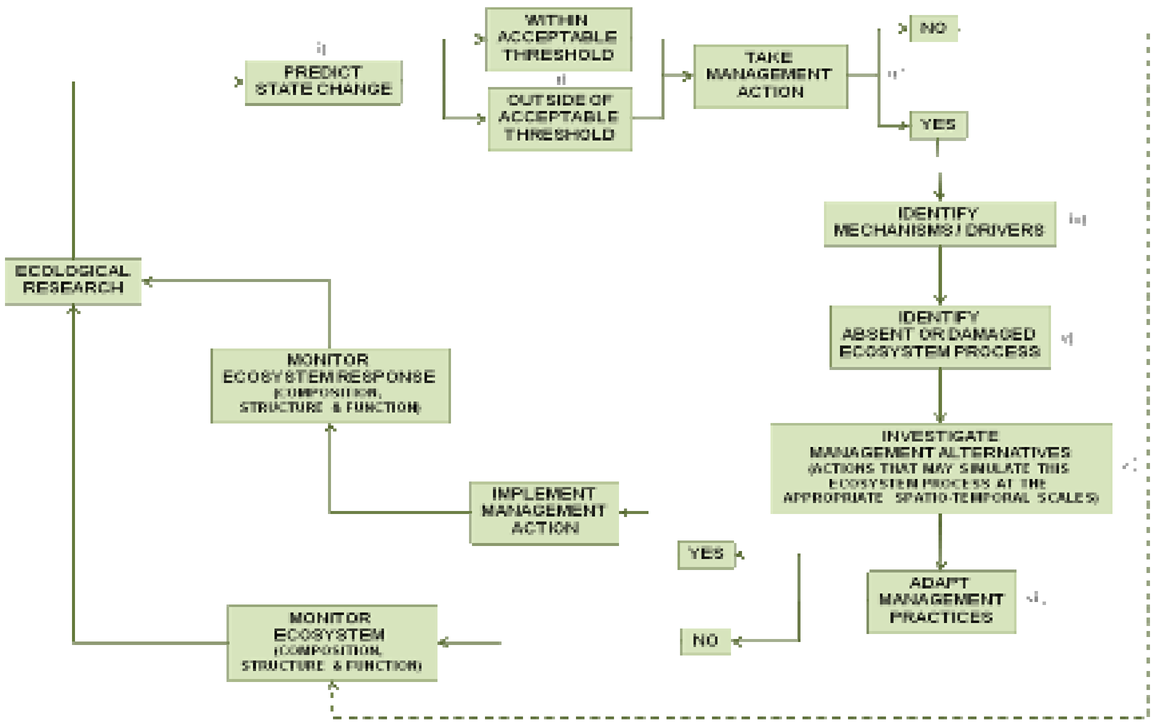


Figure 16: An adaptive management approach for conservation decision makers: i) First predict a state change; ii) then separate desirable (natural) and undesirable (unnatural) change); and iii) decide if management intervention is necessary; then iv) identify the mechanisms responsible; and v) identify the absent or damaged ecosystem process that must be restored by management intervention; then vi) decide what management intervention will best simulate this specific ecosystem process; and finally vii) adapt management practices appropriately (Figure 16).

The selection of appropriate management actions related to human activities and elephant densities are of particular concern to today's conservation biologists (Zeng *et al.* 2005; Scholes & Mennell 2009). Often seen as conflicting forms of land use, conservation and tourism form an unusual alliance. Conservation is intended to minimize the threat of human disturbance and preserve ecological integrity. Tourism on the other hand acts as an indirect amplifier of this threat by promoting an increase in visitor numbers and human activities within conservation areas (Wall 1993). Realistically however, most protected areas would not survive without the revenue generated by tourism and tourism activities. As a result, tourism can create both threats and opportunities for protected areas, depending on the way it is planned, implemented and managed in relation to biodiversity sensitivity and value (Goodwin 1996; de los Monteros 2002).

Density-driven elephant impact is another major environmental concern for most southern African protected areas. Elephant management issues are, and will remain, contentious. Moreover, these issues are subject to intense public and political scrutiny, driven by emotive responses and personal values (Scholes & Mennell 2009). Under natural conditions resource availability and thus distribution is varied across the landscape thereby influencing elephant movement, numbers, intensity of landscape use and ultimately the level of elephant impact (van Aarde *et al.* 2009). However, in protecting conservation areas, elephants (along with other species) are fenced in and prevented from moving seasonally in response to natural pressures (Mabunda *et al.* 2003). This may have undesirable outcomes for other biodiversity values. In response SANParks has suggested that by restoring or mimicking natural resource distribution (e.g. water), the spatial and temporal distribution of elephant impact may be normalized (KNP 2009). However to achieve this, the influence of naturally occurring high and low elephant densities (i.e. elephant "hotspots") on biodiversity should underpin the identification of zones of potential management intervention. As a result, the classification of KNP into zones of high to low biodiversity sensitivity-value may help managers spatially evaluate the risks of human and/or elephant impact and prioritise management actions. For example, highly sensitive or valuable biodiversity areas, existing within pristine wilderness and which are subjected to potentially high levels of elephant impact, should be flagged as areas of management concern.

4.2. WILDERNESS (OR USE) ZONATION

4.2.1 BACKGROUND

The KNP Recreational Opportunity Zonation (ROZ) plan of 1997 (revised in 2003), was designed to identify a disturbance gradient in the landscape as a result of human activities as a spatial plan for tourism development (Freitag-Ronaldson *et al.* 2003). Unfortunately due to the prevailing viewpoint of the time, biodiversity maintenance was not the driving principle behind this zonation, because it was assumed that ecotourism use did not pose a threat to ecosystem maintenance (Freitag-Ronaldson *et al.* 2003).

Today, the environmental impact caused directly or directly by tourism activities have become a growing concern (Hunter & Green 1995; UNEP 2001), although the appropriate or maximum acceptable level of human activity for protected areas is imprecise (Zeng *et al.* 2005). During the development of the ROZ, the intangible qualities of wilderness were recognized through the express needs that park visitors have to seek solitude, remoteness, and peace in addition to simply viewing wildlife (Freitag-Ronaldson *et al.* 2003). In light of receding government subsidies to conservation agencies, the coexistence of conservation and tourism is an increasingly difficult balancing act. As pressures to generate revenue increase, tourism demands on wilderness areas increase and with it the threat of human disturbance. Adequate protection of the last surviving tracts of wilderness, inside or outside protected areas, should therefore be globally prioritised (Alexander 2008). This is especially important as wilderness areas serve as a benchmark against which to measure environmental health (Sinclair 1998). They also provide ecosystem services such as water provision, nitrogen fixation, pollination, and carbon sequestration and are often strongholds for biodiversity (Mittermeier *et al.* 2003).

The National Environmental Management: Protected Areas (NEM:PA; Act No. 57 of 2003) and Biodiversity Acts (NEM:BA; Act No. 10 of 2004), define Wilderness Areas as *"an area designated in terms of Section 22 or 26 for the purpose of retaining an intrinsically wild appearance and character or capable of being restored to such and which is undeveloped and roadless, without permanent improvements or human habitation"* (Act No. 57 of 2003). Similarly, the US Wilderness Act (Public Law 88- 577 of 1967) defines Wilderness as being *"designated in terms of section 22 or 26 for the purpose of retaining an intrinsically wild appearance and character or capable of being restored to such and which is undeveloped and roadless, without permanent improvements or human habitation; a place where the earth and its community of life are untrammelled by man, where man himself is a visitor who does not remain.*" In another interpretation, wilderness is *"an area little affected by current civilization where nature and natural processes are in charge and where people can isolate themselves from other people"* (Fenton 1996, p. 17). While according to the IUCN, wildernesses are: "*Large areas of unmodified or slightly modified land, and/or sea, retaining its natural character and influence, without permanent or significant habitation, which is protected and managed so as to preserve its natural condition"* and *"Ecosystems where since the industrial revolution (1750) human impact (a) has been no greater than that of any other native species, and (b) has not affected the ecosystem's structure. Climate change is excluded from this definition"* (Chape *et al.* 2003).

The profundity of ethical issues and personal value judgements surrounding the term 'wilderness' makes one universally acceptable definition impossible (Alexander 2008). The term was derived in the 19th century from an interest in preserving wild places for their intrinsic, aesthetical and spiritual value (Kalamandeen & Gillson 2007), largely in response to expanding habitat transformation and urbanisation. However, critics believe the concept to be flawed, that humans are not separate from nature, and that ecosystems artificially fossilised in an intermediate state will deteriorate *(*Aplet *et al.* 2000; Alexander 2008). In 1933, Shelford wrote: "*Primitive man, who could not remove the forest or exterminate the animals, is probably properly called a part of nature*" (Shelford 1933, p. 241).

It has since become accepted that early humans were a natural component of the past wilderness although they were undoubtedly modifiers of their own surrounding terrestrial ecosystems (Alexander 2008; Lockwood 2009). However, we must be mindful not to confuse today's modern man with yesterday's primitive man, who inhabited the earth in small numbers, were often nomadic and sustainably utilized natural resources rather than over-exploiting them. The current high and rapidly growing human densities of today are placing increasing pressure on conservation in general, and shrinking land potentially available for wilderness conservation (Harcourt *et al*. 2001; Carver & Fritz 2001; Mendel 2002; Alexander 2008). Therefore, effective protected area management should aim to balance management actions and tourism activities with the protection of biodiversity through spatial planning and zonation (Mazzotti 2001).

4.2.2 ZONATION

A park zonation plan is an important conflict management template for the development of a spatial activities framework to guide and co-ordinate conservation, tourism and visitor experience initiatives in and around protected areas (KNP 2009). The zonation process itself is essentially an expert based GIS approach aimed at land use planning for conservation activity zoning within protected areas (Holness & Skowno 2008). Areas are identified along a spectrum of environmental modification, from pristine wilderness to completely urbanised environments (Hendee *et al.* 1990; Carver & Fritz 2001). Although it is difficult to identify the exact point along this gradient at which wilderness is separated from other land uses (Carver & Fritz 2001), a number of proxies have been identified. These include 1) habitat or landscape diversity, 2) rarity, 3) naturalness (essentially the effects of human influence or disturbance), 4) area of land (minimum sizes ranging from 100 000 km^2; 10 000 km^2; 250 km^2; 20 km^2 to 5 km^2), and 5) threat of human interference (proximity to human influence or presence) (Margules & Usher 1981; Lesslie & Maslen 1995; Carver & Fritz 2001; UNEP 2001; Carver *et al.* 2002; Mendel 2002; Sanderson *et al.* 2002; Howard & Howard 2005; Burgess *et al.* 2006).

According to the Global Methodology for Mapping Human Impacts on the Biosphere (GLOBIO; UNEP 2001), environmental modification or disturbance is directly related to the distance from human infrastructure (i.e. settlements; mechanised access; transmission lines). Therefore, in order to measure this influence, proximity to human disturbance (or infrastructure) is spatially quantified using a GIS. In the case of the KNP, where there is a century long legacy of development, it is logical that existing wilderness areas are equal to the sum of land, minus the present and neighbouring influence of existing human activities and infrastructure (i.e. KNP's current human footprint). In order to pre-classify the wilderness continuum found in the KNP, the existing human footprint is buffered by pre-defined distances relative to the level of disturbance associated with each infrastructure type and thereby ensuring the integrity of surrounding natural areas (Table 4).

Table 4: Classification and associated buffer distances of existing infrastructure.

CATEGORY	GENERAL INFO	BUFFER DISTANCE
Tourist Main	Tarred	2 km
Tourist Secondary	Gravel	1 km
Management Primary	Gravel	500 m
Tourist 4x4	Dirt track	200 m
Management Secondary	Dirt track	100 m
Powerline	Dirt track	1 km
KNP Boundary	Boundary fence	500 m

Although seemingly innocuous, roads in particular have been well documented to contribute to habitat fragmentation (Hafla *et al.* 2008). Roads may physically occupy a small fraction of the landscape in terms of surface area but their influence frequently extends far beyond their immediate boundaries (Hafla *et al.* 2008). A recent assessment of road impacts (Fahrig & Rytwinski 2009) found that the number of documented cases of negative environmental effects outnumbered the positive by a factor of five. In assessing the extent of road impacts for spatial planning different authors have attempted different approaches. For example, Aplet *et al.* (2000) applied a variable road system buffer with 5-point Wilderness Quality Scores (WQS; 1 being poor and 5 being good) i.e. 2 km (WSQ=1); 2-5 km (WSQ=2); 5-10 km (WSQ=3); 10-25 km (WSQ=4); and greater than 25 km (WSQ=5). The South African National Protected Area Expansion Strategy buffered all major roads by a standard 50 m (Jackelman *et al.* 2007), while the South African National Spatial Biodiversity Assessment varied buffer distances (10 – 30 m) according to the road types (Rouget *et al.* 2004). Considering that maximum erosion impacts occur between 0.5–1 km from roads, and 80% of impacts occur up to 2 km from roads, the following classification was applied for buffering tourist access roads: all adventure roads (suited to 4x4 or SUV vehicles) - 200 m, well maintained gravel roads - 1 km, and tarmac roads - 2 km (Shi *et al.* 2007; Lagabrielle *et al.* 2009). Moreover, tarmac roads are specifically buffered by 2 km as they are also often the nexus point of human activity, for example tourist camps and staff villages. As a result, the 2 km buffer distance should compensate for noise pollution and general density-dependant human influence associated with these hotspots of human activity. Primary and secondary management roads are buffered by 500 m and 100 m respectively, while all overhead power lines are buffered by 1 km to account for the two-track service road and additional visual disturbance. Finally, external influences are controlled by a 500 m buffer around the park boundary as the first line of defence to safeguard valuable and sensitive biodiversity features. After removing all buffered road areas, the KNP is fragmented into 162 patches with an average patch size of 15 200 ha, of which five are < 1000 ha, 88 are between 1000 ha and 5000 ha, 30 are between 5000 ha - 10 000 ha and 39 patches are > 10 000 ha (Figure 17).

Each fragment is then zoned according to a visitor use category, developed for SANParks to reflect a gradation of wilderness quality: Wilderness; Remote; Primitive; Low Intensity Leisure and High Intensity Leisure (KNP 2009; Table 5). Ultimately 62 wilderness areas were demarcated with a mean size of 13 642 ha (min 3 361 ha; max 33 969 ha; sum 84 5791; standard deviation 7 207 ha).

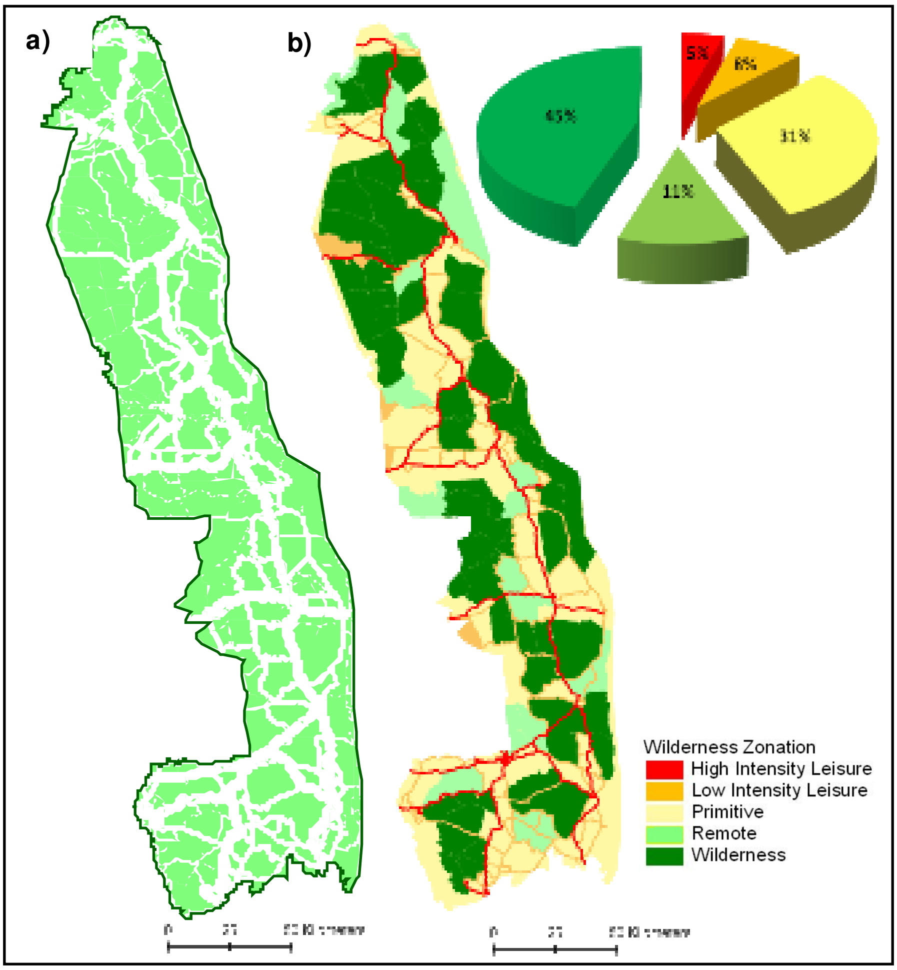


Figure 17: Wilderness zonation process for Kruger National Park, a) Buffered road areas removed from the protected area landscape, resulting in 162 patches with an average patch size of 15 200 ha. Five of the patches are < 1000 ha, 88 are between 1000 ha and 5000 ha, 30 are between 5000 ha - 10 000 ha and 39 patches are > 10 000 ha; and b) Final Wilderness Zonation illustrating the spectrum of wilderness qualities from pristine wilderness; remote; primitive; low Intensity leisure to high intensity leisure (tourist rest camps) areas.

Table 5: Zonation categories and the associated infrastructure that may be developed in each (adapted from KNP 2009)

USE ZONE	TYPE OF ACCESS	INFRASTRUCTURE
WILDERNESS	Only pedestrian. By booking only. One group at a time.	Heritage sites only.
REMOTE	Only pedestrian. By booking only. One group at a time.	Heritage sites; redundant infrastructure ear-marked for rehabilitation; management tracks.
PRIMITIVE	Controlled access. Self drive (mostly by 4x4) or on foot. Access routes restricted to visitors with bookings for facilities.	Ranger houses and field staff accommodation, permanent pickets, staff facilities, administration offices, rifle ranges, airstrips; small bush camp type facilities or controlled access concession sites.
LOW INTENSITY LEISURE	Self drive. Roads mostly accessible to sedan vehicles. Some parks may exclude safari vehicles and busses from this zone.	Larger basic camp grounds & small camps with fully equipped self contained, self catering units. Camps without additional amenities such as shops, petrol station and restaurants. Picnic sites with toilets, information centres; gravel roads.
HIGH INTENSITY LEISURE	Self drive. Accessible to busses and safari vehicles.	Large camps with fully equipped self contained units. Interpretative centres, restaurant, shops and petrol; tarred roads.

4.3. ELEPHANT IMPACT ZONATION

4.3.1 BACKGROUND

Elephant numbers in the KNP have reached an all time high over the last century (Whyte *et al.* 1999), with a continuous growth rate of ~ 7% per annum. As a consequence of these rapidly rising numbers of elephants in protected areas in southern Africa in general, the potential threat to the ecosystems of these areas, and the people living adjacent to and dependant on these areas has also increased (Scholes & Mennell 2009). As a result, an assessment of elephant ecology, dynamics, populations, impacts and other factors was necessary to determine potential population management options. In response, Scholes & Mennell (2009) have published *"the most systematic and comprehensive review of savanna elephant populations and factors relevant to managing them to date.* A comprehensive discussion is available in du Toit *et al.* (2003), Joubert (2007) and Scholes and Mennell (2009), and as a result, I will not go into detail about the complexities of this highly controversial topic.

The challenge for all conservation managers lies in accommodating the myriad of potential natural state changes of even healthy ecosystems into a single management plan (Sinclair 1998). Modern conservation approaches attempt to manage for heterogeneity and endeavour to compensate for the heterogeneous patchwork of species or ecosystem states (Sinclair 1998).

To complicate things further, disturbance is not always undesirable, as two types of disturbance exist: (1) the relatively predictable, small-scale, often biotic disturbances that may promote diversity by generating local heterogeneity; and (2) large-scale disturbances which may both reduce diversity (Wood 2001), or promote diversity by creating a heterogeneous bio-physical template (Parsons *et al*. 2005). All types of disturbance are vital for the maintenance of overall diversity and should be recognised and accommodated by managers who should strive to prevent biodiversity loss (Wood 2001). How much and what type of disturbance is too much, is difficult to answer. For example, how many elephants are too many and what type of elephant impact is detrimental to ecosystem integrity? The remainder of this chapter will focus on the delineation of elephant "hotspots" (areas with consistently high densities of elephant) as a means of estimating spatial disturbance patterns in order to determine priority areas for conservation at different scales (Ceballos & Ehrlich 2006). Bearing in mind that the delineation of elephant hotspots does not on its own constitute high or low impact, this needs to be considered in combination with other biophysical factors, including biodiversity value and sensitivity.

4.3.2 ZONATION

The identification of naturally occurring high and low elephant density areas or hotspots should form an important component in the development of the KNP Elephant Impact Zones. In combination with BIOSEVA, the dry season distribution and density patterns of elephants was investigated using annual aerial survey data. These data have been collected annually between June and August since 1985 and include information pertaining to population numbers and herd structures. I therefore derived annual density and distribution patterns of elephants using census data from 1985 to 2009 (n = 25) plotted as an event theme directly from an access database using an SQL connection into ArcView (ESRI 2000). A kernel density calculation was computed annually for the period 1985 - 2009. The kernel method calculates the density for each cell by summing the population value, distributing this value out from each point within a user defined search radius (10 km) comparable to the average daily area typically used by elephants during the dry season (Young *et al.* 2009). Persistent elephant distribution and density hotspots were identified by averaging these annual grids over a 25-year period and reclassifying values into six classes above and below the mean elephant population density at intervals of one standard deviation (Figure 18; Applegate 1995). Although only dry season accounts of elephant distribution and density patterns is possible from available data, resource limits are narrower and therefore the risk of undesirable levels of disturbance is higher during this period. In identifying these areas of consistently high or low elephant density over the last 25-years, in combination with the location of valuable and sensitive biodiversity (BIOSEVA) areas, may help focus elephant management spatial planning in the KNP. For example, insight into current density patterns will aid decision makers identify and induce spatial and temporal variation in elephant landscape use by restoring the spatial limitations of the landscape (Leggett 2006; Valeix *et al.* 2007). The theory behind this being that elephant impact (intensity of landscape use) may be controlled through the manipulation of elephant movement patterns by varying resource limitations spatially and temporally.

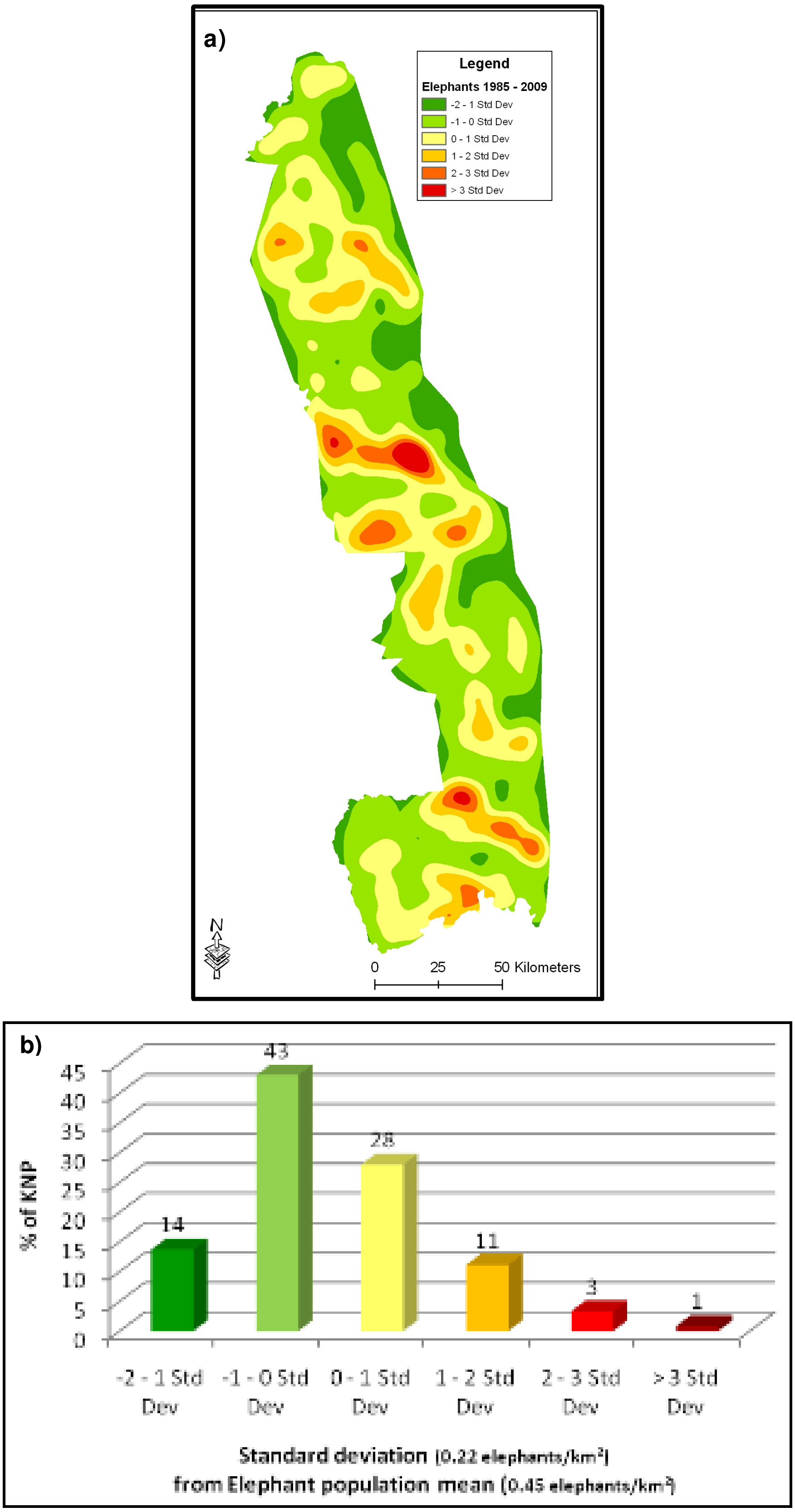


Figure 18: Distribution and density patterns of elephants over 25-years of dry-season aerial surveys (1985-2009; (a), classified according to six intervals of standard deviations from the mean elephant density (b).

4.4. RESULTS

Effective protected area management should aim to minimize undesirable impacts on biodiversity through management practices specifically tailored to individual landscape characteristics (Sala *et al.* 2000). These include for example, land use and elephant distribution-density characteristics. As a result, all wilderness areas and elephant hotspots were extracted from the Wilderness Zonation and Elephant Distribution-Density grids respectively. Wilderness areas constituted all patches larger than 3000 ha once the human footprint area had been removed. Elephant density zones are classified as all areas with densities greater than one standard deviation from the mean. Subsequently, individual wilderness and elephant grids are combined using an arithmetic sum overlay operation in the raster calculator, with high values of BIOSEVA as follows (Figure 20):

(([a] >= 4) + ([b] == 1) + ([c] >= 4))

a) Jenks classification of Biodiversity Sensitivity-Value (BIOSEVA)

b) Wilderness Zonation.Zone == "Wilderness"

c) Classification of elephant distribution and densities from 1985-2009 (+/- standard deviation)

As a result, 48% of KNP is comprised of either wilderness areas, elephant hotspots or high BIOSEVA; 24% is comprised of the combination of either elephant hotspots within wilderness areas and vice versa, wilderness areas with high BIOSEVA and vice versa; or elephant hotspots with high BIOSEVA and vice versa; and 4% is comprised of wilderness areas, with high BIOSEVA, which is also subjected to high levels of elephant impact (elephant hotspots) (Figure 19).

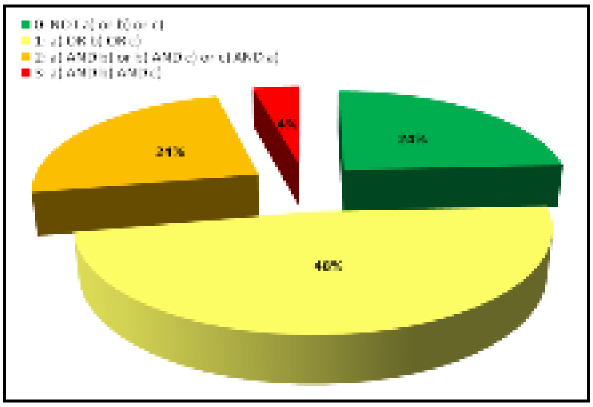


Figure 19: Percentage of a) wilderness areas and b) elephant hotspots represented by different levels of c) biodiversity sensitivity and value.

Therefore, it is this 17% (BIOSEVA wilderness areas); 19% (BIOSEVA areas threatened by high elephant densities); and in particular the 4% (BIOSEVA wilderness areas threatened by high elephant densities) that must be prioritised in terms of protection and impact reduction efforts.

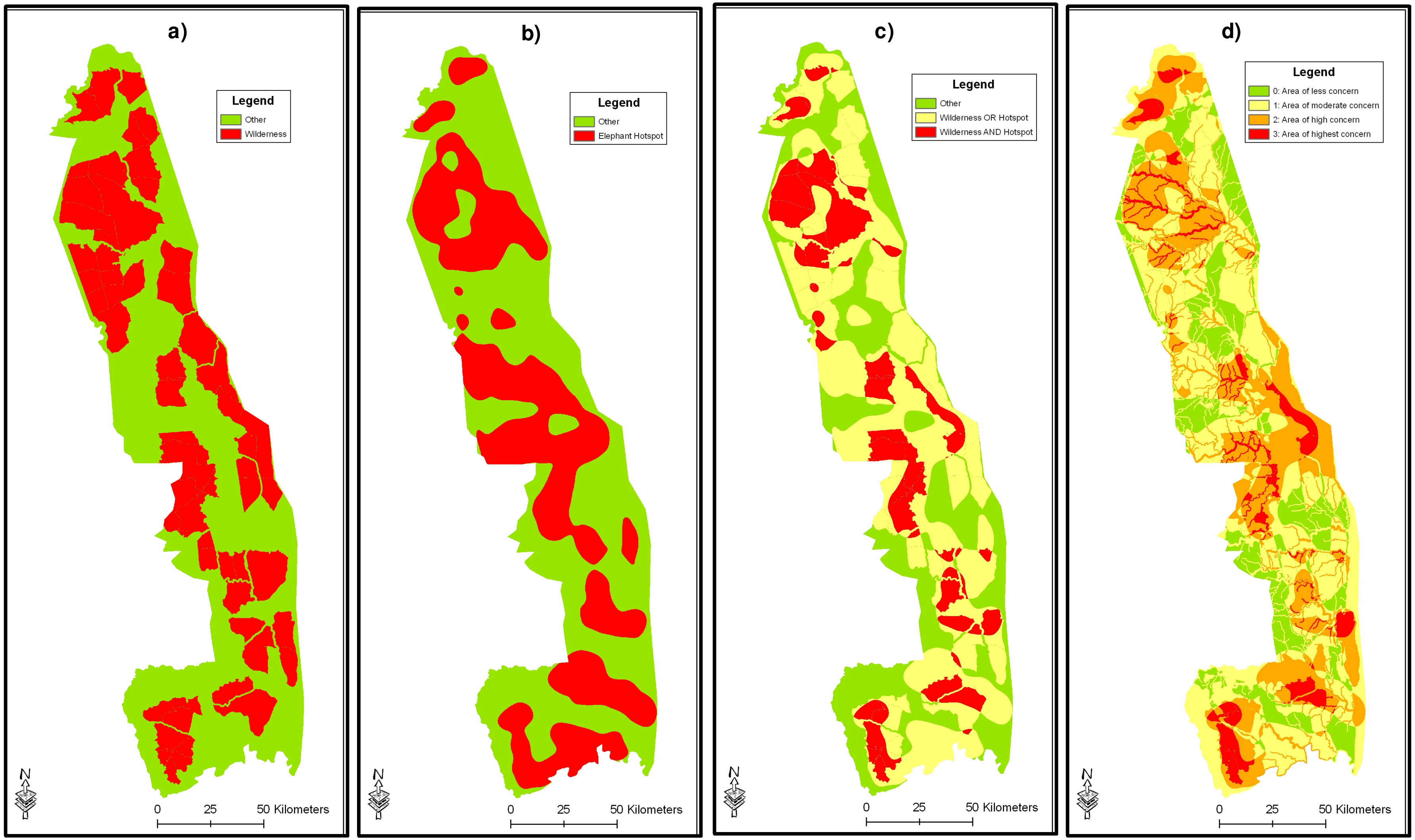


Figure 20: a) Wilderness areas and b) elephant hotspots c) combined to represent elephant hotspots in wilderness areas that d) contain high levels of biodiversity sensitivity and value.

~ **CHAPTER 5** ~

DISCUSSION AND CONCLUSION

Traditional biodiversity assessment and conservation planning techniques have in the past been restricted to the identification of protection priorities outside of protected areas. This frequently resulted in an implementation gap between conservation planners and practitioners (Balmford & Cowling 2006; Copeland *et al.* 2007). Here conservation planning techniques are used within an established protected area (KNP) as a decision support tool for planning and prioritising local protection efforts and management actions. The KNP was stratified into six zones of biodiversity sensitivity-value according to their intrinsic landscape characteristics. Highly sensitive and valuable areas made up **11% (6-5)** of KNP's surface area, **32% (4-3)** was moderately sensitive and valuable, and the remaining **57% (2-1)** was scored relatively low in terms of biodiversity sensitivity and value (Figure 15). Interestingly, rivers appear as areas of particular concern in the BIOSEVA, which is consistent with ongoing concern over poor water flow and quality measurements, habitat loss due to sedimentation and a number of other factors (McLoughlin *et al.* 2010). However, these results may assist in leveraging additional support from catchment management agencies in order to sustain minimum in-stream flow requirements (see McLoughlin 2010 for a discussion).

Protection efforts may now be prioritised on the basis of the value of biodiversity in an area, and its sensitivity to a particular tourism, management activity and/or other disturbance factor. However, the process of prioritization of research efforts and management actions are dependent on the nature of the management concern. In a National Park system, obviously biodiversity is an important constituent, however due to the sheer size of KNP, wilderness is another valuable and non-renewable resource deserving special protection. Not only do these areas act as strongholds for biodiversity, they also provide ecosystem services and enhance ecosystem resilience. By removing the area of influence of all existing infrastructure from the landscape, the remaining 45% of KNP may be divided into 62 intact wilderness areas (**22 areas < 10 000 ha**; **29 areas between 10 000 – 20 000 ha**; **8 areas between 20 000 – 30 000 ha**; and **3 areas of > 30 000 ha**). In combining wilderness with BIOSEVA, wilderness areas that are valuable and sensitive in terms of biodiversity represent **17%** of KNP and should be prioritized for protection. In this way any tourism, management or research developments that may threaten the integrity of these areas may be avoided.

In addition to concerns over human land use, elephants are selective agents of disturbance, acting as spatial and temporal drivers of ecosystem change. Even though the raging elephant debate is laden with ecosystem and ethical complexities, elephants are habitat engineers and when their dispersal is restricted it is not hard to recognize that disturbance may become severe and undesirable. In order to quantify the risks associated with elephant land use, persistent and prolific patterns of high elephant densities (elephant hotspots) must be identified.

Using a dry-season snapshot, annual aerial census kernel density grids were used to classify KNP into six zones of persistent elephant concentrations over 25 years. Areas **>1 standard deviation** from the 25 year mean of elephants per km^2 constituents **15%** of KNP; **28% is between 0 - 1 standard deviations**; and **57% is < 0 standard deviations**. Whether these areas are practical for the development of elephant management zones (similar to that of Whyte *et al.* 1999), where elephant numbers and distribution patterns are altered in favour of biodiversity maintenance by varying resources spatially and temporally in the landscape (e.g. water provision) remains to be seen.

When hotspots overlap with areas of high biodiversity sensitivity and/or value, protection priorities should be set for these areas by managers. Within KNP, **17%** of the highly valuable and sensitive biodiversity is subject to persistently high concentrations of elephants. Once the decision to intervene has been made (Figure 16), appropriate management actions must be assessed in relation to the desired outcome. For example, in order to reduce elephant numbers, managers will need to simulate natural mortality, reduce conception rates and/or decrease the reproductive lifespan of elephants. Future improvements to an elephant impact zonation for KNP include: a predictive component of modelling elephant movements in response to environmental and management changes, addition of a rainfall component to investigate patterns of elephant density and distribution under different rainfall cycles (e.g. extreme high and low rainfall years).

As biodiversity is dynamic over both space and time, some areas will have higher levels of biodiversity and be more or less susceptible to disturbance than others at different times. In order to quantify these spatio-temporal patterns of biodiversity, these analyses should be re-run with each review of the management plan (i.e. 5 years). Therefore, the next step is to generate a working model, using ModelBuilder in ArcMap (ESRI 2006), of BIOSEVA which can be re-run relatively easily and consistently. For example, as new and improved information becomes available, BIOSEVA should be re-run using spatially explicit records and avoid landtype summaries. Additional information to be incorporated into future BIOSEVA include: visual and sound sensitivity analyses; better process surrogates (Rouget *et al.* 2004); neighbourhoods of habitat heterogeneity (as a proxy for biodiversity extent) using iterative remote sensing classifications and the inclusion of vegetation or habitat structural heterogeneity using moving window algorithms on remotely sensed products like the MODIS tree cover.

This may be especially useful in light of the fact that the exact distribution of all species will never be known. As an alternative, habitats may be used as surrogates for species (e.g. habitat suitability mapping) or biodiversity (e.g. habitat heterogeneity begets biodiversity). Also, it may be useful to compare these wilderness mapping results with those of GLOBIO (UNEP 2001) by cross-tabulating the areas and quantifying similarities and dissimilarities.

Improvements can always be made to any model including the BIOSEVA, wilderness and elephant impact zonations, however at present they represent the best available estimate of biodiversity status (11%) and potential human (17%) and elephant (19%) impact. With these areas of high biodiversity sensitivity and value identified, management plans must be mindful of the consequences of future park developments and /or new manipulative elephant management strategies for the biodiversity template.

The use of BIOSEVA is not restricted to Conservation Development Frameworks or elephant management issues but can also be applied to concerns involving any disturbance or threat, for example invasive species and prolonged fire frequency, to name a few. Moreover, decision makers should not limit themselves to the reclassified BIOSEVA values (1-6) but should rather interrogate individual component scores further (Figure 6). In this way the spatial dynamics of biodiversity across KNP may be investigated in more detail and protected areas management may succeed in minimizing impacts and maintaining ecosystem integrity. Fortunately, the recent emphasis shift from static plans to deliverable goals will undoubtedly help integrate systematic conservation planning back into the local decision-making process and ensure plans are updated regularly in response to conservation actions (Knight *et al.* 2008).

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