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Systematic Conservation Planning for South Africa's Forest Biome:

An assessment of the conservation status of South Africa's forests and recommendations for their conservation

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Abstract

This thesis reports on the first substantial data collation, analysis and interpretation for a systematic conservation plan for the entire South African forest biome. This was done by addressing the following questions: What is the current state of the forest biome? How well are forests protected? How threatened are they? What are the conservation priorities, and what needs to be done to improve forest conservation?

The study is broad and integrative, using information from various published and unpublished sources, as well as expert judgements. The general framework of systematic conservation planning was used along with the software tools typically used for irreplacebility analysis. Rule based modelling, expert judgements and GIS modelling, were used to develop indices of threat, vulnerability, fragmentation, degradation, connectivity and irreplacebility, at the scale of forest patches, forest clusters and forest types. An index of subsistence resource use of forests was modelled using population density, extent of electrification, forest accessibility and the buffering effect of plantations and woodlots. Using these indices, priorities for conservation were identified. Species richness and numbers of red data and endemic species, were also evaluated for each forest type. Forest patches as well as forest clusters were used as planning units, while forest types were used as surrogates to represent forest biodiversity .

South African forests have by far the highest number of tree species per unit area of any temperate forest in the world. A high proportion of species occurring in forests are threatened and endemic. At least 56 forest occurring vascular plants and approximately 88 forest occurring faunal species are listed as IUCN red data species. The current forest protected area network of South Africa, does not adequately protect representative samples of forest biodiversity pattern and process. Approximately 25 % of the total forested area occurs within formal statutory protected areas, but most of this is made up of just a few forest types. Of the 21 forest types assessed, six have less than 10% formal protection. Three forest types, Eastern Scarp, Pondoland Scarp and Kwazulu-Natal Dune forests stand out as being highly vulnerable to biodiversity loss, of these; Pondoland Scarp forests have the lowest level of formal protection, and the highest number of endemic species, making this forest type, the highest conservation priority in the country.

The limitations of the mathematical selection algorithms (C-plan and MARXAN) to incorporate ecological heuristics and context specific information within reserve selection were recognised. In response to this, a rule based modelling approach was used, that enables ecological heuristics to guide the selection of priority forests, This provided pragmatic, but not necessary mathematically optimal solutions to network reserve design.

The traditional (largely silvicultural) focus of forest management and reserve planning in South Africa, has tended to view forests as geographically and functionally distinct ecosystems, without adequate consideration of landscape scale processes and requirements for connectivity. For long term conservation of forest biodiversity, planning requires to occur across multiple scales, and with a broader and longer term view than what has been the traditionally focus.

Forest conservation needs to involve both on, and off-reserve strategies. This should include: expansion of the formal protected area network (so as to adequately represent all forest types), improvements in management of existing reserves, and regulation of land use change within forested catchments and catchments linked to forests. This should form part of integrated land management strategy that directly involves communities in forest conservation programmes.

Executive summary

The purpose of this thesis was to determine forest conservation priorities for the development of a forest protected area network. Information was obtained from a number of sources including quantitative spatial data sets, expert judgments and literature reviews The general framework of systematic conservation planning was applied. key outputs being an assessment of threats, protected area gap analysis and lists of forest conservation priorities at the level of species, patches and forest clusters.

Chapter 1 discusses the rationale, aims, approach and limitations of the study, and provides a brief background to forests in South Africa. Importantly it is pointed out that the highly fragmented nature of the forest biome, its wide geographic spread, and occurrence within a diversity of land management authorities, makes forest conservation inherently complex. An important limitation of this study was the lack of comprehensive geo referenced data on all forest occurring species. To address this shortcoming, the national forest types were used as surrogates for the distribution of forest biodiversity.

Chapter 2 evaluates the state or 'health' of forest habitat by considering the amount of forest that has been lost, the amount that has been degraded, and the level of fragmentation of each forest type. No accurate calculation of the overall loss of forests can be made from the data currently available, however experts estimate that most forest types have lost from 15% to more than 35% of their original extent. Coastal forest types have experienced the highest loss, mostly due to agricultural expansion and coastal development, with estimates being as high as 65% loss for the coastal forest belt since European colonisation. Important causative agents of forest degradation in South Africa include historical logging, (widespread until 1939), contemporary selective logging of mature trees, and subsistence harvesting. Available literature, as well as expert surveys, suggests that a large proportion of the forest biome has been significantly degraded. No quantitative data was available to verify the extent of this degradation, but experts estimate that between 40 and 60 % of the area of a number of forest types may be significantly degraded. Currently, the most significant cause of forest degradation is uncontrolled harvesting of subsistence forests products by rural populations, particularly in communal areas. South African forests are naturally highly fragmented, primarily the result of the repeated and often

drastic changes in palaeoclimate. This fragmentation has been greatly exacerbated by anthropogenic induced transformation of both forest habitat and the surrounding matrix.

Chapter 3 evaluates the state of forest species by considering the overall richness of forest plants, as well as the comparative richness, endemism and uniqueness of forest types. The conservation status of threatened forest flora and fauna are also described. Based on vascular plant richness, the most species rich forest groups are the Scarp and Mistbelt forests, with the Eastern Scarp and Pondoland Scarp forest types having the highest levels of species richness, endemism and phylogenetic richness - these forests are therefore of extreme conservation importance. A global comparison of temperate forest tree species richness shows South African forests to have the second highest number of tree species, second only to the East Asian forests, but these are orders of magnitude larger than South African forests. If species richness is considered per unit area of forest, South African forests have by far the highest number of tree species of any temperate forest in the world. Overall 56 species of vascular plants, listed as IUCN red data species occur in forests. Of this, two species are extinct in the wild, four are critically endangered, eight endangered, 20 vulnerable and 22 are near threatened. Approximately 88 forest occurring faunal species are listed as IUCN red data species. Of this, 11 are critically endangered, 21 endangered, 32 vulnerable, and one recorded extinction. Forests have the highest proportion of threatened vertebrate species of any biome in South Africa: overall, approximately 13% of all forest-occurring vertebrate species are threatened.

Chapter 4 evaluates different levels of South African forest protected areas. The assessment focused on how the current protected area network protects forest pattern. Approximately 25 % of the total forested area occurs within formal statutory protected areas, but the distribution of these protected areas are highly biased to just a few forest types, with more than half of all protected forest area made up of just three forest types. Many forest types remain poorly represented within the formal protected area network. Of the 21 forest types assessed, six are below the 10% minimum formal protection level set by IUCN.

Chapter 5 assesses current threats to forests. Using GIS analysis combined with rule based modelling; levels of different threats (at the scale of forest patches, clusters and forest types) were modelled. This included an index of the subsistence resource use pressure on forests, as modelled from population density around forests, extent of electrification, forest accessibility and buffering effect of plantations and woodlots. Threats arising from population pressure and matrix transformation are considered to be the most important threats to forests. Contemporary coastal and urban development as well as mining have, and can still, cause significant loss of valuable forest habitat, but areas affected tend to be localised. The combination of land use change, habitat fragmentation and climate change are predicted to impact synergistically on forest biodiversity. Three forest types, Eastern Scarp, Pondoland Scarp and Kwazulu-Natal Dune forests stand out as being highly vulnerable to biodiversity loss, of these; Pondoland Scarp forests have the lowest level of formal protection, making this forest type, the highest conservation priority in the country.

Chapter 6 evaluates 16 000 plus forest patches to identify forests most in need of conservation action. Two approaches were used: a quantitative approach using systematic assessment, and a qualitative approach using expert judgements. The quantitative systematic assessment used C-plan to derive irreplaceability values for forest patches, and GIS rule based modelling to predict levels of threats for forest patches. The qualitative assessment relied on the experience and informed judgement of forest experts to select priority forests. The priority forests derived from the systematic assessment were reviewed by a panel of experts. Experts were in agreement with almost all forests selected by the systematic assessment, but felt that some of the most important forest had been excluded. Forests from each of the two approaches were combined into an integrated list of priority forests. This chapter also briefly discusses two important issues pertinent to conservation planning. These include: combining expert judgements with systematic assessments; and the incorporation of socioeconomic costing into the process of reserve selection.

Chapter 7 shifts focus at two levels, firstly the scale of planning is shifted from forest patches to forest clusters; and secondly the method of selection is shifted away from the dichotomy of either qualitative (expert) or quantitative (systematic) selection to an integration of both approaches using expert system rule based modeling. It is contended that systematic conservation planning has placed unrealistic emphasis on the need to find mathematically optimal solutions to reserve network design, and this at the expense of more pragmatic approaches that include ecological heuristics and context specific information. The challenge is posed to design a forest protected area network that incorporates forest ecosystem processes operating across a range of time and space scales. It is argued that this requires a shift in thinking from the traditional silvicultural perspective of forests as ‘stand alone island patches’, to an ecosystem-landscape perspective of forests, where forests are considered as part of a broader interconnected landscape system. By using planning units of a size broadly similar to the spatial scales at which forest ecosystem processes operate, planning units serve as spatial surrogates for these processes. So for example, using forest clusters as conservation planning units, will automatically included most landscape scale processes. The selection of priority forest clusters was initially conducted using an iterative computer selection algorithm (MARXAN), which selected priority clusters based on calculated irreplaceability values. However, a number of the larger forests, considered by forest experts as high priorities, had been excluded by the MARXAN selection process. Analysis of this revealed that MARXAN tended to favour selection of smaller forests to make up targets rather than the larger (biologically more valuable patches), due to the high ‘cost’ of the later. In response to this, a less mechanistic, but mathematically non-optimal approach, was developed, based on using ecological heuristics to guide slection. Ecological heuristics are ecological principles derived either from context specific information and/or from established ecological theory. Ecological heuristics were incorporated within a semi automated selection algorithm using expert system type rules. These rules also draw on the results of irreplaceability analysis.

Chapter 8 provides a synthesis of the main findings and a summary of the forest conservation priorities across three major spatiotemporal scales. It is concluded that the current forest protected area system in South Africa is inadequate to ensure the persistence of forest biodiversity. Key reasons include: the low levels of formal protection for many forest types; forest protected areas that are not designed to protect landscape scale processes; the absence of appropriate land use regulations in land surrounding forests (the matrix); and absence of effective community based conservation programmes. Forests conservation needs to include both ‘on and off-reserve’ conservation strategies. On-reserve conservation include expansion of the formal protected area network, and improved levels of conservation management in existing forest reserves. Off-reserve conservation include measures to maintain land surrounding forest patches in near natural states, and management of catchments linked to, or containing forests, so as to account for the cumulative impacts of land use change on sensitive ecosystem. Of critical importance to the conservation of many forests is the need for programmes that promote community based conservation, alternative sustainable livelihoods and rural poverty alleviation.

CHAPTER 1: INTRODUCTION

1.1 Rationale

Healthy forest ecosystems are ecological life-support systems. Forests provide a wide range of goods and services vital to human health and livelihood (Lawes *et al*., 2001; FAO, 2003; MEA, 2003; Lawes *et al.,* 2004b; FAO, 2007). Many of these goods and services are traditionally viewed as free benefits to society, or 'public goods' - wildlife habitat , species conservation, watershed services, carbon storage, and scenic landscapes, for example. Lacking a formal market, these natural assets are traditionally absent from society's balance sheet; their critical contributions often overlooked in public, corporate, and individual decision-making (Krieger, 2001).

Systematic conservation planning provides an effective way to seek and identify the most efficient and effective types of reserve design to capture or sustain the highest priority biodiversity values (Margules and Pressey 2000). Essentially, it entails a semi structured planning procedure to guide the process of selecting suitable conservation areas that can meet the objectives of biodiversity representation and persistence. A framework for systematic conservation planning was first formally described by Margules and Pressey (2000), and later modified by Cowling and Pressey (2003), and Cowling *et al*., (2003b).

Although systematic conservation planning has been widely applied to many regions and biomes of South Africa (see for example Cowling, 1999; Cowling *et al*, 1999b; Goodman, 2000; Cowling *et al*., 2003b, Knight and Cowling 2003; Von Hase *et al*., 2003; Desmet, 2004; Rouget *et al.,* 2003b; Smith *et al*., 2006; Berliner and Desmet, 2007; Pence *et al.,* 2007; Berliner and Desmet, 2008), its application specifically to the forest biome, has been largely ignored, and forest conservation in South Africa has been largely based on no, or ad hoc planning. This thesis provides the first country wide and systematic conservation planning exercise for the forest biome.

For a number of reasons, forest conservation in South Africa is inherently complex, making a systematic approach particularly necessary. Reasons for this include: the high diversity of forest habitats, with over 24 distinct forest types; the discontiguous nature of the biome (over 16 000 patches, spread across a wide geographic area from the Limpopo province in the North, to the Cape Peninsular in the South); the diversity of land management authorities in which they fall; the extreme rarity of the vegetation type (forests make up less than 0.4 %, of the surface area of the country); the economic importance of many forests to rural communities and the associated high vulnerability to over exploitation. As such, it is critical that the limited resources available for forests conservation are optimally employed. The purpose of this work is therefore to support efficient and effective forest conservation.

1.2 **Aims**

The key aims of this thesis are: a) to profile the current state of the forest biome in South Africa, and b) using the general framework of systematic conservation planning, identify forest conservation priorities that will support the development of a protected area network representative of forest biodiversity, and that ensures its persistence.

1.3 **Approach and limitations**

The overall approach of this study was to determine forest conservation priorities by integrating available quantitative spatial data (using rule based modelling with GIS analysis), literature reviews, and rapid qualitative expert judgements. The general framework of systematic conservation planning was used (as described for example by Margules and Pressey, 2000). Key outputs include: an assessment of threats to forest, protected area gap analysis, and lists of forest conservation priorities at the level of species, patches and forest clusters. C-plan was used to select priority patches while MARXAN was used to select priority forest clusters; with both modified to include expert judgments. It is contended that systematic conservation planning has placed excessive emphasis on the need for mathematically optimal solutions, at the expense of using ecological heuristics to select reserve networks.

Although forests have many values; in particular they are recognised as providing ecosystem services important to human livelihoods, this study considers forests essentially from the perspective of their biodiversity value only. However, because it is widely recognised that biodiversity underpins the many other values attributed to forests (Dudley *et al.,* 2002; FAO, 2003; Gross 2006; FAO, 2007), this study regards biodiversity value as a general surrogate for all forest values.

Typically, systematic conservation planning entails the analysis of quantitative data (Margules and Pressey, 2000), however, given the paucity of available quantitative data, a combination of both quantitative and qualitative approaches are often required. In particular, the degree to which ecological heuristics and context specific constraints can be incorporated within an exclusively quantitative analysis is limited. To circumvent these shortcomings and facilitate the incorporation of local and expert knowledge, a rule based modelling approach was used which integrated both quantitative analyses with qualitative expert judgements. Expert judgments can provide a rapid and cost effective approach to identifying priorities, it is never the less recognised that experts often differ in judgements, the accuracy of which may also be difficult to determine.

An important limitation of this study was the lack of available geo referenced forest species data. To compensate for this limitation, forest types, (as identified by Von Maltitz *et al.,* 2003) were used as biodiversity surrogates. The broad distribution of these forest types within South Africa are presented in Figure 1. Biodiversity surrogates are attributes of species believed to represent the distribution and abundances of species and species assemblages (Hunter, 1999). Biodiversity surrogates may include forest types (Lindenmayer and Franklin, 2002), other plant communities (Desmet, 2004), broad habitat units (Cowling and Heijnis, 2001) or ecoregions (Dinerstein *et al.,* 2000). Surrogates are usually essential in conservation planning, as it is impossible to comprehensively document all biodiversity (Margules and Pressey, 2000).

Importantly, plant species data was only available to me at the level of forest types, and not individual patches; and faunal data, only at the level of the whole biome. This implies that species data was not used by the mathematical selection algorithms to differentiate between forest patches of the same forest type (although, these would of been considered within the expert selection process, and that targets could not be set specifically for occurrence of rare or endangered species The limitations of this assumption of surrogacy need to be recognised. Not all patches of a specific forest type contain the same species, and often there may be a relatively large overlap in species between certain forest types. Forest species distribution ranges tend to change gradually with species dropping out as one moves along a north to south, and east to west gradient, (Geldenhuys, 2000). Despite these limitations, forest types provide a useful approximation for the majority of forest species, but should not be used as an absolute predictor of species occurrence, in particular for rare or unique species.

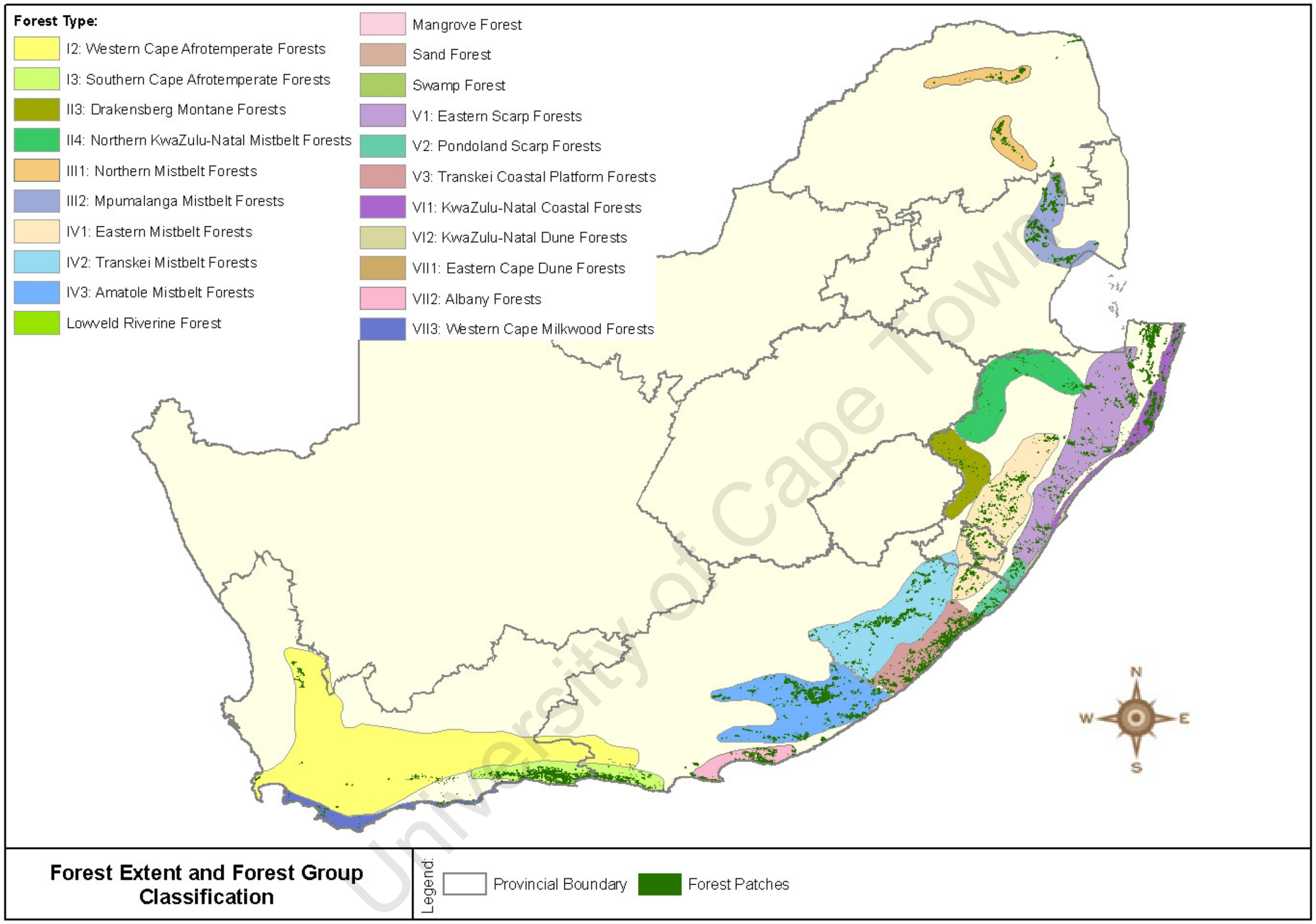


Figure 1. Distribution of forest types in South Africa (after Von Maltitz *et al.,* 2003 using NFI data). The coloured shapes show the approximate area of distribution of each forest type. Forest patches shown as green dots. Approximate scale: 1: 10 000 000.

Gap analysis was conducted by determining the extent to which forest types were represented within the current configuration of protected areas, while rule based modelling was used to integrate spatial data (such as population pressure, wood usage, agricultural potential, matrix transformation, and forest patch area to perimeter ratios) into indices of threat for forest patches, clusters and forest types. Qualitative expert judgments were used to determine the current state of forest habitat (forest loss and degradation) while quantitative analysis (using FragStats) was used to determine indices of fragmentation of forest types.

Irreplaceability of forest patches were determined using C-plan (Anon. 1999), while a separate analysis using MARXAN (Ball & Possingham, 2000), calculated irreplaceability of whole spatial groupings of forest patches (referred to as forest clusters). Both are criticized by me for being overly mechanistic in the selection process and driven by the objective of mathematical optimization, rather than the need to use sound ecological principles to identify conservation priorities. As a response to this, an expert system, rule based approach was used, that formalises ecological understanding (heuristics) into selection rules used to guide the selection of priority forests.

Because of the breadth of the study, (the biodiversity of the whole forest biome considered across major spatiotemporal scales), no new field data was collected, rather, a wide range of existing data sources were integrated into predictive models.

Although the data used was the best available at the time of data analysis (2004 to 2006), some of the data sets are outdated. In particular, land cover data (used in assessing matrix transformation) was for the year 2000, and population data (used in threat assessment) was for the year 2001. It is certain that land transformation has increased since 2000, although it is less clear to what extent rural populations may have changed since 2001. Particularly, considering that South Africa's annual population growth rate of 1.6 % (Stats SA, 2002) has predicted to decline to 0.6% for 2000 projected to 2010 (UNSD, 2007). In addition, urbanization has resulted in an overall negative rural population growth rate of -0.4 %, for 2000-2005 (UNSD, 2007).

1.4 **Overview and background**

Forests are an important and rare vegetation type in South Africa. They make up approximately only 0.4% of the surface area of the country. They form part of two global biomes: the warm temperate evergreen forest biome (Afrotemperate forests) and the subtropical coastal forest biome (Mucina and Geldenhuys, 2006). They occur as an archipelago of patches scattered along the eastern and southern escarpment mountain ranges and coastal lowlands of South Africa. Being highly fragmented and discontinuous, forest are usually considered as relicts of a once more widespread biome, the extent of which has fluctuated considerably over the last 180 000 years, in response to palaeoclimatic changes (Vogel, 1990; Partridge *et al.,* 1990; Lawes, 1990; Partridge *et al*., 1993; Eeley, *et al.,* 1999; Lawes *et al.,* 2000). Despite the small surface area, forests make a disproportionably high contribution to the conservation of South Africa's biodiversity (Geldenhuys and MacDevette, 1989), being second only to fynbos in terms of plant species richness per unit area (Gibbs Russel, 1985; Gibbs Russel ,1987; and this study). In addition, they provide essential habitat to at least 13% of all South Africa's IUCN red listed vertebrate species (EWT, 2002). Globally, South African forests are recognised as having the highest tree diversity of any temperate latitude forest, with between three and seven times more tree species than other forested areas of the southern hemisphere, this despite covering the smallest area in comparison to other counties. Furthermore, when it comes to the richness of genera and families of trees, South African forests are unparalleled (Silander, 2001; Cowling, 2002)

Past human activities, including fires, historical logging and clearing for agriculture, have resulted in significant loss of forests across the country (King, 1938; King, 1941; McCracken, 1986; Geldenhuys, 1994; McCracken, 2004; Lawes *et al.,* 2004b; Mucina and Geldenhuys, 2006). The exact extent of these losses is uncertain. The extensive logging and deforestation prevalent during the colonial era, has largely halted, but ongoing forest degradation, primarily through non sustainable use and land transformation in and around forests is placing increased pressure on forest biodiversity. Not only have many forests lost their full complement of original fauna (Castley and Kerley, 1996; Lawes *et al.,* 2000; De Villiers and White, 2002; Hayward *et al.,* 2005), but the floral composition and structure of many forests have been disrupted by both the legacy of historical logging (Lawes *et al.,* 2007c), as well as ongoing non-sustainable selective harvesting practises (Obiri et. al., 2002; Lawes and Obiri, 2003; Boudreau, *et al.,* 2005). The impacts of these selective harvesting practices on forest dynamics are largely unexplored and unknown (Lawes and Obiri, 2003).

Perhaps one of the biggest threats to long term persistence of forest biodiversity in South Africa, particularly within the context of predicted impacts of climate change, (see for example Taylor and Hamilton, 1994) is habitat loss within the forest matrix. This study has found that over one third of the forest matrix had been transformed to agriculture and plantation forestry. In many cases this has resulted in forest patches becoming ecologically isolated, and has led to the disruption of landscape level ecological processes effecting forests, such as natural disturbance regimes (of fires, wind, herbivory etc), changes in geo-hydrology, as well as the metapopulation dynamics of some forest species. While it may be still too early to detect the full impacts of matrix transformation on forest biodiversity, limited evidence suggests that metapopulations of forest dependent species are coming under increased pressure (Swart and Lawes, 1996; Lawes *et al.,* 2000; Wethered and Lawes, 2003). The effects of matrix transformation on patchy metapopulations are well established in landscape ecological models (Hobbs, 1993; Forman,1995; Wiens, 1997; Monkkonen and Ruenanen, 1999; Turner, 2001) and have been empirically established in a large body of research, particularly in tropical forests (see for example Camargo and Kapos,1995; Foster *et al.,* 1999, Lindenmayer *et al.,* 2005). Importantly, the impact of habitat change on forest biodiversity will be exacerbated by climate change. While in the past many species may have responded to past climatic challenges by shifting their distribution ranges across natural landscapes, (Balmford, 1996; Eeley *et al.,* 1999), today the increasingly human altered landscapes will limit their ability to do so.

1.5 **Data sources**

Because no complete, country wide forest cover data set was available at the time of analysis (2004 to 2006), calculating the current extent of forest cover required combining a number of separate data sets. The National Forest Inventory (NFI, 2005) data set was used as the prime source, but was incomplete. To address gaps in this, the following sources were also used: KwaZulu-Natal Wildlife forest coverage (provided by Dr. P. Goodman); data from Mpumalanga Parks Board (provided by Mr. Mervin Lotter), as well as data from the beta version of the national vegetation map (Mucina and Rutherford, 2004).

The prime data source, for forest cover, the National Forest Inventory (NFI, 2005) was commissioned by the Department of Water Affairs and Forestry from a consortium of consultants with the aim of producing a digitised map of the countries forests based on TELKOM digital orthophotography at scale of 1:10,000. The forest data sets provided by the provincial conservation authorities are derived from ortho-corrected SPOT2 & SPOT4, 20 m imagery captured in 2005.

Land transformation around forests was obtained from National Land Cover classified satellite imagery (NLC, 2000). The additional data sources used to model threats to forests are given in Table 22, section 5.2.2.

CHAPTER 2: THE STATE OF FOREST HABITAT

2.1 Overview

The ‘state’ or status of an ecosystem, habitat, forest type, or biome, broadly equates with the concepts of: ‘ecosystem health’ (see for example De Leo and Levin, 1997; Noss, 1999; Andreasen *et al.,* 2001 and others); ecosystem integrity (Karr, 1992, Noss, 1995; Noss 2000b); and biodiversity intactness (Scholes and Biggs, 2005; Biggs *et al*., 2006).

The objective measurement of the integrity of an ecosystem can only be defined relative to a predefined initial state. Scholes and Biggs, (2005) consider ‘biodiversity intactness’ as a measure of the change in abundance across all well-known elements of biodiversity relative to their inferred pre-colonial state. Importantly, declining ecosystem health is more than just a change in species population frequencies (which may be impossible to distinguish from natural fluctuation patterns), rather it involves changes in the functional attributes of ecosystems. These manifest as systemic changes, often with multiple knock-on effects, as may occur for example, with dramatic changes in nutrient cycling, seed dispersal, pollination, and natural disturbance regimes of a forest. Importantly these functional changes can be caused by changes in the spatial configuration and extent of the habitat, primarily thorough effects associated with habitat loss and fragmentation (Saunders *et al.,* 1991; Solé, *et al*., 2004).

Because detecting changes in ecosystem functioning may be costly or difficult to measure directly, it has become common practice to use measurable indicators, or ‘surrogates’ of ecosystem state (Noss, 1990; Noss, 1999; Smeets and Weterings, 1999). This approach has been particularly useful in rapid assessments and in state of the environment reporting within the DPSR (Drivers-Pressures-State-Response) framework, as described for example by OECD (1998) and CSD (2001). Although there is no consensus regarding what indicators should be used to measure ecosystem health, they generally involve measures of both changes in ecosystem quantity and quality, as measured by indicators of ecosystem structure, function and composition (Franklin, 1981; O'Neill *et al.,* 1986; Noss, 1990).

In this thesis I differentiate between the state of the forest habitat and the state of forests species. This chapter evaluates the state of forest habitat by evaluating how much forest area has been lost, degraded and fragmented (the state of forest species is dealt with in chapter 3).

2.2 How much forest remains and how much has been lost?

2.2.1 Introduction

Various approximations of remaining total extent of the forest biome made in the past have mostly underestimated the total extent of forests (see Table 1, below).

Table 1. Estimations of total remaining extent of forested area in South Africa

Source	Remaining forest area (km ^{2})
FRD forest biome Map (Anon, 1987)	3 023
Old SA vegetation map (Low and Rebelo, 1996)	4 025
Von Maltitz *et al.* (1999) complied from Cooper (1985); Cooper and Swart, (1992); Geldenhuys (1991); Thompson (1999)	3 500
NLC 2000. Thompson, (1999)	5 386
New SA vegetation map Mucina and Rutherford (2006)	4 479
National Forest Inventory, DWAF (2005)	4 973

Understanding patterns and processes of habitat change is essential for managing and conserving forest fragments in anthropogenic altered landscapes (Laws, *et al.* 2004 a). Understanding how much of a habitat type needs to be conserved to ensure biodiversity persistence is integral to conservation planning (Margules and Pressey 2000; Pressey *et al.*, 2003, Driver *et al.,* 2004; Desmet and Cowling, 2004). To do this requires knowledge of the original extent of the habitat and how much has been lost.

Forests in South Africa have a long history of non-sustainable utilisation, being one of the first biomes to undergo heavy exploitation with the colonisation of the Cape. Most of the destruction took place at the hands of European settlers in the period 1860–1940, (King, 1938; King, 1941; McCracken, 1986; McCracken, 2004; Lawes *et al.,* 2004b; Mucina and Geldenhuys, 2006; Lawes *et al.,* 2007c).

Limited quantitative data is available to accurately determine forest loss across the whole country, and early estimates of forest loss (given by King, 1938, King, 1941; Rycroft, 1942; Rycroft, 1944) cannot be accurately tested. However, evidence suggests that in certain areas indigenous forests were significantly larger than at present (Moll, 1972; Wager, 1976), but at least for some areas, these losses have been exaggerated, and more likely to be in the order of 10 to 15 % loss, mainly through boundary contraction (Prof. Mike Lawes, personal communication). See also Lawes *et al.,* (2004a) who compared digitised aerial photographs from 1944 and 1996 for the Karkloof-Balgowan archipelago in KwaZulu-Natal, and found only a 5.7 % decline for this period. This is contrasted to previous reports of Rycroft, (1944) for the period 1880–1940, that estimated a loss of up to 80% for this area. Direct comparisons between these two approximations are however difficult to make given the different time period over which the comparisons were made. Lawes *et al.* (2004a) points out that despite a relatively low estimation of forest loss for the period considered, most of the loss could be attributed to disappearance of the smaller patches (< 0.5 ha), leading to an overall increase in isolation of remaining patches, this being further exacerbated by transformation of the grassland matrix to plantation forestry.

The difficulties of measuring historical forest loss are further complicated by uncertainty regarding the current extent of forested area in South Africa. Accurate mapping of the extent of the forest biome of South Africa has been confounded by spectral image confusions between forests and other vegetation types, in particular thicket, dense woodland and plantations (Geoterraimage, 2005). In addition differences in scale, minimum size of mapped patches, and geographic extent of the mapping initiatives, have also led to different estimations of total forest extent. (For further discussion on difficulties of forest spectral image classifications see Thompson, 1999). To illustrate the difficulties of spectral image interpretation for indigenous forests, I overlaid the forest cover of the National Forest Inventory (NFI, 2005) with the cover used in the national vegetation map (Mucina and Rutherford, 2004) and compared this with Google earth satellite imagery for Manubi forest, in the Eastern Cape (see Figure 2, below). Although far from a quantitative comparison, it clearly shows the difference between the two interpretations of forest cover, pointing to the importance of ‘ground truthing’ when mapping vegetation.

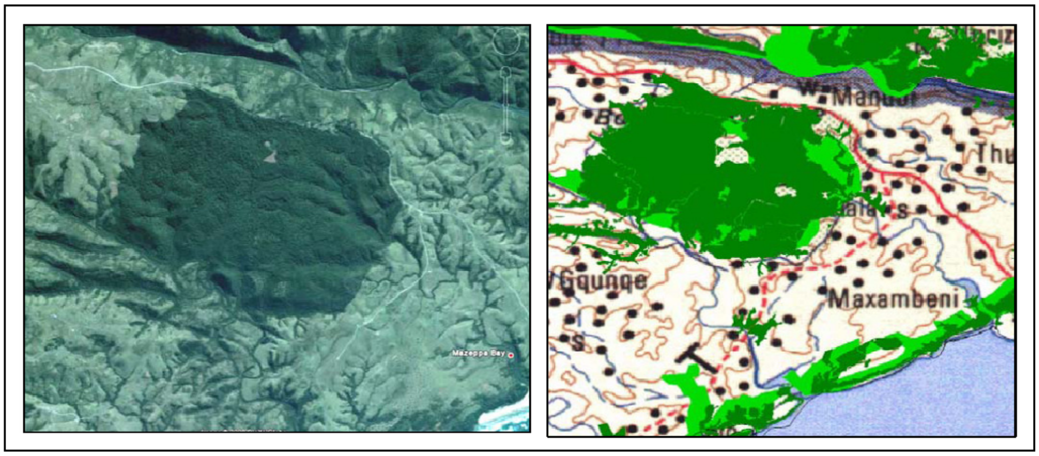


Figure 2. Comparison of satellite imagery, (map on left) with mapped forest cover, (map to the right), showing two different interpretations of indigenous forest cover for Manubi forest, in the Eastern Cape. The two forest covers are from the National Forest Inventory, in dark green, and the national vegetation map of Mucina and Rutherford (2004), in light green. These differences arise from different image interpretation of indigenous forest as opposed to secondary forest, woodland, scrub, plantations and shadows.

2.2.2. Method

Because this study made use of existing data sources, i.e. no primary data was collected in the field, and because no definite forest cover layer existed at the time of analysis, calculating the current extent of forest cover required using a combination of a number of different datasets from different sources. (see section 1.5)

Because much forest loss in South Africa occurred during European colonization, analysis of historic photographic imagery (that date back to about 1940) is of limited use. In the absence of quantitative data, expert judgements were used to approximate forest loss for each of the twenty one forest types considered. Because of the large degree of uncertainty, broad categories were used. Forest experts from the Department of Water Affairs and Forestry[1] were asked to estimate forest loss in one of three broad categories: high (>35 %), medium (15 to 35 %) and low (<15 %).

[1] Estimates provided by Dr. Armin Seydack, Izak van der Merwe and Theo Stehle forest ecologists from the Department of Water Affairs and Forestry (Unpublished DWAF report).

2.2.3 Results

By combining a number of available data sets, this author estimated the current remaining extent of the forest biome to be 4 927 km ^{2}

The degree of total loss of forest biome is difficult to determine, however expert estimations of loss for forest types point to losses of more than 35 % for at least five forest types (see Table 2).

Table 2. Estimated percentage area loss since European settlement and currently remaining areas of South African forest types.

Forest Type	Estimated % loss	Remaining area (ha)
KwaZulu-Natal Coastal	> 35	21 089
KwaZulu-Natal Dune	> 35	12 396
Mangrove	> 35	2 393
Pondoland Scarp	> 35	12 284
Transkei Coastal Platform	> 35	61 484
Albany	15-35	22 046
Eastern Mistbelt	15-35	41 842
Eastern Scarp	15-35	33 750
Licuati Sand	15-35	24 276
Lowveld Riverine	15-35	11 401
Transkei Mistbelt	15-35	30 250
Western Cape Milkwood	15-35	2 500
Amatole Mistbelt	< 15	64 221
Drakensberg Montane	< 15	1 926
Eastern Cape Dune	< 15	10 941
Limpopo Mistbelt	< 15	5 323
Mpumalanga Mistbelt	< 15	32 772
Northern KwaZulu-Natal Mistbelt	< 15	19 204
Southern Cape Afrotemperate	< 15	74 848
Swamp	< 15	3 022
Western Cape Afrotemperate	< 15	4 731

2.2.4 Discussion

Various factors are believed to have contributed to the decline of forested area, including historical timber harvesting, clearing of forests for agriculture and timber plantations, as well as the increased incidence of fires (Cooper, 1985; Lawes, *et al.* 2004a; Geldenhuys, 1994; Mucina and Geldenhuys, 2006). However, no accurate calculations of the total extent of loss can be made from data currently available. Lawes (2002) estimated an overall decline in the forest biome of approximately 40%, while Low and Rebelo (1996) estimated the original extent of the forest biome to be 7148 km ^{2} , which would give a loss of approximately 31 % , if the current extent of the forest biome of 4 926 km ^{2} is used. However, these approximations are based on the biome climatic potential (Rutherford, 1995) which is highly unreliable as it does not account for abiotic factors that may of limited original forest occurrence, such as fire.

Estimates of forest loss vary, depending on forest types and experts. Coastal forest types appear to have the highest losses, this being mostly due to past agricultural expansion and coastal development. Experts (see table 2) estimate coastal forests loss to be in excess of 35 %, but some believe this to be a lot higher. For example Lawes (2002) believes that forest loss may be as high as 65% for the costal forest belt in KwaZulu-Natal, and Cooper (1985), estimated that up to 90% of the coastal forests in KwaZulu-Natal have been cleared for agriculture.

Calculations of how much forest remains and how much has been lost, are further confounded by the fact that forests regenerate when causative agents of degradation such as fire, and overharvesting are kept out, and the distinction between secondary and primary forest is not always clear. Succession may eventually return forests to an original state, but it is not clear when intermediate stages can be considered as 'mature forest' (Geldenhuys, 2002). The original use of the term 'secondary forest' was reserved for forests that have regenerated after complete clearing (Collet, 1994), however, Chokkalingham and De Jong, (2001) use the term to refer to any forest that has regenerated either after compete clearing, or after being degraded. Geldenhuyse, (2002) points out that in certain areas, such as the lower slopes of the Soutpansberg Mountain and along the eastern escarpment of South Africa, forests, particularly those adjacent to plantations, have expanded due to control of wild fires by plantation managers. Similarly, expansion of forest margins has been observed in some communal areas of the Eastern Cape, where high grazing pressure around forests keep fuel loads too low to support hot fires (Professor Christo Fabricius Nelson Mandela Metropolitan University's George Campus, personal communication).

It is evident that overall, the forest biome has declined significantly as a result of man induced changes, but even with improved methods of remote sensing and image analysis, it is unlikely that we will ever know with any degree of certainty how much forest area has been lost. It is possible that some of the earliest aerial photographs available for South Africa (dating to around 1940) could be used to determine forest loss for certain areas (but only from this date onwards) as was done by Lawes *et al.,* (2004a) for the Karkloof-Balgowan forest archipelago. Unfortunately, these historical images do not cover all forested areas.

Previous surveys have typically concentrated on mapping only the larger forest patches, often excluding forest smaller than 50 hectares (for example the survey of Cooper, 1985; and Anon, 1987) or those smaller than 10 ha (National Forest Inventory, DWAF, 2005). As pointed out by Lawes *et al.* (2004a) much of the forest loss is from smaller patches that are particularly vulnerable to surrounding land use changes, and while their loss may be small in terms of area, their loss has contributed significantly to forest fragmentation and a decline in landscape level connectivity. It is important therefore that future mapping be at a scale that is sufficiently detailed to detect functional changes in the forested landscape.

2.3 How degraded are our forests?

2.3.1 Introduction

As part of the evaluation of the current state of the forest biome, an understanding of causes and extent of loss of forest integrity or degradation is particularly important. The purpose of this section is therefore to use expert judgments to provide an estimation of the extent of degradation of each forest type.

There is no globally agreed operational definition of forest degradation and perceptions of what constitutes a degraded forest will vary greatly depending on the objectives of the forest managers. For example, forest managed for biodiversity conservation will be considered degraded if species have been lost, but these same forests may still retain full functionality for carbon sequestration, wood production, soil conservation or recreation (CPF, 2008). I have used the definition of degradation applied by the South African National Biodiversity Institute for setting endangerment ratings for forests, as 'a significant change in forest structure, function and composition that will take several decades to recover if pressure is removed '. Using this definition, forest degradation can be measured by changes that may occur in canopy and understory structure; species composition; and ecosystem functioning, (as may be brought about by disruption of recruitment processes, removal of dispersal agents or disruption of natural disturbance processes).

Important causative agents of forest degradation in South Africa include historical logging, widespread until halted in 1939, (Cawe and McKenzie, 1989; Lawes *et al*. 2007c;), contemporary selective logging of mature trees (Symes, *et al.,* 2004; Henderson and Downs, 2006) and non sustainable subsistence harvesting (Cooper, 1985; Geldenhuys and McDevette, 1989; Cooper and Swart, 1992; Du Plessis, 1995; DWAF, 1999, Shackleton *et al.,* 2004a; Shackleton *et al.,* 2004b; de Villiers and White, 2002; DWAF, 2003, Von Maltitze *et al.,* 2003; Lawes *et al*., 2004a; DWAF, 2005a, Mucina and Geldenhuys, 2006; Lawes *et al*. 2007c). Forest degradation may also be the result of changes in natural disturbance regimes such as fire, wind, light and herbivory. These can cause changes in natural forest margins and ecotones (Noble, 1993; Everard, 1994; Martin *et al.* 2007); disrupt natural gap phase dynamics (Pickett and White, 1985; Pickett *et al.,* 1989; Midgley, *et al.,* 1997; van der Merwe and Seydack, 2005; Everard *et al.,* 1995); change rates of decomposition and nutrient cycling (Kotze and Lawes, 2007); and alter succession regeneration (Lawes *et al.,* 2007b). For example, plantation forestry in upstream catchments have caused considerable hydrological changes that impact on forests (Everard *et al.,* 1994; Schulze, 2000; Armstrong, 2003; Lawes *et al*., 2004b; Dye and Versfeld, 2007; Lawes, 2007b), and have changed natural disturbance processes such as fire, wind and predation (Seydack, 2004; Kotze and Lawes, 2007b). At a species level, overharvesting of selected species causes changes in population structure leading to possible recruitment bottle necks (Midgley *et al.,* 1995; Midgley, 1999; Lawes, 2007c) as well as possible metapopulation extinctions (Swart and Lawes, 1996). At a landscape level, degradation can also be measured as a decline in ecological connectivity leading to increased metapopulation isolation and vulnerability to local extinctions, that may occur for example with the loss of smaller forest patches, often important as stepping stones between larger patches.

2.3.2 Method

The extent and degree of degradation of South African forests is difficult to quantify, as such estimations have had to be made based on available literature and on expert judgments. A rapid assessment approach was used whereby an e-mail survey was sent to ten forest experts who were asked to give upper and lower degradation estimations for each forest type. Using a zero to five scale, where 0-20% = 1; 21-40% = 2; 41-60% = 3; 61-80% = 4 and, 81-100 % = 5. Upper and lower means for each forest type were then calculated.

2.3.3 Results

Results of the expert survey to determine the extent of forest degradation are presented in Table 3, below.

Table 3. Upper and lower percentage estimates of the extent to which forest types have been significantly degraded. Estimations are the mean of ten forest experts (Berliner *et al.,* 2008).

Forest Type	Mean upper and lower % area estimated by experts as significantly degraded (n=10)	
	Lower	Upper
KwaZulu-Natal Coastal	41	60
Lowveld Riverine	41	60
Swamp	41	60
Transkei Coastal	21	40
Pondoland Scarp	21	40
Eastern Scarp	21	40
Eastern Midlands	21	40
Drakensberg Montane	21	40
Licuati Sand	21	40
Mangrove	21	40
Eastern Mistbelt	21	40
KwaZulu-Natal Dune	21	40
Transkei Mistbelt	0	20
Albany Coastal	0	20
Limpopo Mistbelt	0	20
Western Cape Milkwood	0	20
Mpumalanga Mistbelt	0	20
Eastern Cape Dune	0	20
Amatole Mistbelt	0	20
Western Cape Talus	0	20
Southern Cape Afrotemperate	0	20
Western Cape Afrotemperate	0	20

2.3.4 Discussion

From the literature available (Cooper, 1985; Geldenhuys and McDevette, 1989; Cooper and Swart, 1992; Castley and Kerley, 1996) as well as expert surveys, it is evident that a large proportion of the forest biome has been significantly degraded, however no quantitative data is available to verify the extent of this .

Currently, the most significant cause of forest degradation is uncontrolled harvesting for subsistence forests products by rural populations. As can be expected, forest types found in communal areas tend to have the highest levels of this degradation. It is well documented that rural poverty is associated with high dependency on subsistence forest resources (Shackleton *et al*., 1999; Pandey, 2002; Arnold and Persson, 2003; INR 2003; Shackleton *et al.,* 2007). While poverty is likely to increase dependency on forest resources, this has been exacerbated by the break down in traditional resource use control mechanisms in many communal areas of South Africa. The democratisation of South Africa has been accompanied by loss in the powers of traditional leaders who have historically been responsible for regulating resource use of local forests (DWAF 1999; Von Maltitz *et al*., 1999; Obiri and Lawes, 2002; de Villiers and White, 2002; DWAF, 2003; Von Maltitz *et al.,* 2003). Rising levels of rural poverty suggest that this trend will increase in the future (INR, 2004, DWAF, 2005a).

Quantitative methods of monitoring forest degradation are urgently needed in South Africa. Potential methods include remote sensing (see for example Lambin, 1999), fixed point digital photographic assessment, (Hall, 2001) and indices of integrity based on key indicators species (see for example Carignan, and Villard, 2002; Biggs *et al*. 2006)

2.4 How fragmented are forests?

2.4.1 Introduction

Habitat fragmentation significantly increases vulnerability to biodiversity loss (Saunders *et al.,* 1991; Murcia, 1995; Laurance, 2000) and is therefore an important consideration when setting conservation planning priorities.

Globally, but particularly in the tropics, fragmentation is one of the greatest threats to forest biodiversity, (Saunders *et al.,* 1991; Solé, *et al*. 1991; Bierregaard *et al.,* 2001; Lawes *et al*., 2006; Kirika *et al.,* 2007). The problems associated with habitat destruction, that caused the fragmentation in the first place, are further compounded by the inability of remaining forest fragments to support viable populations, this especially so for larger vertebrates. Where fragments have become too small to support viable populations, or where dispersal between fragments is restricted and distances between fragments are large, local extinctions occur (Saunders, *et al.,* 1991; Solé, *et al*. 2004; Lawes, *et al*. 2006). In addition, fragmentation increases the area exposure of the forest to the surrounding matrix, leading to numerous edge effects that alter the conditions of the outer areas of the fragment, greatly reducing the amount of true forest interior habitat (Murcia, 1995; Krüger and Lawes, 1997; Laurance, 2000).

The high degree of natural fragmentation of South African forests are primarily the result of the repeated and often drastic changes in palaeoclimate, particularly since the Quaternary period that lead to successive periods of shrinkage and expansion of forests (Deacon and Lancaster, 1988; Partridge *et al.,* 1990; Lawes, 1990; Partridge *et al*. 1993; Eeley, *et al.,* 1999; Lawes *et al.,* 2000). In more recent times, human impacts in the form of fires, clearing and harvesting (Cooper, 1985; Geldenhuys and MacDevette, 1989; Geldenhuys, 1991; Geldenhuys, 1994; Lawes *et al.,* 2006) have exacerbated fragmentation. This being particularly evident by complete loss of many smaller forest patches (Kotze and Lawes, 2007), shrinkage of larger patches (Cooper and Swart, 1992), and the transformation of the forest matrix (Lawes *et al.,* 2004b; Berliner *et al.,* 2006). Coastal forests, historically contiguous, have been particularly affected by anthropogenic fragmentation over the last 100 to 150 years mostly due to expansion of plantation forestry and sugar cane farming.

Because South African forest faunal and floral communities have evolved under fragmented conditions and have experienced a number of ecological or climatic extinction filtering events (*sensu* Balmford 1996), it has been suggested by Lawes, (1990) and Lawes *et al*. (2007a), that remnant species may be inherently resilient to further fragmentation effects. Certainly, as can be expected with fragmentation related extinction filtering, faunal communities in South Africa's Afromontane forests are dominated by generalist species (Wirminghaus and Perrin 1993; Wethered and Lawes, 2005). Historically fragmented forests tend to show less edge and area-effects than recently fragmented forests (Laurance, 1991; Laurance, 1997; Laurance, 2000, for Australian forests, and Lawes *et al.,* 2005, for South African Afrotemperate forests). However, palaeoclimatic induced fragmentation occurred in the absence of any anthropogenic disturbance of the landscape (in particular matrix transformation), and species movements between patches would have been unimpeded. Today transformation of the matrix makes species movement between patches difficult, and in many cases distances between patches have increased, with the loss of smaller forest patches that would have served as stepping stones between larger forests (Kotze and Lawes, 2007).

The purpose of this section is to investigate patterns and differences in forest fragmentation between the forests types considered

2.4.2 Method

Fragmentation can be measured in various ways. Essentially these measures can be grouped as either structural or functional metrics (McGarigal *et al*. 2002). Structural metrics are defined as measures of the physical composition or spatial configuration of the patch mosaic without explicit reference to ecological process (for example edge to area ratio). Functional metrics measure landscape pattern functionally relevant to the organism or process under consideration (for example inter-patch connectivity). Measurements of fragmentation tend to be sensitive to changes in the grain size (spatial resolution), and the extent of the landscape under investigation (Turner, 2005).

Three metrics were used to compare relative degrees of fragmentation of South African forest types. These included: mean patch edge to area ratio; mean patch size; and mean isolation index. Input data for fragmentation metrics was calculated from GIS analysis of the forest cover layer (see section 1.5).

Mean patch edge to area ratio

A mean edge to area index was computed for each forest type by taking the sum of the forested area and dividing it by the sum of the perimeter distance of all patches of a forest type (see section 7.2.2 as well).

Mean patch size and size class distribution

Mean patch size and their distribution across size classes was calculated for each forest type.

Mean isolation index

Isolation/proximity refers to the tendency for patches to be relatively isolated in space from other patches of the same class, and an isolation index is a measure of how clumped or scattered patches are across its distribution range. Because the notion of 'isolation' is vague, there are many possible measures depending on how distance and patches are defined. The simplest measure is the mean nearest-neighbour distance over all patches, however because small patches may provide less landscape connectivity than larger patches, an alternative measure was used that incorporated forest area. This was done by computing the total additional forested area in 5 km buffers around each forest. Because the area of each buffer varies according to the perimeter of the patch, the total areas were adjusted by dividing by the perimeter of each buffer. FRAGSTAT software (McGarigal *et al.,* 2002), running within ARC GIS was used to calculate the fragmentation indices.

Aggregation of indicators

To enable integration into a single fragmentation index, each metric was transformed into common units according to their 'fragmentation effect'. This was done by scaling values along a linear one to five scale, according to fragmentation effect (i.e. 1 = very low fragmentation effect, and 5 = very high fragmentation effect). The assumption was made that each metric contributed equally to the overall fragmentation effect, allowing for the mean of all three to be computed for each forest type.

2.4.3 Results

Forest patch sizes

The approximate area of 492 700 hectares of forest biome are made up of an estimated 16171 patches, giving a overall mean patch size of 30.4 hectares, however this varies considerably between forest types The distribution frequency of forest patches across size classes is heavily skewed towards the smaller patch sizes, and for most forest types over 80 % of patches are smaller than 50 hectares (see Table 4, below).

Table 4. Numbers of forest patches in size classes, and mean patch size for each forest type.

Forest Type	Numbers of forest patches in size classes (ha)									Mean patch size (ha)
	>1	1-50	50-100	100-200	200-400	400-800	800-1600	1600-3200	>3200	
Albany	0	447	25	19	4	3	2	1	1	44
Amatole Mistbelt	326	1320	99	67	38	19	3	1	1	34
Drakensberg Montane	0	128	5	2	0	0	0	0	0	14
Eastern Cape Dune	1	93	22	8	3	1	1	0	1	84

Eastern Mistbelt	81	1154	84	46	22	11	4	0	0	30
Eastern Scarp	18	1355	39	14	11	2	1	3	1	23
Kwazulu-Natal Coastal	216	485	22	12	4	7	4	0	1	28
Kwazulu-Natal Dune	48	151	15	7	3	3	0	1	1	54
Licuati Sand	0	288	29	20	8	5	5	2	0	68
Lowveld Riverine	0	34	16	8	7	1	3	1	0	163
Mangrove	1	5	0	2	0	0	2	0	0	239
Mpumalanga Mistbelt	74	740	38	43	17	6	6	1	0	35
Northern Kwazulu-Natal Mistbelt	0	274	10	5	0	0	1	0	0	18
Limpopo Mistbelt	187	364	34	18	7	7	3	1	0	31
Pondoland Scarp	49	306	26	9	6	1	1	1	0	32
Southern Cape Afrotemperate	52	1246	89	66	34	18	8	7	0	50
Swamp	0	26	4	2	2	1	1	0	0	84
Transkei Coastal Scarp	1519	1843	156	59	31	12	4	1	0	17
Transkei Mistbelt	217	1126	81	42	15	4	0	0	0	20
Western Cape Afrotemperate	0	247	8	1	2	3	0	0	0	18
Western Cape Milkwood	0	139	9	1	1	0	0	0	0	17

Isolation index

An isolation index for each forest type was calculated by calculating the mean area of neighbouring forest patches, (within 5 km buffers) adjusted for differences in buffer area by dividing by buffer perimeters. The lower the forested area within neighbouring buffers, the higher the isolation index. Results are presented in Table 5.

Table 5 Index of isolation for each forest type. Calculated from mean area of neighbouring forest patches within the 5km buffers, divided by perimeter of buffers

Forest type	Isolation index
Lowveld Riverine	0.077
Drakensberg Montane	0.066
Low Escarpment Mistbelt	0.106
Western Cape Milkwood	0.089
Western Cape Afrotemperate	0.143
Swamp	0.154
Eastern Cape Dune	0.291
Eastern Scarp	0.372
Albany Coastal	0.521
Licuati Sand	0.548
Mangrove	0.522
Eastern Mistbelt	0.516
Kwazulu-Natal Dune	0.692
Southern Cape Afrotemperate	1.379
Mpumalanga Mistbelt	0.987
Pondoland Scarp	2.047
Transkei Mistbelt	0.939

Amatole Mistbelt	2.930
Transkei Coastal	3.732
Limpopo Mistbelt	3.573
Kwazulu-Natal Coastal Average	7.052

Edge to area index

The mean edge to area ratio for each forest type (in meters of forest perimeter per hectare of forest area), is given in Table 6 for each forest type.

Table 6. Mean edge to area rations of forest types. Calculated from mean total length of forest perimeter (in meters) divided by mean area (hectors) for each forest type

Forest type	Edge to area (m/ha)
Lowveld Riverine	114.8
Eastern Cape Dune	146.5
Swamp	160.2
Drakensberg Montane	163.6
Low Escarpment Mistbelt	165.1
Albany Coastal	167.5
Licuati Sand Average	181.2
Western Cape Milkwood	191.2
Western Cape Afrotemperate	195.9
Eastern Scarp	201.7
Southern Cape Afrotemperate	240.0
Eastern Mistbelt	264.0
Kwazulu-Natal Dune	386.6
Pondoland Scarp	387.0
Mpumalanga Mistbelt	414.7
Amatole Mistbelt	434.2
Mangrove	456.2
Transkei Mistbelt	464.5
Limpopo Mistbelt	616.6
Kwazulu-Natal Coastal	636.9
Transkei Coastal	801.5

Aggregated scaled fragmentation index

An aggregated fragmentation index for each forest type was derived by taking the arithmetic mean of the scaled indices of edge to area ratio, isolation index and mean patch size for each forest type. This provides a relative ranking of the degree of fragmentation of each forest type. Results presented in **Error! Reference source not found.**.

Table 7. Fragmentation of forest types as measured by three fragmentation metrics: edge to area ratio; isolation index, and mean patch size. Values scaled from one (low fragmentation effect) to five (high fragmentation effect). Forest types ranked in order of most fragmented.

Forest type	Mean edge to area scaled index	Mean isolation scaled index	Mean patch size scaled index	Aggregated fragmentation index
Drakensberg Montane	1.0	5.0	5	3.7
Transkei Coastal	5.0	0.1	5	3.4
Western Cape Milkwood	1.2	3.9	5	3.4
Lowveld Riverine	0.7	5.0	4	3.4
Western Cape Afrotemperate	1.2	3.2	5	3.1
Mangrove	2.8	0.9	5	2.9
Kwazulu-Natal Coastal	4.0	0.1	4	2.7
Limpopo Mistbelt	3.8	0.1	4	2.7
Transkei Mistbelt	2.9	0.4	4	2.4
Amatole Mistbelt	2.7	0.2	4	2.3
Pondoland Scarp	2.4	0.4	4	2.3
Eastern Scarp	1.3	1.4	4	2.2
Eastern Mistbelt	1.6	0.7	4	2.1
Swamp	1.0	2.2	3	2.1
Kwazulu-Natal Dune	2.4	0.6	3	2.0
Low Escarpment Mistbelt	1.0	4.0	1	2.0
Licuati Sand	1.1	1.2	3	1.8
Albany Coastal	1.0	1.3	3	1.8
Mpumalanga Mistbelt	2.6	0.5	2	1.7
Southern Cape Afrotemperate	1.5	0.6	3	1.7
Eastern Cape Dune	0.9	1.6	2	1.5

2.4.4 Discussion

Forest types most vulnerable to the negative effects of fragmentation are those with relatively small patch sizes, with a high edge to area exposure, and where patches are far apart and isolated by transformed matrices. An important aspect of fragmentation, particularly in inherently patchy habitats, is the degree of isolation and hence ecological connectivity between forests patches. Although inherently fragmented, many forests have become increasingly isolated due to land transformation of the surrounding matrix. According to these criteria, Drakensberg Montane, Transkei Coastal, and Western Cape Milkwood forests are the most fragmented forest types.

Fragmented habitats are inherently more vulnerable to biodiversity loss than intact habitats and this is likely to be more so if the fragmentation is recent and can be attributed to anthropogenic causes (Saunders *et al.,* 1991; Solé, *et al*. 1991). With current data available, it was not possible to differentiate between anthropogenic induced and natural fragmentation due to past palaeoecological changes. In some cases it may be possible to infer anthropogenic fragmentation where forest patches are surrounded by transformed matrices, however this cannot exclude the possibility that these forest were already fragmented, perhaps by fires, prior to the matrix being transformed by humans.

Anthropogenic transformation not only changes the spatial configuration of forest patches relative to each other (Haila, 1999) but also the functional and ecological relationships between patches (Ricketts, 2001). Transformed matrices act either as dispersal barriers or as selective filters to the movement of organism between patches. In either case, this ultimately leads to changes in species composition and ecosystem structure (Fahrig, 1985, Ricketts, 2001; Wethered and Lawes, 2003; Banks *et al.,* 2005).

Fragmentation highlights two issues pertinent to the conservation and management of forest biodiversity. Firstly, forest types that are highly fragmented will have relatively small core areas (the portion of the forest unaffected by edge effects). To illustrate the importance of this, compare the fragmentation indices for Transkei Coastal forests with those of Amatola Mistbelt forests. Both have a similar total area, but with very different edge to area ratios (see Table 4**Error! Reference source not found.**, above). The former will have less core habitat than the latter, and effectively less intact area available to meet conservation targets. In highly fragmented forests, edge habitat can make close to half of the total area. For example, Lawes *et al.,* (2004a) found that as much as 41% of KwaZulu–Natal's Afromontane forests could be considered as edge habitat. This was based on the calculation of Kotze and Samways (1999) that edge effects for epigaeic forest invertebrates extend as far as 32 meters into the forest interior. Not only does this emphasise the importance of conserving the largest forest patches, with the lowest edge to area ratio, but also suggests that area requirements for conservation targets should be set according to core area, and not total area, (or in other words, conservation targets may need to be adjusted upwards where edge to area ratios are high). Secondly, where forest types consist of many isolated patches (i.e. showing a high isolation index), such as Drakensberg montaine forests, the smaller patches, although they seldom contain unique species, (Wethered and Lawes 2003), should be given a higher conservation value than what may be implied from their size alone. This is because of their functional value in maintaining ecological processes within the landscape, in particular their role in providing ‘stepping stone’ habitat for species dispersal and re-colonization (Forman, 1995; Lawes *et al.,* 2004b; Turner, 2005; Kotze and Lawes, 2007). In addition, isolated forests are highly sensitive to factors that affect species dispersal, such as transformation of the forest matrix.

CHAPTER 3: THE STATE OF FOREST SPECIES

3.1 Overview

Understanding the conservation status of habitat and the species sheltered therein is an important step for planning their conservation (Given and Norton 1993; Burgess, *et al*., 2006). South African forests provide critical habitat for a large number of species, many of which are rare or endangered (Geldenhuys and Macdevette, 1989; Castley, and Kerley, 1996; Castley, 1997). This chapter provides a quantitative and comparative assessment of botanical richness, and the conservation status of rare and threatened flora and fauna occurring in South African forests. An evaluation of species richness, phylogenetic richness (as indicated by generic and family richness), endemism and numbers of unique species within forest types provide important information for setting conservation priorities particularly at the level of species.

3.2 Forest biome plant species richness

3.2.1 Introduction

Regional and local diversity of South African forest tree species declines in a south-westerly direction, from the northeast of the country to the Cape peninsular (Geldenhuys, 1989; Geldenhuys & MacDevette, 1989; Geldenhuys, 1992; Midgley *et al*., 1997). For example, coastal forests in KwaZulu-Natal have higher richness than Coastal forest of the Western Cape. This pattern is consistent with a pole ward decline in richness observed for many other groups of organisms (Midgley *et al.,* 1997; Rosenzweig, 1995). Alpha diversity also declines with increasing altitude, so that coastal forests invariably have higher diversity than montane forests at the same latitude (Geldenhuys and MacDevette, 1989; Geldenhuys, 1992; Geldenhuys, 1993; Midgley *et al*., 1997; Mucina and Geldenhuys, 2006).

With exception of the succulent Karoo and nama Karoo biome, forests occur adjacent to all of South Africa's major biomes, they are therefore exposed to a high diversity of plant species on the margins and ecotones, as such forest margins may contain species common to adjacent biomes (Mucina and Geldenhuys, 2006).

The aim of this section is therefore to: compare vascular plant species richness of the forest biome relative to other vegetation biomes in South Africa; describe the plant species richness of the forest biome; and compare phylogenetic richness, species richness, endemism, as well as species uniqueness across each of the forests types.

3.2.2 Methods

All biome data, except for the forest biome was obtained from EWT (2002). The log transformed version of species numbers to area (of biome or forest type) were used to determine relative plant species richness (after Arrhenius, 1921; Williams, 1964; Rosenzweig, 1995; Cowling *et al.,* 1997).

Forest plant data derived from a number of different sources were integrated into a Microsoft Access database (Berliner and Wright 2008). The two most important unpublished data sources include: plant releve' data used for forest subtype classification, (compiled by Prof L Mucina, and arranged into forest types by Dr. P. Desmet) and the data sets used by Von Maltitz *et al*. (2003) for the national forest type classification.

Database queries were used to determine overall forest plant species richness (at family, genus and species level), overall tree species richness (at family, genus and species level), species richness per forest type, as well as numbers of unique occurring species for each forest type.

Because species accumulate at a declining rate with increased area sampled, the relationship between numbers of species to area sampled is not linear (Arrhenius, 1921; Williams, 1964; Rosenzweig, 1995; Cowling *et al*., 1997). To facilitate the comparison of species richness of forests types and biomes, the standard power function of the log of species to the log of area was used, with the degree of deviation from the fitted curve taken as an indication of the relative richness of each forest type or biome.

The comparative richness of South African forests relative to other temperate forest in other parts of the world was evaluated using the tree species richness data from Silander (2001).

3.2.3 Results

Comparison of forest biome plant species richness with other biomes

In Figure 3 below, the total numbers of vascular plants occurring within each of the major vegetation biomes of South Africa are presented as a bar graph.

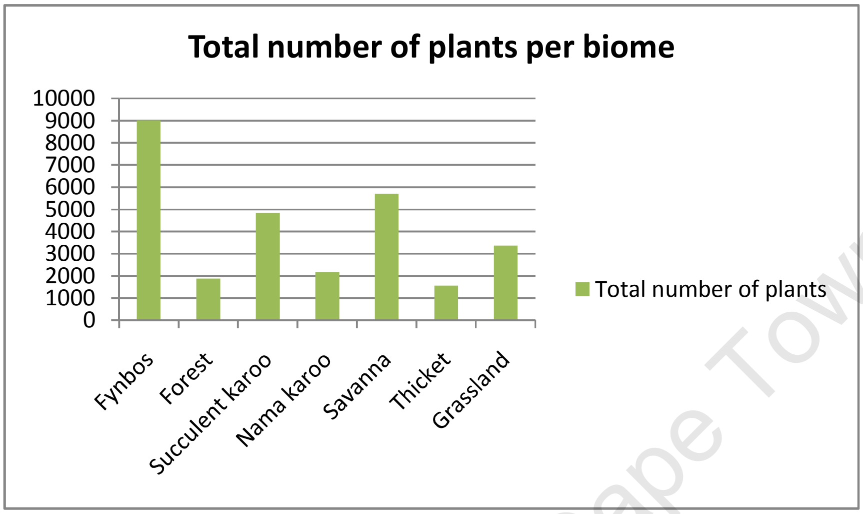


Figure 3. Total number of vascular plant species per biome. (All biome data, except for forests, are from EWT, 2002. Forest data is from this study).

Because the relationship between numbers of species to area sampled is non linear, the relative biome richness was determined by the degree of deviation from the fitted regression curve for the log of species number to the log of biome areas (see Figure 4, below).

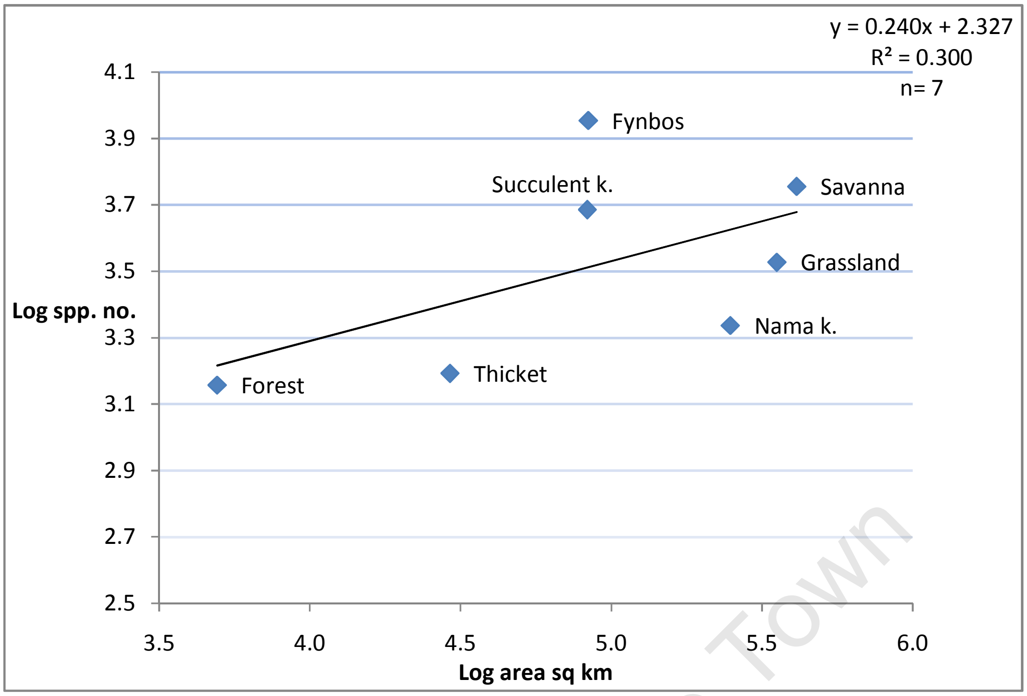


Figure 4. Linear regression of the log of number of species to log of area for South Africa's major vegetation biomes. All biome data, except for forests, from EWT (2002), forest biome data is from this study.

Back transformation of residuals derived from the regression curve in Figure 4, shows fynbos as the most species rich biome (approximately 18 species per km ^{2} higher than predicted by the regression curve), followed by succulent Karoo (approximately 7 species per km^2 higher); savanna (approximately 4 species per km^2 higher); forests (approximately 2 species per km ^{2} less, than predicted); grasslands (3 species per km^2 less, than predicted); succulent thicket (about 4 species per km^2 less than predicted), and nama karoo (about 9 species per km ^{2} less than predicted by the regression curve).

Description of forest biome plant species richness

The total numbers of vascular plant species, plant families and plant genera occurring in the forest biome of South Africa are given in the Table 8, below.

Table 8. Plant species richness of the forest biome,

Level	Total number	Ratios
Families	176	
Genera	780	Genera : Family: 4.4
Species	1885	Species: Genera: 2.4

Forty seven percent of all plant species occurring in forests come from just fifteen (8.5 %) of all occurring forest plant families; these families are given in Table 9, below, along with numbers of genera and species.

Table 9. Top fifteen forest occurring plant families (sorted by total number of species).

Family	Genera	Species
Fabaceae	53	122
Poaceae	50	102
Rubiaceae	33	99
Asteraceae	29	76
Celastraceae	18	64
Acanthaceae	20	55
Lamiaceae	11	52
Apocynaceae	25	50
Anacardiaceae	10	47
Euphorbiaceae	22	45
Orchidaceae	26	37
Gesneriaceae	1	37
Malvaceae	14	35
Cyperaceae	11	35
Pteridaceae	5	25

The most species rich plant genera occurring in forests are given in the
Table 10 below (these genera account for around 17 % of all forest occurring species).

Table 10.Top fifteen forest plant genera sorted by total number of species.

Genus	Species
Streptocarpus	37
Rhus	35
Plectranthus	32
Asplenium	24
Pavetta	22
Asparagus	21
Acacia	20
Ficus	19
Combretum	18
Gymnosporia	17
Maytenus	17
Diospyros	16
Aloe	14
Senecio	14
Eugenia	12

Description of plant species richness of forest types

Eastern Scarp forests have the highest number of species as well as the highest phylogenetic diversity, (as shown in numbers of families and genera) of all forest types considered (see Table 11, below).

Table 11. Plant species richness of forest types, ranked according total number of species. Taxon ratios for each forest type also given.

Forest Type	Families	Genera	Species	Genera: Family	Species: genera
Eastern Scarp	119	356	631	3.0	1.8
Pondoland Scarp	96	280	468	2.9	1.7
Transkei Mistbelt	110	278	459	2.5	1.7
Mpumalanga Mistbelt	94	228	385	2.4	1.7
Low Escarpment Mistbelt	87	212	331	2.4	1.6
Western Cape Afrotemperate	88	179	279	2.0	1.6
Albany Coastal	77	181	268	2.4	1.5
Eastern Cape Dune	61	153	245	2.5	1.6
Southern Cape Afrotemperate	91	166	241	1.8	1.5
Eastern Mistbelt	64	132	226	2.1	1.7
Kwazulu-Natal Dune	72	150	197	2.1	1.3
Licuati Sand	54	140	195	2.6	1.4
Kwazulu-Natal Coastal	56	135	194	2.4	1.4
Limpopo Mistbelt	65	129	188	2.0	1.5
Transkei Coastal	63	121	177	1.9	1.5
Amatole Mistbelt	69	119	161	1.7	1.4
Drakensberg Montane	65	105	157	1.6	1.5
Swamp	62	96	120	1.5	1.3
Lowveld Riverine	34	68	88	2.0	1.3
Western Cape Milkwood	24	28	29	1.2	1.0
Mangrove	12	15	15	1.3	1.0

Comparison of plant species richness of forest types

A comparison of species richness of forest types can be made by taking the linear regression of the log of the number of species (occurring within each forest type) to the log of the area of each forest type. Points above the regression curve represent forests types with higher richness than what is predicted from the fitted curve; for the given area of a forest type, and those below, lower richness (see Figure 5, and Table 12, below).

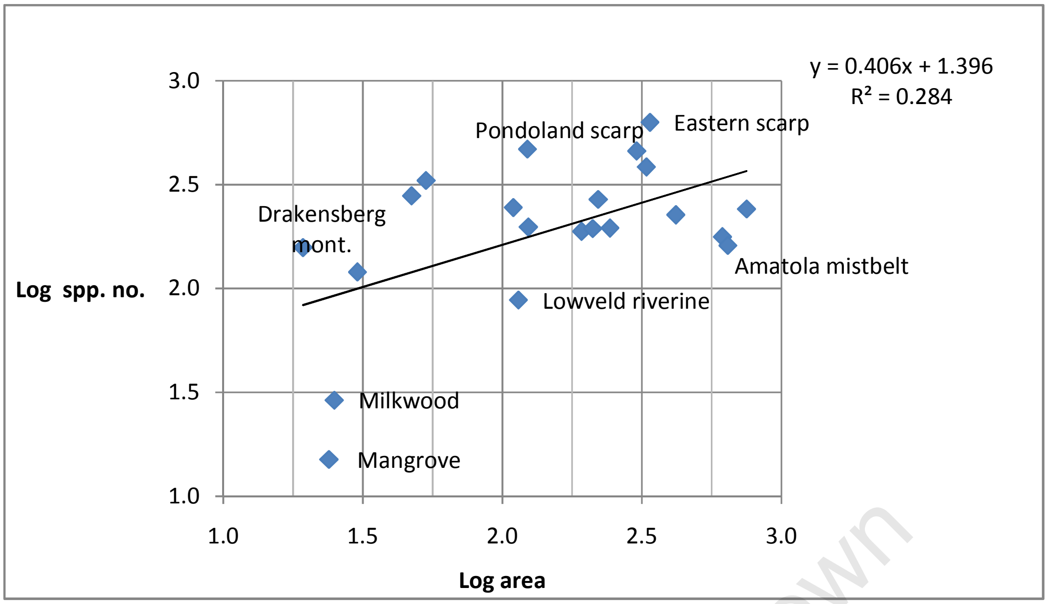


Figure 5. Forest type species-area relationships using log transformation to determine relative species richness (vascular plants only).

Table 12. Plant species richness by area of each forests type. Ranked using deviation from fitted log transformed species-area curve (derived from figure 5).

Forest type	Area (km2)	No of plant species	Deviation from fitted curve (log species no)
Pondoland Scarp	122.8	468	0.43
Low Escarpment Mistbelt	53.2	331	0.42
Eastern Scarp	337.5	631	0.38
Western Cape Afrotemperate	47.3	279	0.37
Drakensberg Montane	19.3	157	0.28
Transkei Mistbelt	302.5	459	0.26
Mpumalanga Mistbelt	327.7	385	0.17
Eastern Cape Dune	109.4	245	0.17
Swamp	30.2	120	0.08
Albany Coastal	220.5	268	0.08
KwaZulu-Natal Dune	124	197	0.05
Limpopo Mistbelt	192	188	-0.05
KwaZulu-Natal Coastal	210.9	194	-0.05
Licuati Sand	242.8	195	-0.07
Eastern Mistbelt	418.4	226	-0.11
Southern Cape Afrotemperate	748.5	241	-0.18
Transkei Coastal	614.8	177	-0.28
Lowveld Riverine	114	88	-0.29

Amatole Mistbelt	642.2	161	-0.33
Western Cape Milkwood	25	29	-0.50
Mangrove	23.9	15	-0.78

Unique occurring plant species per forest type

The number of species and the proportion of species unique to a forest type vary widely. Although there is a relatively high overlap in tree species between some forest types, others are high in overall species number and in numbers of unique species, these forest are of high conservation importance. In Table 13, below the total numbers of plant species and those that are unique to the forest type are shown.

Table 13. Number and proportion of plant species unique to each forest type.

Forest Type	Total plant species number	Number plant species unique to forest type	% unique plant species of total occurring
Mangrove	15	10	67
Lowveld Riverine	88	39	44
Licuati Sand	195	72	37
Western Cape Afrotemperate	279	102	37
Pondoland Scarp	468	166	35
Western Cape Milkwood	29	10	34
Low Escarpment Mistbelt	331	108	33
Swamp	120	37	31
Eastern Scarp	631	161	26
Southern Cape Afrotemperate	241	52	22
Transkei Mistbelt	459	91	20
Drakensberg Montane	157	31	20
Mpumalanga Mistbelt	385	62	16
Albany Coastal	268	39	15
Amatole Mistbelt	161	19	12
Kwazulu-Natal Dune	197	23	12
Limpopo Mistbelt	188	18	10
Eastern Mistbelt	226	21	9
Transkei Coastal	177	13	7
Kwazulu-Natal Coastal	194	12	6
Eastern Cape Dune	245	11	4

3.2.4 Discussion

Forest biome species richness

Although the forest biome does not have the plant species richness of some of the other vegetation biomes of South Africa, it is nevertheless one of the smallest, and if considered per unit area of biome it has the highest density of plant species per unit area (0.38 plant species per km^2, with the next highest being the fynbos biome with 0.11 plant species per km^2). However, because of the non linear relationship between species accumulation and area sampled, this comparison is more correctly represented using the deviation from the linear region of the log form of species to area. Using this method, the forest biome ranks fourth out of the seven biomes considered, in terms of relative plant species richness.

A comparison of tree species richness of the South African forests with tree species richness of other temperate forests of the world, shows south African forests to have the second highest number of tree species, second only to East Asian forests, but these are orders of magnitude larger than South African forests (see Table 14, below). If species richness is considered per unit area of forest, South African forests have by far the highest number of tree species of any temperate forest in the world (Silander, 2001; Cowling, 2002).

Table 14. Tree species richness of South African forest compared to temperate forest from other parts of the world. All data used is from Silander (2001), except for South African forests (marked with *), where data is from this thesis. Forest biome maximum extent uncertain (Note that numbers of tree species for South Africa is likely to be higher than indicated in the table, as not all species in the database used, provided information on growth form i.e. trees, shrub etc).

Region	Families	Genera	Species	Genus : family	Species : genus	Forest biome maximum extent ($Km X10^{3}$)
South Africa*	86 +	266 +	501 +	3.1	1.9	?
S. E. South Africa	88	280	598	3.2	2.1	20-50
Europe	21	43	124	2.0	2.9	3300-3910
East Asia	67	177	876	2.6	4.9	3210 -3720
Eastern North America	46	90	253	2.0	2.8	3560-3720
Western North America	24	47	131	2.0	2.8	700-1000

Chile	29	40	83	1.4	2.1	330-370
Southern Brazil	25	45	77	1.8	1.7	<100
Southeast Australia	37	78	331	2.1	4.2	300-700
New Zealand	47	74	212	1.6	2.9	230-250

Plant species richness per forest type

Geldenhuys (1992; 1997) suggest that the relatively high degree of tree species shared between forest types, implies that many of these species must of established before major fragmentation of the biome (at least since the late Miocene), and that forests survived in areas now considered as dispersal corridors during periods of forest retreat and fragmentation. Furthermore, the number of dispersal corridors (mountain chains, escarpments, river valleys, coastal dune systems) meeting in a particular forest, appears to be one of the most important variables determining the numbers of woody plants. This view is not held by Lawes (2007a) and Lawes (personal communications, June, 2009), who believes that forests were isolated by climate change and that species richness is rather a consequence of selectively climatic filtering combined with allopatric speciation, primarily a result of the isolation. Further, this would not be possible if forests were a loosely connected network of corridors in the late Miocene. That forests appear to lie along corridors of potential dispersal activity is coincidence borne of topography and climate limits on where forests can and will establish.

The alternative views of Lawes, (2007a) and Geldenhuys (1992; 1997) are not necessary incompatible, as it may well depend on the dispersal ability of the taxa concerned. Geldenhuys (1992; 1997) dispersal corridors seem feasible to explain the spread of wind dispersed trees, that are likely to follow wind patterns along topographic corridors, whereas Lawes's, (2007a) allopatric speciation theory, will hold for either poorly dispersed taxa (such as invertebrates), highly prone to isolation, and therefore more like to undergo speciation; or for highly mobile taxa like mammals, that are less depended on topographic corridors for dispersal.

The species richness ranking of forests types remains largely constant irrespective if ranked by number of families, genera or species. This implies that overall species richness of forest types are good predictors of relative richness at higher taxonomic levels (i.e. phylogenetic richness). A comparison of species richness between forest types brings up interesting questions regarding the phylogenetic relationships and palaeoecological origins of the forest types. The most species rich forest types tend to be the Scarp and Mistbelt forest types. This is not surprising as these forest types contain species from both Afromontane and coastal forest groups well as many unique species. Forest such as Eastern Scarp and Pondoland Scarp, have exceptionally high levels of endemism, as well as species, and phylogenetic richness, and are therefore of extreme conservation importance.

Taxon ratios

Of particular interest to conservation biologists is the conservation of phylogenetic diversity. A relatively simple way of evaluating forest types according to their phylogenetic diversity is by considering the ratio of species-to-genus and species-to-families. Interpretation of differences in these ratios is by no means clear. Typically, they haves been used to describe community patterns and to infer levels of competitive interactions among species within genera (MacArthur and Wilson 1967; Järvinen, 1982), as well as rates and time available for evolutionary speciation (Rohde, 1992; Schluter, 2000) . A low species-to-genus ratio has been interpreted as a product of strong intra-generic competition (Elton, 1946), which limits con-generic coexistence (Darwin, 1859). Consistent with this hypothesis is the observation that species-to-genus ratios are usually smaller for islands than mainland communities (Elton, 1946). However, Gotelli and Colwell (2001) have shown that subtaxon to taxon ratios are an increasing function of sample size, and would be expected to decrease in small communities, regardless of the level of competition. This is also born out by the strong correlation that I found between the total number of plant species occurring in a forest type and the ratio of genera-to-families, or species-to-genera (correlation coefficient of 0.83 and 0.85 respectively, $p < 0.05$). According to Eeley *et al.* (1999), scarp forests have acted as refugia during periods of unfavourable climatic change. This may account for species accumulations in scarp forest that have both higher total species number, and higher species to genera ratios than any other forests.

Species to genus ratios of South African forest were found to be relatively similar across different forest types, but were slightly higher in forest types with higher species diversities (see Table 11). Species to genus ratios of Southern African forest are generally low compared to tropical forest (see table 14, and Geldenhuys, 1990). High species to genus ratios are thought to reflect greater evolutionary time and evolutionary rates (Rohde 1992). This is supported by the observation that certain species rich genera, such as *Isoglossa and Plectranthus,* for example, tend to be shrubs or small trees with relatively short generation times (Midgley *et al*., 1997). Lawes (personal communication, June, 2009) believes that high intra-generic competition, associated with isolation, could be a partial explanation for the low species to genus ratio of South African forests relative to other temperate forests

Unique species

The results of the comparison of numbers of unique plant species occurring within each forest type are interesting in that some forest types with the lowest total numbers of species, such as Mangroves, Sand, and Lowveld Riverine forests, also have the highest proportion of species that are unique to those forest types. There is also a strong correlation between total number of species occurring in a forest type, and the total number of species unique to that forest type (correlation coefficient of 0.82, $p < 0.05$).

3.3 The conservation status of forest plants

3. 3.1 Introduction

Trends in numbers of red data species are potentially useful as indicators of conservation status of forest types, and for the biome as a whole (Butchart *et al*., 2004; Butchart *et al*., 2007). The aim of this chapter is to describe the conservation status of forest types by determining the number of plants in each of the IUCN red list categories.

3.3.2 Methods

The database developed for South African forest species (Berliner and Wright, 2008) was cross referenced against national (Raimondo *et al.,* 2009) and international (IUCN, 2007) red data listings. Although all available data sources were used, the plant data base is not complete, particularly with regard to non-vascular plants (bryophytes and lichens).

3.3.3 Results

Overall 56 species of vascular plants listed as IUCN red data species occur in forests, two are extinct in the wild, four are critically endangered, eight endangered, 20 vulnerable, and 22 are near threatened. The full list of threatened forest plant species are given in appendix E. A summary for each forest type is given in Table 15, below.

Table 15. Numbers of red data vascular plants occurring in each forest type (status according to the South African National Biodiversity, 2007 Interim Global Status assessments). Abbreviations: EW, extinct in the wild; CR, critically endangered; EN, endangered; VU, vulnerable; NT, near threatened.

Forest Type	EW	CR	EN	VU	NT	Total	SA endemics
Pondoland Scarp	1	1	5		12	19	35
Eastern Scarp	1	1	2	6	5	15	13
Transkei Coastal			1	3	3	7	8
Transkei Mistbelt		1		1	2	4	4
Albany Coastal		1		1	1	3	4
Mpumalanga Mistbelt				3		3	4
Western Cape Afrotemperate			1	1	1	3	7
Amatole Mistbelt				2		2	1
Kwazulu-Natal Dune				1	1	2	3
Drakensberg Montane					1	1	3
Eastern Cape Dune					1	1	2
Southern Cape Afrotemperate					1	1	1
Swamp				1		1	0
Eastern Mistbelt						0	0
Kwazulu-Natal Coastal						0	0
Licuati Sand						0	0
Limpopo Mistbelt						0	0
Low Escarpment Mistbelt						0	1
Lowveld Riverine						0	1
Mangrove						0	0
Western Cape Milkwood						0	0

3.3.4 Discussion

Forest red data plants occur across all growth forms, although almost half are shrubs or small trees (see Figure 6, below). Seven of these are *Encephalartos* species, while these are not usually considered true forest species; they are often associated with forests, occurring on forest margins or rocky outcrops adjacent to forests.

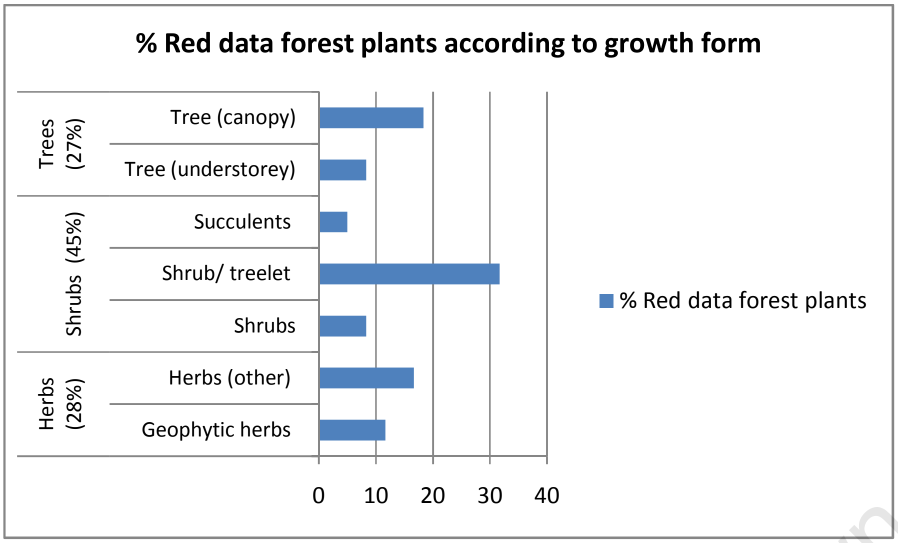


Figure 6. Percentages of red data forest occurring plants by growth form.

Most of the red data plants occur in coastal and scarp forest types, with just two forest types, Pondoland scarp and Eastern scarp containing more than half of all recorded forest red data plants. This assessment again, highlights the conservation importance of Eastern Scarp and Pondoland Scarp forest, both with low levels of formal protection (see chapter 4), and facing increased pressure, primarily from subsistence harvesting (see chapter 5).

3.4 The conservation status of forest fauna

3.4.1 Introduction

Forests provide important habitat to a large number of vertebrate and invertebrate animal species that use them for food, shelter, breeding and foraging.

Many of the mammals and a significant proportion of the birds occurring in South Africa's forests are not confined to forest habitat and often are widely distributed within different forest types (Geldenhuys and MacDevette, 1989; Castley and Kerley, 1996). In addition, many forest birds make seasonal movements between, and into forests, for example the Spotted Ground-thrush (Zoothera guttata) migrate to forests at lower altitudes during winter (Barnes, 2000). The fact that birds may move large distances between forests, is significant as birds play an important role in pollination and seed dispersal of many forest plants (Koen and Crowe, 1987; Koen, 1992; Geldenhuys, 1996). Bird species richness is relatively low in forests, compared to other habitats, and like plants, also shows a decline from north to south (Koen and Crowe, 1987). For example, moving southwards from the Eastern Cape, 54 bird species have been recorded in Dwesa forest, 43 in Alexandria forest, 35 in Diepwalle forests (Knysna), and only 15 in the forest patches of the Cape Peninsula (Cody, 1983).

Forest mammals generally occur solitary (e.g. leopard, bushbuck) or in small groups, and are often nocturnal. Mammals play important roles in forest dynamics and natural disturbance regimes. For example, herbivores play important roles in controlling understory plant growth (Lubbe, 1990), seed predation and seed dispersal (Koen and Crowe, 1987; Geldenhuys, 1996).

Surveys indicate that South African forests have very high invertebrate diversities with high levels of endemism. For example, the forest biome has the highest number of endemic spiders of all biomes, with 47 families represented by about 312 species (SANSA, 2007). In just one region, the coastal dune forests of Richards Bay, KwaZulu-Natal, over 96 different species of spider were identified (Dippenaar-Schoeman and Wassenaar, 2006).

A large proportion of forest fauna, even those considered as typical 'forest species' will make use of the forest - matrix ecotone and surrounding areas (Geldenhuys and MacDevette, 1989; Armstrong, 2003; Lawes, 2007b; Wethered and Lawes, 2003). These species are often considered as 'forest occurring' or 'forest associated', rather than 'forest dependent' species (Kikira *et al*., 2007; Lawes, 2007 b). See Table 16, below.

Table 16. Approximate number and percentages of forest fauna categorized as forest dependent (species that live and reproduce only in forest habitats), or forest associated (species that inhabit forest but utilise other habitats). Data derived from Lawes *et al*., (2007).

Group	Numbers of species	% forest dependent	% forest associated
Birds	150	29	71
Mammals	84	19	81
Frogs	32	44	56
Sum		28	72

Many forest associated fauna, because they typically move between patches and the matrix, play particularly important roles in seed dispersal of many forest species. For example up to 90% of all tree species in tropical forests are dispersed by frugivorous animals (Tabarelli and Peres, 2002), approximately 33 % for mixed tropical-subtropical forest (Hall and Swaine, 1981) and around 30% for Southern Cape Afrotemperate forests (Gelenhuys, 1993).

The aim of this section is to determine the conservation status of forest occurring fauna, to make comparisons between biomes and across different forest types.

3.4.2 Methods

A database of faunal species occurring in South African forests was compiled from various available sources (refer to notes after Table 17). These were then checked against national and international red data listings, and integrated into an Microsoft Access database (Berliner and Wright, 2008). Comparisons were then made across biomes.

Because of incomplete data, the species richness of forest fauna could not be determined, however, sufficient information was available to evaluate the conservation status of on red data forest faunal species.

The data presented here was the most comprehensive available to me at the time of analysis. However it needs to be pointed out that it is far from complete and not all taxonomic groups, in particular forest invertebrates, have been well surveyed or assessed for red data status.

3.4.3 Results

Approximately 89 forest faunal species are listed as IUCN red data species, 11 are critically endangered 21 endangered, 32 vulnerable, with one extinction, Eastwood's Long-tailed Seps (*Tetradactylus eastwoodae*), last collected in 1914 from Woodbush forest (Branch, 1988). A full list of threatened forest fauna are given in appendix F, and have been summarised in Table 17, below.

Table 17 Numbers of red data animals occurring in South African forests. Threat status are according to global IUCN, 2007 classifications, except for mammals, where South African red data book (Friedmann and Daly, 2004) was used (compiled by Berliner and

Wright, 2008). Note: forest dependent species are considered to be species that breed predominantly in forests, or whose distribution overlaps at least 80% with those of forests

Group	EX	CR	EN	VU	NT	Total	Endemic to SA	Forest dependent	Forest dependent and endemic
Mammals[1]		1	5	7	8	21	6	11	3
Birds[2]		0	2	6	8	16	14	5	4
Amphibians[3]		2	3	2	2	9	9	5	5
Reptiles[4]	1		3	3	3	10	9	8	7
Crustacea (crabs)[5]				1		1	1	1	1
Gastropoda (snails)[5]		5	4	4		13	13	13	13
Odonata (butterflies)[5]		1		1	2	4	4	4	4
Diplopoda (millipedes)[5]		1	5	7		13	13	13	13
Onychophora (velvet worms)[5]		1	0	1		2	2	2	2
Sum	1	11	22	32	23	89	71	62	52

Original data sources:
1 Friedmann and Daly, 2004.
2 Barnes, 2000.
3 Minter, *et al.,* 2004
4 Branch, 1988, except for chameleons: Reisinger *et al.,* 2006, and Dr.K.Tolley pers. coms.
5 IUCN Global 2007 classifications from: www.iucnredlist.org

Red data species make up a significant proportion of the total number of faunal species occurring in forests, and for mammals and amphibians, these proportions are the highest of any biome (see Figure 7, below).

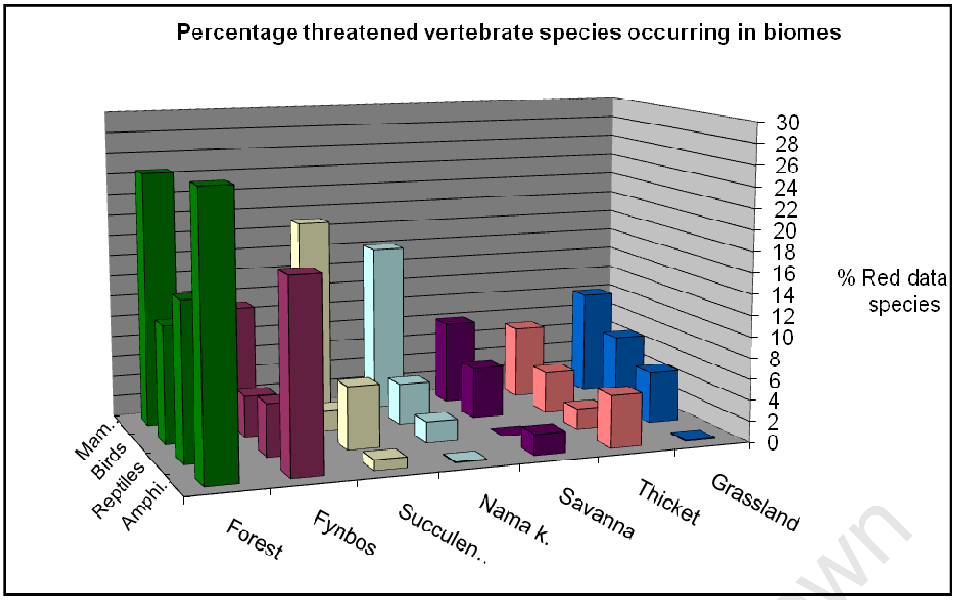


Figure 7. Percentage of vertebrate species occurring in each biome that are listed as threatened. (Forest data compiled by Berliner and Wright, 2008, data for other biomes from EWT, 2002).

Relative to other biomes, forest make up an extremely small area of the country, however, when the numbers of threatened vertebrates are considered per unit area of biome, forests come out as having by far the highest concentration of threatened species per unit area (see Figure 8, below).

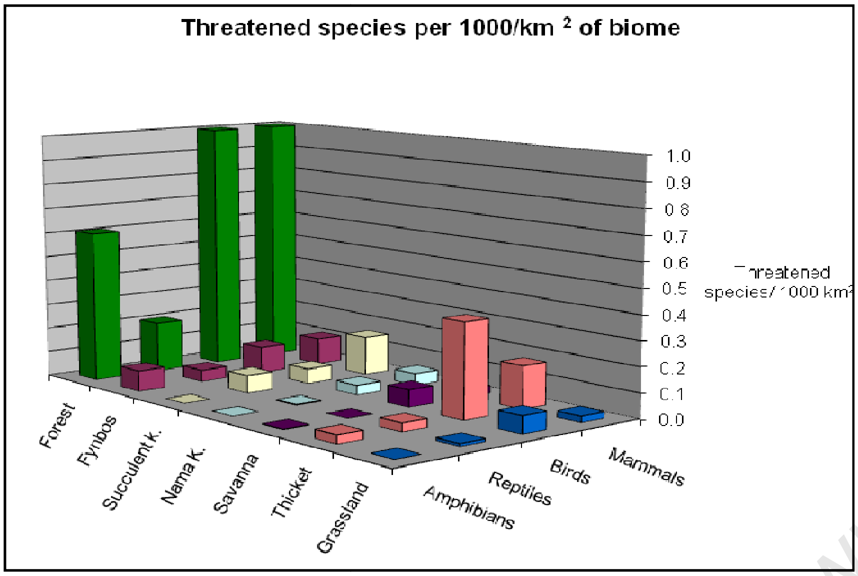


Figure 8. Threatened vertebrate species per 1000 km^2 of biome (all biome data from EWT, 2003, except for forests from data base compiled by Berliner and Wright, 2008).

3.4.4 Discussion

South African forests are important habitat for a surprisingly high number of threatened faunal species. In comparison to South Africa's other biomes, forests have the highest proportion of its vertebrate species that are threatened. Overall, approximately 13% of all vertebrate species that depend on the forest biome for their survival are listed as threatened by the IUCN. Approximately 26% of all forest amphibians, 25% of forest mammals, 15% forest reptiles and 11% forest birds are threatened.[2] Forests also have the highest concentration of threatened species per unit area.

Less is known about forest invertebrates than for other taxonomic groups. Although a number have been listed as red data species, the threat status of many more are still unknown. Often with restricted ranges and poor dispersal capabilities, forest invertebrates are particularly vulnerable to local extinctions (Samways, 1993).

[2] A forest dependent species is defined by Lawes (2007) as 'a species that breeds exclusively in forests'. For this analysis, I have defined a forests species to be a species that requires forest habitat for long term population persistence (as determined by an expert, or where the species distribution ranges overlaps by at least 80% with mapped forests).

CHAPTER 4: FOREST PROTECTED AREA GAP ANALYSIS

4.1 Introduction

Ensuring adequate representation of biodiversity within a formal protected area network is a key aim of conservation planning (Cowling, *et al.,* 1999a; Cowling, *et al.,* 1999b; Cowling and Pressey, 2003; Margules and Pressey, 2000). To date, no comprehensive assessment of the protection status of South African forests has been done (DWAF, 1997), earlier efforts to assess this (e.g. Edwards, 1974; Huntley, 1978; Low, and Rebelo, 1996) were hampered by incomplete forest cover data. The national protected area network has also changed considerably since these assessments. The analysis presented here therefore represents the first comprehensive protected area gap analysis for South African forests.

Until recently, no nationally accepted classification for indigenous forests of South Africa was available .The completion of the first objective, formalized bioegographic-floristic classification of forests (Von Maltitz *et al.,* 2003; Geldenhuys and Mucina, 2006) has provided the opportunity to use forest types as biodiversity surrogates to systematically assess levels of formal protection.

The need for a systematic assessment has become increasingly urgent due to the considerable flux in forest ownership and management currently in South Africa. Current trends are for the devolution of forest tenure and management from the Department of Water Affairs and Forestry, to national and provincial conservation authorities, and even to local communities (DWAF 1997, 1997; Neil, 2000).

The aim of this study was therefore to undertake a systematic assessment of the forest conservation area network, and how well the diversity of forest pattern is represented. Using forest types as forest biodiversity surrogates, the following questions were asked:

- To what degree are forest types represented within the protected area network?
- To what extent have conservation targets been achieved for forest types?
- How does formal protection compare across different biomes?

- To what extent are the High Conservation Value forests (HCVF)[3] represented within the statutory protected area network?
- Can targets be achieved for formal protected areas by transfer of state forests to provincial or national conservation authorities?

4.2 Methods

At the time of the analysis, no complete digital forest cover was available. To address this shortcoming, forest spatial data for different regions of the countries forests were integrated into a single shape file. The prime data set used was the South African Department of Water Affairs and Forestry's National Forest Inventory data (NFI, 2005). However, due to significant gaps in this data, data provided by provincial conservation authorities (KwaZulu-Natal and Mpumalanga) were also used, as well as data from the South African national vegetation map (Mucina and Rutherford, 2004) used for some of the azonal forests (riverine, mangrove and swamp forests). All forests, except the azonal forest types, were mapped using aerial photography at approximate scale of 1: 10 000 (DWAF, 2003). Azonal forest types were mapped at a coarser scale of 1: 250 000, based on the National Land Cover Satellite data (NLC, 2000; Mucina and Rutherford, 2004). The protected areas spatial data layer for South Africa was compiled using a combination of national and provincial data sources (Berliner *et al.,* 2006). This was later updated to be in line with the National Protected area coverage compiled by The South African National Biodiversity Institute (Rouget *et al.,* 2004).

A common problem with an assessment of reserve effectiveness is the question of what constitutes an effective protected area. It is well recognised that different types of protected areas contribute to biodiversity conservation to different degrees (IUCN, 1994; IUCN, 2003) and with different levels of effectiveness (IUCN 2000). In keeping with the classification of Rouget *et al.* (2004) a distinction was made between Type 1 and Type 2 protected areas. Type 1 protected areas are reserves that have been declared under national legislation, and include national parks; provincial nature reserves; wilderness areas and specially proclaimed state forests. Type 2 protected

[3] The HCVF concept was initially developed by the Forest Stewardship Council (FSC) for use in forest management certification. They include forests of outstanding significance or critical importance, (Jennings and Jarvie, 2004). For the purpose of this study HCVF are defined to be those forests that are 100 % irreplaceable, as determined using C-plan conservation planning software

areas include all other conservation areas (excluding state forest) such as private nature reserves, national heritage areas, community conservation areas, and municipal reserves. This analysis considered South African Department of Water Affairs and Forestry (DWAF) state forests as a separate category to Type 1 and 2 reserves, because few are actively managed as protected areas, but are nevertheless state owned. Although all forests are theoretically protected under the National Forest Act, few state forests are effectively managed as conservation areas and can not be considered as a valid protection category (Castley, and Kerley, 1996; DWAF, 2005a).

All spatial data analysis was done using the Geographical Information System Arc View 3.2. The High Conservation Value forests were derived from the Irreplaceability ratings as determined using C-plan (Anon, 1999). All data sets were later imported into an Microsoft Access data base to facilitate analysis.

Part of gap analysis is the determination of target shortfall. Target setting is a corner stone of systematic conservation planning (Margules and Pressey, 2000). Targets aim to address the question of how much of a particular ecosystem or habitat needs to remain intact to ensure species persistence. Targets used for the forest gap analysis were those derived by Berliner *et al*. (2006). Although the approach was roughly based on species- area curve relationships as described by Desmet and Cowling (2004), values were adjusted according to a number of criteria including:

- index of the species diversity of forest types (using the species area curve's z- value);
- relative rarity of the forest type;
- index of fragmentation, and;
- estimated historic reduction (since 1890).

For further details of the calculations used to derive targets, refer to appendix A.

4.2 Results

4.3.1 Levels of protection for forest types

The areas of each forest type falling within the three protection categories considered are given in the Table 18, below.

Table 18. Percentage of forest types, by area falling within three types of protection. Forest ranked according to combined totals. (Note: Type 1 protected areas include national and provincial parks, wilderness areas and special forest protected areas; Type 2 protected areas include municipal reserves, private reserves and natural heritage sites; State forests are indigenous forest falling with state land managed by the Department of Water Affairs and Forestry)

Forest type	Total area (ha)	Percentage of area under protection type			
		Type 1	Type 2	State forests	Combined
Mangrove	2 393	73.9	0.0	0.6	74.5
Lowveld Riverine	11 401	67.9	0.1	0.0	68.0
Northern Mistbelt	19 204	3.6	0.0	63.9	67.5
Swamp	3 022	67.2	0.0	0.0	67.2
KwaZulu-Natal Coastal	21089	61.3	0.0	0.0	61.3
Transkei Coastal	61 484	4.8	0.0	56.5	61.3
Transkei Mistbelt	30 250	0.0	0.0	57.5	57.5
Southern Cape Afrotemperate	74 848	56.4	0.1	20,5	56.4
Western Cape Afrotemperate	4 731	50.2	0.0	2.1	52.4
Mpumalanga Mistbelt	32 772	47.2	2.8	0.0	50.0
Drakensberg Montane	1 926*	47.3	0.0	0.0	47.3
Amatole Mistbelt	64 221	1.4	1.0	44.6	47.0
Pondoland Scarp	12 284	10.4	0.0	33.0	43.5
Eastern Mistbelt	41 842	10.7	0.8	31.9	43.4
Licuati Sand	24 276	42.2	0.0	0.0	42.2
Albany Coastal	22 046	34.8	2.4	0.0	37.2
KwaZulu-Natal Dune	12 396	22.7	0.6	13.0	36.3
Eastern Scarp	33 750	25.8	0.8	9.3	35.9
Low Escarpment Mistbelt	5 323	14.9	7.6	0.0	22.5
Eastern Cape Dune	10 941	8.3	0.6	0.0	8.9
Western Cape Milkwood	2 500	2.0	0.0	0.0	2.0
Total	492 700	25.2	0.7	27.1	53

* The area given for Drakensberg Montane forests comes from the original national Forest Inventory data set used in this analysis. Recent work done since this, by Lawes *et al* (2007b), found 7 025 hectares of this forest type,

Currently 25.2 % of the total indigenous forest area falls within formal statutory Type 1 protected areas, while 53 % of forested area falls with some form of protection (all three protection categories combined). Forest types with the least overall protection (with less than 30% under any form of protection) include, Eastern Cape Dune, Low Escarpment Mistbelt, and Western Cape Milkwood.

4.3.2 Conservation target achievement for forest types

Of the twenty one forest types considered only two have achieved Type 1 conservation targets (see appendix A). These include: Albany Coastal and Southern Cape Afrotemperate forests. A further three come within 25 % of achieving targets, these are: KwaZulu-Natal Coastal; Western Cape Afrotemperate and Drakensberg Montane. The remainder have less than 25 % of their targets in Type 1 protection (see Figure 9, below).

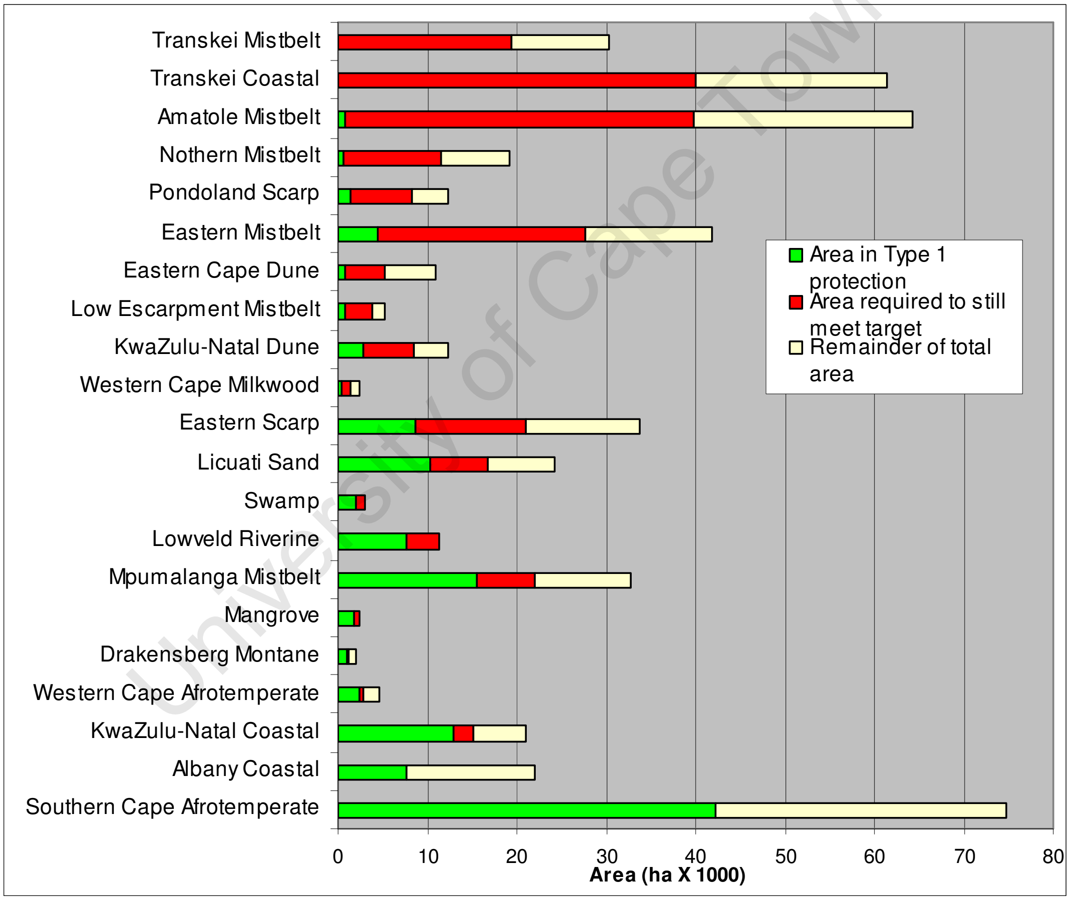


Figure 9. Area of forest types under Type 1 protection (in green), and outstanding area required to meet targets (in red). Forest types ranked from top to bottom according to lowest target achievement.

4.3.3 Comparison of formal protection at biome level

In comparison to the other biomes, forests have the highest proportion of total area under formal protection. Forest, fynbos and deserts are also the only biomes to have achieved national biome level conservation targets as set by the Department of Environmental Affairs and Tourism (see Table 19); however, these targets are significantly lower than those used in this study (see appendix A).

Table 19. Type 1 protection levels and national conservation target achievement for the major vegetation biomes of South Africa. All biome data, apart from forests, are from Holness *et al.* (2008).

Biome	Biome area (km^2)	Biome PA target (%)	Percentage of biome under strict protection
Albany Thicket	29133	10.4	7.1
Desert Biome	7164	18.2	22.3
Forests	4927	25.0	25.2
Fynbos Biome	83952	15.0	19.8
Grassland Biome	354493	13.5	2.0
Nama-Karoo Biome	248196	11.2	0.7
Savanna Biome	412663	10.3	9.2
Succulent Karoo Biome	83287	12.3	5.2

4.3.4 Formal protection of High Conservation Value Forests

High Conservation Value (HCV) Forests are forests of outstanding significance or critical importance (Jennings and Jarvie, 2004). For the purposes of this analysis I have defined HCV forests as those forest patches with a 100% irreplaceability, which in most cases are the large intact forests. The degree to which forest types have their HCV forest represented under Type 1 protection is illustrated in Figure 10, below.

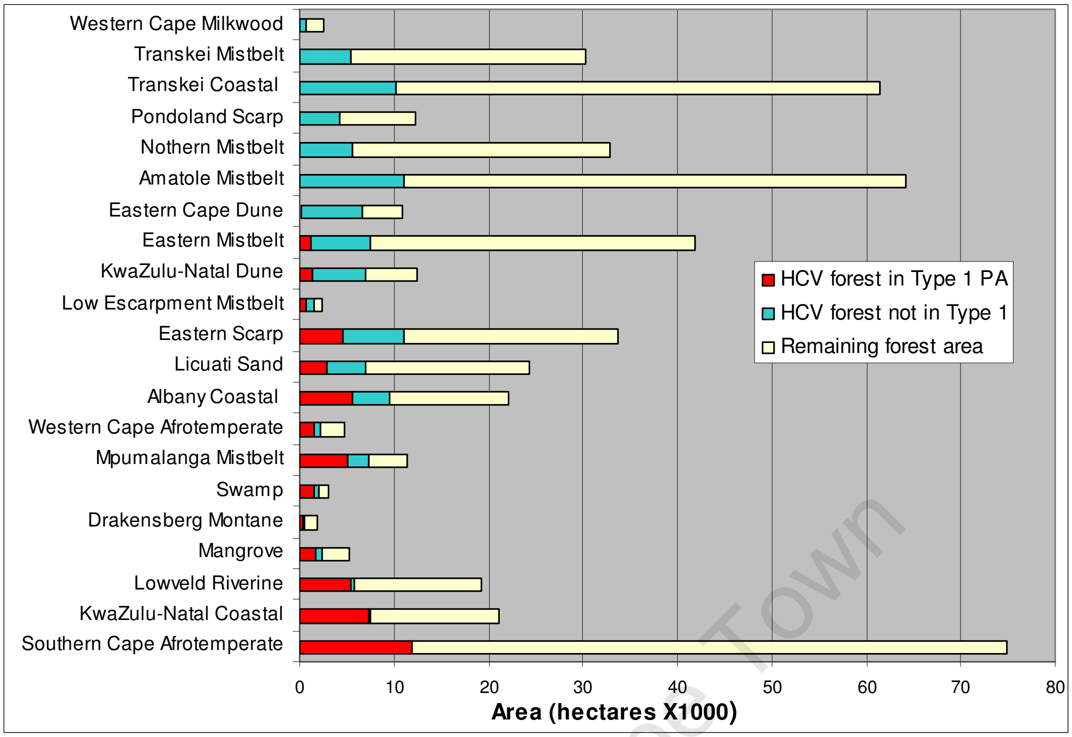


Figure 10. Levels of Type 1 protection of High Conservation Value (HCV) forests for each forest type, ranked from least to most protected.

Overall only 40 % of forested area considered as HCV are formally protected (in Type 1 protected areas); however this is biased towards a few forest types with just six forest types making up around 75 % of the HCV formally protected forests. Seven forest types have no formal protection of their HCV forests.

4.4 Discussion

At the national biome level it would appear that forests are better protected than other biomes, however this highly aggregated data does not reveal how well forest biodiversity pattern and process are being effectively protected on the ground. For example, just over half (53 %) of all South African forest falls within some form of state protection, but much of this is made up of state forest with little or no 'on the ground' conservation management, and despite being theoretically protected under the National Forestry Act, are often subjected to over harvesting and even total destruction (Goodman, 2000; Obiri and Lawes, 2002; Obiri, *et al.*, 2002; Lawes *et al.*, 2004b; DWAF 2007). In addition, Type 1 protected forests (25.2 % of the biome) are heavily biased towards just four forests types that make up 66 % of the Type 1 forest area (see Figure 11, below).

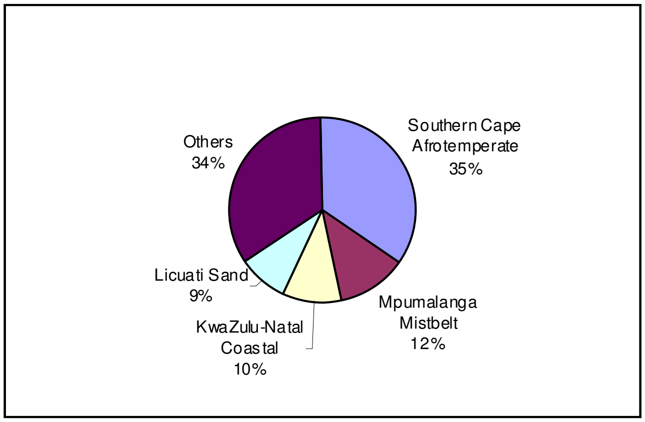


Figure 11. Percentage representation of forest types under Type 1 protection.

Despite these shortcomings, in comparison to other African countries the overall level of formal protection of South Africa forests are significantly higher than the mean for 43 other African countries, this being in the order of about 16 % 'dedicated primarily to biodiversity conservation' (State of the World's Forests, 2007).

When considered against the conservation targets used in this study, almost half of all forest types have more than a 70% target shortfall. While the exact level of targets necessary to ensure forest biodiversity persistence may be uncertain, using the internationally accepted minimum level of formal forest protection of 10 % as endorsed by the IUCN and others (Davey, 1998; Dudley *et al*, 2002; Dudley *et al* 2003; IUCN, 2003; FAO, 2003; Schmitt *et al.,* 2008), at least six forest types fall below this minimum. These include Limpopo Mistbelt, Transkei Coastal Scarp, Transkei Mistbelt, Amatole Mistbelt, Eastern Cape Dune, and Western Cape Milkwood forests. A further two, Pondoland Scarp and Eastern Mistbelt forest are at the 10% level. These forest types urgently need improved representivity within Type 1 protected areas.

The sheer number and wide geographical spread of more than 16 000 forest patches present a formidable challenge to forest conservation planners. Of critical importance is to ensure that forests with the highest conservation value of each forest type are represented within the formal protected area network. Overall 60 % of HCV forests fall outside of formal Type 1 protected areas. Of concern are the forest types with little or no HCV forests falling within Type 1 protection, and for these forest types, some of the most important forests fall outside of the formal protected area network (see Table 20, below).

Table 20. Degree of Type 1 protection of High Conservation Value (HCV) forests within each forest type.

Degree of HCV forested area protected in Type 1	Forest type
Absent or very low (Less than 5%)	Amatole Mistbelt
	Pondoland Scarp
	Transkei Coastal
	Transkei Mistbelt
	Western Cape Milkwood
	Northern Mistbelt
	Eastern Cape Dune
Low to good (5 -70 %)	Eastern Mistbelt
	KwaZulu-Natal Dune
	Eastern Scarp
	Low Escarpment Mistbelt
	Licuati Sand
	Albany Coastal
	Mpumalanga Mistbelt
	Western Cape Afrotemperate
	Swamp
Good (> 70 %)	Drakensberg Montane
	Mangrove
	Lowveld Riverine
	KwaZulu-Natal Coastal
	Southern Cape Afrotemperate

The results of this study support the views of Pressey and Tully (1994); Rouget *et al.* (2003b) and others, that past reserve acquisition has been driven more by opportunistic and 'ad hoc reservation' than systematic assessment of conservation needs and meeting of conservation targets.

Finding suitable land to improve representation of forest types within a formal forest protected area network is complicated by the assortment of land ownership and land tenure regimes that forest fall within. Currently the management authority of many forests is uncertain. Recently a number of DWAF state forests have devolve management to provincial and national conservation authorities (DWAF 2005c). This will have a significant impact on the conservation status of some but not all forest types. For example, the 2006 transfer of state forests in the Knysna, Tsitsikamma and Woody Cape district to the South African National Parks have resulted in two forest types exceeding (Southern Cape Afrotemperate) and attaining (Albany Coastal) Type1 conservation targets. Obviously, forested land under state control would be the most cost effective areas to expand the protected area network. However, even if all state forest land were to be transferred to provincial or national conservation authorities for inclusion within formal Type 1 protected areas, a number of forest types would still have very low representation levels. This is because approximately 45 % of the area of South African indigenous forests occurs on land not directly owned by organs of the state; with roughly half of this falling into communal land and the remainder on private land. Forest types occurring predominately on communal or private land are the least protected and also the most vulnerable. For a number of forest types, particularly those occurring predominantly on communal or private lands, the only viable option for ensuring protection and achieving targets, is to expand the non statutory protected area network to include community managed conservation areas. In Table 21, below, the potential contribution that state forests could make to meeting forest type targets are assessed.

Table 21. Potential contributions that state forests could make to meeting targets. Forest types are divided into four groups, according to the contributions that state forests could make to increasing levels of formal protection.

Forest Type	% Target	Current target shortfall	% Private and communal	% of target met if all state forests transfer to Type 1 PA
A: *Forests with high target shortfall and where targets cannot be met by state forests*				
Western Cape Milkwood	55.76	96%	98%	0%
Eastern Cape Dune	48.46	83%	91%	0%
Low Escarpment Mistbelt	71.74	79%	77%	0%
Licuati Sand	69.27	39%	58%	0%
Swamp	100	33%	33%	0%
Lowveld Riverine	100	32%	32%	0%
Mpumalanga Mistbelt	66.99	30%	50%	0%
Mangrove	100	26%	25%	2%
Drakensberg Montane	63.5	26%	53%	0%
B: *Forest types with high target shortfall, targets can be partially met by state forests*				
Amatole Mistbelt	62.12	98%	53%	73%
Pondoland Scarp	66.61	84%	56%	59%
Eastern Mistbelt	66.45	84%	57%	57%
Kwazulu-Natal Dune	69.2	67%	64%	28%
Eastern Scarp	61.61	58%	64%	26%
C: *Forest types that targets can be almost completely met by state forests (90-100%)*				
Transkei Mistbelt	64.17	100%	42%	90%
Transkei Coastal	65.01	100%	39%	94%
Northern Mistbelt	59.56	94%	32%	114%

D: Forest types that have met or almost met targets				
Western Cape Afrotemperate	60.08	16%	48%	22%
Kwazulu-Natal Coastal	71.69	15%	39%	0%
Albany Coastal	35	1%	63%	0%
Southern Cape Afrotemperate	49.08	-15%	23%	278%

An additional complexity for forest protected area expansion (particularly for group A forests in Table 21), is that a significant area of forested land is under land ownership dispute. At least 10 % of indigenous forests have land restitution claims (calculated from data in DWAF, 2005). A number of these claims have been successful, but many remain unresolved. This analysis found that about 45 % of land claims on forest land are in existing Type 1 protected areas, and 38 % are on DWAF state forests. A number of High Conservation Value forests fall within communal lands; these should be prioritised for implementation of a community based conservation approach (see for example World Bank, 2002).

Although not adequately accounted for in this analysis, it needs to be pointed out that a number of private forestry companies conserve numerous patches of natural forest on their land, and in some instances these forests have been given national heritage status. It estimated that 41 000 ha of indigenous forest are managed by private timber companies (DWAF 2003). The conservation of these usually small patches, (despite being surrounded by plantations) can make an important contribution to forest conservation.

Despite the positive trends of DWAF state forests being transferred to national or provincial conservation management authorities, in a number of cases, these administrative changes have lead to confusion regarding ownership, management and user rights to local communities. In some communal areas of South Africa this has been further exacerbated by the erosion of traditional tribal authority and the associated loss of authority to regulate resource use (DWAF 2005).

Finally, the assignment of legal protection status is no guarantee of effective conservation (WWF, 2004). In South Africa many proclaimed forest reserves are still subjected to illegal harvesting. For example, two priority forests Ngome and Ngoye (Ongoye), in Kwazulu-Natal (both under Type 1 protection) are under pressure from bark and other medicinal plant harvesting, as well as poaching (Boudreau & Lawes, 2005; Boudreau *et al.*, 2005).

CHAPTER 5: ASSESMENT OF THREATS TO FORESTS

5.1 Introduction

Distinction can be made between the root causes and proximate causes of biodiversity loss. Root causes of biodiversity loss, are usually socioeconomic, while proximate causes are the activities that impact directly on biodiversity (Stedman-Edwards, 2000; Wood, *et al*. 2000; Diamond, 2005), see Figure 12, below.

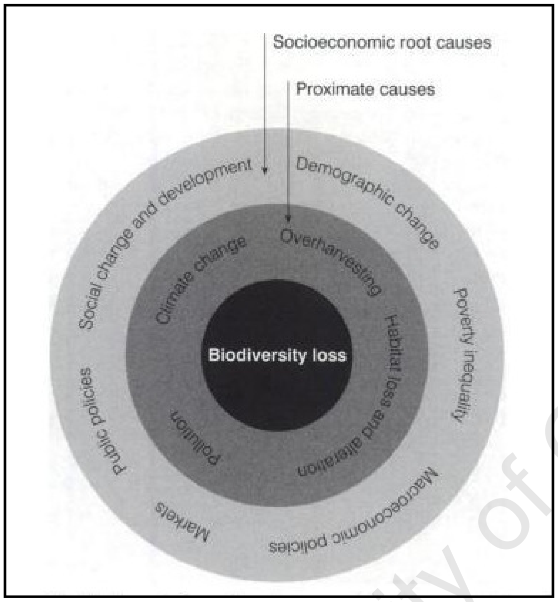


Figure 12. Proximate and socio-economic causes of biodiversity loss (after Stedman-Edwards, 2000)

South African forests are rare, highly restricted and occur embedded within landscapes that are under increased human pressure. Historically, the major causes of forest loss were logging and clearing for agriculture (King, 1938; King, 1941; McCracken, 1986; McCracken, 2004); while predominant contemporary causes, include clearing for timber plantations, coastal development and non sustainable harvesting of forests products by rural communities (Cooper, 1985; Geldenhuys and McDevette, 1989; Cooper and Swart, 1992; Mander, 1998; DWAF, 1999, Shackleton *et al*., 2004a; DWAF, 2003, Von Maltitze *et al.,* 2003; Lawes *et al*.2004; Mckean , 2004 ; DWAF, 2005a, Mucina and Geldenhuys, 2006; Lawes *et al*. 2007). Although illegal clearing and logging of forests have largely halted, and the establishment of the plantation timber industry has reduced demands for commercial indigenous timber, increased levels of rural poverty have stimulated dependency on forest resources, in particular for fire wood, food, medicinal plants and building materials (Pandey, 2002; Arnold and Persson, 2003, INR, 2003; Williams, 2004; Shackleton *et al, 2007)*

Despite forests being able to withstand certain levels of harvesting, before biodiversity is negatively impacted (Boudreau & Lawes, 2005; Boudreau *et al*., 2005), the mid- to long-term impacts of selective timber harvesting on forest dynamics are largely unknown (Lawes and Obiri, 2003), as are the levels of harvesting of non woody forest products that are sustainable over the long term (Mucina and Geldenhuys, 2006).

Land transformation, in particular from timber plantations, have significantly altered landscape level geo-hydrological processes (van der Zel, 1995; Dye and Versfeld, 2007), as well as natural disturbance regimes originating from fire, wind and herbivory (Lawes *et al.,* 2004b; Lindenmayer, and Franklin, 2006; Lawes *et al.,* 2007b; Odion and Sarr, 2007). In South Africa, many smaller forest patches are being eroded, or have been complete destroyed by fires, grazing and harvesting for wood, bark and other medicinal plants (Cooper, 1985, Geldenhuys, 1994; DWAF, 2003, Lawes, *et al*., 2007c; Kotze and Lawes, 2007) leading to fragmentation and loss in landscape level connectivity between patches, and an increase edge ecotone (Forman, 1995, Laurance, 2000; Kotze and Lawes, 2007).

An evaluation of levels of threat and vulnerability faced by biodiversity is an important component of establishing conservation priorities (Margules and Pressey 2000; Cowling and Pressey, 2003; Wilson *et al.,* 2005; Burgess *et al.,* 2006). Because the mathematical quantification of threats is typically complex, a qualitative rule based expert system modelling approach was used. The following steps were followed. First, a conceptual model that identifies the cause and effects of threats was developed; second, key indicators or surrogates that could be used to quantify the relative magnitudes of these threats were identified, and thirdly, rules were developed that enable threats to be scored and integrated into overall threat ratings for each forest type.

To supplement the modelling approach described above, an alternate method of establishing the relative importance of the different threat factors on forests was done by quantifying the frequencies in which threats were listed as threat agents responsible for listing of red data forests plants.

Finally, the relative vulnerability of each forest type to biodiversity loss was approximated by considering three indicators of threat: population pressure, transformation of the forest matrix, and numbers of threatened plants occurring within each forest type.

The threat of global climate change on forest biodiversity, although globally of increasing concern, was not specifically included in this analysis, but is discussed under section 8.2.4 Expanding research of the forest biome.

5.2 Methods

5.2.1 Developing a conceptual model of threats

A conceptual model describing the cause and effect chains of biodiversity threats to South African forests was developed from discussion with forest experts and from information synthesised from various literature sources. Threats were then described in terms of the components: *drivers, triggers and modifiers*.

5.2.2 Data used to derive key indicators of threats

Despite the potential complexity of mapping biodiversity threats, at the broad scale biodiversity loss correlates with a number of relatively easily and spatially quantifiable phenomena, that can be used as surrogate measures of threats to biodiversity (Marguels and Pressey, 2000; Rouget *et al.,* 2003b). These typically, include: population density; agricultural potential of the land (Rouget *et al.,* 2003b; Reyers, 2004; Wilson *et.al*, 2005); and poverty related wood usage (Wood *et al.*, 2000; Arnold and Persson, 2003; Shackleton *et al.,* 2004a; Shackleton *et al.,* 2007).

The conceptual model of threats was simplified (in line with availability of data) into key threats and indicators. The data sources used to determine these indicators are presented in Table 22, below.

Table 22. Data sources and indicators used to model threats.

Threats	Spatial data used	Derived indicators
Population pressure and Subsistence resource use pressure	• Population density (from, National census 2001 data, and national communities database) • Accessibility to forest resources modelled using topography, road access, road penetration (from GIS intersections). • House hold fuel use, i.e. reliance on wood as prime source of fuel (from :National census 2001 data)	• Subsistence Resource Use Pressure Index (SRUPI). • Accessibility index. • Index of house hold wood use dependency
Agricultural pressure (land transformation)	• Agricultural Research Council (ARC) land capability spatial data.(Schoeman *et al.*,2000) • National Land Cover (NLC, 2000)	• % arability index (of forest buffer area). • Threat of agricultural transformation index.
Urban expansion	• Coastal forest types. • Forests 15km from an urban area.	• Threat of urban expansion.
Mining	• List of forests currently being mined or proposed for mining, (DWAF pers comms).	• Threat of mining score
Transformation of forest matrix	• National land cover 2000 (NLC, 2000; Fairbanks *et al.,* 2000)	• Transformation in 5km buffers; timber plantations in forests cluster matrix areas

5.2.3 Modelling population pressure per area of forest

Population pressure on forests was approximated using spatially explicit population data (derived from the national communities' data base, DWAF 2002). Population polygons were intersected with forest patch buffers to calculate a forest population loading (numbers of people per hectare of forest within 5 km and 1 km forest buffer areas).

5.2.4 Modelling subsistence resource use

Although the forest population loading index can give a general idea of pressures acting on forest resources, there are a number of other factors that may influence the actual harvesting pressure on forests. These include the accessibility of the forests, and the availability of alternatives to the forest resources, as well as the need for fire wood to supply household fuel needs. An understanding of the key factors driving subsistence harvesting was synthesised from discussions with experts as well as reviews of literature, from this a conceptual model was developed that formed the basis of a rule based expert system model.

5.2.5 Modelling agricultural pressure

The threat of transformation from agriculture and timber plantations is directly related to the suitability or arability of land surrounding forest for these activities. An 'arability index' was derived by extrapolation from the Agricultural Research Council's land capability data (Schoeman *et al.,* 2000). See appendix B for further details.

5.2.6 Modelling development pressure

Forests close to urban centres and coastal areas were considered to have a higher risk of exposure to development pressures. Any forest within 15 km of an urban edge and all coastal forests were considered to be at higher risk from development pressures.

5.2.7 Mining

Current threats from mining to forest are primarily from dune mining in certain areas with coastal forests. Areas currently facing this threat were specifically listed. Scoring rules used, were based on the market demand for the mineral, this varied from 1 (low demand) to 3 (high demand).

5.2.8 Transformation of the forest matrix

The national land cover data set (NLC, 2000) was used to asses land transformation. Three measures of transformation were used: a) within the forest matrix (the non forested area within forest clusters, see chapter 7, for details), and b) within 5 km forest buffers, and c) forest matrix area converted to timber plantations.

5.2.9 Scoring systems used for quantifying and aggregating threats

A numerical rule based scoring systems was used to quantify and aggregate threats. Details are provided in appendix B.

5.2.10 Using threatened species data

Threatened forest plants were identified by linking the forest species database (Berliner and Wright, 2008) with the South African National Biodiversity's red data assessments (Raimondo *et al.,* 2009). Threats were classified and grouped using IUCN's hierarchical classification (Baillie *et al.,* 2004). Species threat data is from Raimondo *et al.* (2009).

The relative importance of a threat factor, was determined by the frequency of occurrences that the threat factor was implicated as being responsible for the decline of the red data species (a species may have more than one causative agents of its decline). These were then summed and expressed at both an aggregated and disaggregated level. For example the causes of decline for the threaten forest species *Stangeria eriopus* is given *as* 'habitat loss, habitat degradation and harvesting' (at the aggregated threats level). This was then disaggregated into more specific threats (or sub threats) as 'harvesting for medicinal plants'; 'development', and 'crop production'.

5.3 Results

5.3.1 Conceptual model of threats

A conceptual model of threats derived is given in Figure 13, below. In this model, *drivers* are the socio-economic forces at the root of biodiversity loss. *Triggers* are the additional factors that may, or may not be associated with the driver, but that are usually required to catalyse the drivers. For example a high population adjacent to a forest, may not necessary have an impact on the forest, but if this same population have high levels of poverty, absence of electrification and poor resource use control (all trigger factors), then chances are that the forest will be impacted. Finally, modifiers, are factors that amplify or attenuate impacts, (such as proximity and accessibility the forest and the presence of substitute resources such as woodlots).

Biodiversity impacts may manifest as changes in forest pattern and/or process. Changes in forest pattern may include changes in spatial extent and number of forest patches or shifts in species composition and age structure, while change in forest processes relate to changes in systemic functioning that may include disruption of nutrient cycling, pollination and seed dispersal). In many cases, there is likely to be a dynamic interplay between changes in forest pattern and changes in forest functioning.

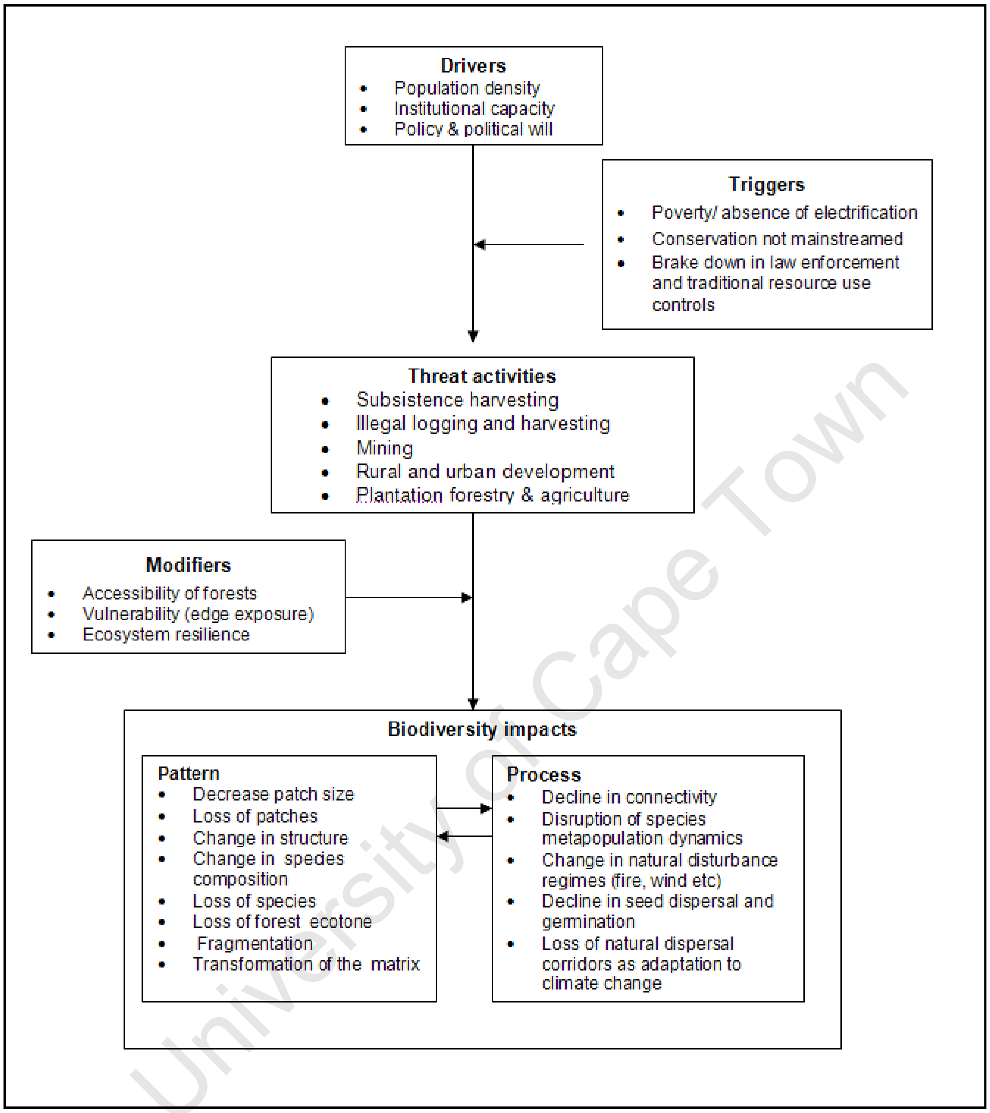


Figure 13. Conceptual model of threats to forests showing drivers, triggers, modifiers and impacts to forest biodiversity in South Africa.

5.3.2 Population pressure

The population density around forests was used to indicate the vulnerability of forest to human impacts. Population pressure was calculated for each forest patch, and then averaged for each forest type for both one and five kilometre forest buffers (see Figure 14, below).

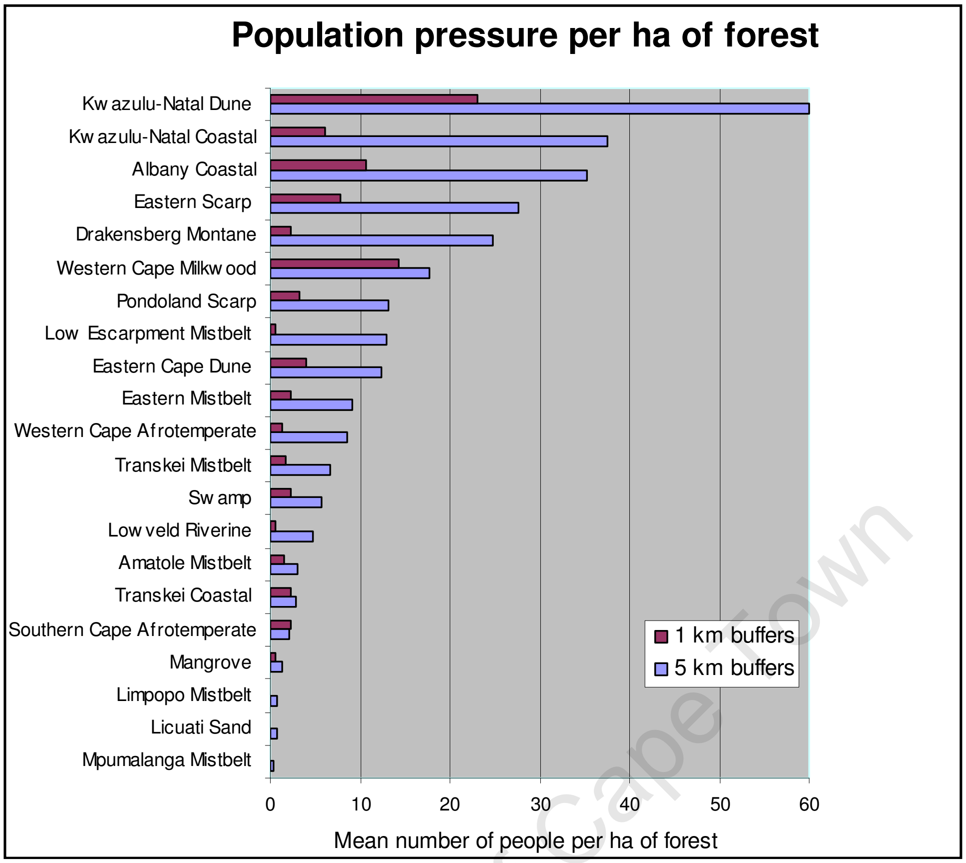


Figure 14. Population pressure on forests expressed as mean number of people per hectare of forest in both one, and five kilometre buffers for each forest type.

The results show that a number of forests types have very high population densities within both one, and five kilometre buffers, notably Kwazulu-Natal Dune and Kwazulu-Natal Coastal Forests. Other forest types such as Eastern Scarp and Drakensberg Montane forests have relatively low population densities within the 1km buffer, but high population densities within the 5 km buffer.

5.3.3 Subsistence Resource use Pressure Index

Forests that are geographical inaccessible, or located near urban centres (where populations are not reliant on primary resources) are likely to have less harvesting pressure than predicted from the population loading index alone. Accessibility is also influenced by the proximity to roads, and the degree to which roads penetrate into forests. In addition, forests located close to woodlots or surrounded by plantations will be partially buffered from harvesting pressure. The availability of alternative fuel resources such as electricity will have a large influence on the use of forests. These considerations were used to model a household wood use index for communities surrounding forest. Using rule based modelling (see appendix B for details); this was then integrated with population density and forest accessibility to give a subsistence resource use pressure index (SRUPI) for each forest patch. This is conceptually described in Figure 15, below.

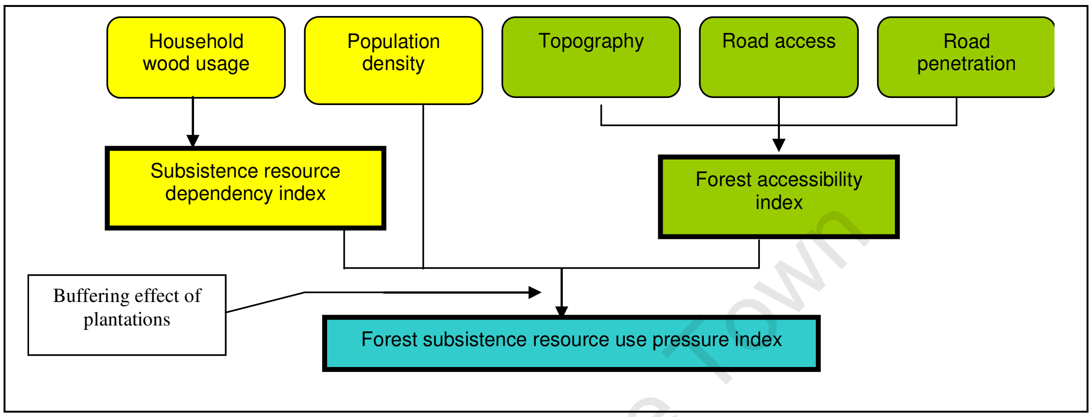


Figure 15. Conceptual model used to derive an index of the Subsistence Resource Use Pressure acting on forests (Yellow boxes represent data extrapolated from national Census 2001; green boxes represent data derived from GIS analysis of digitised topocadastal maps, and white boxes, data from national land cover (NLC, 2000).

Results of modelling the subsistence resource use pressure index (SRUPI), aggregated to the level of forest types are given in Figure 16, below.

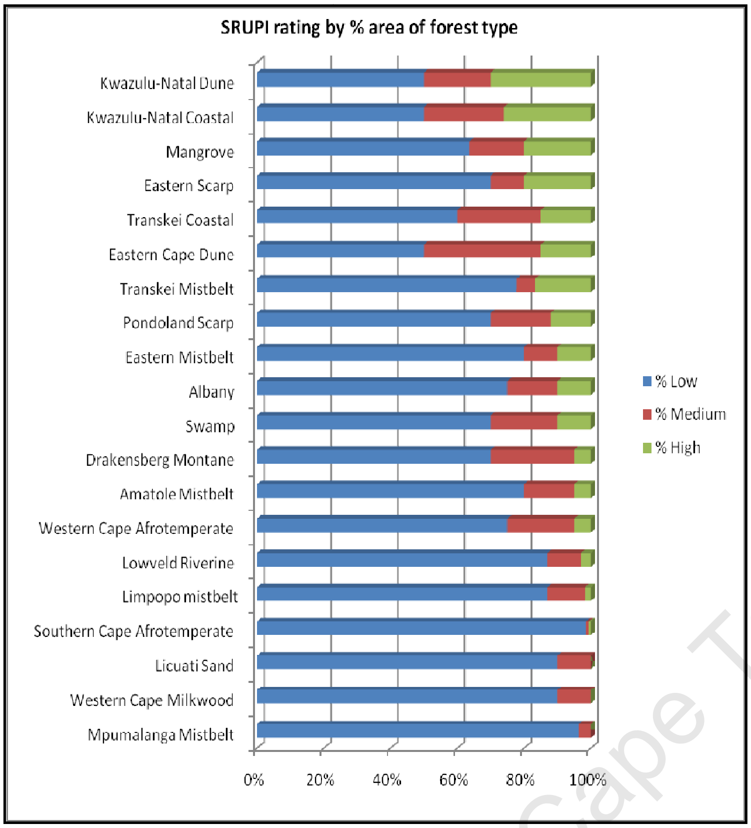


Figure 16. Subsistence resource use pressure index (SRUPI) expressed as the proportion of the total area of forest type that scored either a 'high', 'medium' or 'low' threat rating.

Predictably, forest types located in communal areas of the Eastern Cape and Kwazulu-Natal, areas with high rural poverty, and usually without electrification have the highest SRUP ratings. SRUP values can provide guidance about where livelihoods work could most urgently be targeted.

5.3.4 Transformation of the forest matrix

Three measures were used to determine degree and type of habitat loss around forested areas. Firstly, transformation between closely associated forest patches (i.e. habitat transformation within the forest cluster matrix, see section 7.2). Secondly, transformation of the broader landscape using 5 km forest patch buffers, and thirdly, transformation of the 5 km forest patch buffers due to plantation forestry. Results of this analysis are presented in Figure 17, below.

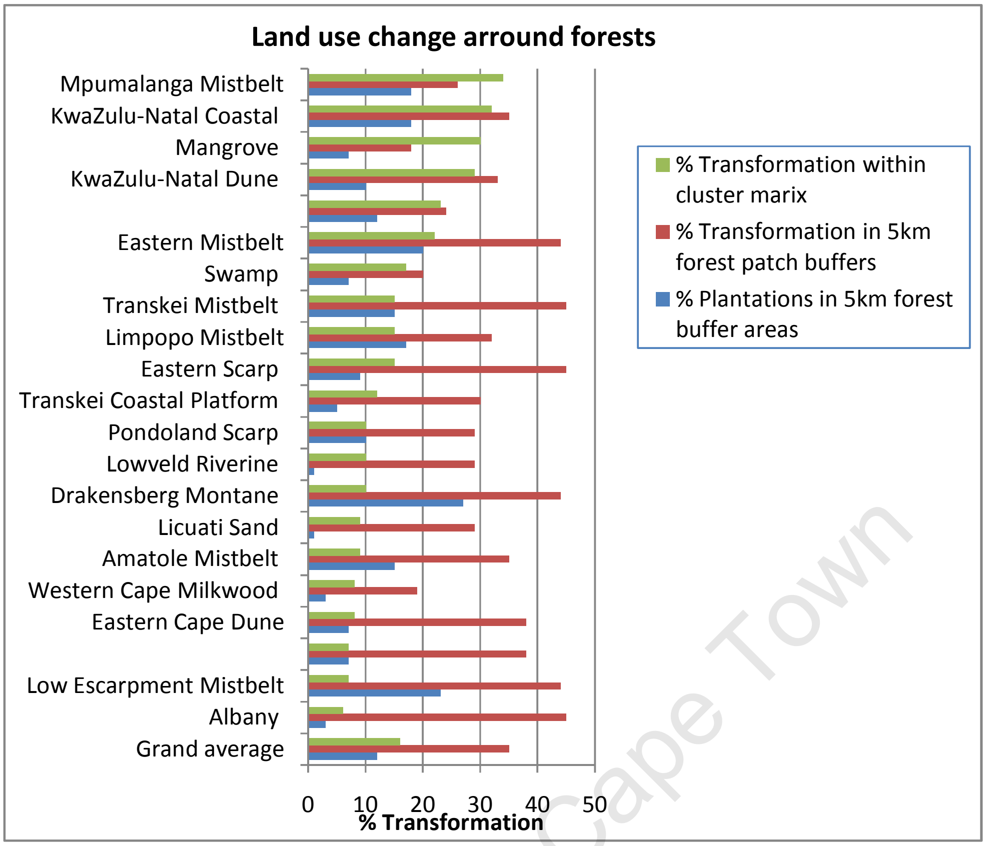


Figure 17. Mean percentages habitat transformation within forest cluster matrix and 5 km forest patch buffers. Percentages of the 5 km forest patch buffers transformed by exotic plantations are also given (calculated from national land cover data, NLC, 2000).

For most forest types, significant forests matrix area has been transformed. For example at the broader landscape level (i.e. within 5km forest buffers) at least six forest types have more than 40 % of surrounding land transformed. Timber plantations are responsible for much of the habitat loss surrounding forests.

5.3.5 Aggregating threats

Using a rule based scoring system (see full description of method in appendix B), the threats of subsistence harvesting , urban expansion, mining, and matrix transformation were integrated into an overall threat rating of either ‘high’, ‘medium’, or ‘low’ for each forest patch and then aggregated to give a percentage of total area within each forest type falling within each category of high , medium or low threat (see Figure 18, below).

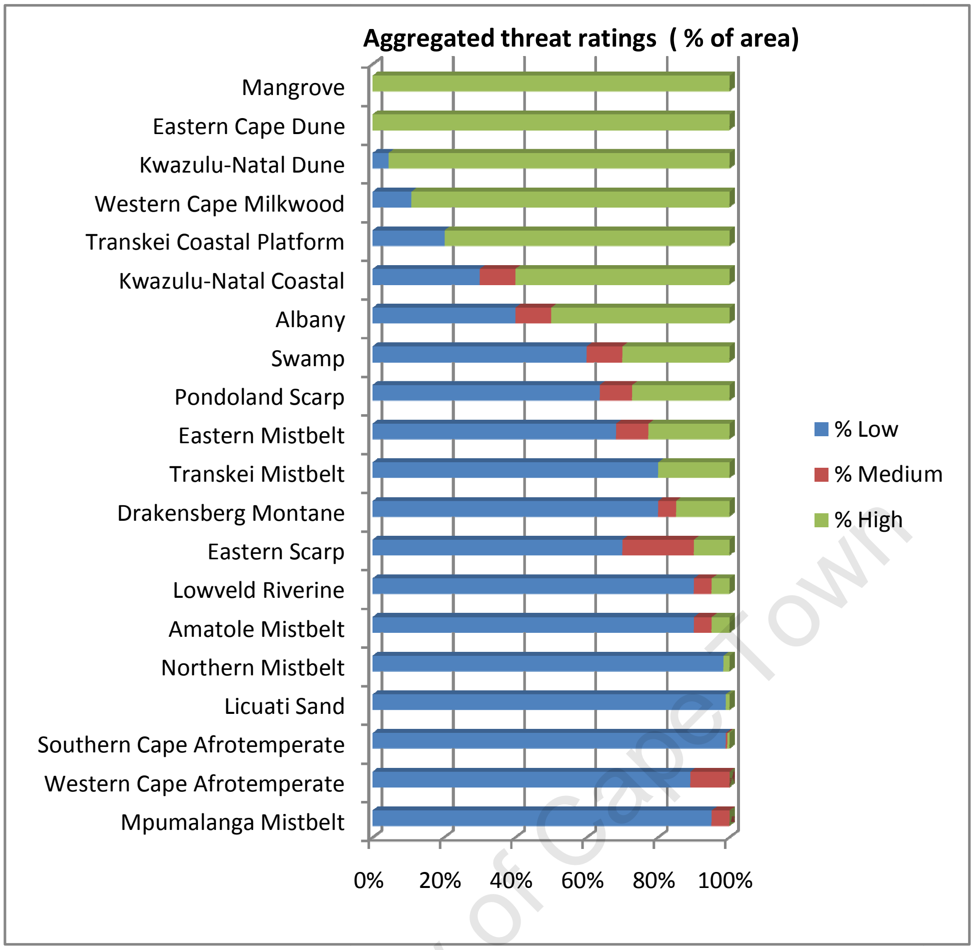


Figure 18. Percentage of forest patches rated as having a high, medium or low aggregated threat rating for each forest type considered.

Approximately 4566 patches were classified as having a 'high' threat rating, representing about 25 % of the total area of the forest biome. The most threatened forest types tend to be those situated close to coastal areas.

5.3.6 Using threatened species data

By adapting IUCN's hierarchical threat classification system, two levels of threats were assessed: the aggregated threat level, such as 'habitat loss' or 'harvesting', and the finer, sub-threat level such as 'medicinal plant harvesting' and fuel wood harvesting (considered as sub threats of 'harvesting'). Results are presented in Figure 19 and Figure 20, below.

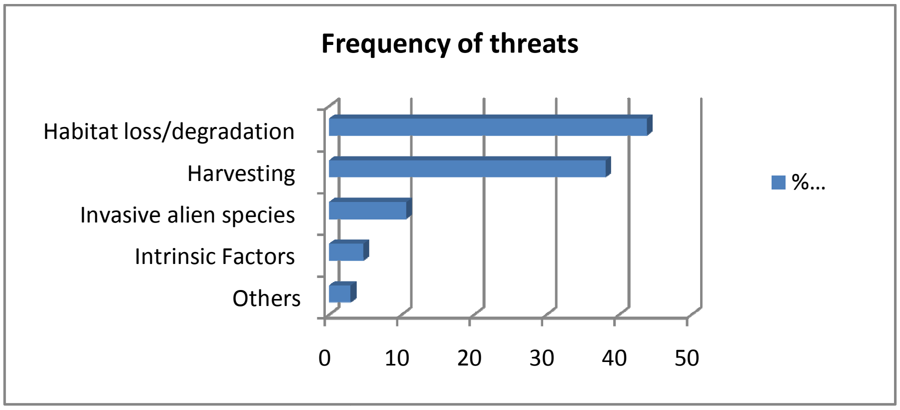


Figure 19. Frequency of occurance of threats responsible for declines in threatened forest plants.

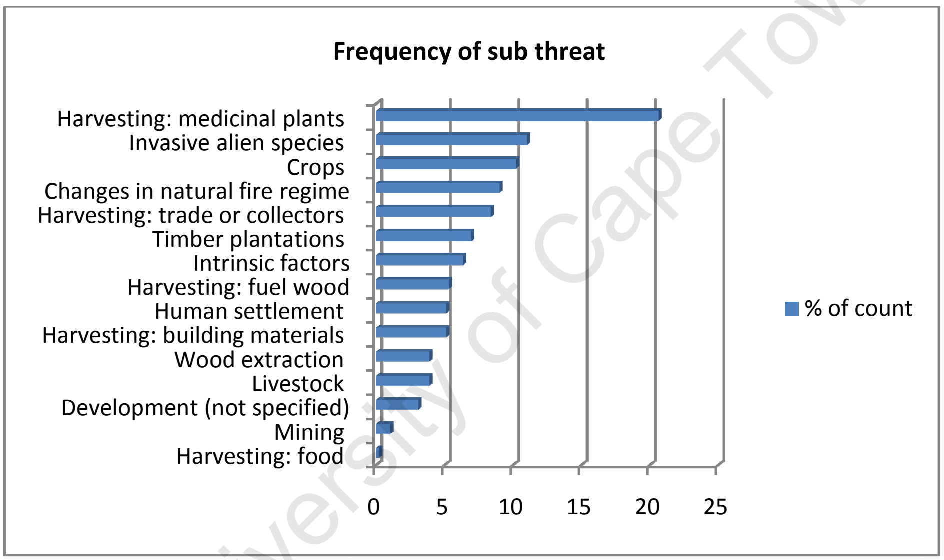


Figure 20. Frequency of occurance of sub threats responsible for declines in threatend forest plants. Defintions of catagories used: ‘crops’ include all forms of cultivation (small scale and comercial), but exclude timber plantations (it’s own category); ‘changes in natural fire regime’ may include too frequent, too few fires, or wild fires; ‘intrinsic factors’ may include any one or more of the following: poor recruitment, poor reproduction, poor regeneration, slow growth , restricted range or pathogens.

5.3.7 Vulnerability of forest types to biodiversity loss

An alternative to the highly aggregated approach used in section 5.3.4 is to present threat indicators separately (only the most reliable indicators were used – those considered to be the simplest and readily quantifiable). Because different threats are quantified using different units, aggregation into a single threat raring is difficult. By standardising each threat factor according to a common one to five scale, the relative importance of each threat factor can be compared and aggregated into a single mean rating for each forest type. The following scaling formula was used: scaled value = (value – (lowest value))/ (highest value) X 5.

The indicators used include: population pressure, transformation of the forest matrix, and numbers of threatened plants occurring within each forest type (data from section 3.4). The results of this analysis are presented below in Figure 21.

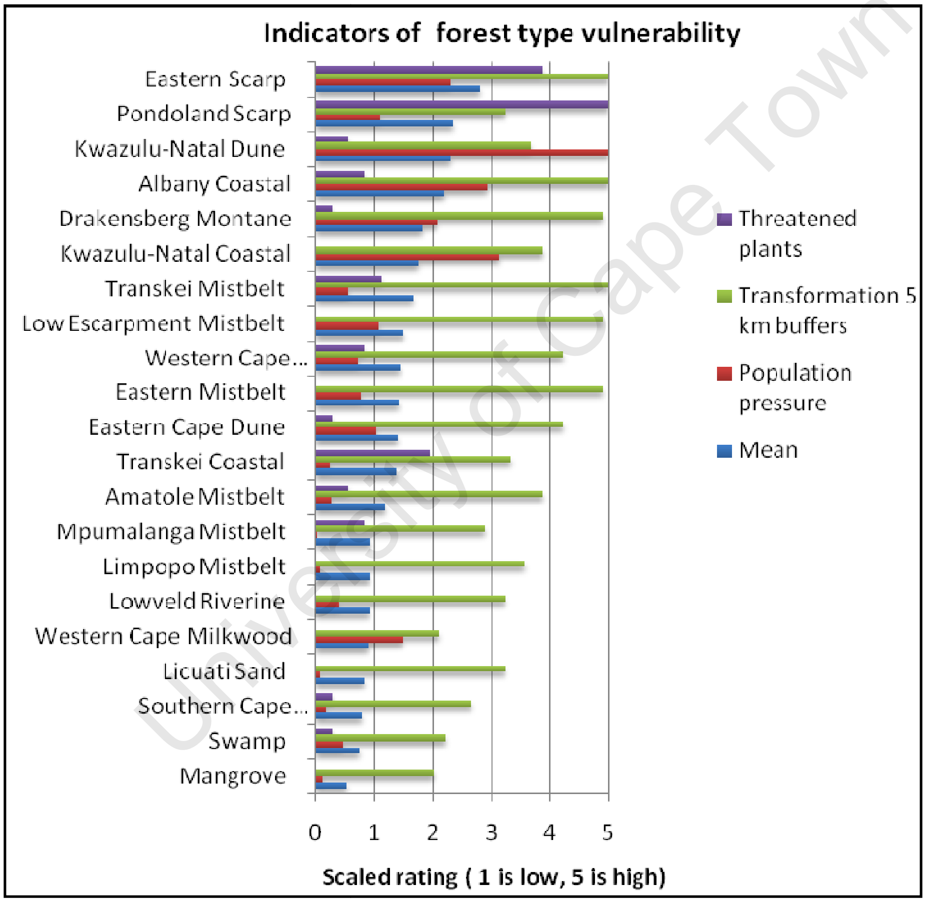


Figure 21.Three indices of vulnerability, normalised along a common zero to five scale. Forest types are ranked according to overall vulnerability by taking the mean of all three scaled ratings

According to this analysis, three forest types, Eastern Scarp, Pondoland Scarp and Kwazulu-Natal Dune forests stand out as being the most vulnerable to biodiversity loss.

5.4 Discussion

Although threats tend to act synergistically, and are often difficult to separate and quantify, it is well known that the biggest threats to forest biodiversity arise from population pressure and habitat change (see for example Dinerstein & Wikramanayake, 1993; Mittermeier *et al.,* 1998; Diamond, 2005). This analysis shows that different forest types show different degrees of vulnerability to these threats (see Figure 21).

Large scale land transformation has occurred over the last few decades across many parts of South Africa (Driver *et al.,* 2003; Driver, 2004), but mostly in areas suitable for agricultural cultivation and timber plantations. Because climatic and soil conditions suitable for supporting plantation forestry are also suitable for indigenous forests (Louw and Scholes, 2002) this form of land use may pose a specific threat to indigenous forests. While legislation forbids direct removal of indigenous forests, this has not prevented the land surrounding forests from being transformed. Matrix transformation exacerbates the effects of habitat fragmentation, particularly in patchy landscapes such as forests (Saunders *et al.,* 1991 Forman, 1995; Murcia, 1995; Hunter, 1999; Laurance, 2000; Lindenmayer, and Franklin, 2006). To what extent matrix transformation has affected South Africa's forests is still not clear. It has been suggested (Lawes *et al.*, 2007a), that South Africa's naturally fragmented Afrotemperate forests may be resilient to further fragmentation effects. However, limited research in South Africa indicates that matrix transformation results in decreased species dispersal ability, and increased vulnerability of isolated metapopulations to local extinctions (Swart and Lawes, 1996; Lawes *et al.,* 2000; Lawes, 2007a). Research in northern hemisphere forests (Hanski and Gilpin, 1991; Taylor, 1991; Wu, 1994; Thomas, 1994, Wiens, 1997; Verheyen *et al.*, 2004) has shown that survival of forest species metapopulations are profoundly influence by connectivity between patches, with the type and extent of land use change, as well as the inherent dispersal ability of the species concerned being important determinants of this. It has also been shown that forests adjacent to transformed lands have significantly altered ecotones compared to forest adjacent to natural grasslands (Cooper, 1985; Everard, 1986; Van Wyk and Everard, 1993; Kotze and Samways, 2004), and that the forest ecotone is important for maintaining certain forest ecosystem processes (Furley *et al.*, 1992; Noble, 1993; Everard, 1994; Martin *et al,.* 2007).

The combination of land use change, habitat fragmentation and climate change are synergistic in their impact on species (Hilty *et al.,* 2006). While species may have responded to past climatic challenges by moving across the landscape, (Balmford, 1996; Eeley *et al.,* 1999), recent human-induced changes to the forest matrix have limited their ability to do so.

Because threats arise from different causative agents and are often systemically connected, aggregation into a single threat index is difficult. Despite this, three forest types, Eastern Scarp, Pondoland Scarp and Kwazulu-Natal Dune forests stand out clearly, as being the most vulnerable to biodiversity loss. Two of these forests types, Eastern Scarp and Pondoland Scarp were rated by experts as having already undergone high levels of degradation (see section 2.3 How degraded are our forests?. In addition, Pondoland Scarp forests with the lowest level of formal protection, (around 10 %), and the highest number of endemic plants (see chapter 3), makes this forest type probably the most important conservation priority in South Africa.

Contemporary threats from coastal development and mining have caused important losses of forests but these are mostly localised and affect a relatively small percentage of the total forested area. As shown in Figure 20, the harvesting of medicinal plants poses one of the most important threats to red data forest plants. Alien invasive plants, although not considered specifically in this study as a major threat (due to difficulties in quantifying impact), may be in some cases important secondary threat factors that exacerbate forest and matrix degradation and habitat loss (personal observation).

CHAPTER 6: SELECTION OF PRIORITY FOREST PATCHES

6.1 Introduction

The indigenous forests of South Africa cover a wide spectrum of macroclimates, land forms and geologies; this variability is reflected in the diversity of different forest types. However, this diversity is not adequately reflected within the current forest protected area network and many unique and important forests remain outside of the formal protected area network (see chapter 4). This situation is not unique to South Africa, and in most countries the selection of protected areas has been determined more by public opinion, political, economic and aesthetic factors rather than on trying to achieve biodiversity representivity (Mittermeier *et al*., 1998; Cowling *et al.,* 1999a; Margules and Pressey, 2000). In addition, most protected area systems have not taken into account the conservation of ecological processes and their importance for biodiversity persistence in their design (Rouget *et al*., 2003b).

To address this shortcoming, the approach of systematic conservation planning was developed to objectively identify priority areas for biodiversity conservation, by considering threats, patterns of biodiversity distribution, and the ecological and evolutionary processes that sustain them (Pressey, 1999; Cowling *et al*., 1999a; Margules and Pressey, 2000; Cowling and Pressey, 2003; Driver *et al*., 2003). However, the systematic approach requires reliable quantitative information on biodiversity distribution that may not always be available, too costly or time consuming to collect. More readily available surrogates of biodiversity patterns, such as vegetation maps (Desmet, 2004), broad habitat units (Cowling and Heijnis, 2001), land systems (Pressey and Taffs 2001) are typically used as approximations that are usually sufficient for planning at broader scales but less so at finer scales. Shortcomings in available data, has also required the use of expert judgment to supplement quantitative assessment.

As many valuable forests occur within communal areas of South Africa and their use forms an important component of rural livelihoods, identifying suitable forest for protected area expansion will need to take into account the socioeconomic context, in particular traditional forest ownership and use rights. Given these complexities, and limitations of available data, the selection of representative priority forests will best be achieved by drawing on both quantitative (systematic) and qualitative (expert driven) approaches.

Apart from the work of Eeley *et al*., (2001), which considered only forests within KwaZulu-Natal; no systematic assessment has been done to prioritise forests across the whole country, this study is the first to attempt to do so. The aim of this chapter therefore is to determine priority forest patches most in need of conservation action.

6.2 Methods

Two broad methods of identifying priority forest were used, quantitative systematic assessment and qualitative expert judgements. These two methods were not totally independent as experts made use of the results of the systematic assessment.

6.2.1 Systematic assessment

Essential to systematic conservation planning is quantitative data on the spatial distribution of biodiversity within the planning domain. Where biodiversity information is incomplete, conservation planners are forced to make use of biodiversity surrogate information to provide approximations of the distribution of biodiversity. Because of the limited geo-referenced forest species distribution data, the forest types of Von Maltitz *et al.* (2003) were used as surrogates to represent forest biodiversity

Spatial data analysis was done within a Geographical Information System, using ARC View 3.2. Additional data analysis was done in spreadsheets and Microsoft Access database. C-Plan was used to calculate irreplaceability values, and MINSETS (the minimal number of planning units to satisfy targets). C-Plan is a conservation-planning computer decision support tool developed by the New South Wales National Parks and Wildlife Service (Anon 1999).

The choice of appropriate planning units is critical to systematic conservation planning, as they represent the units of selection for prioritisation. Forest patches were used as planning units.

The systematic assessment considered priority patches to be those that were 100 % irreplaceable and that are were classified as having a high threat rating.

6.2.2 Expert assessments

Two separate expert assessments were used:

a) *DWAF expert selected forests*

A panel of experts from the Department of Water Affairs and Forestry were provided with the priority list derived from the systematic assessment (using C-plan). This list was reviewed by the experts and further comparisons were also made with forest conservation reviews available in the literature (specifically those of Cooper, 1985; Cooper and Swart, 1992; Eeley and Lawes, 1999; and DWAF, 1992).

b) *Selecting threatened forests ecosystems for listing.*

Section 52 of the National Environmental Management: Biodiversity Act (Act 10 of 2004) provides for the listing of threatened ecosystems or vegetation types. The South African National Biodiversity Institute (SANBI) using the approach of Rouget *et al.* (2004) is in the process of identifying endangered ecosystems and vegetation types for listing under this legislation. The approach is essentially an adaptation of the method used by IUCN to identify red data species (Baillie *et al.,* 2004; IUCN, 2007). It relies on a set of standard criteria each with biome specific thresholds. When thresholds for criteria are exceeded, a specific rating is triggered. The criteria and thresholds used for forests are given in Table 23, below.

Table 23. Criteria and thresholds used for selecting threatened forests and forest ecosystems (forest types). Note criteria codes are those used by SANBI, only criteria used for forest are shown. Abbreviations: CR= critically endangered; EN = endangered; VU = vulnerable.

Code	**Criteria**	**Thresholds**			**Assessment method**
		CR	**EN**	**VU**	
A2:	Ecosystem degradation and loss of integrity	Not valid	If > 40 % of remaining area is degraded	If > 20 % of remaining area is degraded	Expert assessment with supporting quantitative information (on harvesting pressure and forest matrix transformation)
C:	Limited extent and imminent threat	Not valid	Remaining area < 3000 ha	Remaining area < 6000 ha	Threat determined by systematic conservation planning

					assessment (Berliner *et al.*, 2006) in conjunction with expert judgments.
E:	100 % irreplaceable and high threat (Systematically derived priority areas for meeting explicit biodiversity targets)	Not valid	High	Medium	Systematic conservation planning assessment (Berliner, 2006) with review by forest experts from DWAF and provincial authorities.

Candidate forests for listing were drawn from the integrated list of priority forests developed for this thesis, and presented in appendix C. Testing of criteria was done during a series of expert workshops facilitated by SANBI.

6.3 Results

6.3.1 Using systematic assessment to select priority forest patches

Approximately 2% of all forest patches had irreplaceability values of between 0.8 and 1, with 111 (0.55%) falling into the 100% irreplaceability class. This represents more than 40% of the available forest area being assigned an irreplaceability value of greater than 0.8 and with 25% being totally irreplaceable. Results indicate that relative to the total number of patches, few patches are highly irreplaceable, but because they tend to be the larger patches, these patches represent a significant proportion of the forest area (see Table 24, below).

Table 24. Irreplaceability statistics: number of patches selected, area, and percentage of total forest area.

Irreplaceability class	Number of patches (% of total patch number)	Area (km2)	% of total forest area
1	111 (0.55)	1 272.42	25
0.8–0.99	278 (1.35)	855.53	17
0.6–0.8	143 (0.70)	262.04	5
0.4–0.6	171 (0.83)	226.50	4
0.2–0.4	413 (2.01)	416.64	8

The MINSET function is a modifiable algorithm used as a tool to identify an approximate ‘minimum set’ of sites that meet targets (anon, 1999). The following rules were used in the algorithm:

- Select site with highest irreplaceability, if tied –
- Select site with largest area, if tied –
- Select next site.

The algorithm selected a set of 323 sites, with a total area of 4 19.84 km^2. Because the larger patches are selected, they represent a small percentage of the available patches, while at the same time covering the majority of the available forest area. Exceptions to this are Lowveld Riverine, Drakensberg Montane, Northern KwaZulu-Natal Misbelt and Western Cape Milkwood forests where the selected patches represent more than 50% of the available patches. Of the 16 000 plus patches considered, 111 forest patches were 100 % irreplaceable. Of these only 33 patches scored a 'high' threat rating (see chapter 5); these are considered the priority or 'hot spot' forest patches' and are given in Table 25, below.

Table 25. Priority Forest patches identified as 100 % irreplaceable by C-plan, and with an overall threat rating of 'high'. Note that for some forests, multiple patches with the same name were selected.

Forest type	Name or management unit	Size (ha)	Protection status.
Albany Coastal	Langebosch (multiple patches)	341, 2062	None
Albany Coastal	Woody Cape NR (multiple patches)	6005, 1432	largest patch in Type 1 PA
Eastern Mistbelt	Weza/ Ngele (multiple patches)	1053, 854	DWAF special NR
EC Dune	Hamburg coast	1400	None
EC Dune	Kiwane Coastal Forest Reserve	598	None
EC Dune	Mgwalana/ Begha Mouth (multiple patches)	4452, 213	Partially in Type 1 PA
KZN Coastal	Dukuduku	1069	Type 1 PA
KZN Dune	Amatikulu	576	Type 1 PA
KZN Dune	Mtunzini	2521	None
KZN Dune	Sokhulu dune forest	3800	Partially in Type 1 PA
Mangrove	Mhlatuze Richards Bay Game Reserve	975	Partially in Type 1 PA
Mangrove	Mngazana Mangrove	104	Partially state forest
Mangrove	Sokhulu mangrove	1069	Type 1 PA
Mangrove	St Lucia Mangrove	177	Type 1 PA
Swamp	Eastern Shores Swamp	406	Type 1 PA
Swamp	Mnbzwana swamp	303	Type 1 PA
Transkei coastal	Hili/ Ntsubane	1279	State forest
Transkei coastal	Mount Thesinger (multiple patches)	1390, 583, 548, 538,	State forest
Transkei coastal	Mpame	538	State forest

Transkei coastal	Ntlopeni/ Mkomanzi	632	State forest
Transkei coastal	Ntsubane/ Lotana	978	State forest
Transkei coastal	Ntsubane/Uzimpunzi	793	State forest
WC Milkwood	Stanford forest (multiple patches)	99 , 80, 101	None

6.3.2 Additional patches selected by DWAF expert panel

The results of the systematic assessment presented in Table 25, were reviewed by a panel of forest ecologists from the DWAF (Izak van der Merwe, Department of Water Affairs and Forestry, personal communication.). The expert panel was in agreement with all the forests selected by the systematic assessment, but nominated an additional 22 forest patches known to be of high biodiversity value, and under high threat that were not selected by the systematic assessment (note that many of these additional forest, were given 100 % irreplacebility ratings, but did not qualify as priorities, because they were not classified as having a 'high' threat rating by the systematic assessment). The integrated list of priory forest patches given in appendix C, includes both those selected using the systematic assessment and the additional forests identified by the DWAF expert panel.

6.3.3 Threatened forest ecosystems

The forest types and forest patches and complexes that met the criteria described in Table 23, are presented in Table 26, below. The forest type listings were triggered either by criterion A2 (ecosystem degradation and loss of integrity), or criterion C (limited extent and imminent threat), while the forest patches and complexes were triggered by criterion E (100 % irreplaceable and under high threat).

Table 26. Priority forests (clusters or patches) and forest types identified for listing as 'threatened forest ecosystems under Section 52 of the National Environmental Management: Biodiversity Act. Criteria codes that triggered listings are given. Abbreviations: EN = endangered; VU = vulnerable.

Forest types	Criterion A2	Criterion C	Criterion E
Mangroves		EN	
Western Cape Milkwood Forests		EN	
KwaZulu-Natal Coastal Forests	EN		
Lowveld Riverine Forests	VU		

Swamp Forests	VU	VU	
Transkei Coastal Forests	VU		
Pondoland Scarp Forests	VU		
Eastern Scarp Forests	VU		
Eastern Midlands Forests	VU		
Forest patches and clusters			
Mount Thesiger Forest Complex			EN
Sokhulu/Maphelana Forest			EN
Mtunzini Forest Complex			EN
Ongoye Forest			EN
Ngome Forest			EN
Kobongaba Forest Complex			EN
Stanford (Grootbos) Forest Complex			EN
Blouberg Forest			EN
Ntimbankulu Forest			EN
Hlabisa Forest Complex			EN
Greefswald Riverine Forest			EN
Mhlatuze Mangrove Forest			EN
Mngazana Estuary Mangrove Forests			EN
Futululu Forest			EN
Umdoni Forest			EN
Hawaan Forest			EN

6.4 Discussion

This chapter raised two important issues pertinent to the process of conservation planning of forests. These include: the importance of combining expert judgements with systematic assessments and the issue of incorporating socioeconomic costs into reserve selection; these are briefly discussed below.

Using expert judgment in conjunction with irreplaceability analysis

Expert-based judgements and systematic computer algorithms have been considered as two alternative approaches to identifying priority areas for conservation action (Prendergast *et al.,* 1999; Dinerstein *et al.,* 2000; Cowling *et al.*, 2003a; Strager and Rosenberger, 2006). The advantages of drawing on the benefits of both approaches to select priority conservation areas are well recognised by conservation planners (see for example Store and Kangas, 2001; Cowling *et al.,* 2003a; Failing and Gregory, 2003; Strager and Rosenberger, 2006). Both approaches are particularly useful, but where quantitative spatial information is outdated, uncertain, incomplete, or not available, expert judgements become essential. In this study, some forests were not mapped at all, or the situation on the ground had changed completely since the data was collected (for example, the demographics around some forest have changed since population were counted in 2001). In such situations, the inputs of experts familiar with conditions on the ground are invaluable.

To me, the systematic approach has the advantage over expert driven approaches by providing a less subjective, more rigorous and region wide assessment of the options for achieving explicit conservation targets. However, the disadvantages of this approach include a heavy requirement for quantitative spatial data, and an overreliance on simplistic and mechanistic solutions for problems that may involve considerations not easily included within mathematical optimization routines. Further, the real advantages of using expert-driven approaches (although this depends on the quality of experts) is that they are rapid, cost effective and can often incorporate wider, context specific considerations, (such as management and implementation issues). In addition, the explicit involvement of recognised experts in the planning process can provide the added advantage of improving credibility of the plan amongst stakeholders.

It is difficult to evaluate the effectiveness or efficiency of a systematic assessment against an expert based selection. However, what is evident is that the two approaches can and should be used to complement each other. Purely expert driven approaches or combinations with systematic assessments are unlikely to provide mathematically optimal solutions; because by adding or removing planning units to the minset calculated by the systematic assessment you are moving away from the mathematical optimal solution, even though this will probably be improving the overall effectiveness of the reserve design the efficiency of reserve design may decline.

Socioeconomic costs of reserve selection

Conservation is just one goal of landscape management; it has to be considered along with the many other needs of people. The need to develop conservation landscapes that ensure the persistence of biodiversity whilst minimizing impacts on the livelihoods of local people is a well established principle in conservation planning (see for example Dinerstein and Wikramanayake, 1993; Faith *et al.,* 1996; McN*Eeley*, 1997; Margules and Pressey, 2000; Faith *et al.,* 2001; Drechsler and

Watzold, 2001; Driver *et al*., 2003; Luck, *et al.,* 2003; Desmet, 2004). However, as pointed out by Luck *et al.* (2003) few studies have dealt with conflicting socioeconomic interests in a manner that is fully accountable. The need for this is particularly relevant in developing countries like southern Africa, where levels of rural poverty and dependency on natural resources are high (Child, 2004; Shackelton *et al.,* 2007). The establishment of protected areas to promote biological conservation can imposed heavy opportunity costs on local people when protected areas exclude, or even limit, access to subsistence resources (Mayer, 1997; Child, 2004).

The approach used in this study, and many others (see for example Pressey *et al.,* 1996; Pressey and Marguels, 2000; Noss *et al.,* 2001) is to select priority biodiversity areas, based on sites with the highest irreplaceability value and vulnerability/threat rating. However, threats to biodiversity (for example subsistence resource use and agriculture) can also be considered as 'socio-economic opportunity costs' that would be incurred to society by setting aside a particular site for protection. If the aim of prioritization is explicitly stated to select a set of representative sites for protection, that minimizes the overall socioeconomic costs to society, then prioritization should, ideally include some form of cost/benefit analysis that includes socioeconomic opportunity costs of each planning units. For example, this was done by Faith *et al.,* 1996; Faith *et al.,* 2001; and Williams *et al.,* 2003, who used a 'complemetarity value' of each site relative to its weighted cost, to prioritize sites according to the 'net benefit to society'. However, most conservation planning studies can seldom account for socioeconomic costs directly, rather they equate costs with 'threats' to biodiversity conservation, and use this to schedule (the level of urgency) of sites for conservation action (see for example Presey *et al*., 1996; Noss *et al.,* 2001; Pressey, and Taffs, 2001; Rouget *et al.,* 2004).

A conceptual model for potentially applying a cost/ benefit approach to selecting priority forests were developed and presented in the Table 27, below. The approach uses a set of indicators to measure biodiversity and socioeconomic benefits and costs. However, due to difficulties in quantifying many of the indicators this approach was not applied here.

Table 27. A proposed framework for applying cost /benefits analysis to selection of conservation area networks (CAN).

Types of benefits and costs		Indicators	Measures
Benefits	Biodiversity	Representivity.	Degree to which species or vegetation types are represented in the CAN.
		Effectiveness/persistence	Landscape connectivity. Population viability analysis of key species.
	Socio-economic	Tourism and job creation potential. Spinoff potential of small businesses.	Estimated number of jobs.
		Attractiveness for funding (donor or local).	Types of protected areas.
Costs	Biodiversity	Implementation drag and cost of implementation.	Time to implement and Cost of implementation.
		Efficiency.	Unit area per unique species for the CAN
	Socio-economic	Opportunity cost of restricting or not allowing harvesting of forest resources.	Cost of buying resources, or changing to alternatives (fire wood, medicinal plants, building materials etc.).
		Opportunity costs of not allowing other forms of land use (agriculture and mining) in protected areas.	Net profit from agriculture and mining less environmental costs.

Four main problems associated with using cost/benefit analysis in forest conservation planning are identified. These include the following. First, given the extremely limited extent of the forest biome in South Africa, as well as the high degree of fragmentation, almost all of the larger forest patches of each forest type will be 100 % irreplaceable. This provides few options for negotiating alternative protected areas (or planning units) that may have lower socioeconomic costs. Second, accounting for the socioeconomic opportunity costs are seldom straight forward, and direct trade-off with biodiversity benefits may be inappropriate. For example when biodiversity values are closely related to cultural-spiritual values of forests, equating these into monetary terms is problematic. Third, manifestation of opportunity costs and biodiversity benefits usually occur across different time scales and generations. The full biodiversity benefits typically manifest over the long term, and across generations, whereas economic benefits typically are accounted for over the short to medium term, and always within the same generation. Fourth, biodiversity values are not easily translated into marketable economic values (Piece and Moran, 1994), this being part of the problem known as the 'discounting dilemma', because of the difficulty in finding suitable discount values for biodiversity without disregarding the welfare of future generations (Daly and Cobb, 1989; Pearce and Moran, 1994; Gowdy and McDaniel, 1995). For example, under the logic of discounting, it may be economically rationale to exploit all forest timber resources at once, and forgo any biodiversity related benefits to future generations.

In South Africa many forests with high biodiversity value are also important for their socioeconomic value to local communities, but many of these forests are currently facing degradation from non sustainable use. In this regard, the so called 'Participatory Forest Management approach' or PFM, as entrenched within policy directives of the South African Department of Water Affairs and Forestry are particularity relevant. This approach strongly promotes the sustainable use paradigm over strict protectionism. Central to PFM's policy is to 'improve access to forests resources' as well as to 'entrench the right, moral or otherwise for local communities to benefit from forest resources" (White Paper on Sustainable Forest Development, 1996). While these policies, appropriate as they may be within the context of a post apartheid government aiming to address equitable access to resources, they do carry a potential danger of promoting unrestricted access to forest resources without adequately ensuring their protection. It is therefore critical that the Participatory Management Approach firmly entrenches biodiversity conservation as a foundation principle.

CHAPTER 7: SELECTION OF PRIORITY FOREST CLUSTERS

7.1 Introduction

This chapter applies a landscape approach to forest conservation planning. The landscape approach emphasises the spatial configuration of habitat patches, their connectivity, exposure to the matrix, as well as the processes that operate at multiple temporal and spatial scales within the landscape. Unlike the previous chapter where the focus was at the scale of the forest patch, this chapter considers the persistence of forest biodiversity within the context of the broader forest- landscape matrix.

Forest conservation has two key components: the conservation of forest pattern (species and ecosystems) and the maintenance of forest processes (critical ecosystem functioning important for long term persistence of forest biodiversity). These two facets are interdependent in that continued system functioning requires components to be intact. Even if the current forest protected area network was representative of all forest ecosystem types, and these were effectively protected, this may still be insufficient to ensure the long term persistence of forest biodiversity if ecosystem processes are not conserved.

I argue that the current forest conservation area network is a reflection of the past approaches to forest conservation in South Africa, which have largely been rooted in a silvicultural perspective. The silvicultural perspective views forests as commodity-oriented objects and their management is aimed at optimising timber production, while virtually ignoring non timber values (Miller and Seidel, 1990; Curtis and Carey 1996; Simberloff, 1999). The silvicultural perspective essentially, considers the functioning of forests at the scale of the patch or stand, while largely ignoring processes occurring at broader, landscape scales. The result of this approach is a forest conservation area network that is mostly devoid of explicit planning at the level of landscape processes.

Previous studies of conservation priorities for South African forests (Cooper and Swart, 1992, Cooper, 1985; Eeley, *et al.,* 2001) were not explicit in applying either representation or process targets within their methods. Eeley *et al.* (2001) however, has pointed out the need to develop reserve selection procedures that account for processes, as well as pattern, and that our highly fragmented indigenous forests systems could collapse if processes are not maintained. The conservation targets used in the irreplaceability analysis (see chapter 7, and appendix A), are explicit expressions of the area requirements for representing forest biodiversity pattern (based on species-area curves, after Desmet and Cowling, 2004) and do not specifically refer to processes targets. To set explicit targets for forest processes would require an understanding of the complex relationships among multiple factors that determine interactions between spatial patterns of biodiversity and ecological processes. For example, disturbance and succession are key processes that influence spatial and temporal heterogeneity of forest biodiversity (Lindenmayer, and Franklin, 2002 Turner, 2005.), but to translate these processes into prescriptive spatial requirements is complicated by the contingent effects of climate, landform and bioegographic patterns. In addition, current patterns of forest biodiversity have also been influenced by historical impacts of humans on landscape pattern and process (see for example Foster *et al.,* 1998 and Foster *et al*. 1999 and McCracken, 2004).

The challenge then for conservation planners, is to design a forest protected area network that incorporates forest ecosystem processes that operate across a range of time and space scales. To do this, requires a shift in thinking from the traditional silvicultural perspective to an ecosystem-landscape approach in which forests are considered as part of the broader landscape system.

In my view, the central themes of a landscape approach to forest conservation planning should include: integration of ecological processes across multiple scales; incorporation of the forest matrix in planning for the persistence of these processes; and an understanding of the effects of habitat change and fragmentation on these processes. An elaboration of these themes within the context of forest conservation planning is briefly discussed below.

Forest ecological process

Ecological processes effecting forest occur across different spatiotemporal scales. Table 28 below, summarises the most important forest ecosystem processes operating across three major spatiotemporal scales.

Table 28. Forest ecosystem processes operating across three major spatiotemporal scales.

Spatiotemporal scale	Forest processes	References
Evolutionary/ bio-geographic scale (dispersal corridors)	Palaeoclimatic related contraction and expansion of forest biome. Evolutionary processes: speciation, radiation, and extinction events. Species adaptation to climate change.	Eeley *et al.* (1999); Noss (2001); Bennett and Wit (2001); Midgley *et al.*(2003); Pyke (2004); Lawes *et al.*(2007a); Lawes *et al.*(2007b).
Landscape scale (forest clusters and forest matrix)	Habitat fragmentation effects. Maintaining ecotones between forest margin and matrix. Meta population/ meta community dynamics, species dispersal and gene flow. Maintenance of natural disturbance regimes such as fire, wind and herbivory.	Ranney, *et al.* (1981) Swart and Lawes (1996); Cowling (1999a; 2000); Laurance (2000); Bennett and Wit (2001); Rouget *et al.* (2003b), Von Hase *et al.* (2003); Wethered and Lawes (2003).
Ecosystem scale (forest patches)	Forest establishment, regeneration and gap-phase dynamics, seed and seedling predation, seed dispersal, pollination, herbivory, reproductive processes, litter decomposition and nutrient cycling; fire effects on forest edges.	Dunning *et al. (*1992); Geldenhuys (1994); Midgley *et al.*(1990); Midgley *et al.*(1997); Kotze and Lawes (2007).

Importance of the forest matrix

The forest matrix can be defined as the surrounding landscapes that forest patches are embedded within. Many of these areas have important conservation values in their own right, for example the montane grasslands surrounding many Afromontane forest patches are rich in rare and endemic species, (Allan *et al.,* 1997; Armstrong and Hensbergen, 1997; Armstrong and Hensbergen, 1999; Mucina, and Geldenhuys, 2006); and at least a third of all priority forest clusters are situated within matrix vegetation that is listed as either critically endangered or endangered (see appendix D, 5th column). However, in South Africa the values of these areas have seldom been considered as important for maintaining forest biodiversity. Four interrelated arguments are presented below, motivating why I think the forest matrix should be explicitly included as a component of forest conservation planning.

Firstly, given the highly fragmented nature of the South African forest biome (Lawes, 1990; Lawes *et al.,* 2000; Mucina and Rutherford, 2006; Geldenhuys and Mucina, 2006) the survival of many forest dependent species depends on their ability to disperse between forest fragments. Many forest species occur as metapopulations within spatially separated patches (Swart and Lawes, 1996; Lawes *et al*., 2000).

Metapopulation theory has been used to understand population dynamics in fragmented populations (Levins, 1969, Levins, 1970; Opdam, 1990; Hanski and Gilpin, 1991; Wu and Louck, 1995; Picket and Rogers, 1997; Hansen and DeFries, 2007) and the importance of the matrix, in determining dispersal and connectivity between habitat patches occupied by metapopulations, are well recognised (Taylor,1991; Thomas, 1994; Swart and Lawes, 1996; Noon and McKelvey, 1996; Wiens, 1997; Vandermeer and Carvajal, 2001;Wethered and Lawes, 2003; Lawes ,2007b). Maintaining connectivity between habitat patches reduces the likelihood of local extinctions, by permitting exchange of plant and animals as well as their genetic material between metapopulations within a forested landscape (Solé *et al.,* 2004). Thus the matrix plays an essential role in determining metapopulation dynamics of many forest species.

Secondly, the historical patterns of fragmentation of South African forests are primarily the result of the repeated and often drastic changes in climate that have occurred, particularly since the Quaternary period (Lawes, 1990; Lawes *et al.,* 2000; Mucina and Rutherford, 2006; Lawes *et al*. 2007a). This repeated shrinking and expansion of forests into and out of the matrix has contributed towards a close ecological relationship between forest patches and their surrounding matrix. Some forests are essential for maintaining hydrological processes of the surrounding matrix, and vice versa, so that removal of one will compromise the other. The hydromorphic grasslands around Dukuduku forest are an example of this (van Wyk *et al*, 1996; Perrin and Bodbijl 2001a), as well as the palm-veld grasslands often associated with lowland coastal forest (Moll, 1976).

Thirdly, unlike the classical definition of the ‘metapopulation matrix’, as being ‘unsuitable space’ and important only for its ability to maintain species movements between patches, (Levins 1969, Levins, 1970), the matrix of South African forests provide valuable secondary habitat for many forest species. A large proportion of mammals and birds considered as typical ‘forest species’ make use of the forest-matrix ecotone and surrounding areas (Geldenhuys and MacDevette, 1989; Armstrong, 2003; Lawes, 2007b; Wethered and Lawes, 2003). These species are often classified as ‘forest associated’, rather than ‘forest dependent’ species (Kikira *et al*., 2007; Lawes, 2007b). For example, of the larger mammals characteristic of South Africa’s lowland forests, such as vervet monkey (*Chlorocebus aethiops),* blue duiker (*Cephalophus monticola)*, red duiker (*Cephalophus natalensis)*, bushbuck (*Tragelaphus scriptus)*, bushpig (*Potamochoerus porcus)* and banded mongoose

(*Mungos mungo)* (Cooper 1985) only the blue duiker is strictly forest dependent (Bowland, 1990), the others making seasonal or nocturnal use of the surrounding grasslands (Armstrong, 2003). Many of the forest associated fauna play an important role in seed dispersal of forest species (Tabarelli and Peres, 2002; Hall and Swaine, 1981; Gelenhuys, 1993). For Southern Cape Afrotemperate forests, Gelenhuys (1993) found that approximately 30 % of all forest tree species are dispersed by fauna, however Griffiths and Lawes (2007) found this to differ between forest types. Afrotemperate forests having the highest incidence of wind pollination (consistent with the steep topography, seasonally dry environment, and limited resource availability in the habitat); Scarp forests with a high incidence of abiotic (explosive and wind) dispersed seeds; and Coastal forests having the highest incidence of species with fleshy fruits, consistent with zoochory. The close relationships between forests and the matrix are evident in the fact that many species responsible for seed dispersal of forest trees are dependent on both matrix and forest habitat.

Fourthly, anthropogenic climate change presents a potentially severe threat to highly fragmented forest landscapes. Species will be required to disperse, relatively speaking, rapidly through landscapes in order to keep pace with the changing climatic conditions. An important challenge for conservation is therefore to manage fragmented landscapes so as to assist species in tracking changing environmental conditions to which they are adapted (Pearson and Dawson 2005). The combination of land use change, habitat fragmentation and climate change, will be synergistic in their impact on species (Hilty *et al.,* 2006). While species may have responded to past climatic challenges by moving across the landscape, (Balmford, 1996; Eeley *et al.,* 1999), recent human-induced changes to the forest matrix have limited their ability to do so. Not only have land-use regimes within the matrix disrupted ecological connectivity between patches, but also the natural disturbance regimes that maintain forest edges (Kotze and Lawes, 2007; Lawes *et al*. 2007b). During the palaeoecological forest contractions, many species survived because they could move between patches. Today, not only has the matrix become less suitable for species movements, but the distances between patches have also increased, with the disappearance of many of the smaller forest patches (Kotze and Lawes, 2007). These small forest fragments being often also the most vulnerable to anthropogenic disturbances (Midgley *et al*., 1997; DWAF, 2003; Lawes *et al.,* 2004b; Lawes *et al.,* 2005; Lawes *et al.,* 2006).

The effects of fragmentation

Understanding the effects of habitat fragmentation is central to the landscape approach. South African forests are naturally highly fragmented, primarily the result of the repeated and often drastic changes in palaeoclimate. This has been exacerbated by anthropogenic fragmentation causes by loss of many smaller patches (Kotze and Lawes, 2007), shrinkage of larger patches (Cooper and Swart, 1992), and transformation of the forest matrix, with over 35 % of the total forest matrix area already transformed (Berliner *et al*., 2006). Fragmentation disrupts landscape level ecosystem processes and increases the vulnerability of forests to biodiversity loss.

The aim of this chapter is to identify a forest protected area network that will ensure the persistence of forest biodiversity. To do this requires a protected area network representative of forest biodiversity pattern and process. The representation of pattern can be explicitly achieved through forest type targets; while the difficulties of spatially representing processes can only be approximated by using whole forest clusters (including the surrounding matrix) as planning units. This was done by the following steps:

- delineate forest patches into spatially related planning units or clusters;
- assess levels of connectivity between patches within each cluster;
- assess levels of vulnerability of forest clusters;
- determine irreplaceability of forest clusters using MARXAN;
- evaluate results using expert judgements;
- revise and integrate the selection algorithm using a rule based expert system selection algorithm.

7.2 Methods

7.2.1. Matrix transformation

National Land Cover classified satellite imagery (NLC, 2000) was used to determine levels of habitat transformation, within both the forest cluster matrix, and five kilometre buffer areas around each forest patch. The former was used as an indicator of the integrity and connectivity of the clusters, while the latter provided a crude measure of ecosystem integrity of the broader landscape surrounding the forest patches.

7.2.2 Edge exposure

A relatively simple method used to determine changes in fragmentation of patchy habitats is to measure the overall perimeter (edge) relative to the total area (McGarigal, 2002). This ratio is also known as the edge exposure (Laurence, 2001; Solé *et al.,* 2004) and is a function of the shape of the patch. It provides a measure of the amount of forest directly exposed to the surrounding matrix. One way to express this mathematically is to derive an index of forest shape made relative to the edge exposure of a perfect circle, which is the lowest possible perimeter to area ratio of any shape, giving a shape index value of one, and increasing upwards for increasingly complex shapes. This was done using the following formula:

$$SI = P / (2 (A \pi)^{0.5})$$

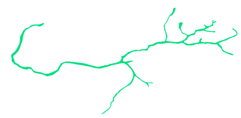
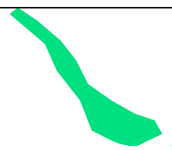
Where SI is the shape index, P is perimeter and A is the area of the patch. To facilitate comparisons between forest types, a scaling formula was applied to the shape index values, so that forest patches with the lowest edge exposure ('near circular' patches) had values that approached zero and patches with high edge exposures (complex shapes) approached one. The following scaling formula was used:

$$SI_{scaled} = SI_i / SI_{max}$$

Where SI_{max} was the highest shape index value obtained for a forest patch. The scaled shape index values are a percentage of the highest value in the resultant set represented between 0 and 1.

Using this scaling formula, a forest patch that is very narrow or with an irregular and complex perimeter, will have a scaled shape index that approaches 1 (high edge exposure), while a forest patch that is close to circular in shape will have a scaled shape index that approaches 0 (low edge exposure). Table 29 illustrates actual examples of different forest shapes and their shape indices.

Table 29. Examples of forest patch shapes and shape indices.

Forest patch shapes	Edge exposure	Shape index
	Very high	1
	High	0.5
	Medium	0.34
	Low	0.1

7.2.3 Vulnerability

Pressey *et al.* (1996) define vulnerability as the likelihood or imminence of biodiversity loss to current or impending threatening processes. Willson *et al.* (2005) extended this definition by distinguishing three dimensions of vulnerability, exposure, intensity and impact. I define 'landscape vulnerability' as the potential for threats to cause biodiversity loss. This can be measured by determining the degree of exposure (in time and/or surface area) and the magnitude of the threats. The perimeter to area ratio is used as a surrogate for the degree of exposure to threats, while the degree of matrix transformation is used as a surrogate for the combined magnitude of threats. Forest patches with a high shape index (high perimeter-to-area ratio), and that occur in highly transformed matrices will be most vulnerable; while forest patches with a low shape index (low perimeter-to-area ratio) occurring within untransformed matrices will be the least vulnerable. Vulnerability was expressed qualitatively as either 'high', 'medium' or 'low' according to the rules shown in Table 30, below.

Table 30. Vulnerability of forest patches expressed as a function of shape index and level of transformation in 5 km forest buffer areas.

Shape index	Transformation %		
	>50	30–50	<30
0 – 0.3	Medium	Low	Low
0.3 – 0.6	High	Medium	Low
> 0.6	High	High	Medium

7.2.4 Forest cluster analysis

The forest patch layer was converted to a grid (100 m resolution). Clusters were delineated using GIS spatial analysis that grouped all patches that were less than 500 meters apart. Clusters were classified according to total forested area, degree of transformation of matrix, river length running through the cluster, and total area.

Maintaining the connectivity of habitat patches is an important component of landscape level conservation planning (Fahrig, 1985; Bennett,1998; Briers, 2002; Nikolakaki, 2004). Forest clusters with high total forest area, low levels of matrix transformation and situated along river corridors were considered to have high ecological connectivity, while clusters with low total forested area, situated in a highly transformed matrix, and without connecting river corridors were considered to have low ecological connectivity. Table 31, below presents the rules used to derive the connectivity indices of forest clusters.

Table 31. Rules used to categorise a cluster connectivity index.

Cluster connectivity index (rating)	Forested area	Matrix	Rivers	Description
0 (Very low)	<25ha	–	–	Isolated forest patches
1 (Low)	>25ha	<50% natural	<1 500m	Forest cluster in transformed matrix
2 (Medium)	>25ha	<50% natural	>1 500m	Forest cluster in transformed matrix with good river connectivity
3 (High)	>25ha	>50% natural	<1 500m	Forest cluster in near natural matrix
4 (Very High)	>25ha	>50% natural	>1 500m	Forest cluster in near natural matrix with good river connectivity

7.2.5 Selecting priority forest clusters

A variety of selection criteria can be used to identify priority conservation areas, but ultimately the method is dictated by the availability of data. Typically, the selection criteria include: the biodiversity benefits/value of the site; economic and social costs of conserving the site; and the probability that that the site will be lost if no action is taken (Hilty *et al.,* 2006). Within the framework of systematic conservation planning, most selection has been based on biodiversity value and vulnerability of the site. Biodiversity value is determined using irreplaceability and complimentarily analysis, which is essentially the gain in representivity of biodiversity relative to targets, when a site is added to an existing set of areas, while vulnerability of the site is determined by analysis of threats (Margules *et al*.,1988; Pressey, *et al.,* 1994; Margules and Pressey, 2000; Possingham *et al*., 2000; Cowling and Pressey, 2003).

7.2.5.1 Using MARXAN

The conservation planning software program, MARXAN (Ball and Possingham, 2000) was used to determine irreplaceability values of forest clusters.

MARXAN calculated irreplaceability on the basis of three factors:

- target achievement (planning units that significantly contributes to achieving targets are favoured);
- planning unit 'cost' (planning units of lower 'cost' are favoured); and
- compactness (planning units connected to each other are favoured in the selection).

Planning units consisted of forest clusters, protected areas and sixteenth degree grid squares. Only Type 1 (statutory) protected areas were considered to contribute to target achievement. Targets used were those derived for the national forest types (given in appendix A). Relative costs of planning units were calculated using: a) the agricultural potential of the site, b) the proximity to development nodes (coast and towns) and, c) the subsistence use value of the forest (see section 5.2.4). GIS rule based modelling was used to aggregate threat values into a single rating (see appendix B).

Compactness was included by using MARXAN's 'boundary costs' feature. This modifying factor favours the selection of planning units that are grouped or connected. It makes conservation sense to minimize outside boundaries because planning units selected on this basis are more likely to function as linkages and corridors between protected areas. This is a particularly useful feature, given the importance of maintaining habitat connectivity in highly fragmented forests. The Figure 22, below shows results of the analysis for a section of the East Cape.

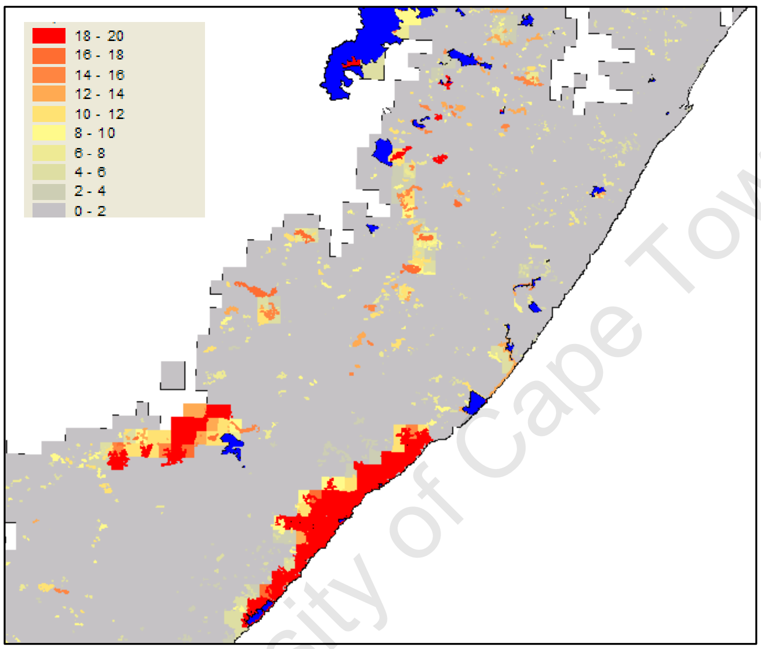


Figure 22. Map of forest irreplaceability for a section of the Eastern Cape Province. The numbers in the legend, relates to how often each planning unit (forest clusters) were selected out of 20 runs. Planning units selected 20 times out of 20 are 100% irreplaceable are shown in red.

7.2.5.2 Review of MARXAN results

Priority forest clusters were those with a 100 % irreplaceability (i.e. selected at least 18 times out of the 20 MARXAN runs), and ranked according to vulnerability ratings, forest clusters with the highest vulnerability being ranked as higher priority

The results of the MARXAN irreplaceability analyse on forest cluster were reviewed by experts from the Department of Water Affairs and Forestry (Izak van der Merwe (Forestry Technical and Information Services, Department of Water Affairs and

Forestry, personal communications). It was noticed that a number of the larger forest clusters known to have very high biodiversity value had been excluded from the MARXAN selection. This prompted the development of an alternative selection algorithm.

7.2.5.3 Using a rule based algorithm based on ecological heuristics

An alternative to using iterative computer selection algorithms, such as MARXAN and C-Plan, is to use rule based heuristics algorithms. Pearl (1984) defines heuristics as algorithms that produce acceptable solutions to problems in many practical scenarios, but for which there is no formal proof of correctness, or the ability to provide optimal solutions. Heuristics are typically used when there is no known method to find an optimal solution under the given constraints of incomplete data.

Reserve selection heuristics were derived using a number of ecological principles pertinent to forest biodiversity conservation. In a similar fashion Schulte *et al.,* 2006 identified a list of ecological first principles that can form the basis of any forest management plan. To compliment these heuristics, context specific information of forest experts were also used. Ecological principles are essentially heuristics that can be expressed as ‘rules of thumb (see Table 32).

Table 32. Description of the ecological heuristics and rationale used to derive criteria for selecting priority forest clusters.

Ecological heuristics	Rational/reference	Derived selection criteria (rules)
Connected patches, or those close to each other, are more ecologically viable than dispersed or unconnected patches.	Island biogeography theory: (MacArthur, and Wilson, 1967).	Use clusters of forest as selection units.
The largest forests are ecologically the most valuable and viable.	Island biogeography theory (MacArthur, and Wilson, 1967; Warburton, 1997; Wethered, and Lawes, 2003).	Select clusters with largest forested area.
The ability for species to disperse between patches improves ecological viability.	Metapopulation theory (Levins, 1969; Wiens, 1997; Lawes *et al.,* 2000).	Select patches that are best connected.
Forests embedded in a natural matrix are more likely to maintain natural ecosystem processes.	Landscape ecology (Pickett and White, 1985; van der Merwe and Seydack, 2005; Bender and Fahrig, 2005; Odion and Sarr, 2007).	Select clusters where matrix transformation is lowest.
Highly fragmented forests are more	Fragmentation theory (Saunders,	Select clusters with

vulnerable and will have fewer species and lower ecological viability.	*et al.,* 1991; Lawes, *et al.,* 2000. Bierregaard, *et al.,* 2001, Wethered, and Lawes, 2003).	largest mean patch size.
Conservation of forest under high threat will have high opportunity costs and low long term viability.	Minimise opportunity costs in reserve selection (Faith, *et al.,* 1996; Margules and Pressey, 2000; Burgess *et al.,* 2006).	Select clusters with lowest threat.
Risk of metapopulation extinction is reduced by selection of a minimum number of separate reserves (instead of one large).	Island biogeography and the design of natural reserves (Diamond and May, 1976).	Minimum number of three clusters per forest type
Achieve representation targets (based on species -area curves) while ensuring minimum area requirement for persistence of processes.	Target based systematic conservation planning (Margules and Pressey, 2000; Rouget, *et al.,* 2003b; Desmet and Cowling, 2004; Pressey *et al.,* 2003; Desmet, 2008).	Select clusters until conservation targets for each forest type are achieved.
Maintain ecosystem process by using catchment transformation thresholds.	Landscape ecology (Andrén, 1994; Fahrig and Merriam, 1994; Monkkonen and Ruenanen,1999; Fahrig ,2001; Fahrig, 2002)	Not specifically used here, but could preferentially select forests occurring in catchments that are still intact.
Protect forests with known occurrences of threatened species.	Threatened forest species are often restricted to certain forest types, with limited occurrence in certain patches.	Not explicitly used, due to poor geo-referenced species data, (but used implicitly in experts selection of priority forests).

The general approach of multi-criteria analysis was used to identify and compare criteria used in the selection rules. Multi-criteria analysis is useful tool to evaluate complex spatial problems (Store and Kangas, 2001). Criteria were ranked in order of importance, and translated into expert systems type rules with selection thresholds for each criteria (using Boolean 'If- THEN' statements). Each criterion is tested according to a sequential hierarchy (e.g. largest forest are best, least transformed matrices are best, least fragmented clusters are best etc.). The heuristics described in the Table 32 above, were translated into a set of reserve selection rules and presented in

Table 33, below.

Table 33. Rule based algorithm used to select priority forest clusters. Rules are applied sequentially according to a hierarchy of criteria. After testing a rule, forests are re-ranked according to the criterion to be tested by the next rule. This is done until targets are met (selection criteria are emphasised in italics within rules).

Rule no.	Selection rules
1	Delineated forest patches into *clusters* (using 500 meter buffers), clusters sorted according to dominant forest type.
2	Within each forest type, rank clusters according to *total size of forested area*.
3	For each forest type, identify the top three clusters with largest forested area: Test: do they have less than 20% of the *matrix transformed?* if yes, select all three and test rule 8 if no, then go to rule 4 .
4	Is the next largest cluster 50% smaller than 3rd largest? If yes, then test rule 5 If no, then test rule 6.
5	Is *cluster matrix transformation* less than 30%? If yes, then select, if no, reject and continue to rule 6.
6	From remaining clusters, select next cluster with *mean patch size* that is at least 15% larger than next, AND that has less than 20% of its matrix transformed, Else go to rule 7
7	From clusters with top three largest mean patch size, select cluster with *lowest threat rating*
8	Have a minimum number of three patches per forest type been selected AND, have *forest type targets** been met or exceeded? If yes, then stop If no, return to rule 6

*Revised forest targets for forest groups are those given in Table 34.

Targets used for this were those derived by Desmet (2008). These were calculated using the species - area curves and without any adjustment factors. Calculations were done at the level of forest group with forest types inheriting the target value from its group (see Table 34).

Table 34. Forest groups with their forest types, showing the revised forest targets (% area of forest required for protection) at the level of forest groups (from Desmet, 2008).

Forest group	% Targets	Forest types
I: Southern Afrotemperate	21.4	I1: Western Cape Talus Forests
		I2: Western Cape Afrotemperate Forests
		I3: Southern Cape Afrotemperate Forests
II: Northern Afrotemperate	21.8	II2: Northern Highveld Forests
		II3: Drakensberg Montane Forests
		II4: Low Escarpment Mistbelt Forests (previously KwaZulu Natal Mistbelt forest)
III: Southern Mistbelt	19.1	III1: Eastern Mistbelt Forests
		III2: Transkei Mistbelt Forests
		III3: Amatole Mistbelt Forests
IV: Northern Mistbelt	21.1	IV1: Limpopo Mistbelt Forests (previously Northern Mistbelt Forests)
		IV2: Mpumalanga Mistbelt Forests
V: Scarp forests	25.0	V1: Eastern Scarp Forests (previously Eastern Mistbelt Forests)
		V2: Pondoland Scarp Forests
		V3: Transkei Coastal Scarp Forest
VI: Southern Coastal	20.4	VI1: Eastern Cape Dune Forests
		VI2: Albany Coastal Forests
		VI3: Western Cape Milkwood Forests
VII: Northern Coastal	17.8	VII1: KwaZulu-Natal Coastal Forests
		VII2: KwaZulu-Natal Dune Forests
VIII: Tropical Dry	9.8	VIII1: Licuati Sand Forest

7.3 Results

7.3.1 Matrix transformation

For most forest types, significant areas surrounding forests have been transformed. At the broader landscape level (using 5km forest buffers) at least six forest types have more than 40 % of their surrounding land transformed. Timber plantations are responsible for much of the habitat loss surrounding forests (see section 5.3.3 for details).

7.3.2 Edge exposure

Analysis of area to perimeter ratios for different forest types shows that the degree of edge exposure differs significantly across forest types, but remains relatively constant between different size classes of the same forest type (see Figure 23 below). This implies that the same forest types will tend to have similar shape patches, irrespective of the size of the patch. The relationship between topographic setting, forest type and forest shape has also been noted by McNab (1996).

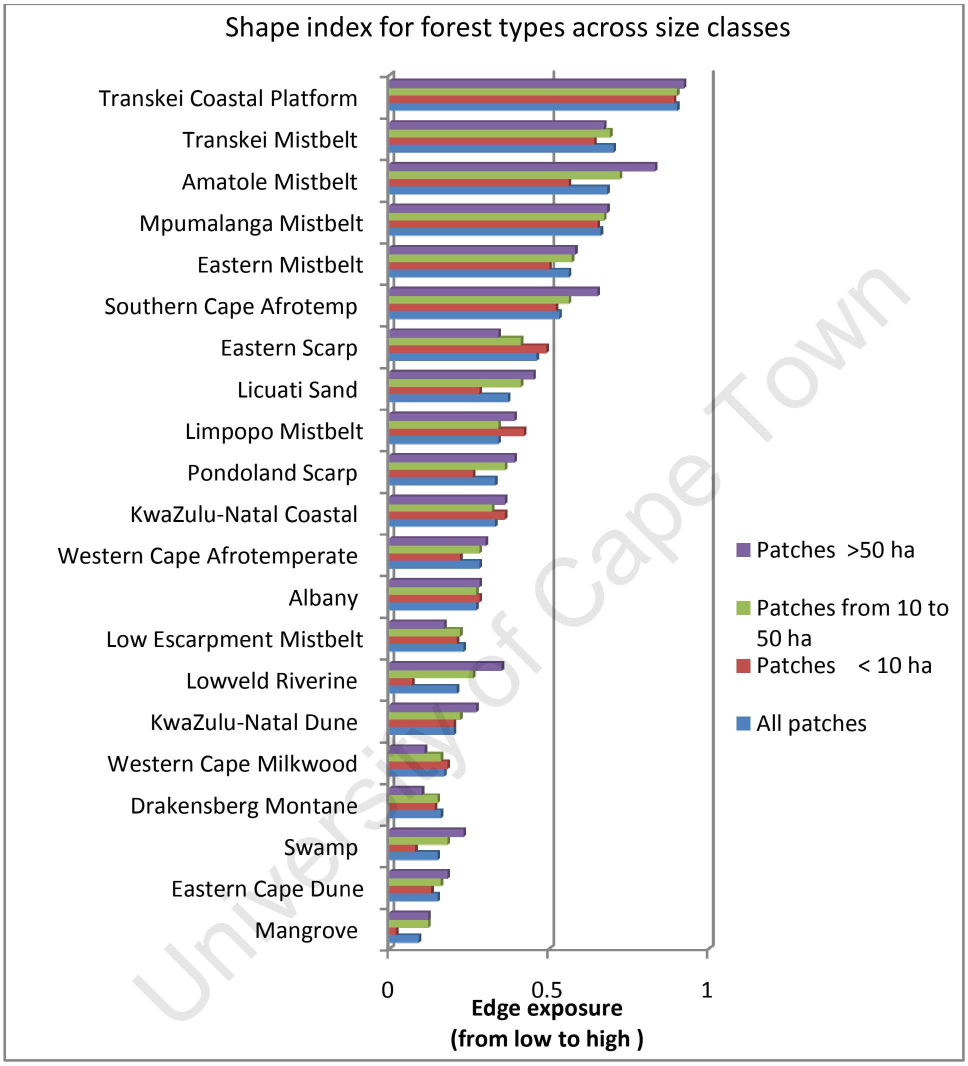


Figure 23. Comparisons of edge exposure for different forest types and across different patch size classes. Edge exposure is measured by the scaled forest shape index, from 0 (low) to 1 (high).

7.3.3 Assessment of vulnerability

The relative vulnerability of each forest type was determined from matrix transformation and edge exposure. Aggregation to forest type level was done by calculating the percentage of patches within each forest type that scored either 'high', 'medium' or 'low', out of the total number of patches (see Figure 24, below).

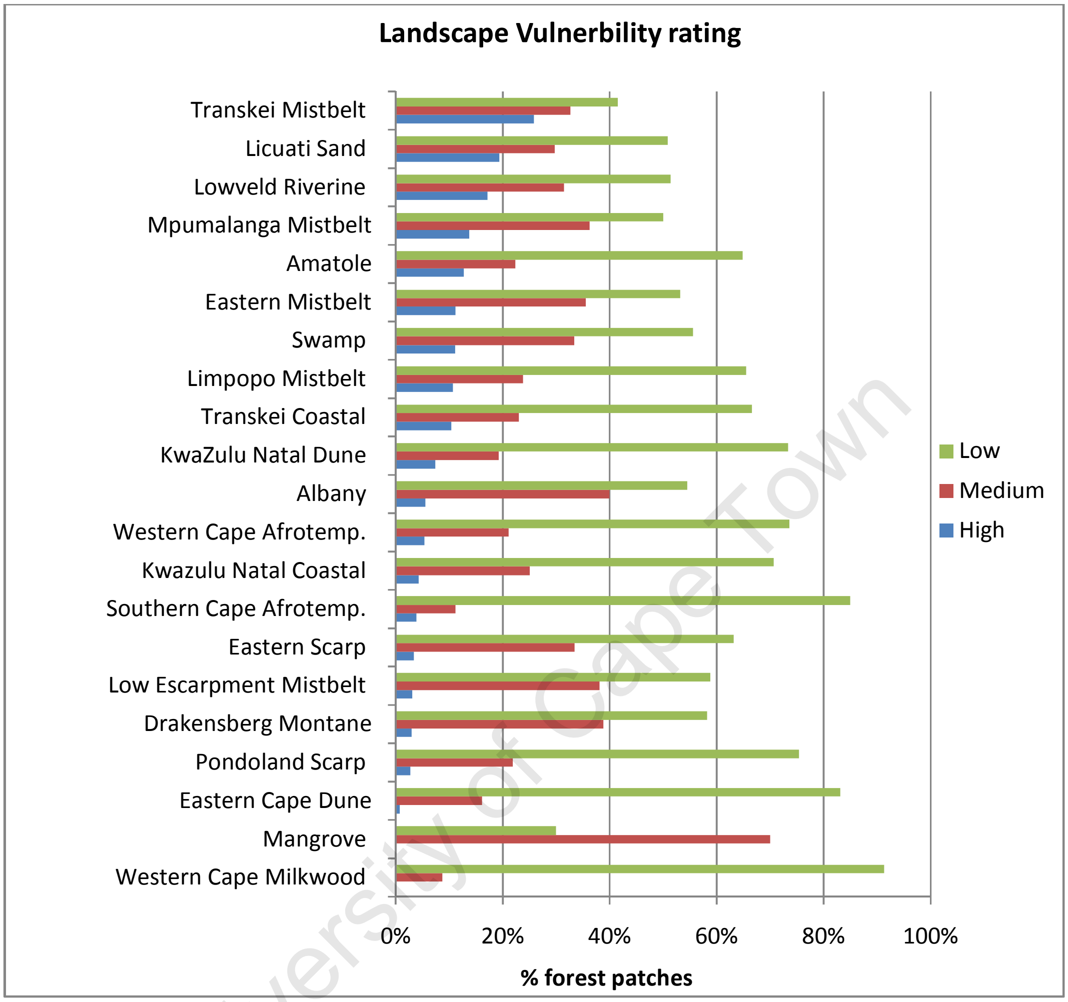


Figure 24. Landscape vulnerability ratings of forest types showing percentages of forest patches falling into each of the following class: 'high', 'medium' or 'low'. Classes based on degrees of matrix transformation and edge exposure. Forest types are sorted according to percentage in the 'high' vulnerability class.

7.3.4 Forest cluster analysis

Cluster analysis reduced the total number of planning units from 16 000 plus forest patches, to 2922 forest clusters. Results of the forest clusters analysis are given in

Table 35, below.

Table 35. Forest clusters analysis showing total area, number per forest type, mean river length, and mean cluster connectivity index. Forest types are sorted according to connectivity index. Connectivity index ranges from 1 (low connectivity) to 4 (high connectivity).

Forest Type	Cluster area (ha)	Numbers of clusters	Sum river length (km)	Mean connectivity index
Lowveld Riverine	53007	34	460	3.3
Limpopo Mistbelt	59132	36	57	2.9
Eastern Cape Dune	29385	33	81	2.5
Swamp	18713	17	46	2.4
Licuati Sand	73545	54	124	2.0
Transkei Coastal	115879	247	375	2.0
Amatole Mistbelt	189929	297	412	1.9
Albany	62914	92	137	1.8
Pondoland Scarp	136863	99	475	1.8
Transkei Mistbelt	118972	230	209	1.8
Southern Cape Afrotemperate	204331	230	572	1.5
Mpumalanga Mistbelt	100843	128	184	1.4
Eastern Mistbelt	153658	458	226	1.3
Western Cape Milkwood	12056	66	7	1.1
Eastern Scarp	121254	469	2	1.0
KwaZulu-Natal Coastal	61300	122	113	1.0
KwaZulu-Natal Dune	30540	65	76	1.0
Mangrove	500	3	2	1.0
Western Cape Afrotemperate	24369	103	109	1.0
Drakensberg Montane	10700	75	27	0.9
Low Escarpment Mistbelt	26196	138	26	0.9
Grand total /average	**1604086**	**2992**		**1.5**

The mean cluster size of forest types shows a statistically significant linear correlation with mean connectivity index (correlation coefficient of 0.71, $p < 0.05$). Because connectivity is largely a function of matrix transformation, and small clusters within a particular forest type are more likely to be embedded in matrices that have undergone higher levels of habitat transformation than larger clusters, smaller clusters tend to have lower connectivity indices than larger clusters (see, below).

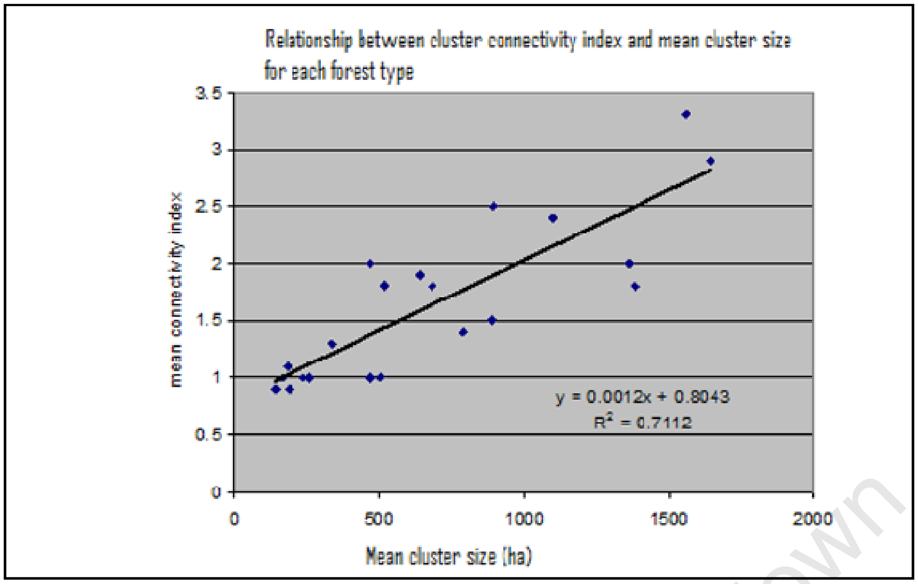


Figure 25. Relationship between the means of cluster connectivity index and cluster size, for each forest type considered.

7.3.5 Selecting priority forest clusters

7.3.5.1 Results of MARXAN analysis

Priority forest clusters identified by systematic assessment are those with 100 % irreplaceability (as determined using MARXAN) and with a high vulnerability rating (see section 5.3.6). They are given in below Table 36.

Table 36 . Priority forest clusters identified using MARXAN, sorted by forest type and then ranked according to vulnerability and then connectivity. Clusters were named according to the largest named patch within the cluster, (a number of clusters do not have names but can be located by their cluster ID).

Cluster ID	Cluster name	Forest type	forest area (ha)	No. of patches	% trans formation in matrix	vulnerability rating	Conn-ectivity index
2461	Pirie/ Amatola	Amatole Mistbelt	26 098	405	12	medium	4
2562	Unknown	Amatole Mistbelt	31	2	27	medium	3
1069	Karkloof	Eastern Mistbelt	4 779	42	29	high	4
1354	Cunningham's castle (N W of Richmond)	Eastern Mistbelt	1 151	17	31	high	4
1569	Bencairnie Forest Reserve	Eastern Mistbelt	973	64	40	high	4
1868	Tabankulu /Kugomo	Eastern Mistbelt	1 074	43	0	low	3
444	Jozini	Eastern Scarp	3 552	7	3	medium	4

536	Ntendeka	Eastern Scarp	3 446	33	6	medium	4
735	Dukuduku	KZN Coastal	12 445	162	18	medium	4
409	Unknown	KZN Coastal	155	1	29	low	3
420	Unknown	KZN Coastal	34	3	18	low	3
312	Kosi bay	KZN Dune/Swamp	114	17	46	high	4
617	Nqutshini (falls outside St Lucia NP)	Licuati Sand	1910	2	16	low	3
338	Unknown	Licuati Sand	6	1	0	low	1
306	Ndumu/Pongola river	Licuati Sand/Lowveld riverine	1 819	19	28	medium	4
563	Tierkloof (East of Ngome)	Northern KZN Mistbelt	141	5	46	high	4
593	South of Ncadu Nature Reserve	Northern KZN Mistbelt	168	12	0	low	3
616	Unknown	Northern KZN Mistbelt	87	2	5	low	3
670	Glencoe	Northern KZN Mistbelt	209	2	6	low	3
2882	Gouna/Blue Lily's Bush	Southern Cape Afrotemperate	49 625	801	22	medium	4
2930	Bergplaas	Southern Cape Afrotemperate	41	11	12	medium	3
3117	Unknown	Southern Cape Afrotemperate	80	4	43	medium	3
3128	Witelsbos	Southern Cape Afrotemperate	88	9	43	medium	3
3134	Unknown	Southern Cape Afrotemperate	21	4	35	medium	1
3165	Unknown	Southern Cape Afrotemperate	60	3	0	low	3
312	Unknown (Kosi bay)	Swamp	314	8	9	medium	3
2144	Gogogo	Transkei Coastal Platform	28	13	0	medium	4
2284	Rebetshane	Transkei Coastal Platform	5 036	215	5	medium	4
2106	Unknown	Transkei Coastal Platform	25	1	0	medium	3
2157	Mpoza/ Maseko	Transkei Coastal Platform	23	1	0	medium	1
2293	Unknown	Transkei Coastal Platform	22	2	11	medium	1
1966	Tsolo	Transkei Mistbelt	3 428	226	32	high	4
2018	Ludaka	Transkei Mistbelt	2 443	194	36	high	4
1768	Buffalo Nek Forest Station	Transkei Mistbelt	1 697	189	17	medium	3
2071	Ngxangxasana	Transkei Mistbelt	1 929	81	13	medium	3
3241	Stillbaai Melhout woud	Western Cape Milkwood	341	1	6	low	3

7.3.5.2 Results of the rule based algorithm

The list of priority forest clusters selected by the rule based selection algorithm are given in appendix D.

7.4 Discussion

Ultimately the job of conserving biodiversity requires broadening the scope of the temporal and spatial considerations beyond the traditional focus of resource management, and adopting a broad scale, long term framework (Soule and Terborgh, 1999). For this, an understanding of the scales at which ecosystem processes operate to mediating the functioning of ecosystems becomes critically important. We can no longer assume that geographically distinct ecosystems such as forest patches, are functionally distinct or self sustaining, and when planning for forest reserves one must consider species movements and process that operate across the broader landscape.

A common problem in conservation planning, is finding suitable spatial surrogates to represent ecosystem processes, (Smith *et al*., 1996; Desmet, 2004; Rouget *et al.,* 2006a) and integrating these with measures to protect biodiversity pattern. Because forest ecological processes occur across a range of different spatiotemporal scales (Spies and Turner, 1999; Lindenmayer and Franklin, 2002; Lindenmayer *et al.,* 2006; Lawes *et al.,* 2007a; Lawes *et al*., 2007b; Kotze and Lawes, 2007), planning for their conservation will also need to occur at multiple spatiotemporal scales (Ritters *et al*., 1997). While knowledge of the spatial requirements of ecosystem processes are incomplete, it is feasible to use spatial surrogates in the form of planning units that are broadly concomitant with the spatial scale at which processes operate. In this regard, the use of forest clusters as planning units provide a practical means to conserve both forest biodiversity pattern, and those processes operating at the ecosystem to landscape scale. This approach is similar to that used by Tran *et al*. (2002); Berliner and Desmet (2007), and Roux, (2007) who used water catchments as large scale planning units to capture hydrological and other process operating at the scale of water catchments.

The incorporation of ecosystem processes are best achieved at a landscape scale (Balmford *et al*., 1998; Soule and Terborgh, 1999; Noss, 2003), however the distinction between area requirements of processes operating at different spatiotemporal scales still needs to be made explicit. For example plant pollination by insects takes place at a spatial scale of a few meters, whereas the area requirements for a mammalian seed disperser may be in the order of many square kilometres.

This study specifically considers process operating across the three broad scales where the focus shifts from species, ecosystems to landscapes. Processes operating at the largest evolutionary/bio-geographic scale, although briefly discussed (see section 7.1) are not dealt with in any detail. To do this would require an understanding of the spatial requirements associated with speciation, radiation, and adaptation to climate change, which goes beyond the scope of this thesis.

It is contended that the mathematical reserve selection algorithms typically used in conservation planning (such as C-plan and MARXAN) have a limited ability to incorporate ecological heuristics, are overly mechanistic, have a high requirement for quantitative spatial data, and do not consider context specific cases involving multiple criteria. This study has identified a need to use flexible reserve selection algorithms that can formally incorporate expert knowledge, and hence draw on a wider range of criteria than what is typically used in mathematical reserve selection algorithms. For example criteria that are often difficult to include within mathematical optimization and complimentarily analysis include spatial reserve design and socio-economic considerations. A multiple criteria analysis approach to conservation planning has been used by Store and Kangas (2001); Noss *et al.* (2002); Tran *et al.* (2002); Moffett *et al.* (2006); Moffett and Sarkar (2006) and others.

It is contended that reserve selection should be based on sets of heuristics, or key ecological principles, rather than relying exclusively on mathematical algorithms with limited ability to incorporate multiple ecological considerations within optimizations routines. For example, landscape ecology provides a number of heuristics (see Murphy and Noon, 1992; Noss *et al*., 1997; Peck, 1998; Theobald *et al.,* 2000, and others), that are particularly applicable to conserving South African's fragmented forest biome. For example, Theobald *et al.* (2000), suggests that large patches that support large populations will be more viable; habitat patches that are continuous (less fragmented) support long-term viability; and patches that are sufficiently close together allow dispersal and thus support long-term viability. In addition, because conservation planners must make decisions with less than complete information, assumptions based on ecological concepts can often substitute for empirical data. Schulte *et al.* (2006) identified 11 forest ecological concepts that they believe can be applied to forest planning and management.

In Chapter 6 (section 6.5) the problem of combining expert judgment with computer based selection was discussed, although not necessary resolved. Heuristics are typically used by human experts, although often implicitly, when identifying conservation priorities. In this chapter the limited ability of iterative computer based selection algorithms (C-plan and MARXAN) to incorporate context specific ecological heuristics within reserve selection was discussed. Expert system type rules are suggested as a means of formalising and integrating these heuristics. Expert systems represent human expertise in the form of rules (Waterman, 1986), and therefore provide an ideal means to integrate human expertise with the computing power of mathematical selection algorithms. Expert systems have been widely used in ecological decision modelling, for example Starfield and Bleloch (1983); Noble (1987); Starfield (1990); Berliner (1990); Starfield and Bleloch (1991) Kalogirou (2002); Filis *et al*. (2003); and Crist *et al.* (2004).

Potential shortfalls of rule based algorithms, is that they cannot provide mathematically optimal solution. However because mathematical optimization will always be limited by the number of variables that can be included, these solutions are in any case, theoretical approximations with limitations. In addition, and as pointed out by Noss, (2003), decisions themselves are matters of public choice, not science so it should not be surprising if the ideal protected area system is hardly ever implemented. In comparison, rule based models are flexible in the number of variables that can be used, and can incorporate context specific considerations. For example, expert review of the MARXAN selected priority clusters (see section 7.3.5.1) revealed that a number of the largest forest patches representative of each forest type had been excluded from the selection. This was explained by MARXAN's use of the 'costs layer'. The MARXAN algorithm tries to minimize costs of the reserve network by selecting the low cost reserve options. Many of the largest forests have high population pressures around them, resulting in high conservation opportunity costs. Rather than selecting these, the MARXAN algorithm selects a number of smaller patches so that targets are achieved with lower overall costs. The problem with this is that the ecological value of forest size is discounted against its current opportunity costs. In keeping with island biogeography theory (MacArthur, and Wilson, 1967) forest size, particularly in highly fragmented systems, is one of the most important indicators of a forest conservation value.

The rule based selection algorithm was developed as a response to MARXAN'S limited ability to represent ecological heuristics. The disadvantage of not using mathematical optimal solutions, are that alternative solutions are likely to be 'land hungry'. However, because area requirements for conservation may be exceeded by those required to maintaining ecosystem services, from the point of view of conservation biology, this ‘overshoot’ is not necessary undesirable. It can also be argued that ‘real world’ constraints make mathematical optimization near impossible, and that improved, effectiveness of the reserve network is more desirable than efficiency (for further discussion on this see Kershaw *et al.,* 1994; Pressey, *et al.,* 1997; Rodrigues *et al.,* 2002). An additional advantage of using rule based algorithms, rather than iterative mathematical optimization algorithms, is in their transparency to non mathematical experts, and because of this, selection rules can be scrutinised according to their ecological, rather than purely mathematical logic. This can also serve to assist in narrowing the gap between theoreticians and practitioners in conservation biology (Prendergast *et al.,* 1999; Cowling *et al.,* 2003a; and Briers, 2002).

CHAPTER 8: SYNTHESIS AND RECOMENDATIONS

8.1 Synthesis of main findings

8.1.1 A summary of conservation priorities for the forest biome

This thesis set out to profile the current state of the forest biome in South Africa and to identify the conservation priorities required for forest biodiversity persistence. An important conclusion of this thesis is that the traditional (largely silvicultural) focus of forest management and reserve planning in South Africa, has tended to view forests as geographically and functionally distinct ecosystems, without adequate consideration of landscape scale processes and requirement for connectivity. For long term conservation of forest biodiversity, planning requires to occur across multiple scales, and with a broader and longer term view than what has been the traditionally focus.

A summary of the conservation priorities identified in this thesis, across three major scales, is given in Table 37.

Table 37. A summary of forest conservation priorities identified in this thesis. Selection criteria used at each of the spatiotemporal scales are provided, as well as reference to look up tables containing further details.

Spatiotemporal scale	Planning units	Selection criteria	Priorities	Look up tables
Species-population	Species	IUCN or SANBI red data species occurring in South African forests.	Threatened forest species.	• Appendix E (Threatened plants); • Appendix F (Threatened fauna)
Ecosystem-habitat	Forest patches	• 100 % irreplaceable patches, under high threat, plus additional expert selected patches.	Integrated list of priority forest patches.	Appendix C
	Forest types	• Vulnerability ranking using: number of threatened plants, matrix transformation, and population pressure. • SANBI criteria for endangered forest types (see section 6.4.3).	• Most vulnerable forest types. • Endangered forest types.	• Table in section 5.4 • Table in section 6.4.3
Landscape	Forest clusters	• Expert rule based selection: using cluster	Priority forest clusters	• Appendix D • Table 34

		size; matrix transformation, threat, contribution to targets, fragmentation (number of patches in cluster) • MARXAN analysis		
Bio-geographic	Forest corridors	Biogeography dispersal corridors	Priority forest corridors	Not specifically dealt with but see section 8.2.7

8.1.2 Key findings regarding the state of the forest biome

- Qualitative, expert estimates indicate that a significant area of forest has been lost, although there is a relatively high degree of uncertainty, estimates are upwards of 35 % loss of the total forest biome since European colonization. Coastal forest types have experienced the highest loss (see section 2.2).
- A large proportion of some forest types have been significantly degraded. Those most affected, being forest types exposed to a history of colonial logging, and those in communal areas exposed to high subsistence harvesting. Up to 60 % of these forest types, as estimated by experts, may have been significantly degraded (see section 2.3).
- South African forests are highly fragmented; this can be measured by the overall high edge to area ratios, high degree of isolation (high distance between patches), and relatively small mean patch sizes. Fragmentation differs widely between forest types. Using a mean of these three metrics, the most fragmented forest types are Drakensberg Montane, Transkei Coastal, Western Cape Milkwood, and Lowveld Riverine forests (see section 2.4 and 7.2.2.).
- Fragmentation of the forest biome has been exacerbated by land use changes in and around the forest matrix. At least six forest types have more than 40% of matrix (surrounding land) transformed. Timber plantations are responsible for much of this (see section 5.3.3).
- Compared to other biomes in South Africa, the forest biome has the highest total number of plant species per unit area of biome. Data compiled for this thesis found a total of 1885 plant species representing 176 families and 780 genera. The total forested area estimated in this thesis is 492 700 hectares (see section 3.2.3.).
- Results show that forests contain the highest proportion and density of red data fauna of any biome in South Africa. Red data forest vertebrates include: 21 mammals, 16 Birds, 9 amphibians and 10 reptiles. Of these, 1 mammal

and 2 amphibians are listed as critically endangered. There has only been one recorded extinction, this of a forest vertebrate. Red data forest invertebrates include 1 crab species, 13 snail species, 4 butterfly or moth species, 13 millipedes and 2 velvet worm species. Of these 5 snail species, 1 butterfly, 1 millipede, and 1 velvet worm are critically endangered. Data on forest invertebrates are incomplete. (See section 3.4.3).

- Many forest plants are listed as threatened particularly those endemic to South Africa. This thesis found that overall 56 species of vascular plants occurring in forests are listed as IUCN red data species, of these, two species are extinct in the wild, four are critically endangered, 8 endangered, 20 vulnerable and 22 are near threatened (see section 3.3.3).
- Forest types with the highest number of red data plant species are also those with the most plant species endemic to South Africa. These include: Pondoland Scarp Forests, with 19 red data species (35 endemics); Eastern Scarp Forests, with 15 red data species (13 endemic), and Transkei Coastal Forests, with 7 red data species (8 endemic), (see section 3.3.3).
- Forest types differ widely in plant species richness and number of unique species (species that only occur in 1 forest type). Many species are common to more than one forest type. Forest types with high numbers of species also tend to also have a high number of species unique to them. The richest forest types, and with the most unique species are: Pondoland Scarp Forest with 468 total plant species (166 unique species), Eastern Scarp Forest with 631 total plant species, (161 unique species), and Low Escarpment Mistbelt forest with 331 total species, (and 108 unique species), (see section 3.2.3.4).
- A few forest types are recognised for having relatively low total numbers of species but with a high proportion of which are unique. These are: Mangrove forest with 15 species (of which 67 % are unique); Lowveld Riverine Forest with 88 species (of which 44% are unique) and Licuati Sand Forest with 195 species (of which 37 % are unique), (see section 3.2.3.4).
- This thesis corroborate the findings of Silander (2001), that South African forests have the second highest number of tree species of any temperate forest in the world, and the highest number of tree species per unit area of forest (see section, 3.2.3).
- The current formal forest protected area network is not protecting representative samples of all forest ecosystems. At least 6 of the 21 forest

types considered have less than 10 % formal protection, and two have no formal protection at all (see section 4.3).

- Forest types exposed to the highest population pressure (as measured by number of people per ha of forest, in 5 km forest patch buffers) are Kwazulu-Natal Coastal and Kwazulu-Natal Dune forests, with 60 and 38 people per ha of forest, respectively.
- Using GIS rule based modelling; a subsistence resource use pressure index (SRUPI) was calculated for each forest patch and aggregated for each forest type. Forest types located in communal areas of the Eastern Cape and Kwazulu-Natal, have the highest SRUP indices. These areas typically have high rural poverty, and no electrification.

8.2 Recommendations

8.2.1 Expansion of the forest protected area network

- It is recommended that the forest protected area network be expanded to include a minimum of 10 % of each forest type. Overall the national biome target set for formal forest protected areas is 20% (according to Holness *et al.,* 2008). This is insufficient to ensure forest biodiversity persistence, and should be increased to at least 50%.
- Increases in the formal forest protected area network should come from those priority forest patches and clusters identified in this study. As a rule of thumb, the largest intact forest patches represent the most valuable refugia for forest biodiversity (MacArthur, and Wilson, 1967; Warburton, 1997; Lindenmayer and Franklin, 2002; Wethered, and Lawes, 2003; Lawes *et. al.*, 2007a).
- Forest protected areas need to be designed from an integrated landscape perspective. This includes maintaining connectivity between forest patches, observing minimum buffer areas around forests as well as the creation of ecological corridors between forest clusters and across altitudinal gradients.

8.2.2 Improved land use planning around forests

- It is recommended, that at least for the priority forest clusters, the matrix area is managed as an integral component of the forest ecosystem. They should thus be afforded special protection to prevent further habitat loss. The demand for suitable arable land, in particular for timber plantations have resulted in significant loss of the forest matrix areas. This poses a serious threat to the long term viability of forest biodiversity, particularly in the face of global climate change, where species may require shifting distribution ranges to adapt to environmental changes. Approximately a third of all the forest matrix area surrounding priority forest clusters, consist of vegetation that is either endangered or critically endangered (see appendix D). Therefore, efforts to conserve these clusters will also benefit the conservation of remaining fragments of rare vegetation types (particularly grasslands) associated with these forests.

- It is recommended that catchment transformation threshold be used to regulate land conversion in priority forested catchments to enable this, forest conservation needs to be integrated with catchment management strategies. Transformation results in fragmented landscapes and loss of ecosystem connectivity. It is well documented that when landscapes are transformed beyond certain critical thresholds, ecological processes tend to collapse, for example natural disturbance regimes and hydrologic cycling show dramatic changes (Andrén, 1994; Fahrig and Merriam, 1994; Fahrig 2001; Fahrig, 2002). Catchment transformation thresholds will ensure that a minimum area of each catchment remains under natural vegetation. Currently, government regulations for land conversion activities with hydrological impacts, also termed 'stream flow reduction activities' (such as plantation forestry and certain crops) require a permitting system that determines the availability of water, based on the estimated reduction in mean annual runoff (MAR) associated with the activity, and in relation to the 'ecological reserve' (the amount of water required for safeguarding and sustaining healthy stream and river ecosystems, see Bosch and Hewlett, 1982; van der Zel, 1995; Dye and Versfeld, 2007). For example the Aforestation Permit System (APS) of the South African Department of Water Affairs uses allowable reductions in stream flow based on downstream demand for each catchment. Unfortunately these restrictions do not consider the cumulative effects of transformation across the entire catchment(s) nor the associated disruptions in landscape level process effecting biodiversity. A review of a wide range of studies by the Environmental Law Institute of America (ELI, 2003), reveal that habitat transformation thresholds vary between 60 to 80 percent of the landscape, i.e. the area required to remain as untransformed to ensure persistence of terrestrial ecosystem functioning and to avoid species loss. Berliner and Desmet (2007) provided recommendations for maximum permissible catchment transformation thresholds for sensitive (priority) sub-quaternary catchments in the Eastern Cape Province. It is intended that catchment transformation thresholds be used in conjunction with other terrestrial land-use planning guidelines.

8.2.3 Promote the development of alternative livelihoods

- Because forests are important to the livelihoods of many rural communities, there is an urgent need to integrate forest conservation planning with rural development and poverty alleviation (see for example INR, 2003; INR 2005). Community based natural resource management strategies need to be employed that promote sustainable forest resource use in conjunction with conservation management (McKean, 2004). In this regard, the Participatory Forest Management (PFM) programme of the Department of Water Affairs and Forestry (DWAF 2005b) urgently needs to be fast tracked, particularly in the forested areas of the communal areas in South Africa.
- Improvements in 'on the ground' conservation management of the existing forest reserves are urgently required. This will involve enhanced capacity for law enforcement, community education; outreach programmes (see for example WWF, 2004), as well as the development of community based forest management structures. Central to this will be funding and training of the forest guards.
- Alternative low impact forms of land use need to be found for forest matrix areas This is necessary to offset socio-economic opportunity costs of conserving whole forest clusters, and to promote sustainable livelihoods,. Potential biodiversity compatible forms of land use that could be employed in forest matrix areas include: ecotourism, bee keeping, game farming, certain forms of permaculture, and multi-crop agro-forestry.

8.2.4 Expanding research of the forest biome

Research priorities for forest conservation are broadly grouped under three headings, these include a general expansion of the forestry research agenda; specific research relating to forest conservation planning and improved understanding of climate change and forests, these are briefly discussed below.

8.2.4.1 Developing a systematic forest research agenda

South African forest research has been predominantly from a silvicultural perspective as such focus has been largely on the woody component, and limited attention has been paid to understanding other taxonomic groups, as well as ecosystem process (in particular those that act across broader spatiotemporal scales than the forest patch). There is therefore a need to expand forest research efforts to be inclusive of all ecosystem components and all spatiotemporal scales. In particular, further research will be required to address the following.

- Finer scale classification of forest sub types, along the lines of Lötter and Beck, (2004), and Mucina *et al*. (2007).
- Inventory and geo-reference forest occurring species, in particular those species of conservation concern.
- Determine sustainable levels of harvesting for target species.
- Develop rapid and cost effective methods to measure and monitor forest degradation and fragmentation.
- Identify suitable indicator species for monitoring changes in forest structure, function and composition.

8.2.4.2 Research supporting forest conservation planning

- Identify optimal size for forest clusters buffers. This will determine the size of the forest matrix and the size of the clusters. This thesis used an arbitrary width of 500 meters to delineate forest clusters, but this could be refined to consider the scales at which ecosystem processes operate. For example, the foraging distance of key forest species; the minimum area required to allow natural fire regimes to maintain forest margins and ecotones.
- Identify forest ecological corridors. This may include techniques such as least costs path (Soule and Gilpin, 1991), tracking of species dispersal corridors used during palaeoclimatic refugia (see for example, Lawes, *et al.,* 2007), and migration pathways (Lindenmayer and Nix, 1993; Bennett, 1998), (see section 8.2.8).
- Improved understanding of spatial and land management requirements to maintain ecosystem processes important for persistence of forest biodiversity (see section 8.2.7).
- Integrating forest conservation planning with other terrestrial and aquatic biodiversity priority setting exercise.

8.2.4.3 Climate change and forests

Dramatic increases over the last four decades in greenhouse gases, such as carbon dioxide, methane and nitrous oxide and the associated increase in global temperature are of increasing concern to conservation biologists (Hannah, *et al.,* 2002; Arau´jo *et al.,* 2004; Lovejoy and Hannah, 2004). Climate is expected to become more extreme with more droughts, flooding and much hotter (IPCC, 2007). Changing levels of CO_2, temperature and rainfall are expected to have significant impacts on the distribution of indigenous forests in South Africa (Eeley *et al.,* 1999; Midgley, 2001; Turpie *et al.,* 2002; Bond *et al*., 2000).

Eeley *et al.,* (1999) showed that the distribution of forest types in KwaZulu- Natal are considerably sensitive to predicted climate change scenarios, with significant shifts in altitudinal and latitudinal distributions. Using a projected warmer and wetter climatic scenario for this province, they predicted a considerable overall increases in potential forest area, although areas suitable to Afromontane forests declined, becoming more fragmented, and migrating to higher altitudes. Scarp forest areas increased but show an altitudinal movement inland and Coastal forests expand substantially along the coast. Sand, swamp and riverine forests, constrained as they are by their substrate, have limited potential for migration, and are considered most at risk to the effects of climate change.

In contrast to Eeley *et al.* (1999), Midgley (2001) using models that predict a general increased in annual temperatures and a general drying over southern Africa, show dramatic decline in the extent of the forest biome over the whole country. Similarly, Fairbanks and Scholes (1999) show significant reduction in the areas suitable for plantation forestry (areas often closely associated with indigenous forests).

What the interaction between elevated CO_2, temperatures and fire, will be on forests is still not clear. It is speculated that increased temperatures may increase the incidence of fire, which in itself will have important implications for the distribution of forests, (Geldenhuys, 1994; Taylor and Hamilton, 1994; Midgley *et al.,* 2001). However, elevated CO_2 has differential effects on the growth rates of trees, shrubs and grasses (Eamus and Jarvis, 1989; Ehleringer, *et al.* 1998). Bond *et al.* (2000) and Bond *et al.* (2003) propose that elevated atmospheric CO_2 could influence tree cover in savannahs by speeding up sapling growth rates relative to flammable herbaceous growth forms such as grasses, thus allowing trees to grow quicker and making them less susceptible to control by fires. Changes in growth rates are likely to affect post disturbance succession and therefore the boundaries between fire tolerant and fire-intolerant vegetation. For example forest margins burnt by shrubland or grassland fires would retreat if forest growth were slow relative to fire return intervals; but under elevated CO2, tree growth rates are stimulated (Ceulemans and Mousseau, 1994), enabling tree saplings to reach fire resistant heights sooner.

Climate models predict a continual warming over the next 100 years or so for Africa, but changes in effective precipitation have proven more difficult to predict, largely because of the complexity of the temperature/water availability balance (Taylor and Hamilton, 1994; Hulme, 1996). Clearly this uncertainty makes prediction of how the forest biome will respond under global warming open to speculation. What is certain however is that the inherently fragmented nature of South African forests, combined with their restricted range, and embedded within heavily transformed and man-managed landscapes, will limit many forest species in shifting their distributions as an adaptation to environmental changes. This makes them extremely vulnerable to climatic shifts. In addition, areas where forests species can be expected to shift their future range may also be those areas most sought after by agriculature and the plantation forestry industry.

Forest conservation planning for climate change will require a protected area network that is large enough to span substantial climatic gradients and that is linked by corridors of natural or semi natural vegetation (Taylor and Hamilton, 1994; Eeley *et al.*, 1999). In addition, areas that have acted as climatic refugia in the past, such as the scarp forest (Eeley *et al.*, 1999; Mucina and Geldenhyse, 2006; Lawes *et al.* 2007a) should be prioritised for protection. An important challenge for conservation is therefore to manage fragmented landscapes so as to assist species in tracking changing environmental conditions to which they are adapted (Pearson and Dawson 2005)

8.2.5 Monitoring and evaluating changes in forest biodiversity

Monitoring and evaluation is an essential feedback component necessary for successful strategic conservation planning (The Montreal Process, 1999; Lindenmayer, 1999; Stem *et al.,* 2005). Given the complexity and difficulty in measuring biodiversity, indicators (or surrogates of biodiversity) are needed to judge the success or failure of management regimes designed to sustain biological diversity (Noss, 1999; Lindenmayer *et al.,* 2001). Of increasing importance is the identification of reliable indicators of global climate change (Hannah *et al.,* 2002). Based on an approach described by Mendoza & Prabhu (1998), and CIFOR (1999) the department of Water Affairs and Forestry have initiated national level principles, criteria and indicator standards for natural forests (Lawes *et al*., 1999; DWAF, 2007). Although this initiative provides a useful framework for aligning and monitoring policy, in the current form, they have limited value as a tool for forest conservation planning.

Suitable indicators need to be identified that can provide useful insights into difficult to measure variables such as forest ecosystem functioning and degradation. Of particular importance, is the need to measure rates and causes of changes in forests structure, function and composition, to quantify the extent and rates of forest degradation, two kinds of forest indicators can be used, these include: indicator species, and structure-based indicators. The later should include stand-level and landscape-level, structural complexity, connectivity, and heterogeneity indices (Lindenmayer *et al.,* 2001). Because the relationships between potential indicator species and forest biodiversity are not well established (Lindenmayer 1999; Lindenmayer *et al.*2001), carefully designed studies are required to test relationships between the presence and abundance of potential indicator species and other taxa, as well as how these changes relate to critical ecosystem processes. It is recommended that sets of indicator species, representative of a range of taxonomic groups, be identified for monitoring changes in forest biodiversity. Suggested criteria for selecting indicator species include: species that are relatively easy to measure or observe, species that are forest dependent, widespread, sensitive to disturbances at ecosystem and landscape levels, species that are harvested or hunted, species that are rare and endemic to South Africa and/or with very limited distributions. Rare species may not necessary be good indicator species, however these species should in any case be included within a species monitoring programme because of their conservation status.

A list of forest dependent species that could potentially serve as useful indicator species for monitoring changes in forests biodiversity are presented in Table 38, below.

Table 38 Forest dependent species that could potentially serve as indicator species for monitoring changes in forests biodiversity.

Taxonomic group	Possible indicator species	Justification
Mammals	Samango monkey (*Cercopithecus mitis labiatus*). Blue duiker (*Cephalophus monticola)*, Red duiker (*Cephalophus natalensis)*,	Metapopulations sensitive to effects of fragmentation, and matrix transformation. (Lawes, 1990, Lawes pers. coms.)
Birds	Spotted Ground-thrush (*Zoothera guttata*); Cape Parrot (*Poicephalus robustus*); Crested Guineafow (*Guttera edouardi*)	Sensitive to changes in forest structure and composition (Symes *et al*. 2004; Barnes,2000); Lawes *et al.,* (2006)
Amphibians	Kloof frog (*Natalobatrachus bonebergi*); Forest tree frog (*Leptopelis natalensis*) Hogsback frog (*Anhydrophryne rattrayi*)	Sensitive to hydrological changes and disruption of forest ecotones (Minter *et al.* (2004). May be useful indicators of climate change.
Reptiles	Dwarf Chameleons (*Bradypodium spp*.)	Poor dispersal abilities, sensitive to changes in forest structure and composition K. Tolley (pers comms)
Invertebrates	Diplopoda (Millipedes): *Doratogonus* spp. Gastropoda (Snails): T*rachycystis, Gulella*, *Natalina* spp. Litter decomposers (e.g. *Talitriator Africana.* Other amphipods rove beetles and certain species of ants.	Often with restricted ranges and poor dispersal capabilities, invertebrates are particular vulnerable to local extinctions. (Samways, 1993).Amphipods were consistently more abundant at highly disturbed forest edges compared to less disturbed interiors (Kotze & Lawes, 2008). Also see Lawes *et al.,* 2005b)

All monitoring information should be centrally located, collated and made readily available for public use. It is important that all management responses to monitoring be conducted within an adaptive management framework. Adaptive management is a structured, iterative process of optimal decision making in the face of uncertainty, with an aim to reducing uncertainty over time by feedback monitoring (Holling, 1978; Walters, 1986; Holling *et al.,* 1995).

8.2.6 Development of endangered species conservation plans

In conjunction with the monitoring of indicators species is the development of comprehensive conservation plans for threatened forest species. Although trends in conservation biology are for planning to be done at the level of habitats or multiple species, there is a need to develop strategic, focused and integrated conservation plans for species that are threatened and on the decline (Simberloff, 1997; Carroll *et al.,* 2001) Such plans would need to include details of the current status, distribution, causes of decline and strategies for recovery, (Noss *et al*., 1997; Lambeck, 1997; Lindenmayer, 1999), as well as spatially explicit population viability modelling (Noss, 2000a, Noss 2000b; Carroll *et al*., 2003) and the analysis of landscape permeability (Gobeil and Villard, 2002).

8.2.7 Planning at multiple spatiotemporal scales

A common problem in conservation planning is finding suitable spatial surrogates to represent ecosystem processes, (Desmet, 2004; Rouget *et al.,* 2003a). It has been suggested (e.g. Balmford *et al*., 1998; Terborgh and Soul´e, 1999; Noss, 2003) that conservation is most effective if conducted at the landscape scale. However, because biodiversity is a multi-scaled concept, and the scale of planning limits the scale at which ecosystem processes can be represented within the planning framework (Forman, 1995), systematic conservation planning ideally, needs to occur across the full range of scales relevant to ensure biodiversity persistence. Applying this to forests, conservation planning needs to consider ecosystem processes as nested scales including process relevant at the scales of individual trees, forest patches, forested landscapes, and hydrological catchments.

The need for multiple spatiotemporal planning scales is supported by the following observations.

- Ecological processes occur across a range of different spatiotemporal scales (Spies and Turner, 1999; Lindenmayer and Franklin, 2002; Lindenmayer *et al.,* 2006; Lawes *et al.,* 2007a; Lawes *et al*., 2007b; Kotze and Lawes, 2007).
- Habitat requirements for different species may be defined and described at different scales (Lindenmayer, *et al.,* 2006). For example an invertebrate's requirements may be at the level of individual fallen trees, while forest mammals may require habitat containing multiple patches within a landscape.

- There are multiple ecological and management scales for the same species, (Forman, 1995). For example Samango monkeys (*Cercopithecus mitis labiatus*) have habitat requirements at the level of the patch (mature fruiting trees), but also require being able to disperse across forested landscapes (Lawes, 1990).
- cumulative impacts can be more accurately assessed within a nested hierarchy of planning units, for example Berliner and Desmet (2007) used hydrological catchments to set land use planning transformation thresholds to account for cumulative impacts of land conversion on downstream aquatic systems in the Eastern Cape (see section 8.2.2).

A few conservation planning strategies relevant to each of the broad spatiotemporal scales at which forest ecosystem processes occur, are suggested in Table 39, below.

Table 39. Suggested conservation planning strategies, across three broad spatiotemporal scales.

Spatiotemporal scale	Conservation planning strategy	Planning units
Ecosystem scale	For each forest type protect representative samples of the largest forests patches. Maintain natural edge -ecotone boundaries.	Forest patches.
Landscape scale	Conservation of whole forest clusters including cluster matrices. Maintain connectivity between patches. Restrict land use change in priority sub catchments, use of transformation thresholds.	Forest clusters and Forested sub catchments.
Evolutionary/ bio-geographic scale	Develop forest corridors linking forest clusters across altitudinal gradients and evolutionary dispersal and refugia routes.	Forest corridors, (chains of clusters), and priority hydrological catchments.

8.2.8 Identifying forest ecological corridors

Predictions regarding the impacts of global climate change on biodiversity, (Noss, 2001; Midgley *et al.,* 2003; Pyke, 2004; Lovejoy and Hannah, 2004; IPCC, 2007), point to the importance of maintaining ecological connectivity across larger landscape scales than what is typically considered in most conservation planning exercises. Essentially this implies that conservation planning for climate change needs to consider the evolutionary and bio-geographic related process such as dispersal and re-colonization pathways from palaeoclimate refugia. This requires maintaining landscape-level connectivity across large geographical regions (Lindenmayer and Nix, 1993). The efficient identification of these 'macro ecological corridors' that maintain connectivity between forest complexes will become increasingly important if conservation efforts are to incorporate climate change within its planning framework.

Although the incorporation of ecological corridors into the forest protected area network was not a specific research objective of this thesis, given its importance, I have provided an example of the kind of planning that will be required for developing climate change resilient forest conservation networks. I have identified one of the most important forest biodiversity regions in the country, the Umtamvuna – Drakensberg forest corridor as an example of this. This corridor provides both horizontal linkages (linking forest patches at similar altitudes) and vertical linkages (linking forest patches across altitudinal gradients). It also links a number of different forest types, and is consistent with the evolutionary faunal dispersal routes suggested by Lawes, 2007a; and Lawes 2007b). See Figure 26 below.

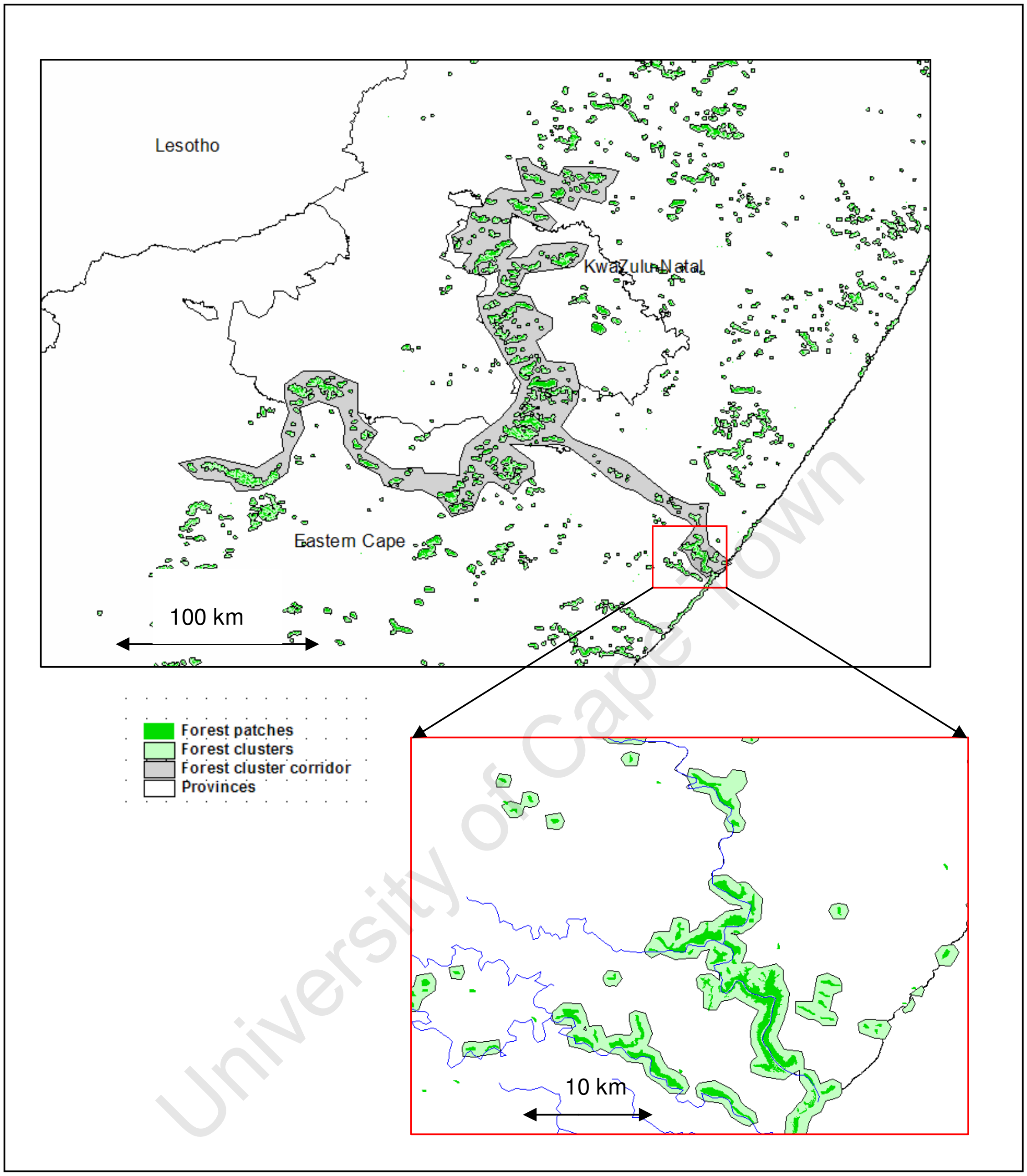


Figure 26 The Umtamvuna to Drakensberg forest corridor providing landscape-level connectivity between forests patches and clusters across a number of different forest types and across altitudinal gradients.

Corridors are often considered as notional or conceptual in that they may not necessary be 'hard edged' or even strictly defined on a map. However what is essential is that areas designated as ecological corridors retain ecosystem connectivity between key habitat patches (Lindenmayer and Nix, 1993). This will require regulating the type of land use within them, so as to maintain ecological permeability and reduced matrix hostility (Fahrig, 1985; Swart and Lawes, 1996; Wiens, 1997; Bennett, 1998; Bennett and Wit, 2001; Briers, 2002). This entails retention of at least the essential components of the original vegetation and limiting the extent and type of land conversion within the remainder.

8.3 Concluding remarks

- South African forests are highly vulnerable ecosystems, and are likely to become more susceptible to irreversible biodiversity loss with expanding agriculture, timber plantations, urban development, and rural population growth. These will be exacerbated by increasing levels of rural poverty as well as global climate change.
- Forest protected area planning and conservation in South Africa has not been systematic, and has been rooted in a silvicultural perspective, that has largely overlooked landscape level ecosystem processes important for forest biodiversity persistence. As such, forest conservation has largely comprised of setting aside 'token' isolated forests reserves without regard for the functional relationships between forest patches and the broader landscape. Currently, forest protected areas are too few, too small, too isolated and not representative of the range of forest types, to maintain natural processes, species populations or biodiversity across forest landscapes.
- The level of effective protection and conservation management, even within many statutory (Type 1) forest protected areas, is typically low or nonexistent, and many forest reserves are currently subjected to illegal harvesting and other forms of unnatural disturbance.
- The absence of appropriate land use regulations of the matrix within forested landscapes has resulted in significant transformation of the forest matrix, and as a consequence, changes in landscape level ecological processes, in particular hydrological systems, natural disturbance regimes, and ecological connectivity, and forest ecotones, have been disrupted. Effective long term conservation of forests will require integrated catchment management strategies to address cumulative impacts on sensitive ecosystems.
- Although it is recommended that the priority forests identified here, should be set aside as reserves, this does not mean that off-reserve management is not needed; on the contrary, reserves will never adequately represent all biodiversity, and are seldom large enough to contain processes across

landscapes. This calls for conservation planning to extend beyond just protected areas, and to include management of surrounding areas as well as cooperative land use planning and management across a range of different land management authorities and land owners.

- There is an apparent lack of political will to ensure the protection of forests in many areas. This is particularly evident in the politically sensitive communal areas of South Africa, where the absence of suitable alternative forms of livelihoods for communities reliant on forests, has resulted in a number of long standing conflicts between forest resource users and conservationists, (the Dukuduku coastal forests are a good example of this, see for example, Carnie, 2005).
- Government promoted initiatives calling for significant expansions of timber and biofuels industries within the communal areas of South Africa (Anon, 2006), pose a major potential threat to the biodiversity of these regions. Habitats that are suitable for the establishment of these industries often coincide with biodiversity sensitive catchments, many of which contain priority forests, (see for example, Berliner and Desmet, 2008).
- For the effective long term conservation of South Africa's forests improvements will need to be made to both 'on, and off reserve' conservation. The former will require expansion of the formal protected area network so as to be inclusive of all forest types, as well as improving the level of effective conservation management within existing reserves. For the latter, improvements will need to be made in the regulation of land use in and around forest clusters, as well as the implementation of integrated catchment management strategies that can address cumulative impacts acting on forests and other sensitive ecosystems.
- Finally, it is of critical importance that forest conservation is integrated with rural socioeconomic development programmes that promote community based conservation, alternative sustainable livelihoods and rural poverty alleviation.

The clearest way into the universe is through a forest wilderness.

John Muir

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Appendix A: Calculating targets for forest types using adjustment factors

Targets are a necessary component of systematic conservation planning. Typically they describe the spatial or quantitative requirements for biodiversity features within the planning unit. This is a brief description of the method used to derive quantitative forest conservation targets.

Data

The forest plant species data used by Von Maltitz *et al*. (2004) to classify forest types, was the main source of data for the determination of forest type targets. Expert opinion was used in cases where insufficient data was available for calculating targets, or where extreme rarity of the forest types necessitated a 100 % conservation target. Expert opinion was provided by forest scientists, from the Department of Water Affairs and Forestry.

Method

Recent research has shown the usefulness of species-area curves in helping to develop conservation targets. For example, Cowling *et al*. (1999b) used species-area curves to adjust the baseline target values for vegetation types in the Cape Floristic Region. This method involves analyzing species-area curves to compare species turnover and relative species numbers for areas of similar size among ecoregional classification units. More recently Desmet and Cowling (2004) showed how the power form of the Species-Area relationship can be used to set conservation targets using sample data. The method they described involves calculating the slope (z-value) of the power form of the species-area curve. This can then be used to estimate the proportion of the area (A) required to represent a given proportion of species (S) by:

$$S' = A'^{z}$$

The vegetation database used to derive the forest type classification was used here to determine the z-value for each forest type. This was done with the aid of

EstimateS software (Colwell 1997) with the bootstrap estimator using the following equation:

$$z = (y_2 - y_1) / (x_2 - x_1)$$

where: y_2 = log(total number of species in a forest type)
y_1 = log(average number of species per sample)
x_2 = log(total area of a forest type)
x_1 = log(average area of samples)

Desmet and Cowling (2004) found the bootstrap estimator provided the most consistent response across datasets to the under- or overestimation of species number. The z-values were then ordered and the ratio between each type and the lowest value was determined. The type with the lowest z-value was assigned the base value of 15%, while the base value for the remaining types was determined by multiplying the z-value ratio for a type by 15%.
The 15% target was based on a study by Pressey *et al.* (1996), which aimed to identify a representative forest network for North Eastern New South Wales, Australia.

The base values for the forest types were then adjusted upwards on the basis of four factors:

i. Relative rarity
ii. Patch fragmentation
iii. Historic reduction (since 1890)
iv. Location within regions/centres of endemism.

More specifically:

i. Relative rarity

This was simply determined by the area covered by each forest type expressed as a percentage of the total forest area. On the basis of these values, each type was assigned into one of three rarity classes (high, medium or low) using a natural breaks classification procedure. Target values were then adjusted as follows:
High = +7.5%
Medium = +5.0%
Low = + 2.0%

ii. Patch fragmentation

This was determined by first determining the mean patch size and mean nearest-neighbour (inter-patch) distance for each forest type (after Benn, 2004). These two sets of values were separately grouped into three classes, namely high, medium and low. A matrix approach was then used to assign each type to a high, medium or low fragmentation category 9see table below).

Table A1. matrix used to assign a fragmentation category for targets

		Mean Patch size		
		Low	*Medium*	*High*
Mean inter-patch distance (nearest-neighbour)	*High*	High	High	Medium
	Medium	High	Medium	Low
	Low	Medium	Low	Low

This approach resulted in types with high inter-patch distances and low mean patch sizes being considered as more fragmented than types with low inter-patch distances and high mean patch sizes.

Target values were then adjusted as follows:

High = +7.5%

Medium = +5.0%

Low = + 2.0%

The same method was used to adjust those process targets which required adjustment on the basis of forest type fragmentation.

iii. Historic reduction (since 1890)

Expert opinion was used to assign each forest type into one of three reduction categories, namely high, medium and low. Mr I. van der Merwe and Mr T. Stehle provided the expert opinion in this instance.

Target values were then adjusted as follows:

High = +7.5%

Medium = +5.0%

Low = + 2.0%

iv. Location with regions/centres of endemism

A GIS layer describing the boundaries of the regions and centers of floral endemism as described by van Wyk and Smith (2001) was obtained. The occurrence of each forest type within these regions was then determined by means of spatial overlays at the forest patch level. The targets for forest types which overlapped with these regions/centres were adjusted by +5%.

The final target percentage for each forest type was therefore calculated as follows:

Final Target = Base + Rarity + Fragmentation + Reduction + Endemism

Each forest type could have its target percentage area adjusted upwards by a minimum of 6% or a maximum of 27.5%.

Results

Table A2 gives the results of Z-value calculations used to calculate the base target for each forest type, and Table A3 summarizes factors used to adjust base targets to get overall target values for each forest type

Table A2**.** Deriving forest type base targets using a 15 % minimum and adjusting upwards depending on relative log transformed species diversity to area relationship ratios (Z-values ratios),). Note that species data was only available for 18 of the 21 forest types considered.

Forest Type	Z-value	Z-value Ratio	Base Target (%)	Sample Size (Number of plots)
Albany Coastal Coastal	0.059766	1	15.00	15
Mangrove	0.102355	1.712586	25.69	66
Eastern Cape Dune	0.113396	1.897325	28.46	21
Western Cape Milkwood	0.130539	2.184156	32.76	13
Swamp	0.142756	2.388563	35.83	114
Southern Cape Afrotemperate	0.151722	2.538588	38.08	255
Northern Mistbelt	0.157629	2.637423	39.56	144
Drakensberg Montane	0.165359	2.766758	41.50	103
Pondoland Scarp	0.165802	2.774169	41.61	69
Transkei Mistbelt	0.176003	2.944857	44.17	170
Transkei Coastal Scarp	0.179354	3.000918	45.01	150
Amatole Mistbelt	0.179795	3.008292	45.12	179
Licuati Sand	0.184344	3.084407	46.27	40
Eastern Mistbelt	0.185075	3.09665	46.45	243
Kwazulu-Natal Coastal	0.186034	3.11268	46.69	129

Kwazulu-Natal Dune	0.188046	3.146349	47.20	143
Low Escarpment Mistbelt	0.198191	3.316099	49.74	52
Mpumalanga Mistbelt	0.199186	3.332749	49.99	206

Table A3**.** Summary of factors used to adjust base targets to get overall target values for each forest type. The factors of rarity and fragmentation class were quantitatively determined, while historic reduction class was approximated using expert opinion. Note that targets marked with a '*' indicate targets set by expert opinion, due to absence of species data, or as in the case of swamp, mangrove, riverine, and sand forest, extreme rarity and sensitivity overrides all other considerations.

Forest Type	Base Target value (%)	Rarity Class	Fragmentation class	Historical reduction class	Target (%)
Lowveld Riverine	-	High	Medium	Medium	100 *
Swamp	35.83	High	Medium	Low	100 (70) *
Mangrove	25.69	High	Medium	High	100 (70) *
Licuati Sand	46.27	High	Medium	Medium	100 (70).
Western Cape Afrotemperate	38.08[1]	High	High	Low	60.
Southern Cape Afrotemperate	38.08	Low	Low	Low	49.
Drakensberg Montane	41.50	High	High	Low	63.5
Low Escarpment Mistbelt	49.74	High	High	Low	71.7
Northern Mistbelt	39.56	High	Medium	Low	59.5
Mpumalanga Mistbelt	49.99	Medium	Medium	Low	679
Eastern Mistbelt Forests	46.45	Medium	Medium	Medium	66.5
Transkei Mistbelt Forests	44.17	Medium	Medium	Medium	64.17
Amatole Mistbelt Forests	45.12	Low	Medium	Medium	62.12
Eastern Scarp Forests	41.61[2]	Medium	Medium	Medium	61.61
Pondoland Scarp	41.61	High	Medium	High	66.61
Transkei Coastal	45.01	Low	Medium	High	65.01
KwaZulu-Natal Coastal	46.69	High	Medium	High	71.69
KwaZulu-Natal Dune	47.20	High	Low	High	69.20
Eastern Cape Dune	28.46	High	Medium	Low	48.46
Albany Coastal	15.00	High	Low	Medium	35.00
Western Cape Milkwood	32.76	High	Medium	Medium	55.76

Note [1]: No species data available, base target set using closest forest type, Southern Cape Afrotemperate forest.
Note [2]: No species data available, base target set using closest forest type, Pondoland Scarp forest.

Appendix B: Scoring system for quantifying threats

Subsistence Resource use Pressure index (SRUPI)
The following formula was used to aggregate threat scores:

SRUPI = ((population density score) + (household wood use score) + (accessibility index score)/3) + (plantation buffering score)

Details of how each variable in this formula were scored are described below.

Population density
Population density in 5km buffers around each forest patch was based on spatial extrapolations of national census, (2001) data. Any forest patch that contained more than 30 people per ha of the 5km forest buffer was considered to have a 'very high' population density and scored a value of 5 (see table B1, below)

Table B1. Scoring system for population densities in the 5km forest buffer areas.

People/ha	Frequency (number of forest patches that fell within range)	Rating	Score
0–1	14 964	Very low	1
1–10	4 023	Low	2
10–20	92	Medium	3
20–30	31	High	4
>30	388	Very high	5

Household wood use
An index of wood usage was extrapolated from National census (2001) data by considering the proportion of households in each forest buffer reliant on wood as the main form of fuel, (as opposed to electricity, gas, paraffin, coal or solar power). A one to five scoring system was used, see table B2, below.

Table B2. Scoring system for household wood use (express as number of households per hectare reliant on wood as the main form of fuel).

Wood use (number of households per ha where wood is main form of fuel)	Frequency (number of forest patches that fell within range)	Wood use dependency Rating	Score
0–0.1	14 140	Very low	1
0.1–0.2	5 991	Low	2
0.2–0.3	948	Medium	3
0.3–0.4	116	High	4
0.4–0.5	25	Very High	5
0.5–0.6	6	Very high	5
>0.6	1	Very high	5

Accessibility of forests

The accessibility of forest is an important factor that will moderate the degree of use of use by humans. Accessibility is determined by topographic position and road access and road penetration into the forests. Rules used to derive an accessibility index score are derived from table B3, below.

Table B3. Scoring rules used to derive an accessibility index of a forest. (Note: where there is no road access, accessibility score is determined by slope only; where there are roads leading to a forest, both slope, and degree of road penetration is used).

Average slope %	Road access to forest		Road penetration in forest		
	No	Yes →	Bad	medium	Good
>25	0		1	2	3
10–25	1		2	3	4
<10	2		3	4	5

Buffering effect of plantations

Plantations and woodlots near forest patches are known to have a buffering effect on subsistence use of forests (Mucina and Geldenhuys, 2006). Where plantations occur within the forest patch buffers, the final SRUPI score was modified by subtracting from 0 to 2 points depending on the extent of plantations (see tableB4, below)

Table B4. Scoring system for the buffering effect of plantations and wood lots around forest patches

% Plantations in buffer	Modifying score to SRUPI
<30 %	0
30–70 %	–1
>70 %	–2

Deriving qualitative ratings for Subsistence Resource use Pressure index

A conversion table was used to convert the SRUPI scores into qualitative ratings at two levels of aggregation (see table, B5, below)

Table B5. SRUPI qualitative ratings.

SRUPI	Rating	Aggregated ratings
<1	Very low	LOW
1–2	Low	
2–3	Medium	MEDIUM
3–4	High	HIGH
4–5	Very high	

Surrounding agricultural land transformation (using an arability index)

The threat of transformation from agriculture and timber plantations is directly related to the suitability of the land for cultivation for agriculture. An arability index was derived by extrapolating from the Agricultural Research Council’s land capability data (Schoeman *et al.,* 2000) into five arability classes, with scores from 1 (low arability) to 5 (high arability). Percentages of each class falling into a 1km buffer area around each forest patch were aggregated into an a overall % arability (proportion of land in Class (I, II , II and IV) expressed as a % of land in class (V, VI, VII)

Urban expansion

Forests close to urban centres and coastal areas were considered to be at risk from development pressures. Any forest within15 km from an urban edge and all coastal forest were considered to be under high threat from urban expansion and scored 5.

Mining

Increasing demand for heavy minerals in certain coastal dune forest of KwaZulu-Natal and the Eastern Cape pose significant threats to these areas. The level of threat was scored according to current demand for the mineral, ranging from 1 (low demand to 3 (high demand).

Aggregation of threats

Combining the different threats into a single index is necessary to make comparisons across planning units. A rule based system was used that incorporated both the highest threats and the additive combined threats so that any of the individual threats scoring ‘high’ would qualify the planning unit for an overall ‘high’ threat status. The cumulative effect of individual threats was considered by combining individual threat scores, and where they exceeded a threshold value of 2.5, a ‘high’ overall threat status was assigned (see table B6).

Table B6. Rules used to aggregate individual threats into a single overall threat rating for each forest patch.

Rule 1	IF [SRUPI] > =4 OR [Threat Transform] >=4 OR [Threat urbanexp] = 5 OR [Threat mining] = 5 THEN [forest threat rating] = ‘high’
Rule 2	IF [SRUPI] <4 OR [Threat Transform] <4 OR [Threat urbanexp] < 5 OR [Threat mining] < 5 AND [combined average threat] > = 2.5 THEN [forest threat rating] = ‘medium’
Rule 3	IF [SRUPI] <4 OR [Threat Transform] <4 OR [Threat urbanexp] < 5 OR [Threat mining] < 5 AND [combined average threat] < = 2.5 THEN [forest threat rating] = ‘low’

Appendix C: Integrated list of priority forest patches

The forests in Table C1, below, represent the integration of priority forests identified by systematic assessment and/or expert judgement.

Table C1 Integrated list of priority forests.

Forest Name	Forest Type	Location (With topographic map code)	Justification	Significant Threats	Protection Status	Size (ha)
Futululu Forest	KwaZulu Natal Coastal Forest	Situated in Greater St Lucia Heritage Park near Mtubatuba KwaZulu-Natal north coast. (map : 2832 AD)	Remaining part of Dukuduku Forest that has not been occupied by informal settlement. Very high biodiversity.	Encroached by informal settlement. Proposed power line.	Situated in Greater St Lucia World Heritage Site/Heritage Park	c. 1030
Umdoni Forest	KwaZulu Natal Coastal Forest and KwaZulu Natal Dune Forest	Situated just south of Pennington on the Kwazulu-Natal south coast . (3030 BC)		Rapid change in land use and transformation of the surrounding landscape, as well as pressure from development.	Private land	c. 210
Hawaan Forest	KwaZulu Natal Coastal Forest	Ethekwini (Durban metropolitan) municipality , near Umhlanga Rocks. (2931CA)		Rapid change in land use and transformation of the surrounding landscape, as well as potential pressure from development.	Private land	c. 100
Sokhulu/Maphelana Forest	KwaZulu Natal Dune Forest	Coastal forest complex between St Lucia mouth and the Mfolozi River. KwaZulu-Natal north coast. (2832 AD and 2832 CB)	Mineral rich area. Forests immediately to the south are mined.	Mineral deposits occur –potential pressure for mining.	Largest part in State forest	c. 3800
Mtunzini Forest complex	KwaZulu Natal Dune Forest	Coastal forest complex between the Mhlatuze river mouth and Mlalazi River mouth at Mtunzini on Kwazulu-Natal north coast. (2831DD)	Forest hosts rare species. Partly transformed by lodge.	Potential pressure for future development.		c. 2521
Ongoye (Ngoye)	Eastern Scarp Forest	Near Mtunzini (2831 DC)	Forest with high diversity and rare and endemic species.	Area at risk from resource utilization and encroachment		c. 3229
Manguzi Forest	KwaZulu Natal Dune Forest	Kosi Bay Maputaland (2632DC)		Large forest under considerable pressure according to Cooper (1985)		c. 489
Ngome Forest	Low Escarpment Mistbelt Forest	Situated near Nongoma in northern KwaZulu Natal (2731CD)	Unique forest with rare and endemic species.	Extensive resource use and potential encroachment despite status	Largest part situated in Ntendeka Wilderness Area	c 2636

Forest Name	Forest Type	Location (With topographic map code)	Justification	Significant Threats	Protection Status	Size (ha)
				as Wilderness area.		
Ngele Forest	Eastern Mistbelt Forest	Southern KwaZulu-Natal (3029 DA)	Unique forest.	Mineral deposits occur – potential for mining.	State forest	c. 1900
Gwaliweni Forest		Near Ingwavuma, KZN (2731 BD + 2732 AC)		At risk from resource use pressures (bark harvesting, hunting etc)		c. 1306
Ntimbankulu Forest		Near Port Shepstone, KZN (3030 CB)		At risk from resource use pressures (bark harvesting, hunting etc)		c. 495
Matabetule/Mzinyati Forest		Near Inanda, KZN (2930 DB)		At risk from resource use pressures (bark harvesting, hunting etc)		c. 252
Mbizane Forest		Near Paddock, KZN (3030 CC)		At risk from resource use pressures (bark harvesting, hunting etc)		c. 231
Nungwane Forest		Near Umbumbulu, KZN (3030 BB + 2930 DD)		At risk from resource use pressures (bark harvesting, hunting etc)		c. 245
Folweni Forest		Near Adams Mission, KZN (3030 BB)		At risk from resource use pressures (bark harvesting, hunting etc)		c. 125
Ndulinde Forest		Near Nyoni, KZN (2931 AB)		At risk from resource use pressures (bark harvesting, hunting etc)		c. 146
Telezi Forest		Near Inanda, KZN (2930 DB)		At risk from resource use pressures (bark harvesting, hunting etc)		c. 95
Nwabi Forest		Near Umbumbulu, KZN (2930 DD + 2931 CC)		At risk from resource use pressures (bark harvesting, hunting etc)		c. 110
Mtamvuna North Forests		KZN (3030 CC)		At risk from resource use pressures (bark harvesting, hunting etc)		c. 234
Dududu Forest Complex		KZN (3030 BA)		At risk from resource use pressures (bark harvesting, hunting).		c. 248

Forest Name	Forest Type	Location (With topographic map code)	Justification	Significant Threats	Protection Status	Size (ha)
Hlabisa Forest Complex		KZN (2831 BB)		At risk from resource use pressures (bark harvesting, hunting).		c. 385
Mfumbe Forest Complex		KZN (3030 AD+BC+BD)		At risk from resource use pressures (bark harvesting, hunting).		c. 550
Mhlatuze Mangrove Forest	Mangrove Forest	In and around Richards Bay harbour. KwaZulu-Natal north coast (2832 CC)	Future expansion of the harbour and pollution threaten part of the mangrove forest.	Harbour development, coal dust pollution, siltation and water pollution.	Land belonging to the Ports Authority, and to private harbour operators, part nature reserve.	c. 390
St Lucia mangrove forest	Mangrove forest	Situated in Greater St Lucia Heritage Park near Mtubatuba. (2832AD)	Largest mangrove in SA	Hydrological changes in the freshwater system.	Situated in Greater St Lucia Heritage Park	c. 200 ?
Nxaxo forest	Eastern Cape Dune Forest	Matiwane district. (3228 DA)	One of the best examples of dune forest	Extensive resource use and potential encroachment. Potential threat from mining activities.		
Ntlaboya forest	Eastern Cape Dune Forest	Matiwane district . (3228 DA)	One of the best examples of dune forest.	Extensive resource use, and potential encroachment. Potential threat from mining activities.		
Kobonqaba forest	Eastern Cape Dune Forest	Matiwane district (3228 DA)	One of the best examples of dune forest	Extensive resource use, and potential encroachment. Potential threat from mining activities.		
Mngazana Estuary Mangrove Forests	Mangrove Forest	Mngazana Estuary (south of Port St. Johns) on Transkei Wild Coast Matiwane district . (3129 DA)	Third largest mangrove with unique fauna, including rare crab species.	Under severe pressure from pole harvesting. Impact from illegal holiday cottages and recreation activities (e.g. wave action of boats) .		c. 104
Hili Forest	Transkei_ Coastal_ Forest	Transkei Wild Coast (3129BC)		Forests under pressure from resource utilisation, stock and frequent fires around the edges.		c. 1279
Umzimpunzi/ Ntsubane Forest	Transkei_ Coastal_ Forest	Transkei Wild Coast (3129BC)		Forests under pressure from resource utilisation, stock and frequent fires		c. 793

Forest Name	Forest Type	Location (With topographic map code)	Justification	Significant Threats	Protection Status	Size (ha)
				around the edges.		
Mpame	Transkei_ Coastal_ Forest	Matiwane (3229 AA)		Forests under pressure from resource utilisation, stock and frequent fires around the edges.		c. 538
Mount Thesiger Forest Complex	Transkei_ Coastal_ Forest	Forest complex adjoining Port St .Johns between the Mzimvubu and Mngazi River mouths Matiwane (3129 DA)		Forests under pressure from resource utilisation, stock, land clearance and frequent fires around the edges.		c. 548
Mount Sullivan (Bovini) Forest	Transkei_ Coastal_ Forest	Matiwane (3129 DA)		Forests under pressure from resource utilisation, stock and frequent fires around the edges.		c. 1390
Ntlopeni/ Mkomanzi Forest	Transkei_ Coastal_ Forest	Matiwane (3129 DA)		Forests under pressure from resource utilisation, stock and frequent fires around the edges.		c. 632
Lotana Forest	Transkei_ Coastal Forest	Matiwane (3129 DA)		Forests under pressure from resource utilization (including hunting). Impact from illegal holiday cottages.		c. 978
Pembeni Forest	Transkei_Coastal_ Forest			Forests under pressure from resource utilisation, stock and frequent fires around the edges.		c. 583
Manubi Forest	Transkei_ Coastal_ Forest		Extremely varied forest with great variety of bird life and exploitable commercial timber.	Forests under pressure from resource utilisation, stock and frequent fires around the edges		c. 762
Pirie Forest	Amatole Mistbelt Forest	Kei, Eastern Cape (3227 CC)		Forests under pressure from resource utilization		
Stanford Forest	Western Cape Milkwood Forest	Between Stanford and Gansbaai near the coast.	Rare and unique milkwood community	Priv ate lodge development – potential to expand	Private Land	c. 198
Blouberg Forest	Limpopo Mistbelt Forest	Blouberg mountain in Lebowa (northeastern part of Limpopo).	High forest with commercial timber	Extensive resource use and potential encroachment		c. 120

Forest Name	Forest Type	Location (With topographic map code)	Justification	Significant Threats	Protection Status	Size (ha)
				on the area.		
Hangklip forest		On Hangklip State Forest near Louis Trichardt/Makhado .(2229 DD + 2329 BB)	Very good example of this forest type	Encroachment of military installations	State forest	c. 286

Appendix D: Priority forest clusters identified using rule based selection

Columns show criteria used in the selection rules, including: % of matrix that is natural, forested area of cluster, number of patches in cluster, and mean patch size. Other information provided include: designated cluster name; protection status of cluster, and if endangered (E) or critically endangered (CE) vegetation types occur in cluster matrix.

Forest types (s)	Name	Location	In PA	E or CE veg. in matrix	% Matrix Natural	Forest area (ha)	No. of patches	Mean patch size (ha)
Albany	Alexandra north	N of Alexandra	N	N	98	9780	24	408
Albany	Forest glade	W of Alexandra	N	N	76	2784	19	147
Albany	Woody cape	SW of Alexandra	Y	N	80	5	421	91
Amatole Mistbelt	Katberg	NW of Hogsback	N	N	92	3628	186	20
Amatole Mistbelt	Keiskammahoek	around Keiskammahoek	N	N	88	26098	405	64
Amatole Mistbelt	Juanasberg forests	N of Alice	N	N	99	1918	41	47
Drakensberg Montane	Kwahlathikhulu forest	N of Monks cowl forest station	Y	N	100	74	2	37
Drakensberg Montane	Monks cowl	S of Monks cowl forest station	Y	N	100	249	8	31
Drakensberg Montane	Edashi forest	near Giants castle	N	N	100	111	1	111
Eastern Cape Dune	Gulu	N and S of Kidd's beach	Y	N	89	602	18	33
Eastern Cape Dune	Keiskamma mouth	N and S of Keiskamma mouth	N	N	92	2562	19	135
Eastern Cape Dune	Mpakweni	N of Fishriver	N	N	97	5185	32	162
Eastern Mistbelt	Qudeni	W of Melmoth	P	Y	92	1876	27	69
Eastern Mistbelt	Karkloof forest	N of Howick	P	Y	71	4779	42	114
Eastern Mistbelt	Ngele (Weza south)	SW of Kokstad	N	Y	97	1003	50	20
Eastern Scarp	Pongolapoort	N of Mkuzi town	N	Y	97	3552	7	507
Eastern Scarp	Ntendeka wildeness/Ngome	E of Vreyheid	Y	N	94	3446	33	104
Eastern Scarp	Hluhluwe complex	W of Hluhluwe town	P	N	96	3276	84	39
Eastern Scarp	Ngoye	SW of Empangeni	Y	Y	100	2613	10	261
Kwazulu-Natal Coastal	Lake st Lucia western shores	W of lake st Lucia	Y	Y	70	5985	63	95
Kwazulu-Natal Coastal	Naylazi forest	W of lake st Lucia	N	N	73	754	7	108
Kwazulu-Natal Dune	Bhangazi coastal forest	S Jeser pt Sodwana	Y	N	92	533	4	133
Kwazulu-Natal Dune/Mangrove	Mtunzini	S coast from Richards bay	P	Y	82	3654	9	406
Kwazulu-Natal Dune/Swamp	Richards bay N/Sokulu S	N. coast- Richards bay	N	Y	79	1891	57	33
KZN Coastal/KZN Dune/ Mangrove	Dukuduku-Sokhulu	E. of Matubatuba	P	Y	82	12445	162	77
Licuati Sand	Tembe/Nhlambeni	N. part in Tembe NR	N	N	97	13491	298	45
Licuati Sand	Munyu	S.W. of Kosi bay	N	N	97	1429	17	84
Lowveld Riverine	Pafuri, Limpopo river	Northern Kruger National Park	Y	N	100	3219	42	77
Lowveld Riverine	Livuvuvhu river	N.W. Kruger National Park	Y	N	100	2342	104	23
Lowveld Riverine	Mkuzi river forest	In old Mkusi Game reserve	Y	N	85	2144	10	214
Lowveld Riverine	Msunduzi river forest	Mkusi Game reserve	Y	N	94	965	3	322
Mangrove	Kosi bay mangroves	at Second lake of Kosi system	Y	N	38	9	1	9

Forest types (s)	Name	Location	In PA	E or CE veg. in matrix	% Matrix Natural	Forest area (ha)	No. of patches	Mean patch size (ha)
Mangrove	Beachwood mangrove NR	in Durban metropol.	Y	Y	65	48	2	24
Mpumalanga Mistbelt	Mpumalunga esarpment	Mariepskop to Onverwacht	N	Y	93	12784	111	115
Mpumalanga Mistbelt	Mac Mac	S.E. of grasskop	N	Y	74	989	32	31
Mpumalanga Mistbelt	Buffelskloof	S. of Lisbon State forest, near Nelspruit	P	Y	70	1722	90	19
Northern Kwazulu-Natal Mistbelt	Ngcaka	Between Paulpetersberg and Wakkerstroom	N	N	100	983	6	164
Northern Kwazulu-Natal Mistbelt	Balelesberge forests	N.W. of Utrecht	N	N	100	221	6	37
Northern Kwazulu-Natal Mistbelt	Glencoe	W. of Dundee	N	N	94	209	2	105
Northern Mistbelt	Letjume forest	Near Buysdorp	N	N	100	3114	30	104
Northern Mistbelt	Blouberg	Blouberg mountains	N	N	100	1171	11	106
Northern Mistbelt	Woodbush/grootbosch	E. of Tzaneen	N	Y	85	7023	108	65
Pondoland Scarp	Umtanvuna to coast	S. of Port Edward	P	N	91	1421	118	12
Pondoland Scarp	Mkabati	Mkabti NR, Mtentu river	P	N	88	531	4	133
Pondoland Scarp	Oribi gorge	Inland from Margate	Y	N	82	467		93
Southern Cape Afrotemperate	Blue Lily's Bush	near Knysna	N	N	78	49625	801	62
Southern Cape Afrotemperate	Witelsboss east	S.E. of kareedouw	N	N	90	970	32	30
Southern Cape Afrotemperate	Thyspunt/Rebelrus	W. of Cape st Frances	P	Y	100	875	11	80
Swamp	Kosi bay swamp South	S.W. of kosi bay	P	N	91	1257	31	41
Swamp	Sodwana bay swamp	W. of Sodwana bay	Y	Y	92	306	5	61
Swamp / KZN coastal	Kosi bay East	E .Kosi bay N	N	N	100	513	9	57
Transkei Coastal Platform	Tyelemanzi	10 km inland of Mpade	N	N	88	814	33	25
Transkei Coastal Platform	Dwesa/Cebwe complex	Dwesa reserve	Y	N	95	5036	215	23
Transkei Coastal Platform	Vata	W. of Nqabara, S. of Dwesa	N	N	95	1117	82	14
Transkei Coastal Platform	Manubi cluster	W. of Mazeppa bay	N	N	87	2048	53	39
Transkei coastal/Pondoland Scarp /mangroves	Wild coast mega cluster: Mpambe, Gxwaleni, Huleka, Mt Thesinger, Lotana, Nsubane, Mngazana mangroves	Mboyti to Mbashe river	N	N	90	35083	1332	26
Transkei Mistbelt	Hili/Mngcozo	N.W. of Umthatha	N	Y	98	1020	46	22
Transkei Mistbelt	Gulawndoba	N.W. of Engcobo	N	Y	87	1929	81	24
Transkei Mistbelt	Umzantsi	Tsomo district	N	N	92	680	32	21
Western Cape Afrotemperate	Orangekloof	Cape Town, Table Mt	Y	Y	100	100	18	6
Western Cape Afrotemperate	Boesmans Bos	E .of Swellendam	Y	N	100	291	10	29
Western Cape Afrotemperate	Grootvadersbosch	E. of Swellendam	P	Y	100	621	1	621
Western Cape Milkwood	Stillbaai Melkhoud	Stillbaai	N	N	94	341	1	341
Western Cape Milkwood	Stanford Milkwoods	Stanford	N	N	98	217	7	31
Western Cape Milkwood	South of Standford	South of Standford	N	N	100	132	5	26

Appendix E. Threatened forest plants.

Table E1. Threatened forest plants, showing: status, endemism, and reasons for population declines (where available). Abbreviations: EW, though to be extinct in the wild; CR, critically endangered; EN, endangered; VU, vulnerable, (note 'near threatened' category is excluded although still considered as red data species). 'Y' = yes, 'N' = No, '?' = uncertainty. For primary data sources see notes at bottom of table.

Scientific name	Conservation status	Endemic to SA	Reasons for population decline
Encephalartos woodii	EW	Y	Unknown, but last plant found in wild in Ongoya forests in 1908.
Turraea streyi	EW	Y	
Encephalartos laevifolius	CR	FSA	Harvesting , collectors, habitat Loss and degradation, Poor recruitment
Encephalartos latifrons	CR	Y	Harvesting , collectors, habitat Loss and degradation, Poor recruitment
Gladiolus cruentus	CR	Y	Habitat Loss, degradation, subsistence harvesting, intrinsic factors - Poor recruitment/reproduction/regeneration
Protea roupelliae	CR	Y	
Albizia suluensis	EN	Y	The habitat fragmentation (agricultural activities and settlement). Too frequent fires. No evidence of recruitment
Encephalartos lebomboensis	EN	FSA	Harvesting , collectors, habitat Loss and degradation, Poor recruitment
Jubaeopsis caffra	EN	Y	Possible that subpopulations destroyed by deforestation
Leucadendron argenteum	EN	Y	
Manilkara nicholsonii	EN	Y	Subsistence harvesting (firewood and timber), agricultural activities and coastal development.
Maytenus abbottii	EN	Y	Subsistence harvesting for firewood and timber, habitat cleared for agriculture and settlement.
Metarungia galpinii	EN	Y	
Rhus rudatisii	EN	Y	
Aloe ciliaris	VU	Y	
Cassipourea flanaganii	VU	Y	
Clivia nobilis	VU	Y	
Colubrina nicholsonii	VU	Y	
Didymoplexis verrucosa	VU	Y	
Encephalartos ngoyanus	VU	FSA	Harvesting , collectors, habitat Loss and degradation, Poor recruitment
Encephalartos altensteinii	VU	Y	Harvesting , collectors, habitat Loss and degradation, Poor recruitment
Eugenia simii	VU	Y	
Gerrardanthus tomentosus	VU	Y	
Impatiens flanaganiae	VU	Y	
Maytenus oleosa	VU	Y	
Pseudosalacia streyi	VU	Y	

Scientific name	Conservation status	Endemic to SA	Reasons for population decline
Raphia australis	VU	N	
Streptocarpus cyaneus	VU	Y	
Streptocarpus denticulatus	VU	Y	
Streptocarpus fenestra-dei	VU	Y	
Streptocarpus kentaniensis	VU	Y	
Streptocarpus molweniensis	VU	Y	
Syncolostemon latidens	VU	Y	
Tephrosia pondoensis	VU	Y	Subsistence harvesting, trampling by livestock

Primary data sources (used in appendix E)

Species list: Von Maltitz, *et al.* (2003); unpublished releve' data collected for DWAF forest subtype classification, compiled and provided by Prof L. Mucina . Data arranged into forest types and summarised for forest target assessment by Dr P. Desmet.

Red data classification and threats: SANBI 2007 listings (www.sanbi.org/biodiversity/reddata.htm); IUCN Global 2007 classifications (www.iucnredlist.org).

Appendix F: Threatened forest fauna

Table E 2. Threatened forest fauna, showing: IUCN status, endemism, forest dependency and reasons for population declines (compiled and integrated by Berliner and Wright, 2008, but see footnote for citations to original sources). Abbreviations: EW, though to be extinct in the wild; CR, critically endangered; EN, endangered; VU, vulnerable; (note 'near threatened' category is excluded although still considered as red data species). 'Y' = yes, 'N' = No, '?' = uncertainty. Definitions of terms and primary data sources given in notes at end of this table.

Taxonomic group		Common name	Scientific name	Status	Endemic to SA	Forest depend-ent	Reasons for population declines
Mammals	*Artiodactyla*	Red duiker	*Cephalophus natalensis*	VU	N	Y	Deforestation and habitat modification, coastal developments
		Suni	*Neotragus moschatus zuluensis*	VU	N	Y	Human habitation, slash and burn agricultural practices
	Hyracoidea	Tree hyrax	*Dendrohyrax arboreus*	VU	N	Y	Deforestation and habitat modification, subsistence harvesting
	Insectivora	Gunning's Golden Mole	*Neamblysomus gunningi*	EN	Y	N	Habitat loss from timber plantations
		Giant Golden Mole	*Chrysospalax trevelyani*	VU	Y	Y	Fragmentation and habitat loss from clearing , timber plantation, grazing impacts. Local exploitation
	Primates	Samango Monkey (SA sub spp)	*Cercopithecus mitis labiatus*	EN	Y	Y	Habitat degradation from selective harvesting (pre 1940 logging), and subsistence harvesting (current) , altered forest structure and food abundance; plantation forestry has increased population fragmentation
		Samango Monkey	*Cercopithecus mitis erythrarchus*	VU	N	Y	See above
	Rodentia	Ongoye Red Squirrel	*Paraxerus palliatus ornatus*	CE	Y	Y	Numbers stable, limited to 1 location (Ngoye forest). Forest subjected to some subsistence harvesting pressure, but not thought to effect squirrels
		Tonga Red Squirrel	*Paraxerus palliatus tongensis*	EN	N	Y	Human settlement, slash and burn shifting agriculture and wood harvesting practices severely affect habitat quality for the species
		Sclater's Forest Shrew	*Myosorex sclateri*	EN	Y	?	Human settlement, forest clearing and

Taxonomic group		Common name	Scientific name	Status	Endemic to SA	Forest depend-ent	Reasons for population declines
							overgrazing
		Four-toed Elephant-shrew	*Petrodromus tetradactylus*	EN	N	Y	Tourism and utilisation of areas , Habitat loss
		Maquassie Musk Shrew	*Crocidura maquassiensis*	VU	N	N	Habitat fragmentation, poor dispersal and recruitment
		Giant Rat	*Cricetomys gambianus*	VU	N	Y	Habitat modification : agricultural activities
Birds		Cape parrot	*Poicephalus robustus*	EN	Y	Y	Low breeding success , captive bird trade, removal of largest trees that they breed
		Spotted Ground-Thrush	*Zoothera guttata*	EN	Y/N	Y	Dune mining, habitat disturbance in protected areas , deforestation in northern part of range, outside SA, subspecies endemic to SA.
		Knysna warbler	*Bradypterus sylvaticus*	VU	Y	?	Clearance of coastal forest , plantation fire management (fire brakes along forest margins) , leading to unnatural ecotones
		Southern banded snake eagle	*Circaetus fasciolatus*	VU	N	Y	Clearing and degradation of coastal forests. Numbers in SA now stable
		Eastern Bronze naped pigeon	*Columba delegorguei*	VU	N	Y	Habitat loss, mostly outside of SA
		Mangrove kingfisher	*Halcyon senegaloides*	VU	N	Y	Clearing and degradation of mangrove forests, e.g. around Durban
		Pel's fishing owl	*Scotopelia peli*	VU	N	Y	Destruction of riparian forests
		Green barbet	*Stactolaema olivacea*	VU	Y/N	Y	Limited range, no evidence for decline
Reptiles	*Seps*	Eastwoods's Long -tailed Seps	*Tetradactylus eastwoodae*	EW ?	Y	?	
	Chameleons	Transkei Dwarf Chameleon	*Bradypodium caffrum*	EN	Y	Y	
		Kentani Dwarf Chameleon	*Bradypodium kentani*	EN	Y	N	
		Setaro's Dwarf Chameleon	*Bradypodium setaroi*	EN	Y	Y	
	Skinks	Woodbush Legless Skink	*Acontophiops lineatus*	VU	Y	Y	
	Snakes	Gaboon Adder	*Bitis gabonica*	VU	N	Y	
	Geckos	Methuen's Dwarf Gecko	*Lygodactylus methueni*	VU	Y	Y	
Amphibians		Ngoni/mistbelt moss frog	*Arthroleptella ngongoniensis*	CR	Y	Y?	Habitat loss and fragmentation due to Afforestation and agriculture
		Table mountain ghost frog	*Heleophryne rosei*	CR	Y	Y	Dams, alien vegetation, too frequent fires, global

Taxonomic group		Common name	Scientific name	Status	Endemic to SA	Forest depend-ent	Reasons for population declines
							warming
		Knysna leaf-folding frog (Knysna banana frog)	*Afrixalus knysnae*	EN	Y	Y?	Habitat loss from: coastal development, plantations, draining, alien vegetation, trampling and cattle grazing
		Hogsback frog	*Anhydrophryne rattrayi*	EN	Y	Y	Habitat loss and fragmentation due to plantations
		Kloof frog	*Natalobatrachus bonebergi*	EN	Y	Y	Clearing of coastal forest (urbanization, plantations, sugar cane) and degradation (stream pollution and wood cutting); destruction of water courses
		Natal leaf-folding frog (Natal banana frog)	*Afrixalus spinifrons*	VU	Y	N	Habitat loss and fragmentation due to timber and sugar cane cultivation, pesticide pollution and trampling by livestock
		Nothern forest rain frog	*Breviceps sylvestris*	VU	Y	Y	Habitat loss and fragmentation due to timber and agriculture, reduction of surface water , altering of fire regimes, roads
Invertebrates	*Crustacea*	Blue River Crab	*Potamonautes lividus*	VU	Y	Y	
	Millipedes	Major Black Millipede	*Doratogonus major*	CR	Y	Y	
		Badplaas Balck Millipede	*Doratogonus furculifer*	EN	Y	Y	Habitat Loss, degradation, invasive species Natural Disasters
		Strong Black Millipede	*Doratogonus infragilis*	EN	Y	Y	Habitat Loss, degradation, intrinsic factors (poor dispersal) Intrinsic Factors
		Ruby-Legged Black Millipede	*Doratogonus rubipodus*	EN	Y	Y	
		Northern Black Millipede	*Doratogonus septentrionalis*	EN	Y	Y	Aforestation, invasive alien species , Fire (a major threat), intrinsic factors (slow growth, low dispersal ability)
		Zululand Black Millipede	*Doratogonus zuluensis*	EN	Y	Y	Aforestation, mining, tourism, harvesting of forest products
		Solitary Black Millipede	*Doratogonus avius*	VU	Y	Y	
		Bearded Black millipede	*Doratogonus barbatus*	VU	Y	Y	Habitat Loss, degradation, natural disasters, Intrinsic factors
		Herberts Balck millipede	*Doratogonus herberti*	VU	Y	Y	
		Hoffmans Black Millipede	*Doratogonus hoffmani*	VU	Y	Y	
		Southern Black Millipede	*Doratogonus meridionalis*	VU	Y	Y	

Taxonomic group	Common name	Scientific name	Status	Endemic to SA	Forest depend-ent	Reasons for population declines
	Natal Black Millipede	*Doratogonus natalensis*	VU	Y	Y	
	Precarious Black Millipede	*Doratogonus precarius*	VU	Y	Y	Habitat Loss, degradation, agriculture, plantations , Fires , intrinsic Factors(slow growth and maturation, limited dispersal ability)
Snails	Not available	*Gulella puzeyi*	CR	Y	Y	Habitat loss
	Not available	*Gulella salpinx*	CR	Y	Y	Habitat loss
	Pondoland Cannibal Snail	*Natalina beyrichi*	CR	Y	Y	Habitat Loss and degradation from infrastructure development, tourism, recreation, Intrinsic factors (restricted range)
	Dlinza Forest Pinwheel Snail	*Trachycystis clifdeni*	CR	Y	Y	Habitat Loss, degradation, human disturbance
	Not available	*Trachycystis placenta*	CR	Y	Y	
	Trumpet-mouthed snail	*Gulella aprosdoketa*	EN	Y	Y	
	Trumpet-mouthed snail	*Gulella claustralis*	EN	Y	Y	
	Not available	*Trachycystis haygarthi*	EN	Y	Y	
	Pondo agate snail	*Archachatina limitanea*	EN	Y	Y	
	Burnup's Hunter Slug	*Chlamydephorus burnupi*	VU	Y	Y	
	Snake Skin Hunter Slug	*Chlamydephorus dimidius*	VU	Y	Y	
	Tongoland Cannibal Snail	*Natalina wesseliana*	VU	Y	Y	
	Not available	*Sheldonia puzeyi*	VU	Y	Y	
Worms	Pink Velvet Worm	*Opisthopatus roseus*	CR	Y	Y	
	Knysna Velvet Worm	*Peripatopsis clavigera*	VU	Y	Y	

Definition of terms:

Threatened species status: recognise or listed by IUCN as extinct in the wild; critically endangered; endangered or vulnerable (note 'near threatened' category is excluded, although still considered as 'red data species').

Endemic to South Africa: the species or subspecies occurs only within South Africa .

Forest dependent: Lawes (2007) defines a forest dependent species as one that breeds exclusively in forests. For this analysis a broader definition was adopted. A forests species being defined as one in which the species population is depended on forest for their survival. This may be in the opinion of an expert, or as where the species distribution ranges (as indicate by best available sources) overlap at least by 80% with those of forests.

Reasons for population declines: this is a brief listing extracted from the literature of the main causative agents that have lead to the species population declines, (provides an explicit link to threats at a biome level). This data was currently not available for reptiles.

Primary data sources:

Mammals

Species lists: Castley, and Kerley, (1996); Von Maltitz *et al.,* (2003); Friedman, and Daly, (2004).

Red data classification and threats: Lawes *et al.,* (2000); Friedman and Daly. 2004; IUCN Global 2007 classifications (www.iucnredlist.org);

Birds

Species list: Wethered and Lawes, (2003); Von Maltitz *et al.,* (2003); Chittenden (2007.

Red data classifications and threats: Brook, (1984); Barnes, (2000); Symes , (2004); Birdlife International (2004); IUCN Global 2007 (www.iucnredlist.org);

Reptiles

Species list: Branch, (1998). Marius Berger and Dr. Krystal Tolley, personal communications.

Red data classification: Branch (1988); Dr Krystal Tolley (Chameleons) personal communications; IUCN Global 2007 (www.iucnredlist.org). Note: latest South African Reptile Atlas (in prep), was not available at time of analysis.

Amphibians

Species list and red data classification: Minter, *et al*. (2004); IUCN Global 2007 classifications (www.iucnredlist.org).

Invertebrates

Species list and Red data classification: Hamer, (2000); Herbert, (1997); IUCN Global 2007 classifications (www.iucnredlist.org); inland Invertebrate Initiative (http://www.nu.ac.za/redlist).