CHAPTER 2: LITERATURE REVIEW

2.1 Introduction

Accelerating rates of habitat destruction in South Africa, owing to high development pressures on the country's rich environmental resources, have cast a new spotlight on biodiversity research; this new interest in biodiversity relates to the present environmental and social imperative of sustainable development (Lovejoy, 1995). Therefore, the inventory of plant and animal diversity has become an urgent task in South Africa. Biodiversity elements form discrete units in the landscape that can be quantified. It is important to know what biodiversity resources we have in our country, and where they are located. This is part of good housekeeping, as the collection, management and communication of biodiversity data form the basis to maintain sustainable development activities (Lovejoy, 1995).

Patterns of biodiversity are investigated at various levels in order to target conservation efforts. Particularly the evaluation of biogeographical patterns of plant diversity forms an important tool in this respect. Existing biodiversity data used for conservation planning is, however, characterized by quality problems, which may result in a predisposed delineation of biodiversity hotspots that do not successfully support biodiversity conservation. The existing floristic data of the western Central Bushveld is both incomplete and biased. Only QDGs covering urban and existing protected areas are well sampled, whereas QDGs of rural and remote areas are rather under-sampled. Consequently, such a species distribution dataset has a limited capacity to serve as tool for the prioritisation of conservation areas.

One solution to this problem would be intensive sampling across the study area, but this is a time-consuming and complex activity, which does not meet the pressure for urgent decision-making required to find solutions for our present environmental crisis (Pearson, 1995). Therefore new methods are necessary to estimate biodiversity patterns across large areas without exhaustive data sampling. The current study represents a start in finding new ways for the estimation of possible species occurrences by the extrapolation of incomplete datasets.

The following discussion thus gives an overview of biodiversity conservation concepts and approaches and how they apply to the present study. Starting with an introduction about what biodiversity is (2.2), how it is measured (2.3), and what the importance and value of biodiversity is (2.4.1), the focus then goes towards the major threats to biodiversity in South

Africa (2.4.2) and the benefits of biodiversity conservation (2.4.3). In the subsequent section a brief history is given about the history and achievements of biodiversity conservation in South Africa (2.5). Since the present study looks at floristic diversity within a bioregion, a definition of the term bioregion as a unit is given (2.6.1), as well as the value of conservation at the bioregional level (2.6.2). A short overview is given on the relevance of conservation planning (2.7.1) and limitations as a result of uncomplete and biased data (2.7.2). Then the discussion rounds off by introducing relevant methods for area prioritization in biodiversity conservation planning (2.7.3) and how spatial modelling using GIS techniques can aid this process (2.7.4).

2.2 What is biodiversity?

The word 'biodiversity' is a contraction of the term 'biological diversity' and literally means the 'diversity of life' derived from the Greek word 'bios' for 'life' (Swingland, 2001). According to Hamilton (2005) biological diversity was first used by Lovejoy (1980) to describe what we now call species richness. In 1985 the contracted expression was introduced for the conference proceedings of the 'National Forum on BioDiversity' in the USA from where it made its way into the political arena at first, but it was soon adopted by scientists to provide justification and funding for the research and conservation of biodiversity (Hamilton, 2005). With the growing concern about the increasing loss of biological diversity due to human activities, the term 'biodiversity' has exploded into our vocabulary and is now widely used by governments, NGOs, media and scientists (Harper & Hawksworth, 1995). After the Rio Convention in 1992, the word 'biodiversity' has been launched world-wide (Duelli & Obrist, 2003).

However, there is no consensus on a universal definition for biological diversity, and thus the term has been loosely applied to depict various contexts of biological diversity; often 'biodiversity' is equalled with species diversity, the number of species and that of the individuals within each of those species found in a habitat, referred to as species richness and species abundance respectively in scientific terms (Hamilton, 2005). But species diversity is only one aspect of biodiversity. Biological diversity exists at many different levels of biological and geographical organization, from genetic diversity within local, regional or global distinct populations of a species, to species diversity of communities, ecosystems and landscapes (May, 1995; South Africa, 2005). According to Long (1996) biodiversity can be observed at any spatial scale, ranging from microhabitats to different habitat patches nested within the various ecosystems as quoted by Swingland (2001).

This complex spatial patterning of species diversity was first described by Whittaker (1960) who developed the concept of α, β and γ diversity (Routledge, 1977); Whittaker (1960) observed that species communities inhabiting the various habitats across a landscape can be hierarchically classified into different components (Routledge, 1977; Christ *et al*., 2003; Hamilton, 2005). According to Hamilton (2005) the first level of the multiple levels of spatial organization of biodiversity is point diversity, the diversity within a microhabitat. Microhabitats are components of homogeneous habitats, whose diversity is referred to as α-diversity. The diversity of different habitats or communities, the β-diversity, makes up the landscape level diversity (γ-diversity) as well as the diversity at biogeographical scale (ε-diversity) (Hamilton, 2005).

In the Convention on Biological Diversity by the United Nations (1992), biodiversity is defined as "*the variability among living organisms from all sources*" including plants, animals, fungi and other microorganisms found in the diverse terrestrial and aquatic ecosystems and ecological complexes in which they occur. A more comprehensive definition is also given by Noss & Cooperrider (1994) who consider biodiversity to be "*the variety of living organisms, the genetic differences among them, the communities and ecosystems in which they occur, and the ecological and evolutionary processes that keep them functioning, yet ever changing and adapting"* (Swingland, 2001).

These definitions relate to the omnipresence of biodiversity on our planet, which can be observed in every drop of water and handful of soil. The Millennium Ecosystem Assessment (2005) refers to this layer of living organisms as our biosphere, where myriads of biota form a dynamic environmental system through their combined metabolic activities. Additionally, these definitions denote not only natural ecosystems but also to human managed ecosystems such as urban landscapes, cultivated lands and rangelands, which harbour their own unique biodiversity that contributes to the ecosystem services in those anthropogenic dominated systems, as well as to the overall biodiversity of a region (Millennium Ecosystem Assessment, 2005). Therefore, biota can be characterized in a multi-dimensional way by its genetic, taxonomic and ecological diversity, which varies over space and time.

2.3 Measurement of biodiversity and the use of biodiversity indicators

Over the past centuries scientists have progressed with studies on the abundance of biological diversity, a large number of species, their functional traits and interactions have been described. We have gained an understanding of where biodiversity is, how its distribution is changing over time, what the drivers of such changes are, and what the consequences of such changes are for ecosystem services and human well-being (Millennium Ecosystem Assessment, 2005).

There still remain considerable gaps in species knowledge due to the enormous and difficult task of biodiversity quantification. The true extent of biodiversity still waits to be unravelled. This is a race against time, as a large part of our biodiversity is in danger of going extinct before we have captured it, mainly as a result of present extinction rates, which—exacerbated by human activities—are expected to have the most profound effect on our biodiversity hotspots "*where many of our most imperilled species are found"* (Hess *et al*., 2006); and secondly, because gaps in datasets and lack of resources to accomplish complete inventories often hinder timely and successful biodiversity planning and conservation.

As a solution to this dilemma, the conservation community has introduced the use of surrogate taxa. It is assumed that the protection of surrogates also capture a wide range of other biota and conservation targets (Marguels & Pressey, 2000; Lawler *et al*., 2003). Surrogate taxa can be classified into three broad catergories: flagship, umbrella and indicator (Hess *et al*., 2006).

2.3.1 Flagship and umbrella species

Flagship species are mostly large and charismatic species, e.g. baobab (*Adansonia digitata*), faced by dwindling population size or endangered status due to habitat destruction and overexploitation; whereas umbrella species are taxa of ecological significance, e.g. camel thorn (*Acacia erioloba*), which are distributed over a wide variety of habitat types and large extents. Both surrogates serve for the protection of other more inconspicious species occurring alongside them in the habitats (Hess *et al*., 2006). Flagship species usually attract public attention for funding and support to set aside areas for biodiversity conservation, and thus '*serve as a flagship in a socio-political context'*; while umbrella species define the size and types of areas that should be conserved (Caro & O'Doherty, 1999). For example many medicinal and useful plants in South Africa with significant cultural and commercial value, such as marula, aloe and hoodia, have become umbrella species for the conservation of the floristic heritage of the South African flora.

2.3.2 Indicator species

Indicator species are defined as "*species or taxonomic groups whose diversity are associated with overall levels of biodiversity"*, which function as an effective umbrella for a wide variety of biota (Landres *et al*. (1988) cited by Hess *et al*., 2006; Bonn *et al*., 2002). Therefore suitable indicator taxa are identified to proactively locate and monitor biodiversity hotspots, track population changes of species whose distribution is linked to a specific surrogate, and to indicate the extent and impact of human activities (Caro & O'Doherty, 1999). Examples of good indicators for determining the degree of anthropogenic habitat degradation is the extent and percentage of weed and invader plant investation.

The use of indicator taxa has become a common approach to measure the diversity of life. Despite a general dispute about their effectiveness, many researchers have provided empirical evidence for the relationship between the richness of certain taxonomic groups and overall species richness in the analysed areas, especially at coarse grains and across large study areas such as bioregions (e.g. Williams & Gaston, 1994; Olsen & Dinerstein, 1998; Caro & O'Doherty, 1999; Hess *et al*., 2006). Studies by Hess *et al*. (2006) found that the variations in indicator effectiveness observed across different studies can be attributed to complex links between grain, extent, geographic location as well as the choice and combination of indicator taxa. In the context of their study, 'grain' is defined as the size of each observational unit within the study area (e.g. hexagons, grid cells and ecoregions). Therefore they conclude that the use of indicator taxa as a conservation tool is only viable if spatial biodiversity patterns coincide across taxa.

According to Duelli & Obrist (2003) the choice of indicators depends on the facet of biodiversity to be evaluated, and is guided by a value system based on various motivations. Common motivations for the measurement and monitoring of biodiversity include amongst others: (1) conservation of biodiversity indicated by rare and threatened species; (2) protection of ecosystem function and resilience based on overall species richness; (3) conservation of cultural heritage indicated by the number of medicinal, useful and cultural species; and (4) biological control specified by the number of problem species.

For each of these value systems, biodiversity is proposed to be indexed by a variety of approaches using several concordant indicators (Duelli & Obrist, 2003). Furthermore the authors highlight that indicators should be ideally a linear correlate to the components of biodiversity under assessment. This means that the chosen indicators need to form a measureable proportion and appropriately represent a target aspect of the biodiversity under consideration.

For example, the conservation value of an area is generally evaluated by the presence of threatened and endemic species. They are good indicators of vulnerability as they are faced by an immediate risk of extinction due to their restricted range. Therefore rarity serves above all as a guide to those species most in need of protection while also providing a means of selecting areas that are biologically important otherwise (Kershaw *et al*., 1995). Rare species are also considered to contribute more to the uniqueness of a habitat than common species (Duelli & Obrist, 2003). Hence, conservationists use them as flagship species for the attraction of public support and funding for biodiversity conservation projects. Although endemic and threatened species are important conservation targets in their own right, reserve networks soley based on them do not guarantee to protection of all biodiversity (Bonn *et al*., 2002). Like Kershaw *et al*. (1995) the authors observed that this approach does not lead to a complementary reserve selection supposed to conserve the full range of habitats with its diverse vegetation types and species assemblages. The value of ecosystem function and resilience is largely determined by the more abundant and widespread species, which have a greater ecological significance than rare species (Millennium Ecosystem Assessment, 2005). As a result, the designation of reserve networks needs to be based on a wide variety of indicator taxa in order to achieve long-term maintenance of biodiversity and sustainable ecosystems. Based on this evidence, the assessment of the conservation importance of the floristic diversity in the western Central Bushveld Bioregion has been based on a range of indicator taxa.

Another question is whether biodiversity indicators are used to quantify aspects of biological diversity itself or as bioindicators for environmental and ecological health of our ecosystems. Depending on the context of research, any one level in the hierarchy of biological organization can give valuable information on biodiversity (May, 1995). The three main attributes of biodiversity as distinguished by Noss (1990) are compositional, structural and functional diversity (Duelli & Obrist, 2003). But the latter two are more difficult to quantify, and mostly are reflected by compositional diversity, such as species richness, which is thus used as a popular and convenient quantifiable measure for species diversity and ecosystem health (Millennium Biodiversity Assessment, 2005). The number of species in an ecosystem contributes to its structure and function by adding trophic levels, ecological niches and functional types (Duelli & Obrist, 2003).

Environmental variables have also been applied to model the occurrence and distribution of species (Ferrier, 2002). A positive correlation between species diversity, habitat heterogeneity and ecosystem function has been proposed by several authors (Cowling *et al.*, 1997; Pausas & Austin, 2001; Pausas *et al*., 2003; Millennium Ecosystem Assessment, 2005; Robertson & Barker, 2006). Dufour *et al*. (2006) believes biodiversity to be influenced by two aspects of spatial heterogeneity: 1) the variability of environmental conditions that affects the habitat types, and 2) the spatial configuration of habitats that affects ecological processes and thus ecosystem function. Therefore, the observation of spatial differences in species occurrences can be based on variations in contemporary ecological characteristics of the observed environments, such as climate, soil, geology and topography (O'Brien, 1993; Pausas *et al*., 2003; Dufour *et al*., 2006). Thus, ecological indicators are widely used to predict the patterns of occurrence of plant and animal species. The present study makes use of environmental factors to explain the floristic patterns observed in the western Central Bushveld.

The quantification of plant species richness is one of the most widespread measures to estimate the level of biodiversity in a given area, as patterns of vascular plant diversity are well documented and among the better-understood (Robertson & Barker, 2006; Dengler, 2009). It involves the measurement of conspicuous and easily quantifiable plant taxa on the level of species, genera, families and plant functional types. Studies of Sætersdal *et al*. (2003) indicate that they serve as reliable surrogates in complementary site selection for the conservation of well correlated faunal groups; particularly vascular plants are a well suited surrogate taxon for invertebrates, fungi, lichens and bryophytes. In the present study, preliminary vegetation studies in the Impala Bafokeng Mining Complex informed investigations of birds and small mammals in the study area. Attributes of vascular plants that make them reliable proxy measures for other taxonomic groups consist of the following: they are generally well described by taxonomists, easily identified in the field, and usually well sampled across the landscape (Sætersdal *et al*., 2003).

There is increasing evidence that numbers of higher taxa (e.g. genera, families) can be used as reliable surrogates for a quick estimation of the overall biodiversity of an area (Gaston, 2000). Several studies have assessed a positive relationship between the number of species and the numbers of higher taxa, e.g. Williams & Gaston (1994) and Balmford (1996a). Furthermore, Balmford et al. (1996b) showed that the use of the higher-taxon approach is a valuable technique for improving the cost and time effectiveness of field surveys. The rationale behind that is obvious: firstly because increasingly fewer taxa need to be counted for higher taxonomical ranks, and secondly because it is easier to distinguish higher taxa than their constituent species. But the benefits are only conveyed if the patterns of species richness correlate well with those of higher taxa; correlations are known to become weaker towards more inclusive levels of taxonomic hierarchy (Gaston, 2000).

However, taxon-based quantification of biodiversity does not capture other important key characteristics of biodiversity such as the variation in species abundance, distribution and ecological function (Millennium Ecosystem Assessment, 2005). Consequently the use of a range of biodiversity indicators is recommended to represent the most important dimensions of biodiversity.

2.4 Biodiversity – threats, values & benefits

2.4.1 Importance and value of biodiversity

The components of biodiversity form the all-important life-support system by supplying a vast array of ecosystem services and natural resources upon which all life depend (Millennium Ecosystem Assessment, 2005). Humans are an integral part of this web of life. "*It is the combination of all life forms and their interactions with each other and with the rest of the environment that has made Earth a uniquely habitable place for humans"* (Secretariat of the Convention on Biological Diversity, 2000).

Biodiversity, ecosystem functioning and human well-being are closely linked: 1) biological diversity is a factor modifying ecosystem processes and services which influences human livelihood and well-being; and 2) biodiversity plays an important role in the maintenance of these ecosystem functions which provide provisioning (e.g. habitat, food, fresh water, clean air and other natural resources), supporting (e.g. nutrient cycling, primary production and soil formation), regulating (e.g. climate, floods and wastes) and cultural (e.g. recreation, education and spiritual needs) services that are essential for human survival and prosperity (Millennium Ecosystem Assessment, 2005).

The relationship between biological diversity and these supporting ecosystem services is based on the multi-dimensional properties of biodiversity—the interplay between genetic, functional and trophic diversity of species, their composition, relative abundance and distribution over habitats and ecosystems (Millennium Ecosystem Assessment, 2005). This variability gives species, its populations and communities, as well as the ecosystems in which they occur, stability, resilience and the ability to adapt to environmental changes, and thus ensure long-term sustainability of ecosystem processes and services.

Natural resources provide the raw material for a wide variety of today's industry: agriculture, horticulture, pharmaceuticals, cosmetics, food and beverage (Secretariat of the Convention on Biological Diversity, 2000). Therefore, the biodiversity and ecosystem services delivered by species and their interactions secure our sources of food, timber and other building materials, medicines and energy, as well as our opportunities for recreation and tourism. The health of human society and economy depends on a continuous supply of natural resources and ecological services, which cannot be replaced artificially. Darkoh (2003) points out that especially in African countries, millions of people are directly or indirectly dependent on biological resources, mainly for meeting their basic subsistence needs, but also for commercial use to support their national economies. Most of the natural resources we depend on are derived from plants; thus, the conservation of our floristic heritage should be a focus area.

Biodiversity holds ample potential for development and the improvement of human well-being (McGinley, 2007). Consequently, it is of utmost importance to conserve our biological and ecological resources by using them in a sustainable way. This is especially true for the southern African savanna where the indigenous vegetation and wildlife are critical for securing a sustainable future; environmentally, economically and socially (Eriksen & Watson, 2009).

On the other hand, biodiversity is a response variable affected by drivers of global change, such as climate, biogeochemical cycling, land-use and introduction of species (Millennium Ecosystem Assessment, 2005). Today human activities are one of the strongest drivers of global change that are the root cause of the unprecedented biodiversity loss in our history.

2.4.2 Major threats to biodiversity in South Africa

The biodiversity we encounter around us today is "*the fruit of billions of years of evolution, shaped by natural processes, and increasingly by the influence of humans"* (Secretariat of the Convention on Biological Diversity, 2000). Human activities have drastically altered natural habitats since the advent of the industrial revolution in the 18th century with major advancements in agriculture, resource extraction, manufacturing and technology. These changes have led to the rapid decline of biodiversity.

Data presented by Heywood & Watson (1995) in the Global Biodiversity Assessment provides evidence that the current extinction rates of plants and animals are about 50–100 times faster than the natural background extinction (Edwards & Abivardi, 1998). Compared to the global standard, extinction rates and the number of threatened species are high in South Africa (South Africa, 1999). The global IUCN Red List contains 12,151 plant species, of which 114 are extinct or extinct in the wild and 8,500 are threatened with extinction (IUCN, 2009). These figures include 58 plant species that has been recorded as extinct or extinct in the wild from the Flora of Southern Africa (Protea Atlas Project, 2010). For example the plant species *Macledium pretoriense* has become extinct from the bushveld in Gauteng as a result of mining and urban development. Moreover, from the 3,435 plant species on the South African Red Data List, 2,084 have been assessed as being threatened to become extinct (Protea Atlas Project, 2010). In other words, 10% of the Flora of Southern Africa (FSA) faces an immediate threat to become extinct. The picture is worsened by statistical proof of drastic upsurges in the number of threatened plants in South Africa in the last two decades of the 20th century: threatened plant increased by 45% in the time between 1980 and 1984, and showed a overall rise of 80% until 1995; hence, South Africa is reckoned to have the highest known concentration of threatened plants and highest extinction count in the world (Wynberg, 2002).

Darkoh (2003) argues that the unprecedented loss of biological diversity in Africa can be attributed to habitat alteration and degradation. Habitat transformation has already affected 16.5% of South Africa's land cover, and some 10.1% are severely degraded (Wynberg, 2002). These figures include about 10% of the Savanna biome.

Because South Africa is one of the countries with the fastest growing populations in the world, it faces challenges to provide an increasing number of people with basic needs, such as food, water, housing, sanitation, waste management, health care and transport (UNEP, 2002).

But this necessitates an increase in food production and natural resource extraction, as well as the expansion of urban areas and associated infrastructure networks, which puts an increasing pressure on our ecosystems and its biodiversity. Therefore the transformation of natural vegetation to other land-uses and over-harvesting of species represents the most important threat to biodiversity in South Africa (Wessels *et al*., 2003). This includes the conversion of natural habitats to farmland for crop and livestock production, timber plantations, human settlements, mining and industrial development. Many plant species are threatened by over-collection for medical, ornamental and horticultural purposes (South Africa, 1999).

2.4.2.1 Agriculture

A large part of South Africa's land area, 86%, is used for crop cultivation and livestock grazing, and only about 10% of the farmland is involved in some sort of conservation activities (South Africa, 1999). Crop and livestock production thus belongs to one of the main drivers of environmental change in South Africa. It has especially contributed to the degradation of vegetation and soil in the past, where rapid population growth and inappropriate government policies have encouraged cultivation in marginal areas and the use of poor agricultural practices in order to increase food production; these past land tenure patterns also forced a large number of people off their land into what was called 'homelands' in order to increase suitable land for the intensification of agriculture (South Africa, 1999; Meadows & Hoffmann, 2002).

The Savanna Biome in southern Africa provides a rich resource-base for a large and growing human population, home to nine million rural and five million urban residents; the growing development pressure has been greatly expanded agriculture and livestock production, which covers even the most fragile Savanna regions (Watson & Dlamini, 2000; Woods & Watson, 2005). Particulary for the savanna woodlands under communal land tenure a substantial loss of tree cover has been documented in many studies (Meadows & Hoffmann, 2002; Clover & Eriksen, 2009; South Africa, 2011; Wessels *et al*., 2011).

Rural communities clear savanna woodlands to create plots for cultivation and grazing lands for livestock, as well as to use the timber for dwellings, fence construction and as fuel wood. Additionally, annual veld burning in the dry season is typically used to decrease the woody layer to improve the veld for livestock grazing (Watson & Dlamini, 2000). But the over-grazing and over-harvesting of useful plant species have greatly contributed to degradation of veld condition: Watson & Dlamini (2000) report that many preferred fuel, construction, craft and medicinal species have become locally extinct already.

In contrast to this, savanna woodlands where cattle are commercially farmed on privately owned land are characterized by extensive bush-encroachment (Hudak, 1999; Watson & Dlamini, 2000). This can largely be attributed to overgrazing, which on the one side reduces grass species that represent a good fuel load for the high intensity fires needed to kill woody invaders, and on the other side results in the dessication and crusting of soil that gives opportunistic woody seedlings a competitive advantage (Watson & Dlamini, 2000). Furthermore, the absence of browsing megaherbivores as important landscape architects in savanna woodlands and low wood harvesting by the ranch owners therefore led to widespread severe bushencroachment accompanied with the loss of native vegetation (Watson & Dlamini, 2000; Palmer & Ainslie, 2002).

2.4.2.2 Urbanization

Urbanization, together with agriculture, is the most important threat to biodiversity around the world, and increasingly so in developing countries (Miller & Hobbs, 2002; Ricketts & Imhoff, 2003; Pauchard *et al*., 2006) including South Africa (Cilliers et al., 2004, 2008; Rebelo et al., 2011); not only because of the direct conversion of habitat, but also as a result of a series of indirect effects such as habitat fragmentation, pollution and waste generation. The footprint of urban areas goes far beyond the boundaries of cities, mainly because dense human settlements have large requirements for resources and ecosystem services. Additionally, urban sprawl and the spread of suburban and exurban development affect biodiversity even in remote areas (Miller & Hobbs, 2002).

Urbanization leads to the homogenization of vegetation, a process where previously distinct plant communities become progressively dominated by a small number of widespread species, mostly ruderal plant assemblages and introduced species (Miller & Hobbs, 2002; Millennium Biodiversity Assessment, 2005; Bigirimana *et al*., 2011). Several of those non-native generalists originate from urban areas, e.g. exotic, domestic and horticultural species that have spread into the wild; others are native pioneer species that become invaders in degraded and human-altered environments. Biotic homogenization is of great concern as it increases local biodiversity while decreasing the regional and global distinctiveness of biotas; furthermore, it obscures the loss of biodiversity (especially that of native species), since many surveys solely measure changes in absolute numbers of species (Olden *et al*., 2004; Millennium Biodiversity Assessment, 2005).

Ricketts & Imhoff (2003) found that species richness is positively correlated with urbanization, which was also confirmed for Africa by Balmford *et al*. (2001). For centuries people settled in areas with highly productive systems where climate and soils support a rich biodiversity. On the other hand, large protected areas are often set up on relative unproductive land, with the most valuable sites in terms of biodiversity located outside of reserves and targeted by developers; for that reason many of the world's biodiversity hotspots have higher than average human population and growth rates, and are rapidly urbanizing (Miller & Hobbs, 2002). Consequently, urban nature conservation and studying the influence of urban areas on the environment are focused by the discipline of urban ecology and has become an important part of biodiversity conservation.

2.4.2.3 Mining

There is a growing awareness that many hotspots of diversity and rarity overlap with hotspots of human development. This is also the case in the mining industry since biodiversity-rich areas are mostly located in areas with rich mineral deposits (Maze *et al*., 2005). The majority of mining activities result in irreversible loss of natural habitat across large areas which wipe out all inhabiting biological diversity. In cases where biodiversity is not directly affected by clearing of vegetation, the pollution of soils with heavy metals and by acidification leads to a loss in sensitive indigenous plant species, which are then replaced by the invasion of tolerant alien plants and weedy invaders.

Competing land-use needs between mining and conservation constitutes a challenge. Due to the development need in South Africa to be internationally competitive and to offer a growing population a sustainable future, the natural environment is often compromised. Mining approval requires detailed surveys of the affected biodiversity as well as proposals for mitigation and rehabilitation. However, impacts are usually assessed on species-level, and the landscape- and ecosystem-wide effects on biodiversity are often neglected. Moreover, the offsets put forward for compensation are seldom the same value as the biodiversity lost (Maze *et al*., 2005).

2.4.2.4 Alien, invader and weed plants

As opposed to indigenous plants which occur naturally in a region, alien plants are mostly exotic plants introduced from other countries (Kurzweg, 2008). Modern transportation methods have increased the mobility of humans and in this way have led to an intentional and unintentional introduction of alien species in South Africa (South Africa, 1999). Many thousand plant species have been introduced from other parts of the world as crop species, for timber and firewood, as garden ornamentals, for landscape restoration, and as barriers and hedge plants (Pimentel *et al*., 2001; Van Wilgen *et al*., 2001). A wide variety of those species have escaped and invaded natural environments to the detriment of native floras. Yet, not only exotic plant species, but also several indigenous plants have become invasive species in disturbed habitats in South Africa, referred to as weeds because they overgrow natural vegetation.

Alien plants are characteristically opportunistic pioneer species that aggressively spread to new habitats and thereby rapidly increasing their range, especially in disturbed areas where they have an ecological advantage over indigenous plants (Kurzweg, 2008). Being habitat generalists, they quickly dominate the natural vegetation by outcompeting and displacing native species. In this way alien plants contribute not only to the transformation and further degradation of habitats, but also to the extinction of rare and threatened species. Many native species are also endangered by hybridization with alien species (Pimentel *et al*., 2001).

Alien species has been recognized as constituting an ecological threat to managed and natural ecosystems. Van Wilgen *et al*. (2008) claim that alien species are known to erode natural capital and compromise ecosystem stability, which finally also threatens economic productivity. Alien plants cause major economic losses in agriculture and forestry worldwide (Pimentel *et al*., 2001). In South Africa, the invasion of weed species contribute to the degradation of natural veld, causing significant economic losses in livestock and crop production (Pimentel *et al*., 2001); as well as significant declines in plant diversity. Detrimental environmental and economic impacts result from genetic changes and loss of species diversity that negatively affect population and community dynamics of native species and thus disturb ecosystem processes and functioning of the Savanna Biome (Van Wilgen *et al*., 2001).

Alien plants put an extra burden on natural resources, especially water and nutrients, which are scarce in many areas of semi-arid South Africa. For instance the water hyacinth, *Eichornia crassipes*, is blocking waterways in South Africa and reducing already scarce water resources (Pimentel *et al*., 2001). It is estimated that alien plants use approximately 3.3 billion m^3 water per year, accounting for 6.7% of the water that could be additionally available as a resource in South Africa's rivers and dams (Van Wilgen *et al*., 2001; Wynberg, 2002).

The above discussed drivers of environmental change affect ecosystems mainly by altering their species diversity, which includes species composition, structure, interactions and the ecosystem processes in which they are involved. But the loss of diversity does not only mean species loss but also a decline in genetic, ecosystem and landscape diversity, which forms the basis for the life-supporting ecosystem services upon which humans depend.

2.4.3 Benefits from biodiversity conservation

In the past the ecosystem services provided by biodiversity were thought of as having little or no economic significance (Edwards & Abivardi, 1998). Freedman (1995) points out that as long as there were sufficient supplies of resources and services from natural systems, there was no need to consider their economic benefits, which led to their rapid degradation by economic activities (Edwards & Abivardi, 1998).

In the light of increasing environmental problems and associated biodiversity decline in the past decades, biodiversity and the services they deliver are now more and more recognized as an essential resource for human survival that needs to be preserved. Edwards & Abivardi (1998) stated that an important achievement in sustaining these conventionally non-valuated resources was to declare them as goods and services that can be quantified economically. The discipline concerned with the evaluation of the costs and benefits of biodiversity and its conservation is called ecological economics. From a utilitarian point of view the contribution of biological diversity to human well-being and sustainable development is being evaluated (Montgomery, 2002). This provides a framework to make choices for setting conservation priorities and finding alternative solutions to get the best benefits for a sustainable future of both humankind and nature.

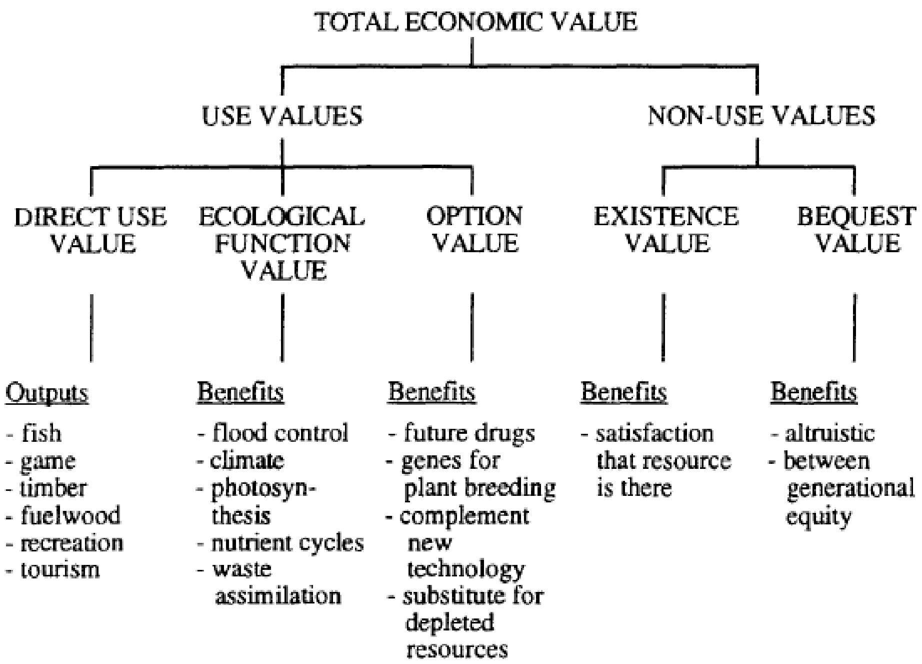

Figure 2.1: Classification of the major categories of values and benefits from biodiversity as presented by Young (1992) (quoted by Edwards & Abivardi, 1998).

People benefit from biodiversity in many ways, either directly by interacting with individual species, or indirectly by the contribution of each species to biodiversity (Montgomery, 2002). There is now a well established classification of the total economic value of biodiversity and the associated benefits as seen in figure 2.1.

The total economic value of biological diversity can be grouped into use and non-use value (Edwards & Abivardi, 1998). Direct use values are derived from biodiversity as a market valued commoditiy, which can be divided into direct extractive benefits such as the production of timber and food from plant species, and direct non-extractive benefits including tourism and recreation (Edwards & Abivardi, 1998; Montgomery, 2002). On the other hand the various ecosystem services biodiversity offer can be classified as indirect use values, for example primary production and climate regulation by vegetation; moreover, there may be many still unexplored benefits from plants and animals (e.g. medicinal and genetic resources), which are grouped under optional use values (Edwards & Abivardi, 1998).

In contrast, non-use values are more difficult to characterize, but they are broadly classified as bequest and existence value according to Edwards & Abivardi (1998): existence value refers to the intrinsic value of species, whereas bequest value is knowledge of species as being of future benefit.

Hence, by summing up all the benefits of biodiversity, it is clear that the conservation of biodiversity found in our natural habitats, their types and relative abundance of native species, are of crucial importance for the resistance and resilience to environmental change such as climate change, and an ensurance for a sustained delivery of ecosystem goods and services in future.

2.5 Historical overview of biodiversity conservation in South Africa

Biodiversity conservation is not an invention of modern times, but can be traced back to the pre-colonial and colonial history of South Africa. The environmental history in South Africa was greatly influenced by its social history (Nell, 2004). Indigenous and colonial African societies had different perceptions about the environment and made use of natural resources in different ways. Therefore the environmental history of South Africa is imprinted by variation in access and conflict over natural resources by the various ethnic groups and societal classes.

For thousands of years indigenous people in South Africa have managed their environment and the natural resources on which they depended on for survival (Burgess *et al*., 2004; Fuggle, 2008). Native communities have cultural values, beliefs and taboos that prevented their natural resource base to become depleted (Izidine *et al*., 2008a). Their respect for the environment was expressed in a number of ways, including proverbs, songs, folklore, myths, rituals, totems and agricultural practices (Makwaeba, 2004). Examples include the setting aside of hunting grounds for royal Zulu members, soil conservation methods of the BaTswana people and totemic animal and plant protection in many tribes (South Africa, 1997, Makwaeba, 2004, Fuggle, 2008). The native community's rules and procedures for resource management were usually controlled and enforced by their traditional leaders. For instance, community laws prohibited the consumption and destruction of highly endangered and ecologically important species which are used for traditional healing (Makwaeba, 2004; Izidine *et al*., 2008b).

Furthermore indigenous cultures protected their sacred lands, long before the establishment of officially protected areas by conservationists (Dudley *et al.*, 2005; Izidine *et al*., 2008b). These sacred sites are regarded by Dudley *et al.* (2005) as the oldest method of habitat protection which forms a large, still unresearched network of nature sanctuaries around the world. Therefore the traditional knowledge rooted in the cultural heritage of South African indigenous societies ensured biodiversity conservation in South Africa before the colonization by European settlers.

Nevertheless, the indigenous nomadic tribes inhabiting the South African interior prior to colonization also contributed to environmental transformation (Nell, 2004). The San and Khoi can be named as good examples. Traditionally the San people lived as hunter-gatherers. Deacon (1989) as cited by Nell (2004) provides historical evidence that the San have employed the burning of vegetation for thousands of years to encourage the establishment of edible plants. Similarly, the Khoi lived a nomadic pastoral lifestyle in which they used fire to support the growth of palatable fodder for their cattle.

With the advent of European settlers in South Africa in the 17^{th} and 18^{th} century, these traditional indigenous resource management systems changed, mainly as a result of the subjugation of native cultures by the colonists that undermined the delicate interactions between the native societies of southern Africa and their natural environment (Nell, 2004); especially as a result of the intensification of hunting activities and ranching of cattle, sheep and goats, as well as the displacement of indigenous populations and deforestation to make way for European agriculture (South Africa, 1997; Fuggle, 2008). Hunting took a sad toll on wildlife in South Africa, which led to the extinction of the Cape elephants and lions before the end of the 18^{th} century. As a consequence of this rapid decline in game, the governor Van der Stel was the first to declare endangered species of antelopes as protected game (Hugo *et al*., 1997).

Nature conservation in South Africa originated as a response to rapid environmental decline and diminishing resources during the 19^{th} century colonization of South Africa. A series of severe droughts in the late 1870s, accompanied by a shortage in land, cattle disease and crop failures, raised concerns about long-term consequences of environmental exploitation in South Africa (Nell, 2004). In search for solutions to prevent further environmental degradation and to manage natural resources, colonial environmental science emerged, first at the Cape colony by establishing posts for botanists, foresters, hydrologists and veterinarians, later augmented by university departments and state-funded research stations employing permanent research staff (Hugo *et al*., 1997).

From the 1860s, several farmers set aside areas of private land to protect wildlife from overexploitation by hunting (Hugo *et al*., 1997). But the first official protected areas in South

Africa were established soon after. The Knysna and Tsitsikamma forest reserves were declared under the Cape Forest Act of 1888, South Africa's first legislation to protect and manage natural resources (South Africa, 1997; Fuggle, 2008). This was followed by the creation of Game Reserves in order to preserve South Africa's wild game: the Pongola (1894), the Hluhluwe, Umfolozi and St. Lucia (1895) and the Sabie (1898) Game Reserves; the latter, which is now incorporated in and known as the Kruger National Park (Hugo *et al*., 1997; Fuggle, 2008). From 1937, the number of National Parks has increased from four to about 17 which raised the percentage of area set aside for conservation to 9% (Hugo *et al*., 1997).

After the formation of the Union of South Africa in 1910, the new government assumed conservation responsibility for land and water by the formulation of legislation to protect and manage various components of the environment. Examples of the several important milestones in the conservation history of South Africa include the promulgation of the following pieces of legislation: the National Parks Act (1926), Soil Act (1940), Water Act (1956) and Planning and Utilization of Resources Act (1967) (South Africa, 1997; Fuggle, 2008). However, according to Fuggle (2008), these independently formulated acts failed in providing an integrated and holistic framework for the management of environmental resources in South Africa; the early environmental legislation was characterized by weak enforcement, partly also due to the lack of public participation.

Some interventions of the new government for the management of natural resources have also resulted in further environmental decline. Nell (2004) reports that state policy in South Africa favoured certain forms of land use over others. For example, agricultural practices such as communal land use and seasonal burning of grass cover was abandoned, while farming was allowed to develop in marginal areas. Thus, '*many historians believe that such state interventions pushed agriculture in South Africa beyond its ecological limits*' (Nell, 2004).

In addition, the 20th century conservation history of South Africa was influenced by the ideological framework of colonialism and apartheid (Fabricius *et al*., 2001; Wynberg, 2002). The conservation approach was characterized by a wildlife-centred, preservationist point of view which holds that nature can only be conserved by fencing off pristine areas from human influence (Khan, 2000; Wynberg, 2002). Thus, the establishment of protected areas was accompanied by re-location and forced removal of local black people, as well as the exclusion of native communities from access to the natural resources within the reserve (Fabricius *et al*., 2001; Wynberg, 2002). Moreover, the segregationist policies of the apartheid regime forced black South Africans to live in designated ‘homelands’. Because ‘homelands’ were mostly marginal areas with scarce natural resources, overexploitation led to environmental degradation in these areas (Nell, 2004). As a result, conservation was viewed with suspicion and mistrust, and received little support.

In the 1970s, there was a global rise in environmental concern due to widespread loss of habitats and biodiversity; the rise in awareness of the consequences of land use change on the environment led to the establishment of the ‘Cabinet Committee on Environmental Conservation’ in 1972, followed by the formation of the ‘Habitat Council’ and the ‘Environmental Planning Professions Interdisciplinary Committee’ representing the civil society (Fuggle, 2008). This gave way for interactive communication between the national government and the civil society in environmental matters, resulting in the declaration of the ‘Environmental Planning Act of 1975’, with the inauguration of Department of Environmental Affairs short after (Hugo *et al.*, 1997). The next important step was the publication of the ‘White Paper on a National Policy for Environmental Conservation’ by the government in 1980 for public review, which then led to the promulgation of the ‘Environmental Conservation Act of 1982’, which represents a significant landmark in the conservation history (Sowman *et al.*, 1995; Fuggle, 2008). It mandated civil society to participate in the development of environmental policies and legislation, and to comment on the compulsory ‘Environmental Impact Reports’ for listed activities (Sowman *et al.*, 1995). Additionally, the act gave way for the appointment of environmental experts to form the ‘Statutory Council for the Environment’ in 1983 (Fuggle, 2008).

In 1978 the concept of conservancies was developed, which is a combined effort of private, communal and business people to integrate land use into regional and local conservation campaigns (Hugo *et al.*, 1997). The co-operative interaction between government, environmental Councils, Committees, NGOs and experts during the 1970s and 1980s gave rise to the amendment of the ‘Environmental Conservation Act’ in 1989. This entailed the introduction of an ‘Integrated Environmental Management’ framework for South Africa in order to link environmental management with land use and physical planning. One important implication of this newly published framework was a shift from preservationist concept in nature conservation and establishment of protected areas to a more integrated and holistic approach, which accommodates the needs and aspirations of society, economy and local communities. As a result the protected area system in South Africa developed into various types of in-situ and ex-situ protected areas, which are managed for a variety of purposes (Hugo *et al*., 1997). One example is the 'National Heritage Programme' founded in 1996 that allows the public to participate in national conservation efforts. According to Hugo *et al*. (1997) landowners can register their land as heritage sites when it has intrinsic natural value and therefore worth being protected and conserved. Another example is the biosphere reserve concept which conceptualizes reserves as consisting of a centre of high conservation level with surrounding zones of decreasing conservation level (Hugo *et al*., 1997).

Democratic political changes in 1994 and South Africa's ratification of the Convention of Biological Diversity in 1995 brought new urgency to enforce a coherent and integrated policy on biodiversity (Wynberg, 2002). As a result, the 'White Paper on Biodiversity' was published in 1997 in order to prepare the ground for the 'Biodiversity Act of 2004'. This specific legislation dealing with biodiversity finally gave the basic legal framework to enforce biodiversity conservation in South Africa.

South Africa has made remarkable strides towards the management and conservation of its rich diversity of species and habitats. The well-developed nature conservation practices and protected area system have gained South Africa global recognition. Today, healthy and diverse ecosystems are regarded as important assets in South Africa that form the basis for economic development and improvement of the quality of life.

2.6 Conservation at the level of Bioregions

2.6.1 What is a Bioregion?

Plant and animal life in the biosphere of the earth occurs in the form of intertwined networks of individuals, populations, communities and interacting ecosystems (Udvardy, 1975). For the purpose of viewing and studying biota, they have been systematically grouped according to their taxonomic, phylogenetic and ecological order. Because biota and ecosystems display spatial characteristics, they can be further grouped into biogeographical order. As a result, terrestrial landscapes can be classified into biogeographical regions on various scales of resolution and by using different ecosystem components (e.g. flora, fauna, and terrestrial and aquatic ecosystems). Biogeographic units are defined by similarities and differences in the occurrence of species, higher taxa and ecosystem units (Udvardy, 1975). The classification of biogeographical regions is a hierarchical approach that involves the delineation of ecological units with increasing use of detail at each higher level of classification (South Africa, 2005a).

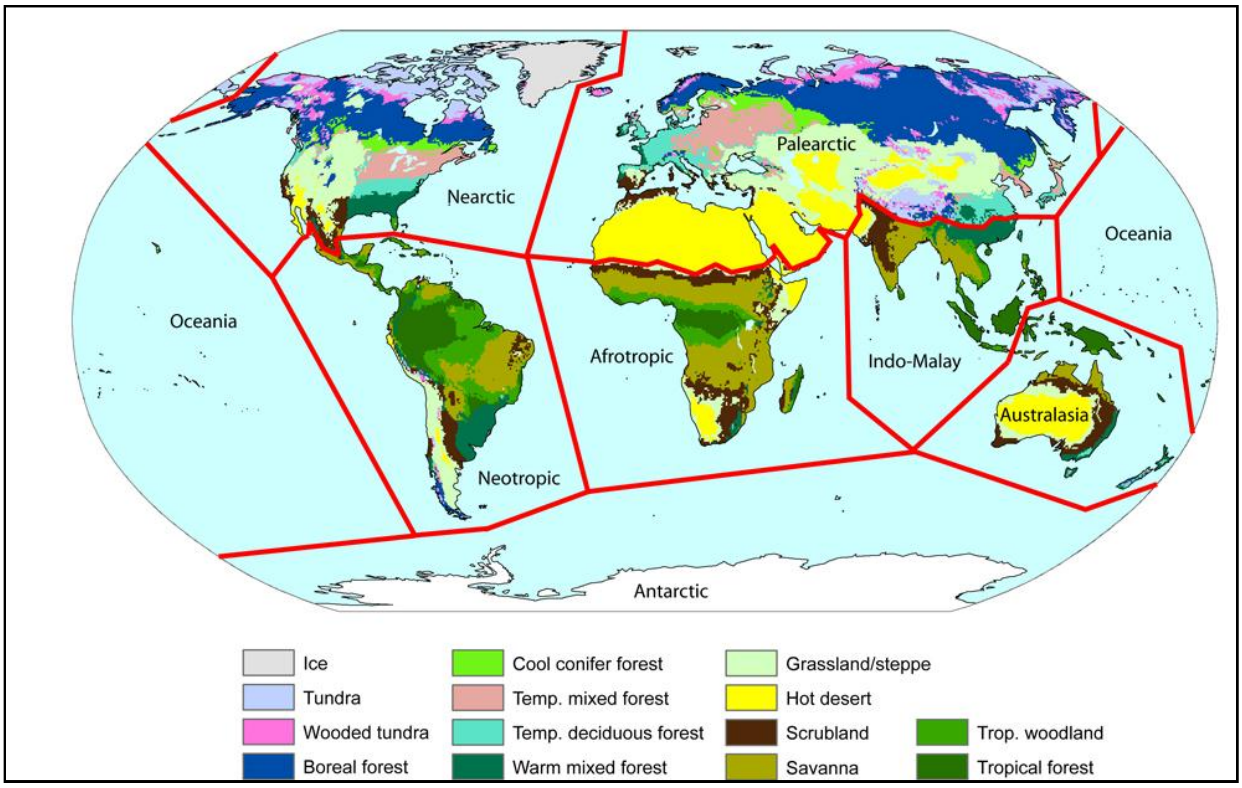

Figure 2.2: The Flora of Southern Africa falls within the Afrotropic Realm as defined by Olsen *et al*. (2001), who classified the world into nine Biogeographic Realms subdivided into 14 terrestrial biomes. Source: Van Vuuren *et al*. (2006).

One of the largest units of division is the ecozone which can be delineated on the basis of the historic and evolutionary distribution patterns of plants and animals. There are five major events in the Earth's history which contributed to the biogeographical history of continents and species, namely 1) plate tectonics and 2) volcanic activities, 3) the rise of mountain ranges, 4) climate change, and 5) sea level fluctuations (Lomolino *et al*., 2006). Plate tectonics is considered as one of the main drivers in shaping the present distribution patterns of life on earth. It contributed to the redistribution of continents and the rise of mountain ranges, which formed barriers for the migration of plants and animals. Therefore plants and animals in the nine terrestrial ecozones of the world developed in isolation over long periods of time, separated by natural boundaries. As they roughly correspond with the floral and zoogeographic kingdoms described by botanists and zoologists respectively, floral and faunal distribution maps were used by Olsen *et al.* (2001) to delineate ecozones. For example the flora of southern Africa falls within the Afrotropic ecozone or biogeographic realm which constitutes the sub-Saharan Africa and Madagascar (figure 2.2).

As introduced above the largest unit of phytogeographical division is the floristic kingdom, which represents the highest rank in the hierarchical classification of phytochoria based on Takhtajan (1986). Van Wyk & Smith (2001) define a phytochorion as a phytogeographic region of any rank with smaller areas nested within successively larger areas—District, Subprovince, Province, Region, Subkingdom and Kingdom.

This classification is established on the assumption that botanical diversity is not evenly distributed, as each plant taxon has its own definite geographical range. In this respect the southern African flora is grouped into the Palaeotropical kingdom together with Madagascar and tropical regions of Asia and Oceania (figure 2.3).

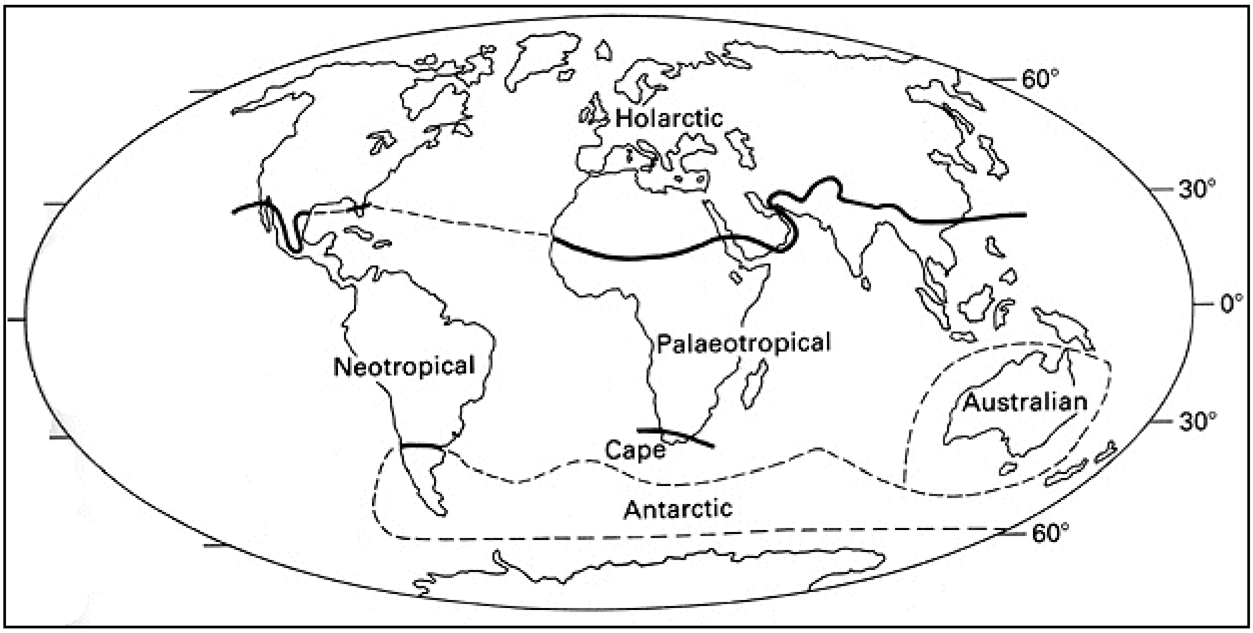

Figure 2.3: Floristic kingdoms after Takhtajan (1986). Source: rbg-web2.rbge.org.uk/nepal/biogeography.html.

The six principal floristic kingdoms of the world can be subdivided into floristic regions based on recurrent patterns in the geographical range of plant taxa (Van Wyk & Smith, 2001). The hierarchical classification of Takhtajan (1986) divides the African subkingdom into five floristic regions or realms, the largest of which is the 'Sudano-Zambesian' phytogeographic region. Floristic uniqueness of a region is largely determined by its floristic elements, which are groups of taxa with similar distributional ranges (Van Wyk & Smith, 2001). According to Van Wyk & Smith (2001) '*endemic taxa represent the geographical element which most naturally characterises the floristic uniqueness of a particular region*'.

On the other hand White (1983) recognizes six non-hierarchical types of phytochoria, five of which are subdivided at the regional level (figure 2.4). The ‘Regional Centres of Endemism’ and ‘Regional Transition Zones’ are the most dominant regional phytochoria found in southern Africa. For instance most part of the western Central Bushveld Bioregion falls within one of White’s ‘Regional Centres of Endemism’, which he defines as a phytochorion with more than 50% of its species confined to it, and a total of more than 1,000 endemic species (Van Wyk & Smith, 2001). White’s non-hierarchical system of phytochoria is the most applicable and widely used floristic classification in Africa. There is a high degree of correspondence between South African biomes and White’s (1983) phytochoria (Rutherford *et al*., 2006). For example the ‘Zambesian’ and ‘Kalahari-Highveld Regional Centres of Endemism’ shows a 33% and 43% correspondence with the Savanna Biome respectively.

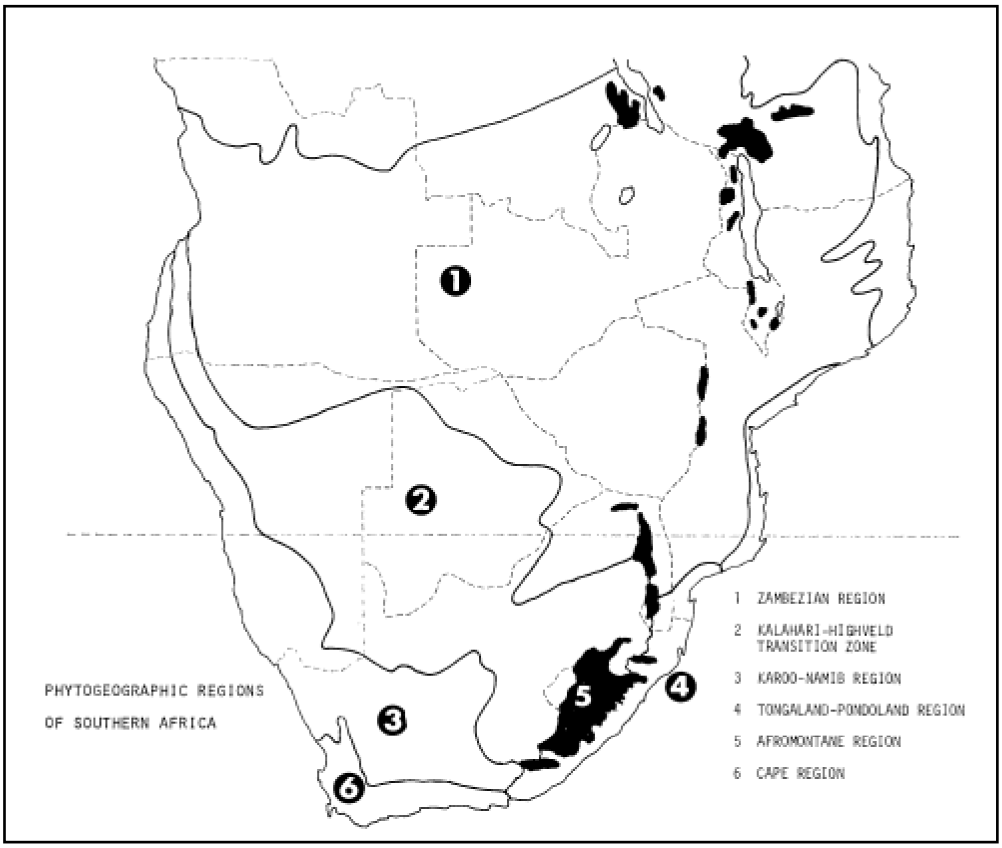

Figure 2.4: Phytogeographic regions of southern Africa based on White (1978). Source: Goldblatt (1978).

Furthermore the world’s land territory can be divided into biomes. There are different approaches to global biome classification. Olsen *et al*. (2001) defined 14 broad categories for global biome distribution (figure 2.2). In South Africa, the ‘Savanna’ biome falls within the ‘*Acacia* Savanna Woodland’ sub-biome within the ‘Tropical and Subtropical Grasslands, Savanna, Shrublands, and Woodland’ biome as defined by Olsen *et al*.’s (2001). There is a fair correspondence between the 'Savanna' biome of Mucina & Rutherford (2006) and Olsen *et al*. (2001)'s broad global biomes—they coincides to 40.2% (Rutherford *et al*., 2006).

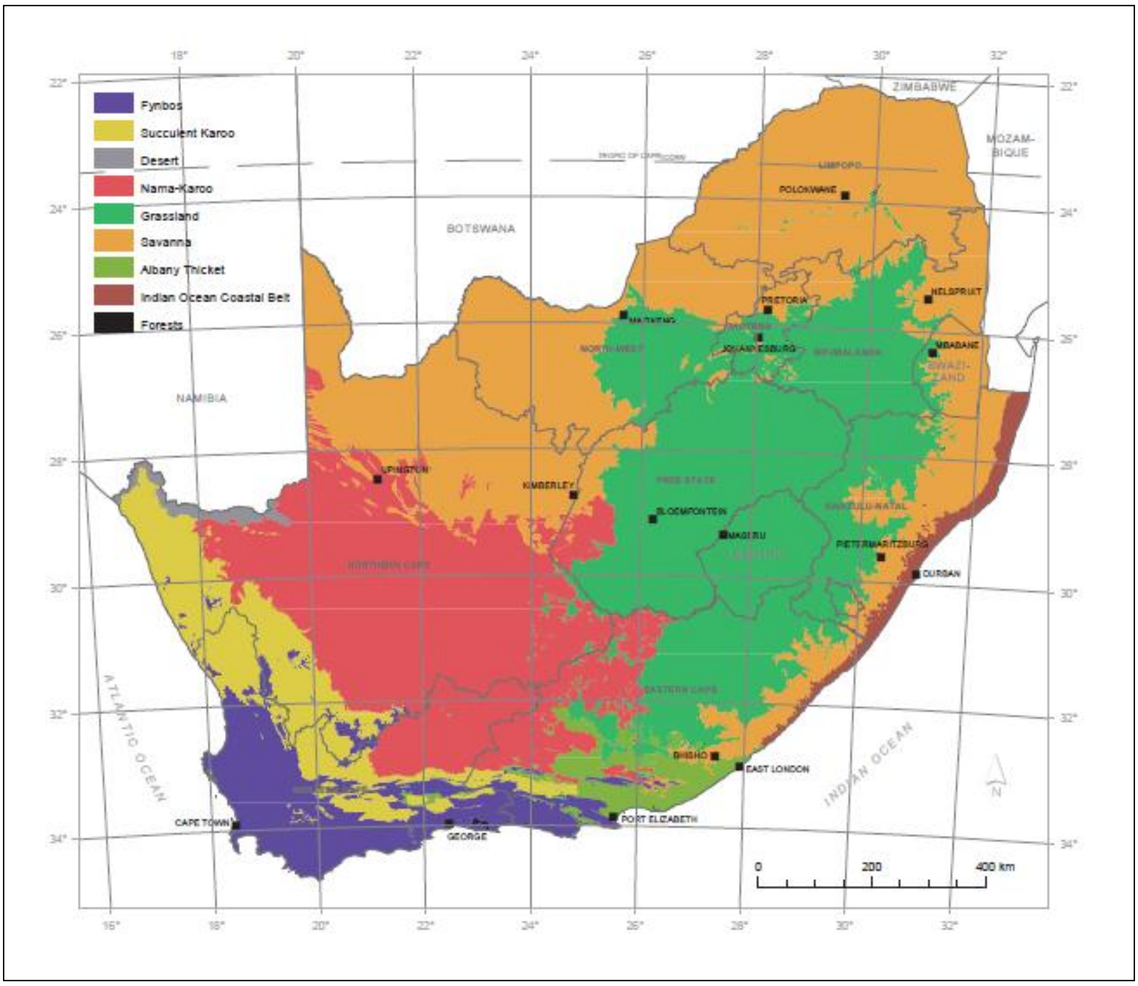

Figure 2.5: Biomes of South Africa as defined by Mucina and Rutherford (2006). Source: www.plantzafrica.com.

Biomes represent the major habitat types found across a continent, classified according to the distribution of dominant life forms. 'Each biome has a characteristic set of plants and animals, as well as a characteristic overall physiognomy' determined by the general outward appearance of the dominant plant life forms (Low & Rebelo, 1996; Tainton, 1999). These distinct biotic communities found in each biome evolved mainly as an adaptation to climatic and soil conditions. Biomes largely coincide with the prevailing climatic conditions. Thus habitat types sharing vegetation of similar type and structure, but with different evolutionary histories, may be found on different continents. For example the savanna biome exists not only in Africa, but also in South America, Australia and India (Pidwirny, 2006). In South Africa seven biomes were recognized by Rutherford & Westfall (1994), with the Desert, Thicket and Indian Ocean Coastal Belt Biomes additionally biomes described by Low & Rebelo (1996), Rutherford (1997) and Mucina & Rutherford (2006) respectively (figure 2.5). The global, broad-scale mapping units of biodiversity can be further divided into regional units, namely bioregions and ecoregions. They are classified on the basis of biological patterns and ecological characteristics in the landscape. However, in the literature there is a lot of confusion and inconsistency regarding the use of the aforementioned two terms. On the one side they are used for various levels of ecosystem or biotic organisation, and on the other side the two terms are often used interchangeable.

Bioregions used in the context of this study as described by Rutherford *et al*. (2006) are defined as spatial terrestrial units on an intermediate level of vegetation organisation between that of vegetation type and biome. For example, the Savanna Biome in South Africa is divided into six bioregions, namely the 'Central Bushveld', 'Mopane', 'Lowveld', 'Sub-Escarpment Savanna', 'Eastern Kalahari Bushveld' and 'Kalahari Duneveld'; each consisting of a characteristic and unique set of vegetation types.

Other biologists have used the term bioregion for biogeographic divisions at various regional scales. The nine bioregions referred to by Rowe-Rowe & Tailor (1996) for KwaZulu-Natal in South Africa are based on the bioclimatic regions of Phillips (1973) and vegetation types of Acocks (1975) as cited by Rutherford *et al*., (2006), and thus display a higher resolution than Mucina & Rutherford's (2006) bioregions for the province.

In contrast, the bioregions of Burgess *et al.* (2004) are used at a hierarchical level higher than that of the South African biomes (Rutherford *et al*., 2006). Ecoregions, as the smallest biogeographical rank, are further grouped into bioregions, and then into realms. For example the 'Cape Floristic Region' is considered as a bioregion, separate from the remaining Afrotropic realm which is grouped as the 'Eastern and Southern Africa' bioregion.

The term ecoregion was coined by J.M. Crowley in 1967 to describe mapped regions of ecosystems in the United States (Omernik, 1987), and from then widely and differently used by various authors including Dasmann (1973, 1974), Udvardy (1975), Omernik (1987, 2004), Olsen *et al*. (2001), Olsen & Dinerstein (2002) and Burgess *et al.* (2004).

Compared to Mucina & Rutherford (2006), Olsen *et al*. (2001) make use of the ecoregion concept for the classification of biogeographic units at the regional level. In cooperation with the World Wildlife Fund for Nature (WWF) and scientists from all over the world, 867 terrestrial ecoregions were identified nested within the eight terrestrial ecozones and 14 biomes mentioned above. This classification reflects floristic and faunal similarities across large areas of land. The 'Southern African Bushveld' ecoregion, for instance, is characterized by its charismatic large mammals, as well as its co-dominant tree and grass layer, which are associated with African savannas (Burgess *et al.*, 2004). There is a 78% correspondence between the 'Southern African Bushveld' ecoregion and the 'Central Bushveld' bioregion (Rutherford *et al*., 2006).

As a result of distinct biogeographic histories across different regions and continents, similar kinds of ecosystems emerged that 'support unique assemblages of species and higher taxa' (Olson & Dinerstein, 2002). Therefore bioregions and ecoregions can be defined as homogeneous biogeographic areas representing natural ecological communities with characteristic flora, fauna, environmental conditions and ecological dynamics. These distinct ecological areas are delineated by similarities in topography, geology, soils, climate, biota, vegetation and hydrology, and often also by human cultural aspects. According to Omernik (2004), there is a significant spatial correlation between these geographical attributes associated with differences in the quality, health and integrity of ecosystems. Thus natural boundaries can be broadly represented by topographic features such as mountains, rivers and oceans. However, there is no absolute spatial coincidence, as environmental conditions usually undergo a gradual change across the boundaries of ecological regions forming transitional zones, the so called ecotones. Bioregions and ecoregions usually span over several habitats and are associated with a mosaic of habitats that don't allow clear boundaries to be drawn; furthermore the boundaries are approximations of the original extent of natural communities prior to major land-use changes (Burgess *et al*., 2004, Olson *et al*., 2001).

Ecological regions can be classified for various purposes using different ecosystem components as stated in the beginning. There is no single biogeographic framework optimal for all taxa and ecosystem components (Olson *et al*., 2001; South Africa, 2005a). Thus the ecoregion classification approach by the World Wildlife Fund only represents a compromise for as many taxa as possible. The goal of this highly inclusive ecoregion classification is the conservation of the rich biodiversity of species and habitats.

Consequently ecological regions can be defined on various other levels emphasizing different terrestrial ecosystem characteristics, or even socio-economic and cultural aspects. For example, ecoregional classifications of river systems identify river ecoregions for the protection of water resources by delineating natural water catchment areas. The Department of Water Affairs and Forestry (South Africa, 2005a) has defined a total of 31 river ecoregions for South Africa, including the 'Bushveld Basin' and the 'Western Bankenveld' which occurs within the study area (see 3.1.2.1, chapter 3).

Brunckhorst (1995) states in his article *'Sustaining Nature and Society: The Bioregional Approach'*, that cultural bioregions should be the ultimate management units for sustainable societies. In this context bioregions can be defined as an area whose limits are not defined by political boundaries, but by the geographical limits of human communities and ecological systems (Brunckhorst, 1995; London, 2000). The consideration of economic and social aspects in a bioregional framework constitutes a novel approach in striving to sustain both, society and biodiversity. It has evolved as a response to failures in the planning and management of regional landscapes. Regional decision-making and management often fails in 'meeting socio-economic needs while conserving biophysical resources', despite sufficient knowledge about ecosystems and anthropogenic effects on them (Brunckhorst, 1995). Therefore bioregional planning "*allows the variously defined and tenured areas of land within a bioregion to be managed in a complementary way*", to ensure long-term sustainability of nature and society.

Bioregionalism considers bioregions as a product of culture-nature interaction (Alexander, 1996). Humans, their culture and political activities are integral parts of ecosystems and the dynamic interactions between its biotic and abiotic components. Therefore, bioregional land-use and conservation planning can be seen as a 'practical land ethic' that ensures that society takes on stewardship over regional natural, human and economic resources. As a consequence, the dynamic and interactive nature of bioregions and their ecosystem components is be conserved, which will give the bioregion the resilience and flexibility to adapt to natural and human induced changes (London, 2000).

2.6.2 The value of conservation at bioregional level

Bioregions are valuable units for the analysis of biodiversity patterns for the purpose of biodiversity conservation. Bioregional classification follows the ecosystem approach, which recognizes that ecosystem components do not function in isolation but rather exist in association with each other. Ecosystems and their biota exhibit regional patterns due to "*spatially variable combinations of causal factors such as climate, soils, geology,* *physiography and vegetation"* (South Africa, 2005a). These regional ecological units thus harbour distinct assemblages of species, ecological characteristics and relationships between organisms and their environments. Therefore conservation on bioregional level will protect the whole palette of its natural communities of species and habitats, as well as the ecosystem services they deliver.

The bioregion serves as framework for the assessment of biodiversity within "*whole landscape ecosystems and their processes"* (Fairbanks, 2000). Biogeographic analysis at the regional scale aims at the identification of areas that are most distinctive or have a high representation value. Biodiversity features that define the distinctiveness of an area include the richness, endemism and rarity of species, higher taxa and habitats, as well as unusual ecological or evolutionary phenomena (Olson *et al*., 2001). Representative areas can be defined either as areas of distinct biodiversity features, or areas that contain a representative collection of species, habitats and processes.

Additionally, the conservation value of an area is determined by the status and intactness of its natural habitats and species communities, the threats faced by its biodiversity, and the degree of protection. Thus the delineation of biogeographic units at the bioregional level can be seen as an excellent tool for the identification of representative areas and hotspots for conservation, which will ensure the persistence of special elements, populations, communities and ecological processes (Olson *et al*., 2001). This then serves as a spatial framework for the selection of priority areas to "*undertake targeted programs of conservation action across large areas of land*" (Burgess *et al*., 2004). Secondly, the bioregional framework can be used to define zones for sustainable ecological management, as the quantity and quality of environmental resources in bioregions also exhibit distinct spatial patterns (Fairbanks, 2000). Conservation prioritization within bioregions improves our efforts to conserve the full representation of a region's biodiversity by making the most efficient use of available funding, land and resources within a considerable time frame.

Additionally, bioregional biodiversity units overcome the shortcomings of global biodiversity maps, which have been ineffective conservation planning tools due to their coarse scale biodiversity classification (Olson *et al*., 2001). Biodiversity assessments at larger scales fail to look at the small and highly distinctive areas, which thus receive insufficient conservation attention. In contrast to that, bioregions reflect a more detailed resolution of biogeographical classification at the level of living landscapes. Because bioregions consist of an interactive assemblage of species, habitats and ecological processes, they are preferred as conservation units over countries. When conservation activities are focused on political units, then there is a risk that important natural regions that extend beyond national borders are overlooked (Burgess *et al*., 2004). Therefore the conservation at bioregional level supports the system of large transboundary conservation efforts and protected area networks which contribute to long-term preservation of ecosystem and its biota.

Burgess *et al*. (2004) holds that maps of biological units of land are of fundamental importance to conservation planning by giving an understanding of what types of conservation interventions will work best in a particular area. For example, the fire-climax savanna woodlands in southern Africa are known to be more tolerant of various kinds of disturbances, because their species are adapted to different habitat patch dynamics over vast areas of land. Savanna ecology is determined by a transition between ‘multiple stable states’ as a result of different intensities of burning and animal browsing (Sharam, 2002). Fire-climax woodlands are characterized by trees with thick, corky, fire-resistant trees and herbs that avoid fire damage either by growing below the level of grass fires or by being fire-resistant (Sharam, 2002; Allaby, 2004). However, repeated and high intensity fires and animal browsing will result in transition of the woodland into the grassland state by reducing the tree layer, which allows grasses to dominate. But the fire-climax woodland will recover over time as tree seedlings establish and mature under moderate fire and browsing pressure. Thus bushveld in the African savanna is best managed by conserving its biota and ecological dynamics in protected areas.

But conservationists have realized that protected areas alone are not successfully conserving biodiversity (Margules & Pressey, 2002; Roe & Hollands, 2004; Cowling & Wilhelm-Rechmann, 2007). One of the major reasons is believed to be the expansion and intensification of land use in areas adjacent to protected areas (Muchapondwa *et al*., 2009). Protected areas are part of larger ecosystems. Therefore, the change in land use and land degradation outside protected areas result in the alteration of ecosystem processes, which over the long-term also affect protected areas and may cause biodiversity loss. Consequently, there is a need to manage regional landscapes to “*maintain the ecological integrity of the protected areas they contain*” as has been stated above (Muchapondwa *et al*., 2009). Thus the bioregional approach will help to meet the biodiversity targets in South Africa by promoting conservation inside and outside protected areas. But the degree of biodiversity conservation in a bioregion depends on the composition of the land use mosaic found in a particular bioregion. Factors affecting land use and biodiversity conservation in bioregions includes ecological, social, technological, economic and political factors (Muchapondwa *et al*., 2009). Thus land use decisions in a bioregion are largely determined by people's value systems and management choices. The resulting land use mosaic among others consists of urban and rural settlements, crop and livestock farming, natural resource harvesting, commercial industries such as mining, and nature reserves.

As a response to these competing land uses, biodiversity conservation in bioregions will only be successful if it is mainstreamed into society. In this regard '*the bioregional context provides an integrative setting*' for local governments to plan for sustainable land and resource use, to minimize or ameliorate impacts, and to monitor the ecosystem and biodiversity status (Bruckhorst, 1996). Additionally, community-based programs as well as public and private initiatives can form complementary efforts to minimize impacts in the bioregion. Therefore bioregional conservation planning integrates and maximises cultural, sectoral and environmental benefits from the region, and at the same time preserves biological diversity and ecological function inside and outside protected areas (Bruckhorst, 1996).

2.7 Approaches to biodiversity conservation planning and management

2.7.1 Introduction

Conservation planning is a rapidly evolving field of study whose stated goal is the preservation of biodiversity—that is: species, habitats and environmental processes. The establishment of protected areas is one of the main tools for achieving the conservation of biological life (Grantham *et al*., 2008). This involves traditionally the proclamation of officially protected areas on the basis of their conservation status as set out in the CBD. Networks of protected areas are created to preserve a representative complement of biodiversity, and are identified by the separation of priority areas from activities that degrade or destroy them (Knight *et al*., 2008).

But after a long history of *in-situ* conservation globally, as well as in South Africa, conservationists have realized that strict biodiversity conservation in formally protected areas will not secure the world's biological wealth (Miller & Hobbs, 2002; Pierce *et al*., 2005). Consequently, the responsibility of conserving biodiversity will fall increasingly on sectors such as agriculture, forestry, mining, as well as urban and land-use planning (Pierce *et al*., 2005). Biodiversity conservation has become the responsibility of society as a whole, but can only be achieved if the needs and desires of people are included in the equation, but in a sustainable way. The extreme preservationist approach of the past with the exclusion of people from access to their local biodiversity resources has often lead to illegal destructive use and overexploitation of natural capital (Kurzweg, 2008).

Consequently, biodiversity can only be preserved over the long term if the conservation of biodiversity in nature reserves is combined with conservation efforts across the landscape. There is an urgent need to mainstream biodiversity concerns into various sectors of society and into the policies and practices of economic companies to achieve measurable conservation and sustainable use of biodiversity. As a result, conservation actions are increasingly guided by conservation plans that are based on systematic biodiversity surveys that provide spatial information on quantitative biodiversity targets (Pierce *et al.*, 2005; Grantham *et al*., 2008).

However, the resources available for biodiversity conservation are small compared to those that are invested in human developments. Nevertheless, in the past few decades a great deal of time, money and effort has been put into the elaboration and advancement of quantitative and spatially explicit techniques, based on species distribution data, for identifying candidate areas for conservation action (Margules & Pressey, 2000; Brooks *et al*., 2001; Knight *et al*., 2008). The purpose of conservation assessment is to supply scientifically defensible information to facilitate the conservation planning process by ensuring both the efficient use of conservation resources as well as an effective application of conservation actions. Grantham *et al*. (2008) argues that conservation planning is only truly successful if further investments in surveys, mapping and modelling leads to improved planning decisions and thus to increased returns on investment. If the costs of conservation actions outweigh the benefits, then resources are better directed towards other biodiversity targets.

2.7.2 Problem of sampling bias in conservation assessment

The identification of conservation priority areas requires uniform sampling efforts throughout a study area, so that the recorded patterns of species distribution and abundance display a true picture, and are not the result of variation in sampling effort (Reddy & Dávlos, 2003). Only in this way true hotspots of biodiversity can be differentiated from areas that seem unique and rich in species due to biased sampling.

However, sampling efforts are mostly not consistent across the landscape. Studies by Robertson & Barker (2006) have shown that the majority of QDGs in South Africa are poorly sampled. This is especially true for the study area; the flora of the western Central Bushveld is largely under-represented by the holdings of the National Herbarium in Pretoria.

These under-sampled grids mostly do not constitute a true representation of the present taxonomic diversity, as low species richness is attributable to insufficient sampling rather than to actual low species richness (Robertson & Barker, 2006). Consequently, under-sampled QDGs, which are actually rich in species or which contain rare and unique species, may be overlooked during conservation planning. Conservation decisions are only as good as the data on which they are based.

On the other side, herbarium collections are not only incomplete for many areas, but are also characterized by quality problems. Sampling efforts are generally biased towards more accessible geographical areas, species that are charismatic and easy to identify in the field, and species that are detectable or present during the season of collection (Robertson & Barker, 2006; Grand *et al*., 2007).

Nevertheless, conservation planners are largely dependent on existing biodiversity data in herbarium and museum collections (Robertson & Barker, 2006). Grand *et al*. (2007) confirmed in their study on the implications of incomplete and biased data on the conservation assessment of the plant family Proteaceae in South Africa, that poor data quality definitely will affect the outcome of conservation plans. The comparison of the complete dataset with introduced sampling biases showed a 1– 5% reduction in species recorded which resulted in 9– 17% larger reserve networks for the biased data. Furthermore, they proofed that biased sampling has a much larger impact on reserve selection algorithms than incomplete sampling alone. For example Grand *et al*. (2007) found that biased sampling failed to detect localized restricted-range species, in contrast to widespread species.

Techniques suggested by Funk & Richardson (2002) to overcome these limitations of collection data include the modelling of known species records using biophysical data, the use of indicator taxa, and the collection of additional information such as historic data from herbaria, sight records and expert knowledge. Most of these approaches are also used in the present study to augment the present species records for plants in the western Central Bushveld held in the electronic database of South African National Biodiversity Institute.

2.7.3 Tools for area prioritizing for biodiversity conservation

Priority areas for conservation vary depending on which survey method was used (Kershaw *et al*., 1995). Conservation areas are basically evaluated in terms of their natural features, such as species, communities and habitats. The study of species distribution has long been a central focus of biogeography and ecology, but now has gained a new momentum and urgency as evidence of the global biodiversity crisis mounts (Reid, 1998). But what geographical regions should be protected in order to sustain the most biological diversity? How can we use biogeographic patterns to assess which areas have the highest priority for conservation?

2.7.3.1 Biodiversity hotspots

The identification of biodiversity hotspots lies at the heart of answering the question of what areas to conserve. Norman Myers defined the term ‘biodiversity hotspot’ in 1988, as he was the first to classify tropical forest hotspots according to exceptional concentrations of plant endemism and serious levels of habitat loss (Reid, 1998; Conservation International, 2007). In 1996, quantitative thresholds has been set up as criteria for the classification of biodiversity hotspots: to qualify as a hotspot an area must contain at least 1,500 species of endemic vascular plants together with at least 70% of the original habitat of the endemics lost (Conservation International, 2007).

Even though this is the formal definition of ‘biodiversity hotspots’, the term is nowadays commonly used to refer to regions of high species richness, including areas rich in endemic and threatened species. According to Reid (1998), geographical areas that rank high on one or more axes of species richness and intensities of threat are designated as hotspots. Therefore, the delineation of biodiversity hotspots is a promising approach that aids in setting priorities for biodiversity conservation. By concentrating on areas with the greatest need for safeguarding biodiversity, conservationists engage in a systematic response to the challenge of halting large-scale extinctions with scarce conservation resources at hand (Myers *et al*., 2000).

For example, biodiversity hotspots are widely used to identify gaps in existing protected areas. Gap analysis usually comprises three steps, the first of which involves the mapping of biodiversity hotspots in the study area. Mapped hotspots of species richness, rarity and threat are compared with existing protected areas and with areas facing immediate threat or degradation. In this way, it is determined which species are already well-conserved and which still need protection. Rodrigues *et al*. (2004a) argue that filling conservation gaps requires the designation of explicit, measurable, and repeatable biodiversity targets (Eken *et al*., 2004). Various selection algorithms (e.g. richness or rarity based selection algorithms) are used to determine a minimum set of areas to reach the conservation target (Reid, 1998).

2.7.3.2 Complementarity

Reid (1998) believes that the approach of complementarity is a more efficient mechanism for maximizing the number of species protected in the smallest number of sites in a given area of land than hotspot analysis. Conservation areas strive to sample the full variation of species composition along important environmental gradients in a region; a goal which is most efficiently accomplished by identifying complementary sets (Van Jaarsveld *et al.*, 1998; Sætersdale *et al*., 2003).

The complementarity selection algorithm usually involves the identification of the species content of existing reserves followed by the selection of further sites in a stepwise fashion to add areas that contribute the greatest number of so far unrepresented species (Reid, 1998; Brooks *et al*., 2001). A common technique, called 'greedy complementarity', is to select the area holding the largest number of new species at every level (Brooks *et al.*, 2001; Fjeldså & Tushabe, 2005). Conversely, Reid (1998) believes that a complementarity selection algorithm is most efficient if it starts with sites containing the most unique species, i.e. species found nowhere else, and thus choosing sites with relative low species richness rather than the most species-rich ones. Brooks *et al*. (2001) goes even a step further by integrating mapped biodiversity threats in the process of complementarity site selection. This process identifies top priority areas as those that are highly threatened and at the same time contribute to a complementary set.

However, in reality there are only a limited number of sites available for biodiversity conservation. Therefore, the best compromise is to combine complementarity site selection and hotspot analysis with the purpose of conserving both representative samples of ecosystems and hotspots of rarity, richness and threat (Reid, 1998). To sum up, by examining how to protect as many species as possible on the smallest available area, scarce conservation resources can be targeted systematically and efficiently.

2.7.4 GIS – a powerful tool for spatial modelling of biodiversity

With the advent of the geographical positioning system (GPS) and satellite remote sensing, spatial explicit data has become available that can be used to describe our environment. Primary biodiversity data was until recently largely available as specimen information such as plant vouchers in herbaria. Researchers use primary data in the form of presence-absence data of species across space and time together with geographical information about soil, geography, climate or other landscape features in their study of biodiversity patterns (Soberón & Peterson, 2004). Today geo-referenced primary data is shared among researchers and available over interlinked and web-based databases.

Given the increased need for the collection, management and communication of data on the status of biodiversity, scientific and technological advances have created the new field of biodiversity informatics. Biodiversity informatics pertains to the use of information technologies for the organization, illustration, analysis and interpretation of primary data regarding life, in particular at the species level (Soberón & Peterson, 2004). The Geographic Information System (GIS) programmes represent such powerful computer-based tools for the handling of spatial biodiversity data. They are designed for entering, storing, managing, analysing and displaying of geo-referenced data (Johnston, 2001; Salem, 2003).

Data from biodiversity inventories are varied and are derived from various sources. It encompasses spatial data such as maps, satellite images and aerial photographs, as well as non-spatial data like species and habitat attributes, and data sources including checklists, floras and faunas, reference collections and indigenous knowledge (Salem, 2003). GIS has the capability to integrate and analyse all of the mentioned types of data for the assessment and management of biodiversity. This is achieved by the ability of GIS technology to perform basic database operations such as data selection and query combined with the visualization and geographic analysis benefits of maps (Johnston, 2001; Geospatial Communication Network, 2010).

Firstly, a GIS functions as a database for spatial geo-referenced features and their attributes, which for example may comprise of distributions of plant and animal species, natural vegetation, soil, geology, hydrology, topography and land cover to name a few. Secondly, the association of spatial entities with attributes allows the combination, comparison and analysis of the diverse data layers to establish relationships between biota and environment (Salem, 2003). For instance biodiversity information can be integrated with environmental and geographic data in spatial overlays and displayed in maps for viewing, interpretation and analysis.

GIS therefore offer another perspective to biodiversity data by adding a geographic dimension which greatly enhances biodiversity conservation by helping to make more informed decisions. Better information means better decisions. However, GIS technology is not an automated but rather an interactive decision-making tool that supports the decision-making process by querying, analysing and mapping of data (Geospatial Communication Network, 2010). GIS outputs assist decision-makers to make considered choices about development and conservation planning by providing various alternatives and by modelling potential outcomes of a series of scenarios.

The benefits of GIS for biodiversity studies and conservation are manifold. First of all GIS maps compiled from baseline data such as relief, soil and vegetation can be used for initial field reconnaissance for efficient planning of fieldwork and sampling design. Thus, sampling sites can be chosen to get the most representative collection of biodiversity data (Gourmelon, 2006).

Moreover GIS technology allows the analysis of bigger datasets across large areas. This makes the assessment of conservation significance possible at a broader, ecologically more meaningful spatial scale, like biogeographical regions (Raal & Burns, 1996). But nevertheless, spatial patterns of biodiversity can be displayed and analysed at different spatial scales. The analysis aims at identifying correlations between different environmental variables and their associations to the conservation status and threat of species.

This finds application in the assessment of the effect of land-use, soil and climate on vegetation or the distribution of plant species. Such surveys form the basis for monitoring of biodiversity and environmental change, the identification and testing of biodiversity indicators, as well as for assessing the effectiveness of existing protected areas for biodiversity conservation. For example Walker & Faith (1993) developed an approach using GIS to test the relative contribution of each nature reserve to biodiversity at different geographic scales (Salem, 2003). Species lists for different geographic locations were linked with geographic and environmental data of nature reserves, followed by the analysis of their complementary contribution to the wholesale biodiversity represented by the reserve network.

The outputs of GIS operations are as diverse as the data inputs, which include tabular, graphical and digital output (Johnston, 2001). For example, various thematic maps can be produced to answer various research questions at a range of scales by choosing from a wealth of techniques for data manipulation and quantitative analysis. An important benefit of maps for conservation is that better decisions can be made with respect to the identification of priority areas. For example, GIS applications allow the modelling of diversity and rarity hotspots and threats from present and future development actives.

To conclude, GIS is a powerful tool for the organization, handling and communication of biodiversity data. Accordingly, the South African National Biodiversity Institute (SANBI) has recently launched a GIS-based biodiversity database system, called BGIS, so as to support the national biodiversity strategy. In the past years many regional biodiversity plans has been drafted to enhance biodiversity planning outside of protected areas (SANBI, 2009), but survey data has to be coordinated with the information that already exists so that it can be used in a complementary manner for improved conservation planning. As a result the online accessible BGIS has been developed as a central hub for the management of spatial biodiversity planning information. The main purpose of the BGIS is to grant easy access to spatial biodiversity data, and "*thereby facilitating its use in biodiversity planning and decision-making across the landscape*" (SANBI, 2009). This is in accordance with the Convention on Biological Diversity which highlights that "*access to good information about biological diversity is the key to mobilizing resources in support of conservation and sustainable use of these biological resources*" (Salem, 2003).