LAND-COVER CHANGE IN THE BERG RIVER CATCHMENT: IMPLICATIONS FOR BIODIVERSITY CONSERVATION

Tristan Stuckenberg

Thesis presented for the degree Masters of the Arts at Stellenbosch University

Supervisor: Dr A Van Niekerk

Co-supervisor: Mrs Z Münch

December 2012

DECLARATION

By submitting this thesis electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the sole author thereof (save to the extent explicitly otherwise stated), that reproduction and publication thereof by Stellenbosch University will not infringe any third party rights and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

Date: December 2012

SUMMARY

Biodiversity refers to the variety of life on earth at all scales of observation. Its persistence underlies ecological and evolutionary processes and is pivotal for the sustenance and future development of human societies through the provision of ecosystem services. Especially since the industrial revolution, anthropogenic land-cover change has placed ever-increasing strain on natural systems through the destruction and degradation of habitat. The Cape Floristic Region (CFR) is a global biodiversity hot spot which contains some of the highest levels of floristic diversity and endemism on the planet. Since European settlement large swathes of this region have been transformed to facilitate socio-economic development, placing tremendous pressure on indigenous biodiversity.

Due to the intimate relationship that exists between land cover and biodiversity it is possible to draw inferences on the current state of the biodiversity of an area, assess the pressures that will likely face it in the future and plan accordingly based on an analysis of land-cover change. As a means of assessing the state of biodiversity in the CFR, this thesis has developed a series of three land-cover maps for the Berg River catchment in the Western Cape province for 1986/1987, 1999/2000 and 2007 using Landsat TM and ETM+ data. Areas of natural vegetation were delineated on the land-cover maps using an object-orientated nearest neighbour supervised classification. Remnants of natural vegetation were classified according to potential vegetation boundaries described by Mucina and Rutherford's map of the vegetation of South Africa, Lesotho and Swaziland.

Contrary to initial expectations, the area occupied by natural vegetation had increased by 14%. However, considerable variation was recorded between vegetation types with certain types exhibiting marked increases in extent while others had been encroached by expanding cultivated and urban areas. An assessment of the accuracy of the 2007 land-cover map showed that significant swathes of natural vegetation were infested with alien invasive species or dominated by particularly resilient species which are not as severely affected by anthropogenic activities as other species. It is concluded that the methodology employed in this study provides a scoping mechanism by which more intensive research may be directed toward areas exhibiting significant land-cover change.

KEY WORDS

Berg River catchment, biodiversity, Cape Floristic Region, geographical information systems, land-cover change, Landsat, remote sensing, vegetation

OPSOMMING

Biodiversiteit verwys na die verskeidenheid lewe op aarde op alle waarnemingsvlakke. Die volhouding daarvan onderlê ekologiese en ewolusionêre prosesse en die verskaffing van ekosisteemdienste is deurslaggewend vir die onderhoud en toekomstige ontwikkeling van menslike samelewings deur. Veral sedert die industriële rewolusie het veranderinge in antropologiese gronddekking toenemende druk op natuurlike sisteme geplaas, grootliks deur die vernietiging en ontaarding van habitatte. Die Kaapse Floristiese Streek (KFS) met van die hoogste vlakke van floristiese diversiteit en endemisiteit op aarde, is ‘n brandpunt van wêreldwye biodiversiteit. Sedert die vestiging van Europese setlaars is uitgebreide dele van hierdie streek omskep om sosio-ekonomiese ontwikkeling te bevorder, wat geweldige druk op inheemse biodiversiteit geplaas het.

Te wyte aan die intieme verhouding wat tussen gronddekking en biodiversiteit bestaan, is dit moontlik om deur middel van ‘n ontleding van gronddekkingsveranderinge afleidings te maak rakende die huidige stand van biodiversiteit in ‘n streek. Sodoende kan bepaal word watter druk ʼn streek moontlik in die toekoms sal moet weerstaan. Vooruitbeplanning kan dienooreenkomstig gedoen word. Ten einde die stand van biodiversiteit in die KFS te beraam, het hierdie tesis ‘n reeks van drie gronddekkingskaarte (1986/1987, 1999/2000 en 2007) vir die Bergrivier-opvangsgebied in die Wes-Kaapprovinsie met behulp van Landsat TM en ETM+ data ontwikkel. Areas met natuurlike plantegroei is met behulp van ‘n voorwerp-georiënteerde naaste-buurman klassifikasie afgebaken. Oorblyfsels van natuurlike plantegroei is volgens potensiële plantegroeigrense, soos beskryf deur Mucina en Rutherford se kaart van die plantegroei van Suid-Afrika, Lesotho en Swaziland, geklassifiseer.

In teenstelling met aanvanklike verwagtinge, het die area wat deur natuurlike plantegroei bedek word met 14% toegeneem. Tog is aansienlike variasie tussen plantegroeitipes opgemerk, met sekere soorte wat opvallende omvangstoename toon, terwyl ander plantegroeitipes deur landbou en stedelike groei vervang is. ‘n Beraming van die akkuraatheid van die 2007-gronddekkingkaart toon dat noemenswaardige stroke natuurlike plantegroei deur uitheemse indringerspesies besmet word of deur uiters weerstandige spesies, wat nie so ernstig as ander spesies deur antropologiese aktiwiteite beïnvloed word nie, gedomineer word. Die gevolgtrekking is dat die metodologie wat in hierdie studie gebruik is ‘n meganisme verskaf waardeur meer intensiewe navorsing op areas wat aansienlike verandering in gronddekking ten toon stel, gerig kan word.

TREFWOORDE

Bergrivier-opvangsgebied, biodiversiteit, Kaapse Floristiese Streek, geografiese inligtingstelsels, verandering in gronddekking, Landsat, afstandswaarneming, plantegroei

ACKNOWLEDGEMENTS

I extend my sincere thanks to the following people and institutions for assisting me in many ways through the course of completing this thesis:

- The staff of the Department of Geography and Environmental Studies, Stellenbosch University for their collective support and use of facilities
- My supervisors, Dr Adriaan Van Niekerk and Mrs Zahn Münch, for their support, input and patience
- Garth Stephenson for his assistance and advice throughout the course of the project
- My fellow students for their support and friendship during my time at Stellenbosch University
- Dr Helen de Klerk for her assistance and friendship
- My family and friends for their support and encouragement
- Mr Nick Helme for his input to the project
- Dr David Eduard van Dijk for his anecdotes, insights into basic ecology and identification of plant species
- Dr Patricia Holmes for her input and insight
- SANBI and CapeNature for making a variety of spatially referenced biodiversity data available online through the Biodiversity GIS programme.

CONTENTS

TABLES

FIGURES

ACRONYMS AND ABBREVIATIONS

ARC	Agricultural Research Council
BGIS	Biodiversity geographical information system
BHU	Broad habitat unit
C.A.P.E	Cape Action for People and the Environment
CD: NGI	Chief Directorate: National Geo-spatial Information
CFR	Cape Floristic Region
CSIR	Council for Scientific and Industrial Research
DEAT	Department of Environmental Affairs and Tourism
DWAF	Department of Water Affair and Forestry
EIA	Environmental impact assessment
EMS	Electromagnetic spectrum
ETM+	Enhanced thematic mapper plus
EVI	Enhanced vegetation index
FAO	Food and Agricultural Organization
GCOS	Global Climatic Observation System
GEOBIA	Geographic Object-Based Image Analysis
GIS	Geographic information system
GPS	Global positioning system
HRG	High resolution geometric
HRS	High resolution sensor
HRV	High resolution visible
HRVIR	High resolution visible infrared
IR	Infrared
IUCN	International Union for the Conservation of Nature
LCCS	Land cover classification system
LCM	Land change modeler
MASL	Metres above sea level
MSS	Multispectral scanner
NASA	National Aeronautics and Space Administration
NDVI	Normalized difference vegetation index
NIR	Near infrared
NLC	National land cover

NSBA	National Spatial Biodiversity Assessment
QDS	Quarter degree squares
RGB	Red, green and blue
RHP	River Health Programme
SADC	Southern African Development Community
SANBI	South African National Biodiversity Institute
SANCO	South African National Civic Organization
SARDC	Southern African Research and Documentation Centre
SLC	Scan Line Corrector
SPOT	Satellite Pour l'Observation de la Terre
TM	Thematic mapper
UNCED	United Nations Conference on Environment and Development
UNEP	United Nations Environmental Programme
USGS	United States Geological Survey
WfW	Working for Water
WGS 84	World geodetic system 1984

CHAPTER 1: LAND-COVER CHANGE AND ITS SIGNIFICANCE FOR THE CONSERVATION OF BIOLOGICAL DIVERSITY

Due to the heterogeneity of topographic and climatic conditions, as well as its relatively large size, South Africa exhibits high rates of species diversity, richness and endemism (Thuiller *et al*. 2006). The country is home to three internationally recognized biodiversity hot spots, namely the Cape Floristic Region (CFR), the Succulent Karoo and Maputaland-Pondoland-Albany Thicket, and more than 20 300 vascular plant species and numerous threatened and endemic animal species (Thuiller *et al*. 2006). South Africa has experienced significant land-cover changes as a result of human endeavour, particularly over the last 100 years, which are thought to have had significant repercussions on the biodiversity of the area (Biggs & Scholes 2002). The designation of protected areas has traditionally focused on factors such as perceived aesthetic appeal and the value of the area in terms of agricultural or mining potential and has often not considered biodiversity and ecological processes (Reyers *et al*. 2001). Considering the richness of biodiversity in the country, the relative underrepresentation of many ecosystems in protected areas and the limited resources with which conservation initiatives operate, it is imperative that priority areas for the conservation of specific species and ecosystems be identified.

The world stands on the precipice of harrowing ecological deterioration in the face of the intertwined problems of population growth, development and environmental degradation (Foley *et al*. 2005). The myriad interwoven biophysical systems that harbour, preserve and perpetuate life on earth bear, and increasingly falter under, the relentless march of civilization. The last several centuries have borne witness to extraordinary rates of biodiversity loss where rampant population growth has colluded with rapid technological advances to place tremendous pressure on local ecosystems. Especially since the United Nations Conference on Environment and Development (UNCED) in 1992, there has been a growing international awareness of the need to conserve biodiversity and align societal interests with environmental concerns. With repercussions that range from the loss and fragmentation of habitat to disruptions in hydrological and climatic conditions, many have come to view anthropogenic land-cover change as the most significant threat to global biodiversity.

1.1 BIODIVERSITY

Biodiversity is a multidimensional concept that encompasses both empirical and conceptual renderings. Empirically, biodiversity is seen as a measure of genetic and species variation within an ecological community or area and the interactions between organisms therein (Hill et al.

2005). As a concept it is aptly defined by the United Nations Convention on Biological Diversity (UNEP, 1992:3) as:

"the variability among living organisms including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems."

However, considerable debate surrounds the use of the term and how it is best measured and the role it should play in conservation planning. This debate is explored in Chapter 2. The remainder of this section will draw attention to the importance of biodiversity and biodiversity conservation with a specific focus on the South African context.

1.1.1 South Africa's biodiversity

South Africa is one of the 17 megadiverse countries which together account for over two thirds of global biodiversity (DEAT 2005). The country only occupies around 2% of the earth's terrestrial surface, yet contributes in excess of 10% to global plant biodiversity (DEAT 2005). More than half of these species are endemic with exceptionally high rates of endemism concentrated in the Fynbos, Succulent Karoo, Nama Karoo and Albany Thicket biomes (DEAT 2005). The country has witnessed severe environmental disruptions as a result of human endeavour which are thought to have had marked repercussions on the biodiversity of the area (Low & Rebelo 1996; Biggs & Scholes 2002; Mucina & Rutherford 2006).

The dominant driving factors have been the expansion of agriculture and commercial forestry which together constitute about 14% of the country's total land surface (Biggs & Scholes 2002; Mucina & Rutherford 2006). Two other crucial factors are the impact of urban expansion, especially around the major urban centres, and the introduction of exotic species, particularly plants, which threaten indigenous species in many areas (Biggs & Scholes 2002; Reyers *et al*. 2007). Land degradation is a major concern in the country although its precise impact on biodiversity varies considerably depending on the nature and extent of degradation and the sensitivity of the affected ecosystems (Reyers *et al*. 2007). The country also exhibits a relatively high population growth rate which stood at 1.4% in 2010 (World Bank 2010) with many marginalized people who are heavily reliant on natural resources for their livelihoods (SADC 2008).

The country's biodiversity is widely used for commercial and subsistence purposes by both the formal and informal sectors (Turpie 2003). In many rural and informal areas, resource harvesting represents a significant component of the livelihoods of marginalized people (SADC 2008). Many species also serve functions in traditional and formal medicines, and as traditional beverages such as rooibos and Honeybush Tea (DEAT 2005). Recently, the export of flowers from the Western Cape has been promoted as a potentially sustainable and empowering industry that has succeeded in generating substantial foreign exchange (Turpie 2003). Promisingly, the sheer diversity of genetic and biochemical resources in the country has raised interests in the potential for bioprospecting and the development of resources associated with a variety of indigenous organisms (Cloete, Nel & Theron 2006).

1.1.2 Significance of biodiversity conservation

The significance of biodiversity lies in its ecological function as a keystone of evolutionary and ecological processes and in its value to humans through the provision of resources and ecosystem services (Gaston 1996). Regarding ecosystem functionality, a greater array or species diversity increases the productivity of a given ecosystem as different species are better able to appropriate different resources (Gaston 1996). Ecosystems which display high rates of biodiversity are also more resilient than those that do not, better enabling adaptation to shifting environmental conditions and thereby ensuring the long-term productivity of a particular area (Gerber 2005).

There is also a consensus that biodiversity and ecosystem functionality are intricately and fundamentally interrelated and that disruptions to particular components of this relationship may have adverse effects on overall ecosystem integrity (Gaston 1996; Wessels *et al*. 2003). The importance of biodiversity to human societies is diverse with benefits that include food, medicine and the regulation and purification of water as well as carbon sequestration, nutrient cycling and soil formation (Gaston 1996; Gerber 2005). It stands to reason that the preservation of biological diversity has significant consequences for the sustainability and future well-being of human societies and the integrity of biophysical systems.

1.2 LAND COVER

Land cover refers to the biophysical cover of the earth's surface. It exerts immense influence on biodiversity and human socio-economic systems and in turn it is influenced by these systems. Thus, land cover can be understood as a manifestation of the history of life on earth and perhaps as a window into its future. An investigation of the relationship between biodiversity and land-cover may be crucial to the judicious management and allocation of natural resources in the future. In light of these considerations this section describes land cover, land-cover change, and discusses the ways in which land-cover is linked to biodiversity and how the relationship between land cover and biodiversity is measured.

1.2.1 Land cover and land use

It is important to distinguish between land cover and land use as the terms are often confused or used synonymously in popular and even scientific literature. *Land cover* is commonly used to describe the various components that characterize the earth's surface: generally biological features such as vegetation and physical features such as soils, water bodies or the built environment. Land cover thus represents the cumulative result of the interaction of natural and anthropogenic processes that impact on the earth's surface (Mannion 2000). Conversely, *land use* describes the function of an area as it relates to human activity, especially its economic or social significance (Mannion 2000). While there is often a significant degree of overlap between land-cover and land-use classifications, differentiation between the two is perhaps best illustrated by examples of incompatible classifications. A nature reserve, for example, may receive the same land-cover classification as an adjacent area of natural vegetation but will be classed as a different land use owing to its economic significance. Much of the confusion surrounding the terms stems from their degree of interrelatedness where human land-use patterns exert an ever increasing influence on land cover through the transformation and alteration of natural landscapes to facilitate human endeavour (Meyer & Turner 1992). In this report *land cover* will render the term a noun while *land-cover* denotes the terms use as an adjective.

1.2.2 Natural land-cover change

Land cover is dynamic with change occurring at differing spatial and temporal scales in response to both natural and anthropogenic influences. Natural land-cover changes operate on varying time scales that may include drastic responses to events such as volcanic eruptions or gradual shifts in vegetation cover in response to climatic change and evolutionary processes. The rates at which these processes occur and the mechanisms that drive them may be difficult to determine owing to the number of interacting factors involved (Nagendra, Munroe & Southland 2004). In this context, natural land-cover changes are best viewed as the ever evolving interaction between the various components of the earth's surface. In the wake of the growing human impacts on almost all aspects of the natural environment, land-cover change is increasingly being considered in terms of social or economic drivers as ever greater areas of the earth's surface are transformed and influenced by human endeavour.

1.2.3 Anthropogenic land-cover change

In contrast to natural land-cover change, anthropogenic land-cover change occurs as societies modify existing land cover to facilitate their development. These transformations and alterations have become so pervasive that they now occupy in excess of 30% of the earth's land surface with even greater areas indirectly affected by these developments (Foley *et al*. 2005). These land-cover changes are a manifestation of socio-economic dynamics and population growth operating within the confines of technological and environmental factors (Mannion 2002; Lepers *et al*. 2005). However, as Houghton (1994) notes, the relationships between the various drivers of anthropogenic land-cover change are often complex and will have repercussions operating at varied spatial and temporal scales.

1.2.4 Land-cover change and the threat to biodiversity

The threat to biodiversity posed by anthropogenic land-cover change consists of two primary components. First is the extent of habitat being directly degraded or lost. This results in an immediate loss of biodiversity as many species will be displaced or eradicated during the transformation while most others will be unable to survive in the newly altered landscape (Parker & Mac Nally 2002; Haines-Young 2009). Second is the impact of habitat degradation or reduction on species persistence and population viability. Species and individual specimens exist within a nexus of mutualistic relationships (Parker & Mac Nally 2002). It follows that disruptions to particular areas or ecological communities may induce a cascading decline in various populations across otherwise disconnected areas through the interconnection of ecological processes such as predation and pollinator relationships (Parker & Mac Nally 2002; Haines-Young 2009). Furthermore, sub-populations in heavily transformed or fragmented landscapes tend to face an exacerbated risk of disruptions to evolutionary and genetic processes and may be too small or isolated to remain viable or reclaim restored areas (Didham *et al*. 2007). Collectively, these factors imply that as the isolation and reduced extent of habitats associated with vegetation cover can have a detrimental impact on the ability of species to persist and propagate in the long term, the immediate effects of land-cover alterations are compounded by long-term hindrance of ecosystem functionality through degradation, fragmentation and contamination (Fahrig 2001; Fahrig 2003; Parker & Mac Nally 2002; Haines-Young 2009).

Many effects of land-cover alteration can extend well beyond the spatial extent of the transformation itself (Wessels *et al*. 2003). The capacity for land-cover alterations to influence hydrological systems over vast geographical areas, the effects of pollution on ecosystems and the contamination of adjacent water courses associated with the use of nitrogen-based fertilizers are oft-cited examples (Wessels *et al*. 2003; Pauleit, Ennos & Golding 2005). Moreover, anthropogenic land-cover alteration has been associated with disruptions to various ecological and environmental functions such as the movements of nutrients through plants, water and soil as well as the movement of soil and water within an area (Crist, Kohley & Oakleaf 2000; Mannion 2002; De Villiers *et al*. 2005). According to De Villiers *et al*. (2005), these effects are compounded by the introduction of pathogens, disruptions to the various hydrological and geochemical processes and to the migration patterns of certain species. Also, the conversion of natural vegetation results in the release of carbon dioxide stored in natural vegetation and soil and generally reduces an area's ability to sequester carbon. For these reasons anthropogenic land-cover transformation and alterations have been identified as significant driving forces of anthropogenic climate change (Mannion 2002).

It is noteworthy that the exact nature of impacts varies considerably between species and between land-cover types and undertaking a comprehensive review of these relationships is often impractical at most scales of measurement (Pauleit, Ennos & Golding 2005; O'Connor & Kuyler 2009). Several authors have pointed out that an inadequate understanding of the sensitivity of particular habitats and ecosystems to degradation and transformation impedes the assessment of the impacts of land-cover change on biodiversity (Holmes & Richardson 1999; Cowling & Heijnis 2001). In particular, the threshold levels of transformation and degradation leading to extirpation within particular habitat or vegetation types are not well established (Holmes & Richardson 1999; Cowling & Heijnis 2001). Additionally, the effects of habitat reduction and fragmentation vary considerably between species and there is only a limited understanding of how to quantify these effects (Didham *et al*. 2007). There is concern that most measures of biodiversity superficially represent the spatial components of evolutionary processes which may have long-term implications for the future viability of ecosystems (Cowling & Heijnis 2001). Evolutionary and ecological processes are difficult to quantify and are consequently seldom factored into spatial biodiversity assessments (Cowling & Heijnis 2001). The relationship between land-cover change and biodiversity loss is thus obscured by incomplete knowledge of the status of biodiversity and ecological processes of many areas.

1.2.5 Measuring the impacts of land-cover changes on biodiversity

Owing to the sheer number of species that may be present in an area it is impossible to fully quantify that environment's biodiversity. Furthermore, no universally accepted standards of defining and measuring biodiversity exist (Reyers *et al*. 2001). Traditionally, the distributions of threatened or indicator species have been used to define biodiversity in what is now termed a species-orientated or taxonomic approach (Oliver *et al*. 2004). However, this approach is limited by biased sampling and incongruencies between the distributions of various taxa and it has tended to overlook less conspicuous species and neglected the intricate relationships existing between species and ecological communities (Stockwell & Townsend-Peterson 2003). Moreover, species data may be unavailable or inconsistent at the level required by conservation or land-use planning (Oliver *et al*. 2004).

The shortcomings of species-orientated measures of biodiversity have shifted the focus toward broader indicators of biodiversity which concentrate on the use of holistic surrogates as measures of biodiversity (Wessels, Reyers & Van Jaarsveld 2000; Oliver *et al*. 2004). These measures are typically defined by a collation of factors such as edaphic and climatic variables, indicator species and vegetation distribution which are taken to underlie the finer aspects of the distribution of biodiversity (Oliver *et al*. 2004; Fischer & Lindenmayer 2007). This approach holds that by identifying examples of as many assemblages and ecological systems as feasible it is possible to define the distribution of, and subsequently preserve, the majority of indigenous biodiversity (Stoms *et al*. 2005). Bailey (1996) argues that there is a need to base such representations on factors that control ecosystem boundaries as opposed to contemporary species configurations in order to compensate for the effects of anthropogenic disturbance. Using a measure of potential biodiversity enables one to examine the impact of land-cover change in areas where disturbance predates available biophysical data. While ecosystems and species occurring over a limited spatial extent may be overlooked in this approach and although the relationships between potential species distribution and abundance is not well established, its value in broad-scale conservation planning is undeniable and it has been particularly effective in identifying priority areas for conservation initiatives (Oliver *et al*. 2004).

Anthropogenic land-cover transformation poses the single greatest threat to biodiversity in South Africa and the world. The manner in which land-cover change unravels in the near future will likely exert considerable influence on presence and persistence of global biodiversity. Consequently the integration of biodiversity and land-cover will prove pivotal to the optimal management of biodiversity in the future. There is consequently a pressing need to develop methods that can expediently assess the state of biodiversity in a given area, identify the current and future drivers of biodiversity loss and provide recommendations on the optimal management of these areas.

1.3 RESEARCH PROBLEM

Amid widespread biodiversity loss and the disruptions to environmental systems around the world, emphasis is being placed on understanding and mitigating the increasingly pervasive impacts of human activities. A major obstacle to the effective design and implementation of such undertakings is that biodiversity must be measured in a way that can be readily evaluated in the light of human impacts. The spatial representation of biodiversity is a pivotal and challenging aspect of conservation planning and ecological assessment. A continuous representation of biodiversity facilitates the assessment of land-cover change impacts as they relate to biodiversity. It allows the determination of suitable habitat extent for a given species or an assemblage of species and assesses the pressures facing them by comparing their current extent to an historical record (Stockwell & Townsend-Peterson 2003).

Vegetation type maps may provide a suitable means by which biodiversity can be assessed in combination with land-cover data as they provide a continuous measure of biodiversity. It is particularly in the Cape Floristic Region (CFR), where rates of floral diversity and endemism are high, that the synthesis of land-cover and vegetation data provides the best coarse-scale means of assessing and monitoring the state of local biodiversity. Little research has, however, been conducted on the capacity of land-cover change analysis for assessing biodiversity change in the CFR.

1.4 RESEARCH AIM AND OBJECTIVES

The primary aim of this research is to map and assess the spatial extent and dominant trends in land-cover changes in an appropriate study area and to examine the ecological impacts of anthropogenic land-cover alteration in terms of biodiversity and habitat reduction.

The secondary aim is to identify, in spatially- and temporally-explicit terms, the dominant drivers of land-cover change and to make recommendations for the optimal management of land resources, paying particular attention to the obstacles that such an initiative is likely to face. The tertiary aim is to evaluate the capacity of land-cover data in conjunction with vegetation-type data, to monitor biodiversity in the CFR.

To achieve these aims, the proposed research seeks to:

1. justify the use of vegetation type as an appropriate biodiversity surrogate;

2. acquire and produce maps of vegetation types and land cover;

3. prepare and standardize acquired data to ensure integrity and utility of different data sets when analysed;

4. assess the accuracy of land-cover maps;

5. establish and quantify the extent of transformation of vegetation to urban, agricultural or other land-cover classes;

6. determine and quantify the impact of anthropogenic land-cover transformation on biodiversity;

7. identify the dominant direct and indirect drivers of land-cover change and comment on likely future land-cover change with the intention of making recommendations for the optimal management of biodiversity;

8. assess the ability of land-cover and vegetation type data to monitor biodiversity in the CFR.

Together these objectives provide an historical analysis of the impacts of land-cover changes on biodiversity in the CFR. Owing to the availability of satellite data as well as practical constraints the study focuses on a portion of the CFR; the Berg River catchment. The Berg River catchment represents a severely transformed area that displays high levels of biodiversity and endemism and faces acute pressure from anthropogenic land-cover alterations and degradation (RHP 2004). In this area, determining the extent and rates of change in the distribution of ecological communities will assist in establishing the status of biodiversity, assessing threats and identifying priority areas for conservation. Furthermore, as the history of land-cover change can provide insights into the trajectory and driving forces that underlie these processes, its interrogation is crucial to the effective design and implementation of conservation and land-use planning. As conservation planning must operate within the constraints of current and likely future land cover-changes, developing means of assessing and understanding these changes and their impacts is vital to the optimal management and planning of biodiversity conservation.

1.5 STUDY AREA: THE BERG RIVER CATCHMENT

The Berg River catchment in the Western Cape province of South Africa, as illustrated in Figure 1.1, is the largest catchment in the Western Cape and is widely regarded as one of the most important due to the rich agricultural areas it supports and the quantity of water it provides to the City of Cape Town. The catchment includes portions of the Bergrivier, Saldanha Bay, Swartland, Witzenberg, Drakenstein and Stellenbosch local municipalities and falls within the West Coast and Cape Winelands district municipalities. The following subsections provide a concise description of the catchment's physical environment and an overview of its socio-economic profile. The environmental issues that affect the catchment are also discussed, with particular emphasis on the state of indigenous biodiversity.

1.5.1 Physical and environmental characteristics

The Berg River rises in the Franschhoek and Drakenstein mountains and flows in a northerly and then westerly direction to discharge into the Atlantic Ocean at St Helena Bay. It is approximately 285 km long and drains an area of 8 980 km². The catchment is subdivided into 12 quaternary catchments varying in size from 2000 km² in the lower catchment to 125 km² at the headwaters. While much of this area is relatively flat, mountains in excess of 1000 m are found in its northern and eastern reaches. The density of drainage channels is remarkably low in the lower catchment but increases significantly in the middle and upper catchment (RHP 2004). The catchment falls within the winter rainfall regime of the south-western Cape, with rainfall generally increasing from the west coast to the mountainous areas in the eastern portion of the catchment (Schulze *et al*. 1997). Mean annual precipitation varies from 300 mm in the lower catchment to 1412 mm in the mountainous upper catchment (Schulze *et al*. 1997). Conspicuous waterbodies include the Voël Vlei and Wemmershoek dams. Prominent wetland areas are the Langebaan Lagoon and the Berg River estuary. The Langebaan Lagoon wetland, an intertidal salt flat and Ramsar site, is well protected and regarded as pristine (RHP 2004). The Berg River estuary gives sanctuary to intricate ecosystems and is known for its aquatic and avian biodiversity. While much of this wetland area retains its natural appearance, the degree to which agriculture and urban development adjacent to the estuary have impacted upon its biodiversity and ecology remains unclear (RHP 2004).

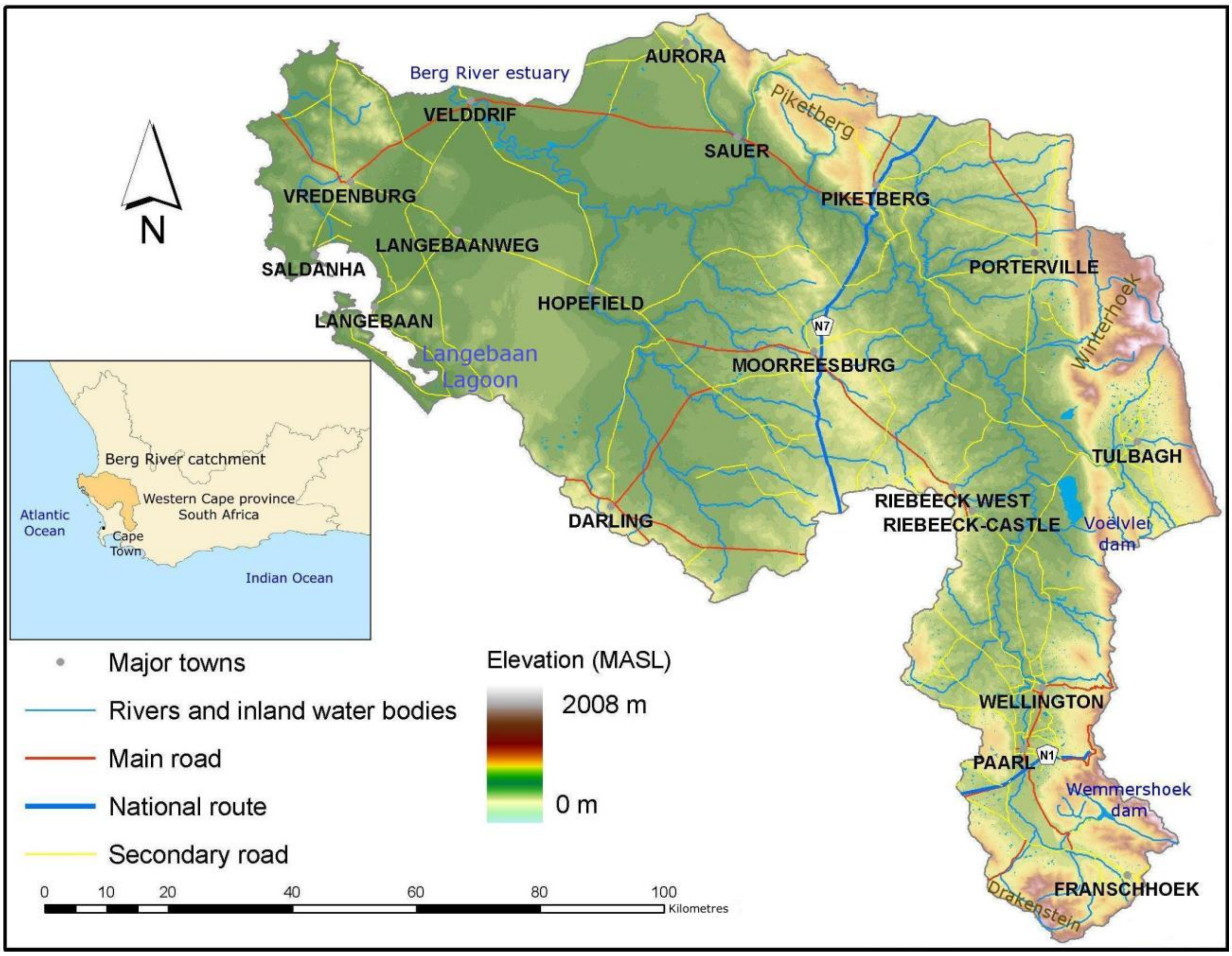


Figure 1.1: The topography, major towns and major roads of the Berg River catchment

The geology of the mountainous and upland areas is characterized by quartzites and sandstones of the Cape Supergroup while much of the rest of the catchment is dominated by shales of the Malmesbury and Klipheuwel Groups, and Cape granites (Clark & Ratcliffe 2007). Although this has resulted in a catchment typified by nutrient-poor lithologies, the mineral content of watercourses increases rapidly as one moves downstream (De Villiers 2007). Soils in this area vary considerably from sandy sediments in the lower catchment to those displaying marked clay accumulations which characterize much of the middle catchment (Clark & Ratcliffe 2007). It is the presence of these rich clayey soils that has made much of the catchment appealing to agricultural development and prompted the extensive transformation of large swathes of low-lying land (Kamish 2008). Shallow, minimally developed soils are also common in the landscape usually found on hard or weathering rock, characteristic of the mountainous regions that flank the catchment and low sporadic granite hills that litter the middle catchment (Clarck & Ratcliffe 2007).

The Cape Floristic Region (CFR), located largely in the Western Cape province, contains a disproportionate number of both threatened and endemic plant species. The main threats to these species are the conversion of natural vegetation to other land-cover classes, largely due to the expansion of agriculture and urban areas, pressure from intensive grazing as well as invasive alien vegetation and inappropriate fire regimes (Von Hase *et al*. 2003). The area constituting the Berg River catchment is entirely located within the CFR and has been subject to extensive transformation, primarily for agricultural. These developments have placed acute pressure on local ecosystems and led to a high concentration of threatened species within the catchment (RHP 2004).

For expedience vegetation in the catchment has been broadly divided into four main groupings: fynbos, renosterveld, sand fynbos and strandveld vegetation types (Figure 1.2). Various azonal vegetation types are also found within the catchment. Fynbos vegetation types are largely confined to fine grained soils at higher elevations on the mountains in the eastern and southern portion of the catchment and the Piketberg (Mucina & Rutherford 2006). Most fynbos vegetation types remain well conserved and have largely avoided anthropogenic transformation owing to their distribution at higher elevations that are unsuitable for agriculture or urban development. Renosterveld and alluvium fynbos dominate the lowlands of the upper and middle catchment and are typically found on fertile clays and silts. This has prompted the widespread clearance of this vegetation to make way for cultivation (Mucina & Rutherford 2006). Renosterveld is one of the most threatened vegetation types in South Africa with some estimates claiming an up to 97% reduction, largely to make way for agriculture. Strandveld and sand fynbos characterize the lower catchment and coastal areas and are usually found on sandy soils having marginal agricultural potential (Mucina & Rutherford 2006).

Thirty-one vegetation types occur within the catchment (Table 1.1 and Figure 1.3). Three vegetation types, Saldanha Granite Strandveld, Saldanha Limestone Sandveld and Saldanha Flats Strandveld, have been identified as being near endemic. Hopefield Sand Fynbos, Swartland Alluvium Fynbos and Swartland Alluvium Renosterveld also have significant portions of their potential extent within the catchment. Swartland Granite Renosterveld, Swartland Shale Renosterveld, Swartland Silcrete Renosterveld, Swartland Alluvium Fynbos and Cape vernal pools have been identified as being critically endangered. Swartland Shale Renosterveld has the largest extent in the catchment of over 3 000 km². Hopefield Sand Fynbos also occupies a large proportion of the catchment with a spatial extent in excess of 1 500 km². Cape Coastal Lagoons, Cape Inland Salt Pans, Cape Seashore Vegetation, Southern Afrotemperate Forest and Western

Altimontane Sandstone Fynbos are found in the Berg River catchment but exhibit a small spatial extent.

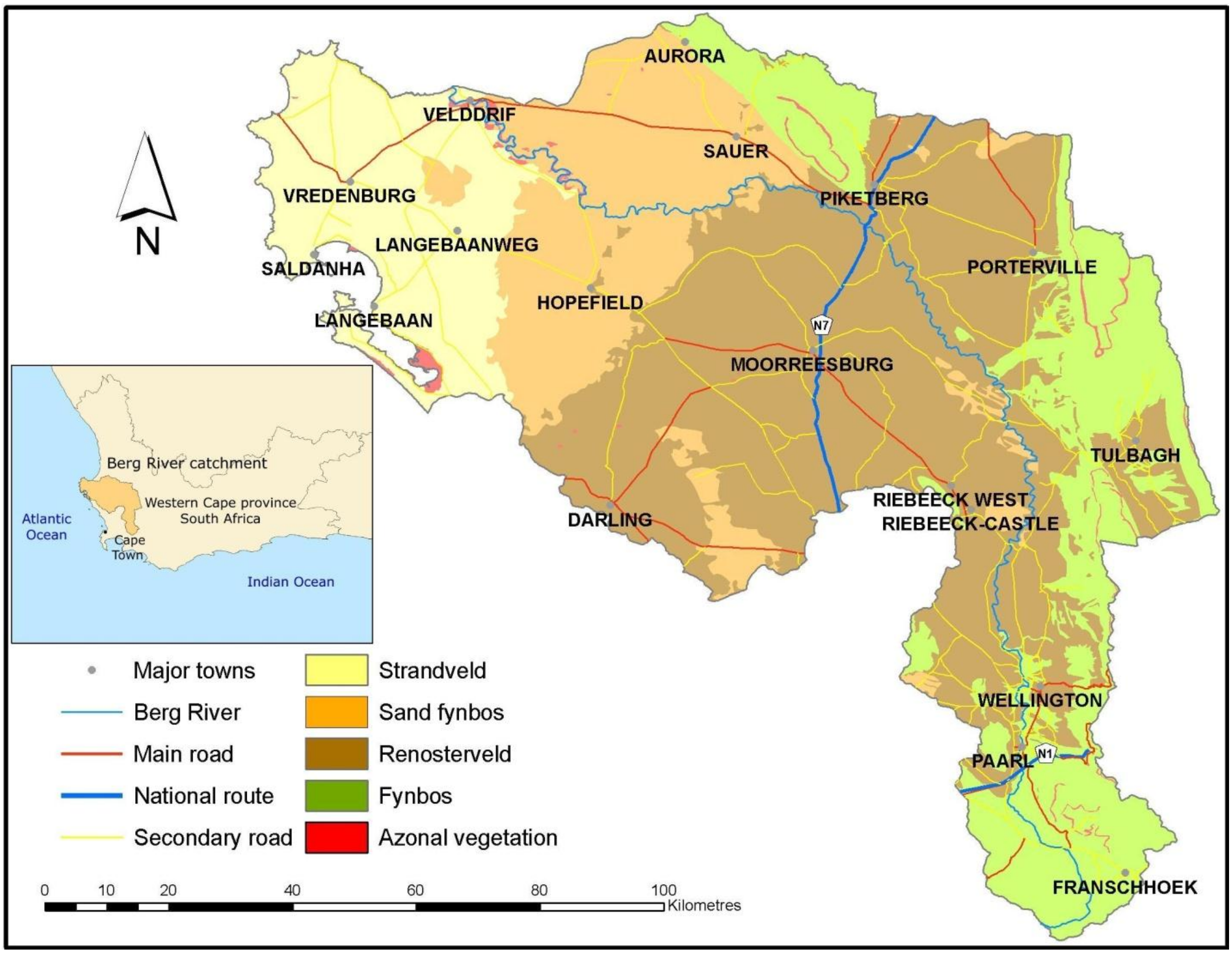


Figure 1.2: Broad vegetation types in the Berg River catchment

The clearing of indigenous vegetation to make way for agriculture has resulted not only in a reduction of the overall species richness but has also been linked to steady increases in the presence of dissolved salts in watercourses throughout the catchment (Flügel 1995; Kamish 2008). It is believed that the removal of deep-rooted indigenous vegetation to make way for shallow root crops, such as wheat, causes the water table to rise, subsequently dissolving salts associated with weathered Malmesbury shale (Flügel 1995). This has an adverse effect on numerous plant and animal species and renders many areas unsuitable or marginal for agriculture. Anthropogenic land-cover alterations in this area have also been linked to increases in the density of various pollutants in the watercourses of the catchment (De Villiers 2007). The high, and steadily increasing presence of inorganic nitrates, phosphates and ammonium pose a serious threat to aquatic biodiversity and a eutrophication risk (De Villiers 2007). Most research associates the increase of these pollutants with agricultural intensification and expansion, industrial development and the expansion of un-sewered settlements.

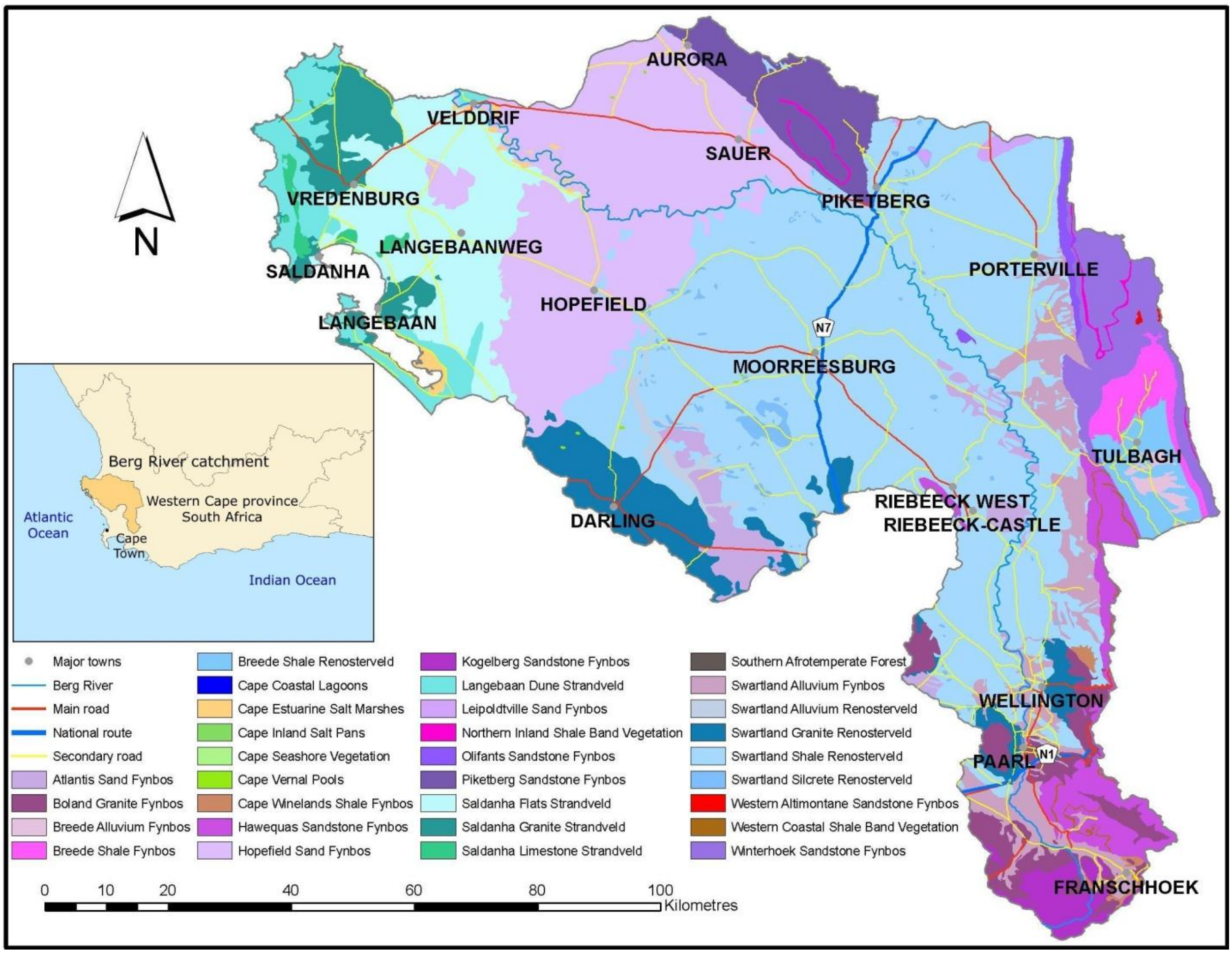


Figure 1.3: Vegetation types in the Berg River catchment

The Berg River catchment represents a severely transformed area that displays high levels of biodiversity and endemism and faces acute pressure from anthropogenic land cover alterations and degradation (RHP 2004). In this area, determining the extent and rates of change in the distribution of ecological communities can be considered pivotal to establishing the status of biodiversity, assess threats and identifying priority areas for conservation. The presence of highly threatened vegetation types in the catchment necessitates an assessment of their current extent and for the identification of potential threats.

Table 1.1 Vegetation types in the Berg River catchment

Vegetation type	Total extent (km²)	Extent within catchment (km²)	Percentage within catchment (%)	Total remaining area (%)	Conservation status	Protection level (%)	Bio-geographically important taxa	Endemic taxa
Atlantis Sand Fynbos	698.0	190.1	27.2	60.3	Vulnerable	2.1	0	6
Boland Granite Fynbos	499.0	253.7	50.8	49.2	Endangered	14.2	0	23
Breede Alluvium Fynbos	510.5	31.6	6.2	43.1	Endangered	0.2	0	17
Breede Shale Fynbos	318.5	116.3	36.5	71	Vulnerable	6.2	0	7
Breede Shale Renosterveld	1046.4	144.4	13.8	69.3	Vulnerable	1.8	0	13
Cape Coastal Lagoons	46.4	0.2	0.4	91.7	Least threatened	29.9	0	0
Cape Estuarine Salt Marshes	102.1	39.8	39.0	86.5	Least threatened	22.8	2	2
Cape Inland Salt Pans	84.6	1.8	2.1	79.6	Vulnerable	20.0	0	7
Cape Seashore Vegetation	227.3	3.8	1.7	98.3	Least threatened	44.5	0	18
Cape Vernal Pools	0.2	0.1	32.5	11.9	Critically endangered	0.0	1	12
Cape Winelands Shale Fynbos	85.7	17.1	19.9	48.8	Endangered	25	0	1
Hawequas Sandstone Fynbos	1051.2	244.5	23.3	95.6	Least threatened	53.1	0	85
Hopefield Sand Fynbos	1797.6	1465.9	81.6	59.6	Endangered	0.4	0	5
Kogelberg Sandstone Fynbos	915.3	104.7	11.4	83.1	Least threatened	57.6	0	187
Langebaan Dune Strandveld	437.7	268.4	61.3	65.8	Vulnerable	28.9	11	1
Leipoldtville Sand Fynbos	2755.4	22.5	0.8	44.9	Endangered	0.0	0	29
Northern Inland Shale Band Vegetation	264.4	24.0	9.1	96.5	Least threatened	17.8	0	13
Olifants Sandstone Fynbos	1058.5	39.8	3.8	92.2	Least threatened	23	0	4
Piketberg Sandstone Fynbos	460.4	306.0	66.5	82.6	Least threatened	0	0	39
Saldanha Flats Strandveld	760.2	697.6	91.8	45.4	Endangered	11.0	8	2
Saldanha Granite Strandveld	234.8	231.6	98.7	30.7	Endangered	9.1	5	15
Saldanha Limestone Strandveld	35.7	35.6	100	59.1	Endangered	0	4	10
Southern Afrotemperate Forest	799.8	0.3	0.04	97.3	Least threatened	59.7	18	11
Swartland Alluvium Fynbos	469.8	415.3	88.4	26	Critically endangered	1.7	0	12
Swartland Alluvium Renosterveld	62.5	51.6	82.5	60.4	Vulnerable	0.0	0	0
Swartland Granite Renosterveld	947.5	396.1	41.8	21.4	Critically endangered	0.5	0	27
Swartland Shale Renosterveld	4945.8	3413.8	69.0	9.6	Critically endangered	0.1	0	34
Swartland Silcrete Renosterveld	99.9	68.9	69.0	10.2	Critically endangered	0.3	0	4
Western Altimontane Sandstone Fynbos	37.5	2.0	5.3	100	Least threatened	34.5	0	11
Western Coastal Shale Band Vegetation	134.7	12.7	9.4	93.9	Least threatened	43.2	0	7
Winterhoek Sandstone Fynbos	1190.0	300.9	25.3	94.9	Least threatened	24.2	0	60

1.5.2 Socio-economic profile

In 2004 the total human population of the Berg River catchment was estimated to be 420 000 with a growth rate of 3% (RHP 2004). The majority (79%) of the population resides in urban areas, the largest of which are Velddrif and Laaiplek near the coast, Mooreesburg, Piketberg, Hopefield and Darling farther inland and Paarl and Wellington in the upper catchment. Population density is moderate at around 47 people per km² and decreases in the western and northern areas with the majority of the area's population concentrated in and around the major urban centres of Paarl and Wellington (RHP 2004). The catchment's road network is sparse with most roads servicing the major urban areas while large swathes of agricultural and natural areas remain inaccessible for most purposes. Future population growth and development is expected to occur primarily in urban areas due to the limited potential for agricultural expansion and other rural economic activities (RHP 2004).

Agriculture is the mainstay of the area with wheat, grapes and deciduous fruit being the dominant crops. Consequently, most industries in the catchment are based on processing agricultural produce and include wineries, canneries and other food-processing plants (RHP 2004). Irrigated and intensive agriculture is concentrated in the upper reaches of the catchment while dryland grain and stock farming dominate the middle to lower catchment. Fisheries are found on the West Coast at Laaiplek and Velddrif and constitute important industries in these areas. Owing largely to the perceived aesthetic appeal of the area, tourism and recreation are substantial and rapidly growing industries which appear set to constitute a more important component of the region's economy in the future.

Although the catchment has been occupied by humans for millennia, anthropogenic land-cover change has been most significant since the colonization of the area by European settlers beginning in the 17th century (Mountain 2003; RHP 2004). The development of the area has extensively reduced the distribution of indigenous vegetation. Due to its agricultural potential, transformation has been especially pronounced in the lowlands, most severely affecting renosterveld and other lowland vegetation types (Von Hase *et al*. 2003; Mucina & Rutherford 2006). Currently, these vegetation types are considered extremely endangered and are minimally represented in the reserve system of the country (Von Hase *et al*. 2003). Much of that which remains of these lowland vegetation types is found in isolated fragments on privately-owned land, impeding its effective incorporation into a reserve system and diminishing the prospects for long term preservation of these vegetation types and the species they harbour (Von Hase *et al*.

2003). Furthermore, the location of lowland vegetation types within an agricultural matrix means that they face continual pressure from factors such as fragmentation and pollution (Von Hase *et al*. 2003).

1.6 RESEARCH DESIGN

This project seeks to map historical land-cover change in a primary catchment (Berg River) to draw inference on its impact on overall biodiversity. The process by which this is achieved entails the application of a set of methods as outlined in Figure 1.4.

In this research, an argument is made that the presence of indigenous biodiversity is inherently linked to indigenous vegetation. On this assumption, vegetation is mapped from historical satellite imagery and its shifting distribution quantified to make pronouncements on the state of biodiversity in these areas. By considering the trajectory of land-cover change over the period of study, in conjunction with various political, socio-economic and biophysical parameters, the likely course of future land cover can be surmised. Subsequently, comment on the future management of this area can be provided.

Conducting the research at a catchment level is, in this instance, pragmatic but has some basis in ecological theory. Wishart (2000) argues that catchments should constitute the primary unit of biodiversity conservation as the geological structure and longitudinal nature of river system presents a natural barrier to many species and harbours local ecological and evolutionary processes. It can also be argued that the biosphere is continuous and that any boundary between ecological communities is artificial. Additionally, as planning and management initiatives tend to focus on administrative areas or catchments forcing conservation initiatives to fit within these confines, catchment level research provides a comparison to real world conservation planning. The use of vegetation types as a surrogate for biodiversity is justified by the high rates of floristic diversity and endemism witnessed in the Berg River catchment and the lack of adequate alternative biodiversity data.

Research Problem
Need for research on the use of land- cover change and vegetation types as indicators of biodiversity in the Fynbos Biome

Research Aim
Generate land-cover maps and examine and assess impacts of anthropogenic land-cover alteration on different vegetation types

Objectives
1. Select and justify the use of vegetation types as biodiversity surrogate
2. Generate and acquire land-cover and biodiversity data
3. Prepare and standardize acquired data
4. Accuracy assessment of land-cover data
5. Quantify the extent of land-cover change
6. Quantify and assess the impact of land cover transformation on biodiversity
7. Identify drivers of land cover and comment on implications of findings
8. Assess the capacity of Landsat-derived land-cover maps and vegetation type data to monitor biodiversity

Methods

Objectives 1 & 2
- Compare and assess different measures of biodiversity
- Identify appropriate measures of biodiversity and the impacts of human activity
- Identify biodiversity data available in South Africa

Objectives 2 & 3
- Obtain historical satellite imagery
- Devise classification land-cover classification legend
- Supervised classification of Landsat-5 and -7 imagery
- Manual editing - visual interpretation - existing data sets (Sanbi, national land cover data) - reference points – SPOT-5

Objective 4
- Obtain historical aerial photographs
- Generate random sample points
- Classify and compare against land-cover maps

Objective 5 & 6
- Assess extent of change
- Assess extent of change in vegetation-type distribution
- Comment on possible ecological implications

Objective 7
- Assess possible causes of change
- Assess management strategies in light of findings

Objective 8
- Comment on the potential and shortcomings of the research findings and compare with other studies in the area

Product
An historical analysis of the impacts of land-cover changes on the biodiversity of the Berg River catchment

Figure 1.4: Research design

However, it must be noted that an exclusive focus on the extent of natural vegetation as a surrogate for broader patterns of biodiversity can be described as artificially binary in that it divides areas into transformed and untransformed with the latter assumed to have no value to biodiversity (Rouget *et al*. 2003). This approach also neglects the differing impacts that various types of transformations can have and ignores vital issues such as connectivity and fails to comment on the structural and compositional integrity of untransformed areas. In this instance, vegetation type data is not intended to serve as an appropriate indicator of species richness and compositional diversity but represents a measure of the overall integrity of natural systems underlying the distribution of indigenous biodiversity.

1.7 REPORT STRUCTURE

This chapter has situated the research within the context of land-cover change and its relationship to biodiversity. It outlined the central aims and objectives of the project and provided a general description of the study area. In doing so it has sought to define the nature and relevance of this research, relating global trends to local circumstance.

Chapter 2 justifies the use of vegetation types as a biodiversity surrogate, provides the theoretical framework in which the research is based, evaluates the potential of alternative methods and ultimately seeks to justify the approach that was adopted. The role of remote sensing and geographical information systems (GIS) in recording and analysing land-cover change as it pertains to biodiversity and highlights the increasingly important role these technologies are playing in biodiversity analysis and monitoring are also discussed.

An overview of the methods and data used in this research is provided in Chapter 3. The accuracy of the derived land-cover maps as well as the limitations and potential pitfalls associated with data derived from remote-sensing devices are also discussed.

Chapter 4 presents and describes the results of the land-cover mapping and the spatial analysis that followed. This includes general land-cover changes and the associated of vegetation-type changes. The final chapter gives a detailed evaluation of the results and integrates the extent of land-cover transformation and its impact on biodiversity. Recommendations are made for future research. The thesis concludes with guidelines for the long-term management of the study area and a critical evaluation of the capacity of the approach used for monitoring biodiversity.

CHAPTER 2: VEGETATION TYPES AS A BIODIVERSITY SURROGATE

The term biodiversity has been deployed to replace well established terms such as species diversity and species richness, offering a rallying point for growing scientific and popular concern over increasing extinction rates and environmental degradation, and to mobilize public sentiment toward conserving species and ecosystems (Gaston 1996). However, the term is imprecise and open to redefinition and reinterpretation in various settings, causing some to question its scientific merit and the means by which it is assessed.

Owing to the complexity associated with considering all aspects of biodiversity at any given time, proxy measures, known as surrogates, are often used in the place of complete biodiversity data. A variety of surrogates have been suggested, each entailing its own set of advantages, limitations and assumptions. The salient findings of a literature study on biodiversity are laid out in the following sections. This chapter aims to explain the rationale behind the use of vegetation types and land cover as surrogates for biodiversity monitoring and to advocate the use of the Mucina, Rutherford & Powrie (2007) vegetation map as a suitable biodiversity surrogate for the study area in particular. Comparable studies will be explored to highlight the differing approaches to assessing and understanding the relationship between land- cover change and biodiversity, and their respective usefulness and limitations. The availability of biodiversity data for South Africa and the implications of this for biodiversity assessment and monitoring in the country are also discussed.

2.1 DEFINING BIODIVERSITY

The most prevalent usage of the term biodiversity is as a synonym for the variety of life on earth. Most definitions are simply expressions of, or expansions on, this basic theme. Biodiversity can be a state or attribute of any given area and refers to the variety within and among biological organisms, assemblages and biotic processes regardless of whether or not they have been subject to human interference. Biodiversity can be examined at any spatial scale ranging from microsites to the entire biosphere (De Long 1996). Major obstacles facing research in this field are applying an operational definition and quantifying biodiversity. The following subsections lay out the history, debate and implications of biodiversity research and describe how the use of the term guides conservation planning.

2.1.1 What is biodiversity?

Hamilton (2005) notes that the term biodiversity is used prolifically, but seldom defined explicitly and despite its phenomenal usage in scientific and popular literature since the early 1980s. Understandings and responses to biodiversity vary greatly with some authors even arguing that it is best conceived as a cluster of related terms or even as a symbol of all that remains unknown or uncertain in biology (Gaston 1996; Bunnell & Huggard 1999). Others present biodiversity as a multifaceted entity, of which only certain aspects can be measured at any given time (Duelli & Obrist 2003).

Initially the term biodiversity was used more in socio-political debates than scientific ones but quickly found its way into mainstream scientific discourse, presumably as a means of securing funding and as a way to bolster the impact of research (Hamilton 2005). Prior to the promotion of the term at the 1992 Rio convention previous work done in related fields focused on distilling human impacts on environmental and ecosystem health — often through the use of indicator species (Ferrier 2002). Although the term is still used to allude to environmental quality, the focus has shifted toward finding indicators of actual biodiversity in the wake of this convention (Ferrier 2002). According to De Long (1996) the term has always had an inconsistent meaning within the field of natural resource management, in part due to a deep-seated dissatisfaction with the lack of a concise and operational definition. Taken in its broadest form, a term such as biodiversity runs the risk of equating itself with the whole of contemporary biology, offering little utility as an entity that can be measured and analysed.

Conceptually, biodiversity is often considered to be multilayered with a hierarchical configuration creating levels of comprehension and analysis depending upon the scale at which it is considered. According to Gaston (1996) the simplest and most widely acknowledged of these divisions is the differentiation between levels of genetic, species and ecosystem diversity. Soulé (1991) identifies five distinct levels of biodiversity: genes, populations, species, assemblages and landscape-scale ecological systems. Advocates of Soulé's classification argue that such an approach is key as it places greater emphasis on intraspecies diversity, which is often ignored in biodiversity assessments and is crucial to the optimal management and conservation of biodiversity. Another classification hierarchy, proposed by Noss (1990), attempts to consider biodiversity as a complex interplay of three interdependent elements: compositional, structural and functional levels. This approach is often favoured by ecologists as it is said to shift the focus toward the interrelationships that exist between all levels of biological organization. Regardless of the debate that surrounds the appropriate configuration of biodiversity, Biggs, Reyers &

Scholes (2006) note that biodiversity is generally considered to denote the diversity of species, the abundance of different types and their distributions. Perhaps the most practical way of defining biodiversity is to consider it as an abstraction of the variation of life at differing levels of biological organization.

In a fashion, biodiversity can be viewed as spatially and temporally dependent. Bunnell & Huggard (1999) describe different processes that affect biodiversity across different spatial and temporal scales and they propose a hierarchical system of concepts related to representing biodiversity at varying scales. For example, a beetle may exist in an ecological space of only a few metres over a period of months while an elephant's domain may encompass thousands of kilometres over decades (Bunnell & Huggard 1999). This is important as the scale at which an assessment is conducted contains its own set of assumptions and compromises. Bunnell & Huggard (1999) also highlight one of the many potential shortfalls associated with the reduction of continuous ecological landscape to discrete entities as being a negation of the relationships that exist between organisms at all levels of organization. A question critical to the application of any surrogate is whether it is likely to inform or mislead research and planning. In this way the validity of any surrogate measure hinges on its ability to accurately represent the spatial or structural components of biodiversity in a given area. In short there will always be tradeoffs where complex issues are simplified for workability.

Despite its eminence, biodiversity remains an ambiguous term shrouded in socio-political connotations. This stands in stark contrast to useful and well defined ecological terms such as species diversity which biodiversity has largely replaced in scientific discourse (Gaston 1996). In retrospect it is perhaps the simplicity of the word that so easily belies the enormous complexity of the patterns and processes to which it refers and the paucity of our understanding of this complexity, even after many decades of research (Ferrier 2002). Still, its value is evidenced by its popularity and ability to draw attention to a wide range of environmental concerns and it is now inextricably linked to a widening focus in conservation that has moved beyond preserving particular species of ecological or social significance and instead seeks to preserve ecological functionality in its entirety (Faith 1996).

Thus it is crucial to define a set of appropriate features to adequately represent biodiversity patterns and processes and indicate how similar, or different, readily mapped areas are regarding their biodiversity. However, perhaps the most crucial aspect to consider when devising or selecting a surrogate is its ability to measure portions of the entity that can be related to a conservation target or goal.

2.1.2 Biodiversity and conservation priorities

Biodiversity assessments are usually conducted with the intention of informing conservation planning and prioritizing future research and management (Sarkar *et al.* 2006). As a consequence, separating biodiversity assessment from its conservation implications is often troublesome and a considerable degree of overlap is usually witnessed between the two. As much of the available literature on measuring and defining biodiversity has been directed at conservation efforts, it is reasonable to discuss the relationship between measuring and monitoring biodiversity and setting conservation priorities (Sarkar & Margules 2002). But it is important to acknowledge that assessing the risk faced by a particular species or ecosystem and setting conservation priorities accordingly cannot be equated to an assessment of overall biodiversity.

Conservation planning is primarily concerned with the identification of species and areas that are to be conserved (Stoms *et al.* 2005). This would usually entail selecting areas that harbour a desired assemblage of species, vegetation communities or environmental characteristics, often under the assumption that this will preserve a broader level of biodiversity. According to Lawler & White (2008) various methods of achieving this have been proposed and they are usually classified as being systematic, dynamic or more opportunistic site selection. Several authors have noted that the designation of protected areas in South Africa customarily focused on factors such as perceived aesthetic appeal and the value of the area in terms of agricultural or mining potential and such designations have often not considered biodiversity or ecological processes (Reyers *et al*. 2001; Lochner *et al*. 2003; Pressey, Cowling & Rouget *et al.* 2003; Reyers 2004). If conservation efforts are to be successful, some knowledge of the distribution of biodiversity in a given area is required. However, the optimal conservation of biodiversity in a given area is hindered by the enormous task of collecting data on every conceivable element thereof and the limited time and funds available to conservation planners (Pressey 2004). Consequently the problem has been addressed by the selection of factors that serve as surrogates for general patterns of biodiversity.

2.2 BIODIVERSITY SURROGATES

In the absence of extensive data on the state and distribution of biodiversity, surrogates are collated to provide practical measures of biodiversity. This process generally involves using species or other ecological measures whose distributions are already defined or can be easily determined. Common examples of indicator taxa include conspicuous organisms such as butterflies, trees and certain bird species while environmental factors such as remotely sensed land cover, vegetation and environmental gradients have been suggested as coarse-scale surrogates (Lawler & White 2008). The dominant types of surrogates, how they are used and the limitations associated with their use are presented in the following sections. However, it is first necessary to differentiate between the terms biodiversity surrogate and biodiversity indicator as the two are often used interchangeably in scientific literature.

2.2.1 Surrogates and indicators

Fundamental to the successful application of a biodiversity assessment is a robust understanding of the theory behind the use of surrogates and indicators. There is a lack of consensus on the precise meanings of the terms biodiversity surrogate and indicator but both are understood to refer to any measure that uses available data to draw inferences about more general patterns of biodiversity and to define unknown biodiversity features from known data (Faith 2003). By following strict definitions, an indicator refers to a specific component of biodiversity taken to broadly represent other components of the biodiversity of a region while a surrogate is a set of indicators that is taken to represent biodiversity in its entirety in the absence of more expansive data. In practice the terms are interchangeable and are frequently used synonymously (Faith 2003). Some authors have noted the recent replacement of the term indicator with surrogate as the latter is assumed to imply a more holistic and integrated appreciation of biodiversity in all its complexity.

When attempting to draw inferences about a vast and multifarious entity such as biodiversity, a lack of knowledge inevitably leads to a series of assumptions and abstractions that facilitate our understanding, research and conception (Santi *et al*. 2009). These are often exacerbated by temporal and fiscal constraints where the most appropriate surrogate that can be used or developed under set conditions is deployed (Santi *et al*. 2009). In this way biodiversity surrogates are used pragmatically, based on the resources at hand, to derive information on an entity which remains elusive.

2.2.2 Why use surrogates?

The rationale behind the use of biodiversity surrogates is fairly simple. The complexity of the biosphere means that it is impossible to possess complete knowledge of the biodiversity of any given region at any given time. However, as an understanding of the distribution and constitution of biodiversity as well as the processes which sustain or threaten it is pivotal to effective conservation planning and the management of natural recourses, gaps in biodiversity knowledge in many parts of the world force researchers and planners to rely on biodiversity surrogates (Ferrier 2002).

As noted earlier, biodiversity is an expansive and contentious term associated with knowledge gaps in many parts of the world. A further concern is whether biodiversity itself is to be represented or whether certain components of biodiversity are to be used as representations of a larger entity (Santi *et al*. 2009). Here a point of contention is how to investigate biodiversity without fundamentally altering its meaning or to provide measurements not wholly consistent with the scope of the term. Remaining cognizant of the differentiation between biodiversity in its entirety and management and measurement-related objectives is central to ensuring the utility of the concept of biodiversity (De Long 1996).

Much in the way that biodiversity is organized in a scale-dependent hierarchy, biodiversity surrogates can be grouped into various categories depending on the scale at which they are applied and the measures by which they are defined. Surrogates tend to be chosen in respect of the scale at which they are applied with coarse-scale surrogates such as environmental or community data used in large areas while taxonomic data tends to be used in localized studies (Mac Nally *et al.* 2002).

2.3 TYPES OF SURROGATES

Traditionally, measures such as indicator, umbrella and flagship species monitoring have been the dominant means used to assess and monitor biodiversity (Mac Nally *et al.* 2002). Currently the focus has shifted toward multilayered surrogates that often combine biological and environmental data. Less-common measures that incorporate socio-economic dimensions such as population growth and resource use, often as a means of assessing the threats that face particular ecosystems or ecological communities, are becoming increasingly popular (Mac Nally *et al.* 2002). Methods directed at conserving biodiversity processes, though rare, are also evident

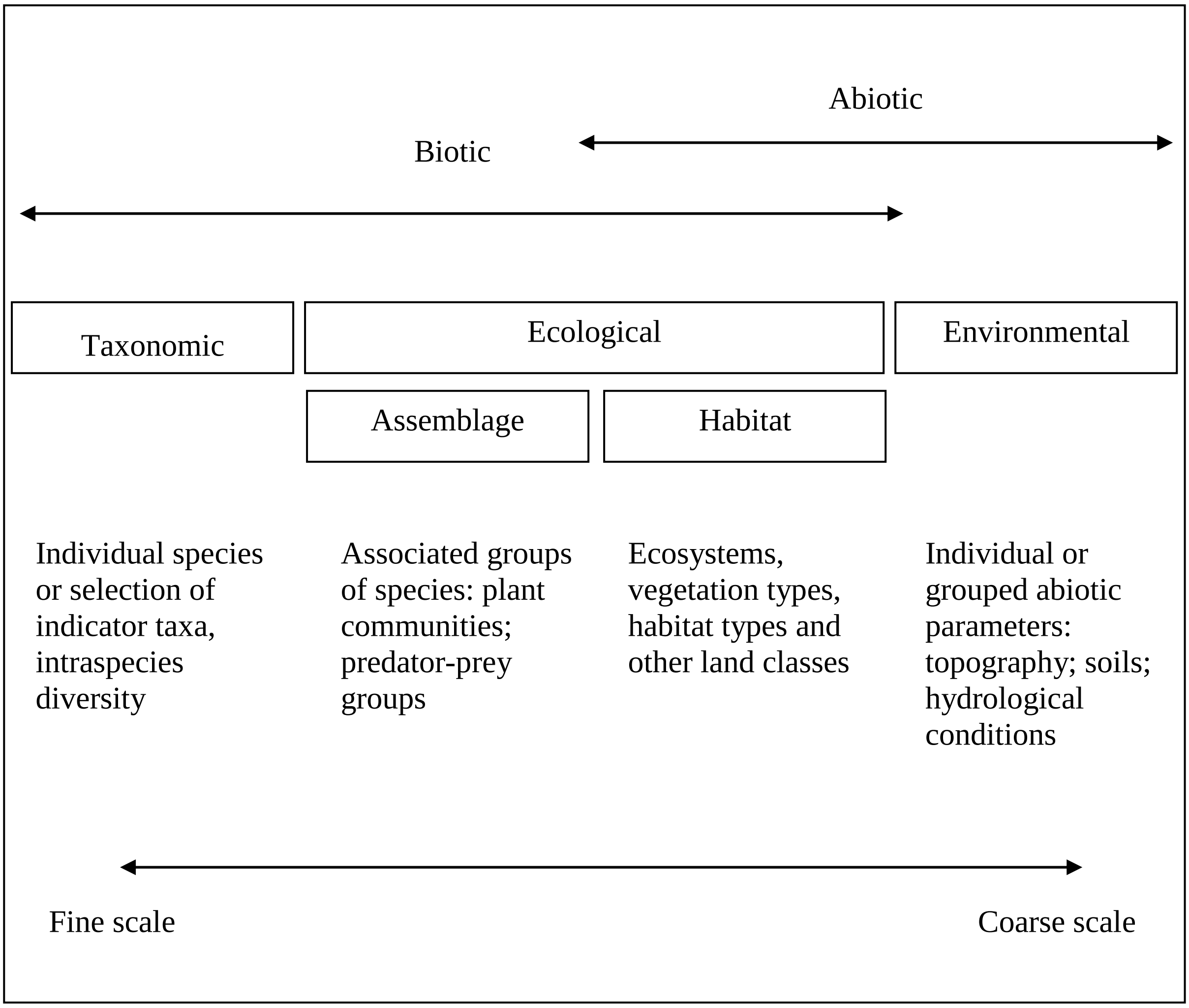
(Grantham *et al*. 2010). It is important to note that any given measure of biodiversity is unlikely to be all-encompassing but should be considered in conjunction with other measures and data.

This research considers a biodiversity surrogate as a measurable correlate to the entity which is to be assessed, that is a particular facet of the biodiversity of a given region. The following subsection describes the dominant forms biodiversity surrogates can take and discusses the potential and limitations of different approaches.

2.3.1 Surrogate classification

Biodiversity surrogates can broadly be divided into three categories: taxonomic, environmental and ecological (Sarkar *et al.* 2006; Grantham *et al.* 2010). *Taxonomic surrogates* use an indicator species or assemblages of species to provide an indication of biodiversity in a particular area. Such an approach stands in stark contrast to *environmental surrogates* which use a set of abiotic parameters thought to underlie the distribution of various organisms and ecological processes to model the distribution of biodiversity. *Ecological surrogates* tend to use a combination of taxonomic, environmental and other biogeographic factors to predict the distribution of unknown facets of biodiversity. Figure 2.1 illustrates the positioning of different surrogates and provides a framework to consider the relationship between different surrogates according to the data from which they are derived and the spatial scale at which they are applied.

In practice, much overlap exists between surrogates with most drawing upon multiple types of available data to make the best use of available resources. As a result, a continuum is best used to position different surrogates in relation to one another depending on the data from which they are derived. Some authors refer simply to fine- and coarse-scale surrogates which are differentiated by the scale at which they are applied and the level of detail they aim to capture (Grantham *et al.* 2010). The so-called 'coarse filter/fine filter hypothesis', described by Stoms *et al*. (2005), holds that by conserving examples of as many habitats and ecological systems as possible it is possible to preserve the majority of indigenous biodiversity but that such efforts should be complemented by more focused assessments where rare or otherwise vulnerable species are involved. Note that this does not constitute an approach in itself; rather it informs the use of multiple surrogates at different scales.



Adapted from Payet (2007:13)

Figure 2.1: Framework for differentiating biodiversity surrogates

In some instances multiple surrogates have been amalgamated or compared to overcome the limitations inherent in the use of surrogate measures of biodiversity. According to Rodriguez & Brooks (2007) and Spangenberg (2007) uncertainties about the choice of biodiversity surrogates remain, largely because they cannot be rigorously tested against empirical data and the biodiversity of only a handful of regions has been documented in sufficient detail to cater for accurate assessments. Currently there is no consensus within the scientific community on which surrogates provide the best overall measure of biodiversity in any given area, they are usually selected to satisfy the objectives of a particular project within the constraints of available resources (Sarkar *et al.* 2006).

2.3.1.1 Taxonomic surrogates

When attempting to represent patterns of biodiversity in conservation areas, biodiversity surrogates used by planners include some of the better-known taxonomic groups, focal species, umbrella species and various species assemblages often dubbed indicator or select taxa. The use of surveyed species to predict the distribution of unsurveyed species and assess local biodiversity is an extensively used technique due to the centrality of species diversity in definitions of biodiversity (Reyers & Van Jaarsveld 2000). The conservation of individual plant and animal species has long been advocated by the International Union for the Conservation of Nature (IUCN) and has subsequently featured prominently in many biodiversity and conservation initiatives (Rodriguez, Balch & Rodriguez-Clark 2007).

The rationale underlying the use of such an approach is the assumption that the distribution of undocumented species will correspond to the distribution of certain documented species. This is to be expected as the distributions of species are likely to correlate with one another due to ecological and evolutionary process such as pollination or predation. Well-researched or highly conspicuous groups of species, such as birds, butterflies or large vertebrates tend to be the focus of such undertakings owing to the ready availability of their distribution data (Gaston 1996; Walther *et al*. 2007; Walther, Van Niekerk & Rahbek 2011). Another approach is the assessment and monitoring of so-called umbrella species which are supposed to provide an indication of the integrity of ecological communities (Sarkar *et al.* 2006). The distribution of these species usually takes the form of point occurrence data acquired from museum collections or field surveys. Point occurrence data can be used in their raw form but are often extrapolated using a variety of statistical techniques to provide a continuous modelled distribution (Ferrier & Guisan 2006; Rodriguez, Balch & Rodriguez-Clark 2007).

The use of taxonomic surrogates is widely considered to have evolved from measures such as environmental health which used sensitive species to measure the impacts of anthropogenic disturbance such as the effects of pollution or other forms of contamination on species abundance (Gaston 1996). A common example would be the use of select groups of aquatic invertebrates which respond clearly to the presence of pollutants, or other forms of disturbance, to assess the well-being of a river system and draw inferences about the other organisms linked ecologically to these species (Gaston 1996). In such an instance this approach may prove highly effective as it is sensitive to ecological processes common to many species. Williams *et al.* (2006) note that the efficacy of species-based surrogacy decreases as the area of interest increases lending support to the idea that taxonomic surrogates are best deployed in small, data-rich and high-priority areas.

However, there is considerable debate over what species should be used and how they are linked to broader patterns of biodiversity. Some argue that particular attention should be given to invertebrates as they are likely to constitute the vast majority of species diversity in any given area (Panzer & Schwartz 1998). Others argue that predatory species, such as eagles, make better surrogates as they depend on prey animals which are in turn dependent on myriad species and ecological processes (Machange, Jenkins & Navarro 2005). It follows that irregularities in predator populations are indicative of changes throughout the food chain. Studies over the course of many years have lent considerable support to the idea that some species may serve as good benchmarks for overall biodiversity (Lawler & White 2008). However, most of these have been carried out in localized areas and are unlikely to yield comparable results when applied to other areas. In other studies different species have been found to display insignificant correlations in their spatial distribution (Lawler & White 2008). This has the potential to significantly skew research findings or render predictions concerning other species specious. This concern is further compounded by the inaccuracies and paucity associated with species data and has led to a marked lack of consensus on how to best select surrogate species or assemblages (Lawler & White 2008).

As measures of biodiversity have progressed, conspicuous shortcomings have detracted from taxonomic surrogates as appropriate measures of biodiversity despite their intuitive appeal (Mac Nally *et al.* 2002). At the forefront of these is that most species in any given area have not been described and for those which have data on their spatial distribution and abundance they are fraught with inaccuracies (Grantham *et al*. 2010). This concern is compounded in the case of rare or otherwise inconspicuous taxa and in data-poor areas, particularly in the developing world (Grantham *et al*. 2010). A common complaint about a species-orientated approach is that it tends to display a strong sampling bias which has the capacity to radically alter research findings and policy pronouncements that follow. Collecting species-level data is also a costly and time-consuming affair requiring an intricate knowledge of the biodiversity of a given area (Lawler & White 2008).The result is that many taxonomic surrogates are unlikely to operate within the constraints of conservation planning or scientific research.

It must also be noted the niche requirements of species are likely to be more flexible than often assumed and can change over time (Spangenberg 2007). Lawler & White (2008) conclude that the limited associations between various taxa produce very little correlation when select indicator taxa are used as a biodiversity surrogate. Often when species-distribution data are used, data-rich areas are often assigned greater biodiversity and risk levels than those where information is scant or inconsistent leading some to assume an inherent bias in this practice (Mac Nally *et al*. 2002; Spangenberg 2007). Additionally, the number of species present in any given area cannot be considered an appropriate indicator of biodiversity in all its dimensions and says almost nothing about genetic and ecosystem diversity (Spangenberg 2007). The use of flagship taxa has been contested on the grounds that no single species can reliably indicate the presence of another and such an approach has the potential to be highly misleading if not used correctly.

Concerns associated with taxonomic surrogates have led some to suggest that effort would be better spent developing more suitable alternative measures of biodiversity and conservation value. Some consider the prolific use of indicator species a 'sentimentalist' approach to conservation where the desire to conserve select, charismatic or popular species cloud the real issues involved, namely the long-term integrity of biological systems, and they argue that measures are needed to assess biodiversity in a more holistic manner (Santi *et al.* 2009). It has also been observed that a taxonomic approach negates the role of higher levels of biological organization and that knowledge of the processes that sustain biodiversity are not considered (Kontula & Raunio 2009). As an alternative it has been suggested that effort be made to consider factors that underlie the distribution of a wide range of organisms as these can capture the essence of biodiversity in a way that taxonomic surrogates cannot.

2.3.1.2 Environmental surrogates

When biological data are limited, abiotic indicators are often used as surrogates to indirectly assess the distribution of organisms and ecological communities. These are considered coarse-scale surrogates and are used to compensate for the inaccuracy or limited availability of species-level data. When compared to taxonomic surrogates, environmental surrogates can be derived quickly and inexpensively, are consistent across large areas and are often the only reliable data available in certain areas (Lombard *et al*. 2003; Grantham *et al*. 2010). Oliver *et al*. (2004) note that the use of land systems or environmental systems is increasingly popular and their findings demonstrate that land systems do represent biodiversity by broadly mirroring patterns of similarity and difference of organisms. However, the range of many organisms overlaps with most conceivable land classes suggesting that a land systems-approach could be complemented by species data (Oliver *et al*. 2004).

The use of environmental surrogates is justified by the dependence of organisms upon a particular set of environmental conditions and ecological interactions; this is the concept of an ecological niche (Margules, Pressey & Williams 2002; Faith 2003; Faith, Ferrier & Walker 2004). A given species will respond to variation in environmental factors such as temperature, moisture and soil nutrients in a particular way that is, in turn, influenced by interactions with other species (Margules, Pressey & Williams 2002; Faith 2003; Faith, Ferrier & Walker 2004). In this way the physiology and genetic makeup of an organism bounds it to a certain set of environmental conditions upon which its habitat depends (Margules, Pressey & Williams 2002). This will result in spatial patterns of distribution regarding abundance or absence corresponding to more- and less-favourable habitats for certain species. It follows that variation in the environment underpins the geographic distribution of species and the ecosystems they represent. An effective environmental surrogate assumes that environmental diversity provides a relative measure of species diversity and that species respond in set ways to changes in environmental variables (Faith, Ferrier & Walker 2004; Grantham *et al*. 2010).

Environmental surrogates have traditionally been represented as discrete land classes or areas of common or overlapping environmental factors (Faith 2003; Faith, Ferrier & Walker 2004). Zones of homogenous climate, geology and topography are common factors used for delineating land classes (Van Niekerk 2010). These factors are often associated with the distributions of well-documented species as there is consensus that such an approach still requires some knowledge of how particular species are distributed if they are to be considered at all effective (Faith 2003; Faith, Ferrier & Walker 2004). In some applications environmental factors have been used as the only criteria by which biodiversity is represented though this approach has been heavily criticized (Keith *et al.* 2009). It has also been advocated that environmental factors provide a sound basis for identifying spatial surrogates for ecological processes (Biggs *et al*. 2008). By preserving a wide array of environmental diversity it is, therefore, possible to preserve the ecological and evolutionary processes that generate and sustain biodiversity.

However, there is marked disagreement over the degree to which environmental factors are responsible for biological diversity, some species exhibit high niche dependence while others appear resilient to environmental and ecological variation (Faith 2003; Faith, Ferrier & Walker 2004; Biggs *et al*. 2008). Geology and climatic factors are often considered good surrogates for mapping the distribution of plant species but not necessarily for fauna (Galley & Linder 2005). Precipitation, altitude and substratum can be considered as refined measures of the broader environmental attributes found to be particularly good predictors of the distribution of certain plant species in the CFR (Galley & Linder 2005; Botes *et al*. 2006). This relationship is largely the product of the restriction of many of the sampled species to certain altitudes in the CFR. Additionally, environmental processes such as hydrological regime are important to maintaining biodiversity, essentially retaining ecosystem integrity to promote long-term biodiversity (O'Conner & Kuyler 2009).

A vital aspect related to the use of environmental diversity as a measure of biodiversity is how similarity and dissimilarity are defined because inconsistency and scale can result in large differences in outcomes (De Long 1996). This is partly due to an overall lack of knowledge about the distributions of most species and this can only be resolved satisfactorily through extensive further research (Faith 2003). However, limited knowledge of species, their conservation status and their distributions could be advanced as reasons for the preservation of habitat heterogeneity as this may be the best way to preserve most biodiversity in the long run and that the conservation of particular species may be counterproductive in that they may have very little ecological importance and divert resources away from conservation initiatives having better overall potential.

The application of environmental surrogates has yielded mixed results. Velásquez *et al*. (2003) developed land units, areas of similar environmental conditions associated with species assemblages, as mechanisms for overcoming the many biases associated with species-distribution data and the limitations of environmental data. It was found that patterns of biodiversity broadly correlated with the derived land units but that this correlation disintegrated at finer scales. Wessels, Freitag & Van Jaarsveld (1999) investigated the use of land facets, areas of uniform topography, soils and hydrological conditions, as surrogates for biodiversity. The results yielded high positive correlations between environmental factors and bird and dung beetle assemblages but correlations between mammals were slight and negligible.

Some areas, such as wetlands and vernal pools, which serve as breeding areas for birds and amphibians and thus essential to biodiversity in certain areas, are unlikely to be considered when environmental data alone is used (Scott *et al*. 1993; Faith 2003). Another pitfall associated with the use of environmentally-derived biodiversity surrogates is that species which exhibit a highly constrained distribution, or one that is not linked to environmental factors, are regularly overlooked and so fail to gain an appropriate level of recognition or protection (Biggs *et al*. 2008). This closely relates to the criticism that environmental surrogates are inclined to overlook habitats and ecological processes at a fine scale (Keith *et al.* 2009). In general, using environmental data involves many limitations and potential sources of error with different studies reporting differing degrees of effectiveness depending on the surrogates used and the area in which they were employed. The boundaries produced by environmental factors such as soil or rainfall regimes are also unlikely to correspond to large compositional changes in species assemblages (Keith *et al.* 2009). A further criticism of the use of environmental surrogates is that correlation is often mistaken for causation and the application of an environmental surrogate risks erroneously associating unrelated factors (Faith 2003; Biggs *et al*. 2008).

To summarise, an extensive review of the biodiversity literature has revealed mixed views on the use of environmental attributes as surrogates for biodiversity with some studies reporting a marked correlation between environmental diversity and biodiversity while others have not. Evaluations of environmental surrogates are apt to remain highly subjective because the biological data required for their assessment are lacking. It is generally accepted that the coarser the scale of application the closer environmental surrogates will correspond to biodiversity and that much of this correlation disappears at finer scales (Faith 2003; Faith, Ferrier & Walker 2004). This implies that environmental surrogates are best used at a broad scale or in conjunction with more detailed data. There has been a growing realization that an exclusive focus on either species or environmental factors is unlikely to represent all the components of biodiversity (Poiani *et al*. 2000; Loreau, Naeem & Inchausti 2002; O'Connor & Kuyler 2009). O'Connor & Kuyler (2009) assert that biodiversity assessment and management have shifted away from narrow measures of biodiversity that neglect the interactions of different elements in a landscape to a holistic perspective that recognizes the need to conserve dynamic, multiscale ecological processes.

2.3.1.3 Ecological surrogates

Measures that seek to integrate biological and environmental data to provide a holistic approach to biodiversity assessment are often referred to as ecological surrogates and they depict biological systems as complex assemblages of species with myriad ecological interactions (Poiani *et al*. 2000). The rationale behind this approach is that an optimal number of species is thought to be subsumed by ecological measures of biodiversity in the absence of complete biological data and this belief has consequently proved to be popular in conservation planning (Sarkar *et al.* 2006; Grantham *et al*. 2010).

How an ecosystem is defined appears to be a significant consideration in the effective implementation of an ecosystem-orientated approach as there are no absolute means of defining and delineating an ecosystem. An ecosystem is generally considered to represent a relatively unique and homogenous arrangement of species, environmental factors and the dynamic processes which develop between them, although definitions may vary considerably (Rodriguez, Balch & Rodriguez-Clark 2007). The foremost proponents of the use of ecological surrogates have been conservationists seeking to determine the threat status of ecological communities as part of their planning efforts. An ecological surrogate can refer to areas of widely differing spatial scales depending on the objectives of its use and the data available (Grantham *et al*. 2010). Ecology can be defined as a system of relationships formed by communities and their environments (Gaston 1996). Subsequently, species assemblages have been proposed as instrumental components of a holistic ecological surrogate. While such an approach inherently places emphasis on interactions between species, conservation efforts have often downplayed relationships and focused on the co-occurrence of species and the area of co-occurrence to devise an ecological surrogate (Mac Nally *et al*. 2002; Keith *et al*. 2009). Although such approaches have been widely adopted, they are seen to be pragmatic rather than scientifically sound.

The conservation of ecological communities is held to protect processes and patterns such as the interactions between species and their environments, although some scholars maintain that the patterns of biodiversity need to be better understood before the processes that bring them about can be alluded to (Gaston 1996; Sarkar *et al.* 2006). It is further claimed that protecting these processes secures ecosystem services and thus provides a socio-economic incentive to study and conserve biodiversity (O’Connor & Kuyler 2009). Ecological surrogates are usually derived from a variety of physical and biological data and often rely on estimating the distribution of various species or assemblages by using environmental data that underpin the distribution of said organisms, such as precipitation, soils or topography (Reyers *et al*. 2001; Fischer & Lindenmayer 2007; Grantham *et al*. 2010). A salient example is an area of comparable hydrological, topographic and climatic conditions which also displays a typical set of species. The selection of data from which an ecological surrogate is collated is guided by a number of factors. These include the availability of data and varied perceptions and understandings of the significance of particular variables in shaping biological distributions (Faith 2000; O’Connor & Kuyler 2009; Grantham *et al*. 2010).

Scott *et al*. (1993) devised GAP analysis as a means of assessing the extent of biodiversity located within a reserve system, largely by predicting the distribution of species from point occurrence data by modelling their likely distribution on environmental factors deemed to determine the spatial distribution of a particular species. Since the introduction of GAP analysis there has been a growing recognition that an amalgamation of species and environmental data perhaps provides the best alternative to the narrow focus of either taxon- and environmentally-orientated approaches.

The use of habitat types is often used as a means of engaging with ecological-type relationships without making direct reference to ecological processes, sometimes referred to as biotopes (Kontula & Raunio 2009). In this sense a habitat type refers to a terrestrial or aquatic area with distinguishing environmental and biological characteristics which differ from surrounding areas (Kontula & Raunio 2009). The broad habitat units (BHUs) developed by the South African National Biodiversity Institute (SANBI) and CapeNature is a local example of this approach where well-established correlates between various environmental and biological data are used to designate areas of comparable biodiversity (Cowling & Heijnis 2001). Fischer & Lindenmayer (2007) propose the association of species and habitats as a functional measure of biodiversity that can be represented spatially. This is achieved by calculating an index of species representivity for various habitats by examining the spatial coincidence of species distribution and habitat or vegetation types (Fischer & Lindenmayer 2007).

The utility of ecological surrogates is their ability to assess loss and degradation at different spatial scales (Rodriguez, Balch & Rodriguez-Clark 2007). Ecological degradation is generally assessed by changes in species composition, structural changes and disruptions to ecological processes such as a decline in ecosystem functionality in response to anthropogenic pressures (Faith 2000; Keith *et al*. 2009). However, measuring these factors is challenging and presents a unique set of difficulties. Defining the threshold levels at which one considers ecosystems to have suffered adverse effects or become extinct is particularly testing as the relationship between biodiversity and ecosystem integrity is highly contentious (Faith 2000; Keith *et al*. 2009). Currently there are no universally accepted criteria for defining these thresholds for ecosystems and there are difficulties associated with defining community extinction as opposed to species extinction (Keith *et al*. 2009).

Ecological surrogates have been particularly effective in identifying priority areas for conservation initiatives but they seldom provide a direct indication of biodiversity loss as the thresholds at which particular species decline in response to ecosystem disruptions are not well established (Muradian. 2001; Fischer & Lindenmayer 2007). Communities usually exhibit greater spatial variability than individual species as their composition varies from place to place, consequently requiring larger areas than species to represent their full diversity (Bunnell & Huggard 1999). While ecosystems and species occurring over a limited spatial extent may be overlooked in this approach and the relationships between potential species distribution and abundance remain uncertain, the approaches value in broad-scale conservation planning is clear (Reyers *et al*. 2001). As a result of these limitations many studies have failed to adequately incorporate the effects of habitat reduction on nested ecological communities and the associated intricate interrelationships. Consequently more detailed analyses of these relationships are called for to build a more comprehensive understanding of the impacts of land-cover change on biodiversity at a finer scale. However, the use of an ecological surrogate – coupled with information on anthropogenic pressures such as land-cover or land-use change – provides a platform from which ecological degradation can be assessed and to focus fine-scaled research.

2.3.1.4 Pressure-based assessments

Biodiversity assessment and conservation are widely regarded as important, if not pressing, concerns. However, much of the scientific literature on biodiversity is related to its systematic scrutiny with the forces driving biodiversity loss receiving less attention (Settele 2005; Spangenberg 2007). Spangenberg (2007) contends that this has inhibited adequate responses to biodiversity loss and as an alternative he suggests that the pressures that face biodiversity be analysed.

A pressure-based assessment focuses on the factors that drive biodiversity loss as opposed to focusing on the extent of biodiversity loss. The rationale behind this kind of approach is that the most effective action that can be taken to preserve biodiversity is to reduce the pressures facing it. Different levels of analysis can be used depending on the scale of a research or planning initiative in conjunction with political instruments within a political or geographic boundary (Spangenberg 2007). Such an assessment includes physical primary drivers such and socio-economic, demographic and land-cover changes as well as secondary drivers such as policies and institutional structures (Spangenberg 2007). Exploring these factors bridges the divide between socio-economic threats to biodiversity and their likely biological impacts which have traditionally occupied separate research spaces and, provided they are of a sufficient temporal depth, could be used to predict threats to biodiversity in the near future. Such an approach should not be limited to socio-economic factors and Settele (2005) has proposed an assessment regime that considers factors such as invasive alien species, pollution and climatic change.

The pressures that face biodiversity are seldom random and are often closely linked to patterns of anthropogenic activity especially land-cover and land-use change (Rodriguez, Balch & Rodriguez-Clark 2007). Acknowledgement of this could be used to hone the scope of assessment and designate priority areas for detailed research so making better use of limited resources. Because of increasing anthropogenic impacts on the biosphere; the integration of pressure-based assessments into biodiversity monitoring and management will likely be decisive if conservation initiatives are to succeed. However, pressure-based assessments must also incorporate biophysical data wherever possible (Spangenberg 2007). While most assessments of biodiversity allude to various pressures that threaten it, few have actively sought to measure these pressures or devise surrogates that can incorporate factors that drive biodiversity loss. Perhaps the most appropriate way of embarking on a biodiversity assessment is to consider as many factors as is feasible.

2.3.2 Conclusion

A variety of surrogates and approaches geared toward biodiversity assessment have been discussed in the preceding sections. Measuring biodiversity is a complex undertaking and no universally agreed upon standards exist. However, several trends were observed in the reviewed literature. First, a move away from narrow taxon-based approaches and toward broader, more inclusive surrogates has been recognized. Second, there is a pressing need to devise measures of biodiversity that can incorporate anthropogenic threats. Vegetation has been identified as a widely applied biodiversity surrogate that combines taxonomic, ecological and environmental data as discussed in the following section. A biodiversity surrogate derived from vegetation data can easily be considered in conjunction with land-cover data as there is a direct relationship between land-cover and the distribution of vegetation. This makes it possible to assess the factors that contribute to biodiversity loss in regard to anthropogenic activity. However, the use of vegetation as a biodiversity surrogate is based on many assumptions and is subject to criticism. This is explored in the following section.

2.4 VEGETATION AS A BIODIVERSITY SURROGATE

Using vegetation as a measure of biodiversity is a popular means of overcoming financial and data limitations in conservation planning and assessment (Lawler & White 2008). Keith *et al.* (2009) state that the pace and magnitude of biodiversity loss has led to widespread recognition that efforts to conserve individual species must be complemented by action directed at ecosystem and landscape scale. This follows from the assumption that because most species depend upon the functions of their habitat for survival, maintaining habitat is the most appropriate way of conserving species. This section describes how and why vegetation is used as a biodiversity surrogate and argues for the adoption of a vegetation-type approach to biodiversity assessment and conservation planning.

According to Lawler & White (2008) the use of all surrogates relies on a predictive relationship between the variable which is to be used and the target which it is to measure. Santi *et al*. (2009) submit that vegetation should be considered a useful tool in this regard because it provides habitat and energy for most species, constitutes the bulk of biomass in most ecosystems and supports crucial functions of the biosphere at all scales. Vegetation is largely responsible for regulating the flow of biochemical cycles, strongly affects soil characteristics and regulates the composition of the atmosphere, and it is also important in local and global energy balances which are essential to vegetation and climate regulation (Santi *et al*. 2009). Vegetation is thus intricately linked to general patterns of biodiversity in most areas and theoretically serves as an appropriate proxy by which to judge the state of biodiversity in a given area.

If consistently applied, a surrogate such as vegetation type maps coupled with accurate land-cover maps, is capable of providing defensible generalizations of biodiversity distribution and status (Ferrier 2002; Santi *et al*. 2009). Indeed, many authors believe that changes in land use, land cover and vegetation are some of the most important broad-scale indicators of environmental and ecological change (Zhang, Zhengjun & Xiaoxia 2009). Rapid advances in GIS and remotely-sensed data on land cover and vegetation provide a platform to coherently integrate disparate studies. Vegetation-based surrogates are popular because they are derivable using predictive modelling and remote sensing, so overcoming the financial and temporal constraints of taxonomic data collection (Ferrier 2002; Santi *et al*. 2009). An approach which uses remotely derived data such as land cover and vegetation types is common in data-poor areas where this may constitute the only expansive and reliable biodiversity data available (Mac Nally *et al.* 2002). Another appeal of this approach is that there is a direct relationship between indigenous vegetation diversity and land cover (Haines-Young 2009). Land-cover changes are highly detrimental to plants and consequently they are likely to have significant impacts on animals too (Santi *et al*. 2009).

The ideal of considering all species compels researchers to acknowledge and attempt to distil the processes governing species persistence. Perhaps the most pressing argument for the use of vegetation types as surrogates for biodiversity is their ability to indirectly represent evolutionary and ecological processes that are not sufficiently understood to be considered on their own (Lombard *et al*. 2003; Santi *et al*. 2009). While the application of vegetation-type surrogates has the potential to capture many aspects of ecology at a coarse scale, it is important to bear in mind that there will always be limitations to and assumptions about the available data. Plant communities have been found to correlate fairly well with overall species diversity, with indigenous mammals and reptiles showing a tendency to follow the decline of indigenous vegetation whereas the responses of amphibians and birds are more difficult to predict (Santi *et al*. 2009).

Vegetation as a surrogate for overall biodiversity allows comparison across a region of interest without gaps in the data and can serve as a reasonable indicator of faunal diversity (Santi *et al*. 2009). Indigenous fauna, especially large mammals, are inclined to follow the distribution of natural vegetation although this relationship varies and is sometimes difficult to predict (Gaston 1996; Santi *et al*. 2009). Panzer & Schwartz (1998) found that plant diversity is apt to correspond fairly well to insect diversity and it provides one of the better biodiversity surrogates, but they stress that wherever possible vegetation should be coupled with other species data. Mac Nally *et al.* (2002) probed the potential of using ecological vegetation classes as a biodiversity surrogate. The results were encouraging and the surrogate performed very well in predicting the distribution of trees, birds and mammals but produced questionable results when considering the distribution of reptiles and amphibians. Gould (2000) used remotely-sensed imagery to delineate vegetation types and then calculated species richness by surveying these vegetation types.

The mixed results yielded by vegetation as a surrogate for biodiversity are due to some species displaying remarkable resilience to the alteration of indigenous vegetation with others requiring specific habitat and ecological conditions to survive (Panzer & Schwartz 1998; Mac Nally *et al.* 2002; Oliver *et al.* 2004). Nevertheless, vegetation is a widely applied surrogate for biodiversity that, despite several shortcomings, remains an integrated and reliable indicator of spatial patterns of biodiversity.

The transformation of natural vegetation to other land-cover types, such as urban or agricultural land, is widely regarded as the single most pressing threat to global biodiversity through the loss, degradation and isolation of habitat and populations (Haines-Young 2009). However, it is important to not conflate a reduction in the spatial extent of a vegetation type with a decline in the structural, functional and compositional features of the population it harbours. Nevertheless, changes in the spatial extent of natural vegetation provide well-founded indications of changes in biodiversity and can be used to focus research on areas experiencing large changes in vegetation cover. De Long (1996) is critical of the use of vegetation as a biodiversity surrogate on the grounds that this does not, in itself, constitute biodiversity. A vegetation type is at best an abstraction delineated by theoretical and practical constraints (Rodriguez, Balch & Rodriguez-Clark 2007). However, given the leniency with which the term biodiversity is used, it is reasonable that vegetation as an aspect of biodiversity facilitates assumptions about more general patterns of biodiversity in the absence of more extensive data.

The utility of vegetation types as a biodiversity surrogate critically hinges on the degree of similarity within and dissimilarity between classes (Oliver *et al.* 2004). There is considerable debate about how mapped vegetation and different vegetation types are classified. The scale at which vegetation is mapped and the means by which it is achieved have significant bearings on a vegetation type map's ability to represent patterns of similarity and dissimilarity among other aspects of biodiversity (Oliver *et al.* 2004). It can be argued that the problem presented by the lack of consensus concerning classification of vegetation types can be overcome by the simple acknowledgement that, provided the classification process is explained, any vegetation classification has inherent value by the differentiation of heterogeneous species assemblages (Oliver *et al.* 2004).

Seen from a conservation perspective, if a wide variety of indigenous vegetation is represented in a local reserve system the chance that most of the indigenous of biodiversity will be accommodated is very good (Gould 2000; Santi *et al*. 2009). Velásquez *et al*. (2003) submit that effective conservation efforts rely on the maintenance of habitat on which individual species and communities depend. A vegetation-type approach makes comparisons between different management scenarios straightforward as assessing their relative representation in a reserve system is simple and inexpensive (Helmer *et al*. 2002; Santi *et al*. 2009). A common concern though is that specific species with specific habitat requirements will be ignored (Panzer & Schwartz 1998). A more promising approach is to consider plant community or assemblage patterns as these are likely to contribute substantially to patterns of species richness and diversity.

Despite several drawbacks the use of vegetation types as a surrogate for biodiversity is a valuable and relatively simple means of biodiversity assessment. The marriage of taxon and environmental data in vegetation type surrogates represents a powerful tool which allows researchers to capitalize on the strengths of different approaches, combining the precision of species-level data with the reliability of environmental data (Faith, Ferrier & Walker 2004; Poiani *et al*. 2000). Results generated using this type of approach are likely to broadly reflect patterns of biological similarity and dissimilarity as well as the various ecological processes that underpin patterns of distribution. In areas where taxonomic-level biodiversity data is limited vegetation types surrogates are likely the best means by which to assess and biodiversity, especially when vegetation type data can be combined with species distribution data. In South Africa the poor quality of species distribution data has led most large scale assessments to rely on vegetation type surrogates. This will be discussed in the next section.

2.5 BIODIVERSITY DATA AND ASSESSMENT IN SOUTH AFRICA

South Africa is regarded as being exceptionally biodiverse but regrettably the task of assessing the state of the country's biodiversity is hindered by the poor quality of the available species-distribution data. Much of the available data is recorded in quarter degree squares (QDS) which may render the results of regional-scale studies moot (Cowling & Heijnis 2001). These data sets also display a strong sampling bias with records concentrated around museums and universities and they are liable to be collected from easily accessible or otherwise conveniently surveyed areas (Van Jaarsveld *et al*. 1998; Cowling & Heijnis 2001). Furthermore, many of the records on which the distribution data are based are not recent, some dating to the early 19th century (Cowling & Heijnis 2001). Species-level biodiversity data are unlikely to be efficacious in South Africa owing to the scarcity of existing records and lack of correlation between species density and the distributions of rare and threatened species (Van Jaarsveld *et al*. 1998).

This section describes biodiversity data available in South Africa and presents large scale biodiversity assessments that have been conducted. The discussion will focus on the CFR, highlight the importance of vegetation data in this area and move to justify the vegetation map by Mucina, Rutherford & Powrie (2007) as the most accurate, detailed and appropriate biodiversity surrogate available for the CFR.

2.5.1 National Scale biodiversity assessments in South Africa

South Africa's biodiversity Act requires the development of a national framework for the management of biodiversity (DEAT 2005). The Act also recommends the regular monitoring of the status of biodiversity. As a result of the dearth and bias of South African biodiversity data, national-scale biodiversity assessments and monitoring projects have combined historical vegetation maps, such as that of Low & Rebelo (1996) and the more recent one of South Africa, Lesotho and Swaziland, compiled by Mucina, Rutherford & Powrie (2007), with land-cover maps to identify disturbed ecosystems, set conservation priorities and direct more intensive research (Reyers *et al*. 2001; De Villiers *et al*. 2005; Reyers *et al*. 2007; O'Connor & Kuyler 2009). The National Spatial Biodiversity Assessment (NSBA) used a version of the Mucina & Rutherford (2006) vegetation map to designate ecosystems. National land-cover (NLC) maps were used to establish the status of individual ecosystems by calculating their untransformed extent.

These assessments have provided a versatile framework for considering the threats that face biodiversity in a given area. However, according to Wessels *et al*. (2003) the rate of change in the spatial extent of ecosystems in South Africa and much of the world is generally not well documented and this could be critical in the future optimal designation of conservation areas. In the South African context an analysis of land-cover change in conjunction with vegetation type data may add integrity to biodiversity assessments. By identifying areas of rapid change, conservation efforts can be prioritized and appropriate planning actions can be taken based on the nature and severity of threats. Given the richness of biodiversity in the country, the relative underrepresentation of many ecosystems in protected areas and the limited resources with which conservation initiatives operate, it is imperative that expeditious and cost-effective methods of biodiversity assessment and monitoring be established.

2.5.2 Biodiversity assessment in the Cape Floristic Region

The CFR is one of five floral kingdoms of the world. It is recognized as having the highest concentration of known plant species of which roughly 70% are endemic (Pence 2008). The exceptional levels of plant diversity observed in the CFR are quite likely the product of the highly heterogeneous edaphic conditions, particularly regarding soil nutrient and moisture content, as well as highly varied precipitation and topography (Thuiller *et al*. 2006). A focus on vegetation types in the CFR will probably be contentious because of the high physical and biological heterogeneity witnessed in the region but it will still provide important information owing to the dominant role vegetation plays in determining the distributions of various species of plants and animals (Younge & Fowkes. 2003; CapeNature 2005; De Villiers *et al*. 2005). The designation of vegetation types will also help protect the environs that serve as incubators for speciation in this area (Thuiller *et al*. 2006).

According to Lombard *et al*. (2003), a potential limitation of the use of vegetation types as a biodiversity surrogate in the CFR is that the sporadic contemporary distribution of many species, especially vertebrates, has been largely determined by historical rather than ecological processes. Many invertebrates also exhibit notoriously patchy distribution but these are more closely linked to vegetation and environmental factors (Lombard *et al*. 2003). The overall conclusion is that a vegetation-type surrogate will serve as a good indicator of plant diversity in the CFR but it should be used with caution when attempting to draw inferences about patterns of vertebrate and invertebrate biodiversity. A crucial unresolved question, according to Lombard *et al.* (2003), is how current abiotic and biotic factors, together with biogeographical history, influence geographical limits of species.

Assessing the threats that face biodiversity relies on the ability to assess the current extent from an historical baseline and the contemporary rate of decline in geographic distribution (Poiani *et al*. 2000). However, defining these parameters presents an array of methodological challenges compounded by the limited availability of relevant data. When assessing biodiversity it is essential that an appropriate historical baseline be established from which to contrast the current state of biodiversity. In many parts of the world the advent of the Industrial Revolution or European settlement is considered to represent points from which to assess anthropogenic declines in indigenous biodiversity (Keith *et al*. 2009). This research used the Mucina, Rutherford & Powrie (2007) vegetation map of South Africa, Lesotho and Swaziland which is a map of potential vegetation and is assumed to represent the patterns of vegetation in this area prior to large-scale human modification beginning in the 17th century.

2.5.3 The vegetation maps of South Africa

To date three prominent vegetation maps of South Africa have been complied namely, Acocks' (1953) *Veld types of South Africa*, Low & Rebelo's (1996) *Vegetation of South Africa, Lesotho and Swaziland*, which has a refinement of Acocks' (1953), and Mucina, Rutherford & Powrie's (2007) *Vegetation map of South Africa, Lesotho & Swaziland*. Mucina, Rutherford & Powrie (2007), the latest and most detailed vegetation map available of the study area, delineates the land surface of South Africa, Lesotho and Swaziland into discrete areas referred to as vegetation types. Mucina & Rutherford (2006: 12) define a vegetation type as:

"a complex of plant communities ecologically and historically (both in spatial and temporal terms) occupying habitat complexes at the landscape scale."

Moreover a vegetation type must appear relatively homogenous in structure and floristic composition and exhibit common ecological processes. While often considered as a refinement of a biome, a vegetation type is defined in terms of dominant and rare species, as well as associations with environmental factors. The map's vegetation units were primarily classified according to a number of factors, namely:

- proximity along ecological gradients;
- dominant ecological factors at landscape level;
- dominant vegetation structure;
- levels of floristic similarity;
- proximity to other vegetation types; and
- potential to exhibit similar characteristics in the absence of anthropogenic influences.

The map was compiled from multiple data sources. In essence the mapping process combined data regarding the distribution of various plant species and assemblages with environmental factors, such as topography, geology, edaphic characteristics and climatic factors that are believed to underlie the spatial patterns of vegetation distribution. Local expert knowledge also played a significant role. For South Africa, Lesotho and Swaziland a total of 435 vegetation types are described with another five identified for the Prince Edward Islands. All the vegetation types are described and include details about the distribution of the mapped vegetation, vegetation and landscape features, geology and soils, climate, lists of biogeographically important and endemic taxa, conservation status, and supplementary remarks.

A common criticism of potential vegetation mapping is the misalignment of potential and actual vegetation cover where mapped vegetation types fail to correspond to actual vegetation cover of the ground. Furthermore, there is much debate about the roles climate, soil, history and topography play in the formation of vegetation and it is not possible to consider these concerns in detail. Pence (2008) notes a general agreement between Mucina, Rutherford & Powrie's (2007) potential vegetation map and recorded communities assessed through field survey but found it necessary to complement the map in some instances by modifying vegetation boundaries and adding several new vegetation types which did not conform to the compositional requirements stipulated by the map.

Several authors have noted the pressing need for uniform definitions and methods for defining and delineating habitats and ecosystems, as well as the need to establish transparent criteria for assessing their extinction risk in a given area. The use of vegetation types as a surrogate for biodiversity facilitates the assessment of risks faced by various ecosystems and species assemblages using data that is readily available. Because a biodiversity assessment carried out in this manner can be systematic, transparent, spatially and temporally explicit, it would represent one of the first truly repeatable and readily comparable attempts to monitor the state of indigenous biodiversity over significant spatial and temporal scales. This approach would also be flexible enough to accommodate other data. Mucina, Rutherford & Powrie (2007) therefore represents a widely available source of reliable biodiversity data that can be used for biodiversity assessment and monitoring and conservation planning in South Africa.

2.6 CONCLUSION

This chapter sought to justify the use of vegetation as a surrogate for biodiversity in the CFR. In doing so it has defined biodiversity and has exposed an animated debate surrounding the use of the term in a scientific context. The reviewed literature has shown that, in the absence of complete knowledge of the distribution of biodiversity, biodiversity surrogates are used as proxies for broader patterns of biodiversity. Providing practical measures of biodiversity for conservation and other planning purposes presents its own set of suppositions and methodological challenges. This includes issues related to the spatial representation of biodiversity and the means by which unknown elements can be inferred from known data. Biodiversity surrogates are differentiated by the scale at which they are applied and the data from which they are compiled. In the past taxonomic data were the preferred forms of biodiversity surrogate, but recently, integrated surrogates that incorporate a wide array of biophysical data are prevalent.

Vegetation is a common surrogate for biodiversity that is representative of broader patterns of the distribution of biodiversity and is indicative of many ecological and evolutionary processes that are pivotal for the long term wellbeing of most species. In areas such as the CFR, where detailed species distribution data is unavailable, vegetation has been identified as an optimal surrogate for biodiversity. This approach is also more likely to benefit from the rapid development of sophisticated GIS applications and more widely available and cost-effective satellite imagery which can cover periods of between two to four decades. Focusing projects on the landscape or ecosystem level is not meant to replace the species-specific approach, rather complement it or provide a pragmatic alternative in data-poor areas. In the CFR the use of vegetation type data as a surrogate for biodiversity has considerable advantages over a species-specific approach owing to the holistic nature of the surrogate and its emphasis on floristic diversity. The use of vegetation type data in conjunction with land-cover data can consequently better inform the development of regional reserves – which can incorporate ecosystem services and cater for nested ecological communities – than species or environmentally orientated data. This also provides a means of rapid assessment and monitoring; allowing for a proactive approach to conservation especially in highly stressed areas where anthropogenic pressure is rapidly increasing.

Vegetation types therefore provide an appropriate surrogate for biodiversity that can readily be combined with land-cover data. The following chapter describes the methods that were applied to generate land-cover maps for the Berg River catchment for a 20-year period and how the generated maps were used to infer changes in biodiversity through the use of vegetation types as a surrogate for biodiversity.

CHAPTER 3: LAND-COVER MAPPING

The classification of raw satellite imagery into land-cover classes was an essential part of this research. Consequently, this chapter will discuss the role of GIS and remote sensing in the generation of land-cover maps. The discussion will focus on the use of multispectral imagery, particularly Landsat-5, Landsat-7 and SPOT-5 imagery. Finally, the chapter will describe the automated classification of satellite imagery using spectral and textural properties in an object orientated environment.

3.1 GIS AND REMOTE SENSING IN LAND-COVER MAPPING

The utility of land-cover maps has been demonstrated in fields as diverse as agriculture, civil engineering and land-use planning (Johnston 1998; Lillesand, Kiefer & Chipman 2004). GCOS (2006) states that land-cover mapping also has applications in societal spheres such as disaster management and service provision. The importance of accurate and timely land-cover data has been highlighted in Agenda 21 of the United Nations Conference on Environment and Development and the World Summit on Sustainable Development in Johannesburg 2002 (GCOS 2006). It is also promoted in existing conventions such as the United Nations Framework Convention on Climate as an essential variable in monitoring climatic change (GCOS 2006). Accurate land-cover data also underpins many applications in hydrological modelling, erosion and sedimentation studies, landscape ecology and sustainable development (Griscom *et al*. 2009).

Consequently, the production of accurate land-cover maps underpins the effective and judicial management of natural and human resources and has developed in line with technological advances and ever expanding knowledge of the forces that shape the earth’s surface. This section provides a background to the use of GIS and remote sensing in land-cover mapping and biodiversity assessment.

3.1.1 Digital mapping and indirect observation

Maps and other abstractions of the earth’s surface have been in use for millennia as repositories of knowledge, to analyse spatial patterns and plan accordingly (Pickles 2004). Early attempts at producing land-cover maps relied almost entirely on cadastral and other local data sources to delineate different land-cover features, but the maps were seldom regarded as reliable or accurate (Falkner & Morgan 2002). The visual interpretation of aerial photography provided a novel platform for mapping land cover that began in earnest in the early 1930s and is now a long established procedure (Falkner & Morgan 2002). In the last three decades spaceborne sensors have revolutionized the manner in which land-cover data are generated.

Remote sensing is the process of detecting and measuring variables from a distance. Recent and rapid technological advances in the field of remote sensing and GIS have facilitated the effective mapping of land cover by the ability of remote-sensing devices to record the reflectance properties of objects or areas on the earth's surface which can be analysed in a GIS and used to infer some of the attributes of a particular area or assign it to a particular land-cover class (Tucker, Townshend & Goff 1985; Petit & Lambin 2002; Hill *et al*. 2005). Together these technologies provide a powerful tool with which to map and analyse the earth's dynamic surface and explore the implications of processes it harbours.

In the past the limited spatial and temporal resolution of remotely sensed data as well as the costs and technical expertise involved have rendered these technologies impractical for some purposes, especially local-scale studies. However the use of remote sensing devices in generating data for larger areas has been thoroughly demonstrated (Tucker, Townshend & Goff 1985; Chen 2002; Stillwell & Clarke 2004; Hill *et al*. 2005; Falcucci, Maiorano & Boitani 2007). Assessments of land-cover change based on remotely sensed imagery have been widely applied on a small scale and have provided a valuable tool in the analysis of land cover as they enable rapid and cost-effective mapping of areas or periods in time that might otherwise be impossible (Wessels, Reyers & Van Jaarsveld 2000; Chen 2002; Falcucci, Maiorano & Boitani 2007).

The use of remote sensing, especially satellite imagery, has facilitated the analysis of global-scale land cover and promoted the identification of threatened areas that were previously beyond the scope of direct methods (Lambin & Ehrlich 1997; Helmer, Brown & Cohen 2000). Moreover, remotely sensed data, especially from satellite imagery, is systematic, explicit and repeatable (Salem 2003; Stillwell & Clarke 2004; Foody 2008). In local or regional studies, satellite imagery is often combined with aerial photographs, land-cover data and direct observation to enhance the precision or detail of an analysis (Weng 2002; Stillwell & Clarke 2004; Hill *et al*. 2005). The potential of newly developed high-resolution satellite sensors for detailed biodiversity assessment is currently being explored as a means of overcoming the limitations of medium and coarse-scale imagery (Hill *et al*. 2005; Zhang, Zhengjun & Xiaoxia 2009).

GIS are efficacious in the analysis of biodiversity and other environmental data and are being increasingly integrated with remote-sensing capabilities (Lillesand, Kiefer & Chipman 2004).

The amalgamation of spatially referenced and attribute data has facilitated the analysis of large volumes of data, an exercise that would otherwise be cumbersome if not impossible (Morain 1999; Salem 2003; Stillwell & Clarke 2004; Foody 2008). Remotely sensed data are particularly useful in areas that lack extensive land-cover or biodiversity data, especially in the developing world where the availability of aerial photography and survey data is limited (Tucker, Townshend & Goff 1985; Lambin & Ehrlich 1997). Used together, GIS and remotely sensed data provide an unprecedented means of generating land-cover data for analysis in conjunction with other relevant data.

3.1.2 Multispectral imagery

The use of multispectral imagery is a wide-ranging and rapidly growing field in remote sensing. A multispectral sensor subdivides the spectral range of electromagnetic radiation into bands, defined as intervals of continuous wavelength collected simultaneously, over a broad range of the electromagnetic spectrum (EMS) which are then recorded and processed to form a series of images that can be combined in various ways (Lillesand, Kiefer & Chipman 2004). While the most commonly used arrangement involves the acquisition of four images in the red, green, blue (RGB) and near infrared (NIR) portions of the EMS, contemporary sensors often record several additional bands (Gibson 2000; Lillesand, Kiefer & Chipman 2004).

The value of multispectral imagery lies in the capacity of different recorded bands to intensify the contrast between different features or areas thereby facilitating their identification (Gibson 2000; Lillesand, Kiefer & Chipman 2004; Keith *et al*. 2009). Common applications of multispectral imagery are water-body penetration for bathymetric mapping, discrimination of soil and vegetation types, forest mapping and vigour assessment, and delineating various anthropogenic and natural features (Lillesand, Kiefer & Chipman 2004). More sophisticated applications are capable of determining chlorophyll absorption, differentiating between plant species, establishing biomass and vegetation moisture content, determining soil moisture, mapping minerals and rock types and various thermal-mapping applications (Lillesand, Kiefer & Chipman 2004; Keith *et al*. 2009). Because many areas display marked seasonal variations, the interpretation of images is often enhanced by reference to images recorded at different dates (Lillesand, Kiefer & Chipman 2004).

Vegetation analysis is generally enhanced by the incorporation of red and infrared bands as these are acutely receptive of physiological changes in vascular plants (Lillesand, Kiefer & Chipman

2004). Measures such as the normalized difference vegetation index (NDVI) and the enhanced vegetation index (EVI) have been used to monitor the health and distribution of natural and agricultural vegetation over time while satellite-derived land-cover maps have been used to quantify land-cover change and make pronouncements on its implications (Wessels, Reyers & Van Jaarsveld 2000; Chen 2002; Stillwell & Clarke 2004; Falcucci, Maiorano & Boitani 2007).

Figure 3.1 shows a portion of the study area displayed in true colour (band combinations 3; 2; 1) as the area would appear when viewed with the naked eye. Figure 3.2 is a false colour display (band combinations 4; 3; 2) of the same area showing vegetation as bright red. This is a result of the absorptive and reflective properties of chlorophyll in the red spectrum and water-filled palisade tissue in the near infrared (NIR) range of the EMS (Lillesand, Kiefer & Chipman 2004). The manipulation of band combinations is a technique commonly used to identify and delineate features on a multispectral image that would not be apparent on a true colour image. A large number of band combinations are facilitated by most image processing software. Some of the prominent multispectral satellites are QuickBird, IKONOS, GeoEye-1, Spot-5, Landsat-5 and Landsat-7.

3.1.3 Landsat imagery

The Landsat programme refers to a series of earth observation satellites originally instituted by the National Aeronautics and Space Agency (NASA) and the U.S. Department of the Interior (Lillesand, Kiefer & Chipman 2004). The programme is now jointly managed by NASA and the United States Geological Survey (USGS) and represents the longest running programme directed at the acquisition of earth observation imagery from spaceborne sensors (NASA 2011). The original satellite, Landsat-1 was launched in 1972 and the latest, Landsat-7, was launched in 1999 (NASA 2011).

Figure 3.1: True colour image of a portion of the Berg River catchment: band combinations 3; 2; 1

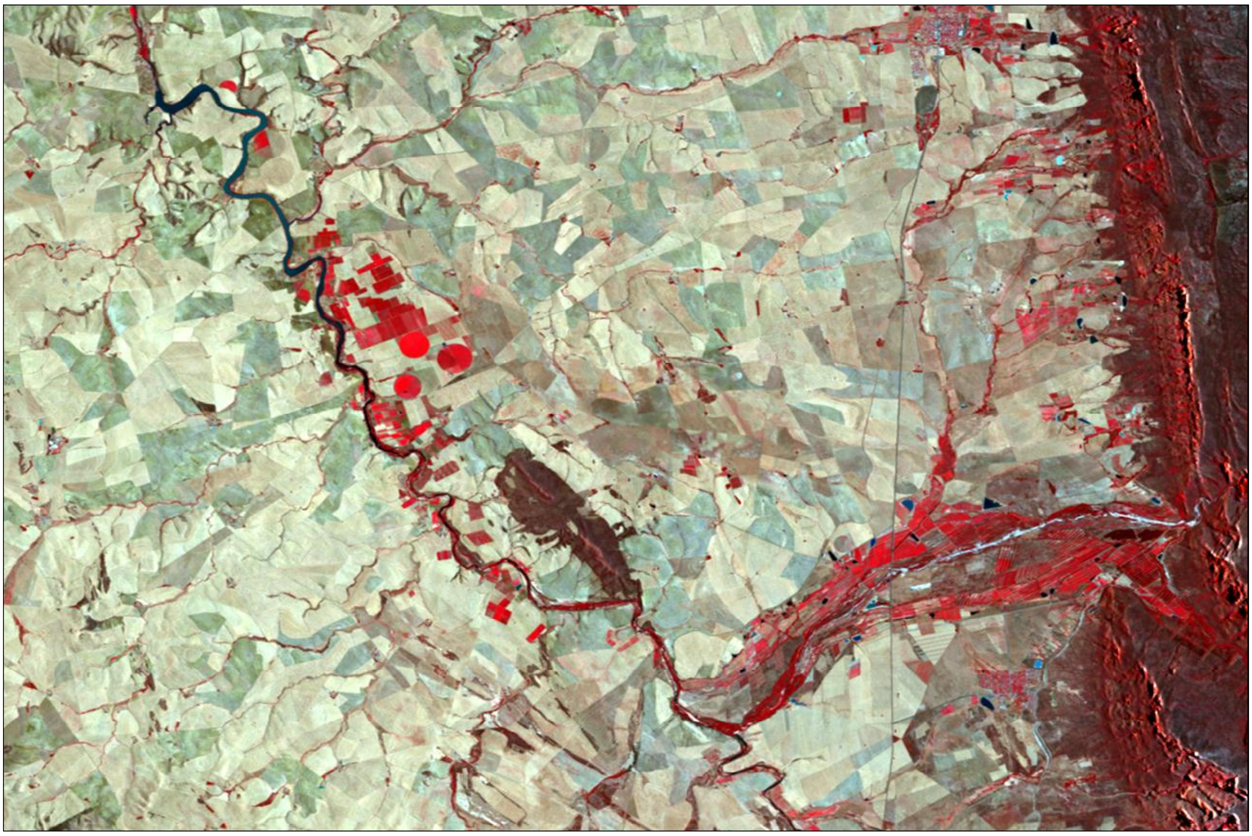


Figure 3.2: False colour image of a portion of the Berg River catchment: band combinations 4; 3; 2

The multispectral scanner (MSS) is an optical sensor that records solar radiation in four spectral bands at a resolution of 80x80 m. The MSS constituted the primary data acquisition apparatus on the first generation of Landsat satellites (1, 2 and 3). The thematic mapper (TM) sensor is a more sophisticated version of the MSS and records reflected and emitted solar radiation in seven spectral bands that range from visible to thermal infrared regions and was featured on Landsat 4, 5 and 6 in addition to the MSS (Lillesand, Kiefer & Chipman 2004). The TM sensor is more finely tuned for vegetation discrimination than the MSS with several bands recoded at narrower wavelengths (Gibson 2000).

The resolution for TM bands is 30x30 m with the exception of the thermal band which has a resolution of 120x120 m. This band is, however, re-sampled to the same resolution as the other bands (Gibson 2000). The enhanced thematic mapper plus (ETM+) sensor carried by Landsat-7 samples at a similar resolution with the exception of the thermal band and is capable of capturing a panchromatic scene at a resolution of 15 m. A comparison of the bands recorded by Landsat-5 and Landsat-7 is presented in Table 3.1. Despite the resolution of Landsat imagery some features, smaller than 30 m, that contrast significantly with their surroundings, such as roads, are sometimes visible while large features that exhibit a reflectance that does not contrast as sharply may not be readily identifiable (Lillesand, Kiefer & Chipman 2004).

False colour combinations and near infrared and infrared bands of Landsat TM and ETM+ imagery are commonly used to assess vegetation health and cover but the platform has found applications in fields such as agriculture, botany, cartography, environmental monitoring forestry, geography, geology, geophysics, hydrology, land-use planning, natural resource management and oceanography (Gibson 2000; Lillesand, Kiefer & Chipman 2004; NASA 2011). A particularly useful aspect of Landsat imagery is that a large geographical area is covered by a scene so that images can be used at a regional scale and map entire areas such as river catchments and administrative regions.

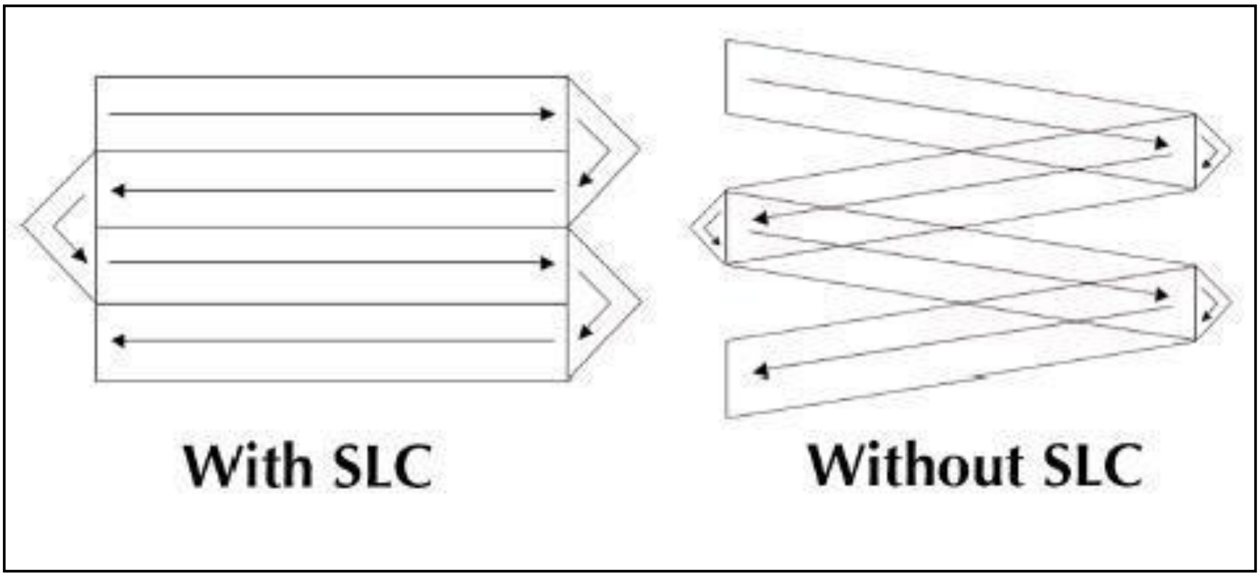
The area covered by a Landsat image is approximately 185x185 km and is skewed eastwards due to the earth's rotation (Lillesand, Kiefer & Chipman 2004). Landsat-5 was launched into a repetitive, circular, sun-synchronous, and near-polar orbit at 705 km from the earth's surface. This orbit enables the satellite to scan the entire earth's surface in a 16-day repeat cycle (Lillesand, Kiefer & Chipman 2004). Like the ill-fated Landsat 4 that preceded it, Landsat 5 carries a TM sensor. Launched in 1984 the satellite remains operational and is widely considered to be one of the most successful earth observation satellites providing thousands of scenes over the course of a 27-year operational history.

Table 3.1: Landsat TM and ETM+ spectral bands

	Landsat-5 (TM)		Landsat-7 (ETM+)			
Band	**Wavelength (µm)**	**Resolution (m)**	**Wavelength (µm)**	**Resolution (m)**	**Nominal**	**Applications**
1	0.45 - 0.52	30	0.45 - 0.515	30	Blue	Water body penetration/ bathymetry, Soil/ Vegetation discrimination, forest type mapping and feature identification
2	0.52 - 0.60	30	0.525 - 0.605	30	Green	Vegetation discrimination and vigour assessment and feature identification
3	0.63 - 0.69	30	0.63 - 0.69	30	Red	Sensing chlorophyll absorption region, plant species differentiation and feature identification
4	0.76 - 0.90	30	0.75 - 0.90	30	NIR	Determining vegetation types vigour and biomass content, delineating water bodies and determining soil moisture content
5	1.55 - 1.75	30	1.55 - 1.75	30	Mid IR	Vegetation and soil moisture content, differentiation of snow from clouds
6	10.40 - 12.50	120	10.40 - 12.5	60	Thermal IR	Vegetation stress analysis, soil moisture discrimination and thermal mapping applications
7	2.08 - 2.35	30	2.09 - 2.35	30	Mid IR	Discrimination of mineral and rock types, vegetation moisture content
8			0.52 – 0.90	15	Panchromatic	Panchromatic image sharpening, feature identification

Adapted from Lillesand, Kiefer & Chipman (2004: 422)

Landsat-7, the latest satellite to be launched under the Landsat programme, follows similar orbits and repeat patterns to Landsat-5 to maintain data continuity (Lillesand, Kiefer & Chipman 2004). However, the Scan Line Corrector (SLC) onboard Landsat-7 malfunctioned on May 31, 2003 (NASA 2011). NASA (2011) describes the SLC as a devise which compensates for the along-track of the satellite allowing the ETM+ sensor to capture parallel scans. As a consequence of this failure the EMT+ sensor now scans the earth’s surface in a zigzagging fashion (Figure 3.3). Attempts to repair the SLC were unsuccessful and its failure is thought to be permanent. In an effort to remedy the problem the sensor was reconfigured in late 2003 to operate its primary electrical harness to compensate for the absence of a functional SLC. The most pertinent effect of the SLC failure is that certain areas in a scene are resampled while others are bypassed. This effect is most noticeable at the edges of a Landsat-7 scene and abates nearer the centre of the scene. While the radiometric and geometric properties of a Landsat-7 scene following the failure of the SLC are comparable to those before it approximately 22% of the data contained within the scene is now lost (NASA 2011).



National Aeronautics and Space Agency (NASA) (2011)

Figure 3.3: Effects of the SLC failure on ETM+ data acquisition

3.1.4 SPOT imagery

Satellite Pour l'Observation de la Terre (SPOT) is a series of high resolution earth observation satellites run by SPOT Image. The first satellite in the series (SPOT-1) was put into orbit in 1986 while the latest (SPOT-5) was launched on May 4, 2002 (SPOT Image 2005). All SPOT satellites share a polar, circular, sun-synchronous and phased orbit at an altitude of around 832 km which allows the satellites to scan the earth's surface in a 24 day repeat cycle (SPOT Image 2005). SPOT-1, -2 and -3 carried the self-same payloads of two high resolution visible (HVR) imaging apparatuses that recorded green, red and NIR bands at a resolution of 20 m and a panchromatic band at a resolution of 10 m (SPOT Image 2005). SPOT-4 was launched in 1998 with a high resolution visible infrared (HRVIR) sensor that incorporated a short-wave infrared (IR) band which was sampled at a resolution of 20 m (SPOT Image 2005).

SPOT-5 was tasked with improving the spatial and spectral resolution of SPOT imagery while ensuring data continuity between the various platforms. SPOT-5 carries two high resolution geometric (HRG) sensors which capture green, red and NIR bands at a resolution of 10 m, a short-wave IR band sampled at a 20 m resolution and a panchromatic band at 2.5 m to 5 m resolutions (Table 3.2). The satellite also included a high resolution sensor (HRS) that captures stereopair images which are used to map relief (SPOT Image 2005). Data acquired through the HRS is used in 3D terrain modelling and associated applications such as flight simulators and the planning of mobile phone networks (Satellite Imaging Corporation 2012). A SPOT-5 image covers an area of 60x60 km in single-instrument mode and 60x120 km in twin-instrument mode (SPOT Image 2005). The platform's high spatial resolution and wide-coverage area have made it appealing for medium scale mapping with applications in urban and rural planning, vegetation monitoring and mapping, and disaster management (Satellite Imaging Corporation 2012).

Table 3.2: SPOT HRG spectral bands

SPOT (HRG)				
Band	**Wavelength (μm)**	**Resolution (m)**	**Nominal**	**Applications**
1	0.50 - 0.59	10	Green	Vegetation discrimination, vigour assessment and feature identification
2	0.61 - 0.68	10	Red	Sensing chlorophyll absorption region, plant species differentiation and feature identification
3	0.79 - 0.89	10	NIR	Determining vegetation types vigour and biomass content, delineating water bodies and determining soil moisture content
4	1.58 - 1.75	20	Short-wave IR	Vegetation and soil moisture content, vegetation assessment and monitoring
5	0.51 - 0.73	5 (2.5 by interpolation)	Panchromatic	Panchromatic image sharpening, feature identification

Adapted from: SPOT Image (2005:3)

Data on land-cover change obtained from remote-sensing devices such as Landsat and SPOT are often combined with data about vegetation types, habitat units and species distribution in a GIS environment to provide an assessment of biodiversity loss in response to land-cover change and to identify important and viable areas for conservation (Wessels, Reyers & Van Jaarsveld 2000; Hill *et al*. 2005). As remotely sensed images for a given area may be acquired at fairly regular intervals in time they have proved particularly effective for monitoring biodiversity and habitats (Hill *et al*. 2005). Remotely sensed data is now used to describe and monitor various biophysical processes, such as net primary productivity, and biological characteristics such as vegetation cover and composition (Haines-Young 2009; Fourie, Van Niekerk & Mucina 2011).

3.2 LAND-COVER CLASSIFICATION

The automated classification of remotely sensed data is a rapidly advancing field that uses various algorithms to place pixels or areas into a particular land-cover class based on their spectral and textural properties (Lillesand, Kiefer & Chipman 2004). The justification behind this approach is time the consumption associated with manually classifying images through heads-up digitization (Lillesand, Kiefer & Chipman 2004). This section details the semi-automated classification of satellite imagery in an object-orientated environment or Geographical Object-Based Image Analysis (GEOBIA).

3.2.1 Image classification in an object-orientated environment

Classification algorithms are often categorized as either unsupervised or supervised. Unsupervised approaches use spectral data to identify areas with common attributes which are assigned to different land-cover classes whereas supervised classification methods use training sites to place pixels or areas into predefined land-cover classes (Lillesand, Kiefer & Chipman 2004). Conventionally, analytic methods of image classification treat individual pixels as discrete units with little consideration for the topological relationships existing between areas (Willhauck 2000). Such an approach is highly susceptible to ambiguous reflectance properties, radiometric effects and data noise (Willhauck 2000; Walter 2004). Consequently, GEOBIA has been developed in which an image is divided into relatively homogenous spectral areas, known as segments. Segments are subsequently treated as singular features or objects (Willhauck 2000; Walter 2004).

3.2.1.1 Segmentation

Segmentation is the process whereby multiple pixels in an image are merged and delineated to form discrete objects and it usually constitutes the first step in an object-orientated classification (Kartikeyan, Sarkar & Majumder 1998; Willhauck 2000; Walter 2004). According to Willhauck (2000), segmentation offers several advantages over a pixel-level classification in that it facilitates analysis of the spectral and textural qualities of an image. The capacity to examine characteristics such as size, shape and the topological relationships that exist between features in a landscape is enhanced by this approach. Hay & Baschke (2010) explains that segmentation aids the integration of continuous remote-sensing data with vector data in a GIS environment.

Various segmentation algorithms exist and they are generally selected according to the task at hand to improve the accuracy of, or simplify, a classification process (Baatz & Schäpe 1999

Kim, Madden & Bo 2010). A segmentation algorithm essentially merges adjacent pixels or smaller objects iteratively to create larger ones based on a user defined-homogeneity threshold, usually the similarity and dissimilarity of spectral properties (Walter 2004; Blaschke 2010). Variations which can be used to optimize the process include varying the selection of initial regions, adjusting and threshold at which regions are merged as well as the threshold at which merging is terminated (Walter 2004; Blaschke 2010).

Segmentation has the capacity to distort the classification process in several ways depending on the scale at which it is executed. Potential pitfalls associated with image segmentation are typically under- or over-segmentation (Baatz, Hofmann & Willhauck 2008). Undersegmentation occurs when surplus areas that are not comparable to the remainder of the segment are incorporated in the output. Oversegmentation occurs when certain variation is erroneously omitted from the output. These errors typically manifest as segments where boundaries do not correspond to a feature on the ground, in segments where multiple land-cover classes are combined into a single feature and land-cover features subsumed by a more dominant land-cover class. Among the many reasons for these errors are radiometric noise and rigidity of homogeneity parameters (Baatz, Hofmann & Willhauck 2008). Various techniques and approaches exist to mitigate the effect of under- and oversegmentation but these aberrations are likely to occur to varying degrees in all segmentation exercises.

3.2.1.2 Supervised classification

Supervised classification schemes are a popular means of classifying remotely sensed images owing to the schemes' customizability, reputation for accuracy and relative ease of use (Stephenson & Van Niekerk 2009). A supervised classification uses some prior acquired knowledge of the area of interest to develop training points broadly representative of various land-cover classes and to guide the classification (Rozenstein & Karnieli 2011). Training points are locations for which the land cover has been established, usually through field survey, the interpretation of aerial photography or personal experience (Rozenstein & Karnieli 2011). These points are used to train a classification algorithm which classifies the image as specified by parameters derived from the training points (Rozenstein & Karnieli 2011).

3.3 ASSESSING THE ACCURACY OF LAND-COVER MAPS DERIVED FROM REMOTELY SENSED DATA

Providing a measure of the accuracy of remotely sensed data is crucial in establishing the validity and relevance of these methods and the data they produce. This is conventionally achieved through the use of reference data which are assumed to provide a more accurate and objective representation of the data being studied. The most common forms of reference data are ground-control points or other forms of remotely sensed data that are considered to be more accurate or objective due to their higher spatial or temporal resolutions (Congalton & Green 2009). Foody (2002) points out that the application of error matrices, sometimes referred to as confusion matrices, is the most commonly used means of assessing the accuracy of remotely sensed data. The compilation of an error matrix involves comparing reference data to corresponding areas on the land-cover map and crosstabulating correct and misclassification between the two data sets.

Field data are a popular source of reference data because samples collected during a field survey assess actual ground data and reveal aspects of a study area that may be undetected in remotely sensed data due to the spatial resolution of the data or misinterpretation of certain features or areas. In cases where field data are unobtainable, aerial photographs are often used as reference data for a land-cover map generated from coarser satellite data as aerial photographs have a greater spatial resolution and their interpretation is a well-established means of accurately assessing land cover (Congalton & Green 2009). However, errors can be generated when the interpreter misclassifies photographs or the photographs themselves are inaccurate or ambiguous and are best used in simpler classification schemes with few or broader classes. As a consequence of these concerns Congalton & Green (2009) argue that aerial photography should, wherever possible, be augmented by field surveys and consultation with experts on the area in question.

Land-cover maps, as is the case with all abstractions, are prone and ambiguities and inaccuracies as they divide the intricately complex surface of the earth into a series of discrete classes (Gibson 2000). Limited access to data and other resources are likely to impede the generation of accurate land-cover maps and this effect is often compounded by considerations such as fiscal and temporal constraints. Remotely sensed data are susceptible to varying atmospheric conditions which may mislead a classification exercise. Furthermore, changes in the sensitivity of spaceborne sensors over time have the capacity to distort comparisons between images captured at different time periods. An accepted standard for the accuracy of land-cover maps derived from remotely sensed data is 85% (Foody 2002). The next chapter describes the methods used to obtain a high level of classification accuracy and how this data was analysed to identify land cover biodiversity change.

CHAPTER 4: METHODS

This chapter describes the processes by which land-cover data were derived, integrated with the designated biodiversity surrogate and analysed to address the objectives specified in Chapter 1. Firstly, the data used and the methods that were employed are outlined. Secondly, the particulars of the mapping and change analysis procedures used are described and discussed. Thirdly, the accuracy of the generated land-cover maps is assessed in view of using them to determine the nature of land-cover change in the Berg River catchment.

4.1 DATA USED

Three land-cover maps were generated from Landsat TM and ETM+ data using a supervised classification algorithm. The supervised classification required the acquisition of reference data to train the classification algorithm and to assess the accuracy of the derived land-cover maps. Ancillary data was used to assess the derived land-cover maps and to guide the post-classification editing that was performed on the land-cover maps.

4.1.1 Satellite imagery

A key consideration of this research was to maintain cost-effectiveness while procuring data of an appropriate spatial and temporal resolution with which to perform an analysis of historical land-cover change at a catchment scale. High resolution satellites such as IKONOS, Quickbird, Worldview-1 and GeoEye-1 were found to have too low an image extent to be cost-effectively applied at a catchment scale. Additionally, the limited temporal extent of images produced by these platforms would limit the scope of the research. Imagery with greater spatial and temporal extents such as the Moderate Resolution Imaging Spectroradiometer (MODIS) and the Advanced Very High Resolution Radiometer (AVHRR) had too coarse a spatial resolution to attain a desirable level of accuracy.

Landsat TM and ETM+ were selected as they offered a suitable spatial and temporal extent and could be acquired at a low cost. An inventory of Landsat imagery, developed by the Centre for Geographical Analysis (CGA) at Stellenbosch University, was used to search for appropriate images for the study. A total of five Landsat scenes representing three sets of imagery covering the entire study area at different time periods were selected. Table 4.1 lists the Landsat scenes that were used to develop land-cover maps.

Table 4.1: Landsat scenes

Landsat scenes prior to merging			
Scene ID	**Path**	**Row**	**Acquisition date**
LT517608319861231	176	83	31/12/1986
LT517508319870109	175	83	09/01/1987
LE717508319991001	175	83	01/10/1999
LE717608320000213	176	83	13/02/2000
LT517508320070217	175	83	17/02/2007

The oldest set represented images that were acquired in the summer of 1986/1987. The latest set of imagery was acquired in the summer of 2007. Unfortunately, no complete set of imagery was available for any given summer season between these dates. A third set of imagery, acquired in the winter and summer months of 1999 and 2000 respectively, was consequently selected.

Three Landsat TM and two Landsat ETM+ images were acquired. To obtain cloud-free images of the entire Berg River catchment it was necessary to merge two Landsat-5 images captured on 31/12/1986 and 09/01/1987, and two Landsat-7 images captured on 01/10/1999 and 13/02/2000. A final Landsat TM image which was captured on 17/02/2007 was found be cloud free and could be used without modification.

4.1.2 Reference data

Field surveys were undertaken to acquire reference data to guide the supervised classification, perform an assessment of the accuracy of the derived land-cover maps and to gain real-world experience of the study area. A total of 819 locations were visited, documented and photographed. The dominant land cover at each location was classified according to the land-cover classification system (LCCS) described in Section 4.2 and a global positioning system (GPS) reading was taken.

It was often difficult to gain access to privately-owned land and this inhibited access to certain areas. As a large area was to be sampled, proximity to roads was critical in designing the sampling scheme. Owing to seasonal variation between the images that were used to generate the land-cover maps, two surveys were carried out: one during the wet season and one during the dry season.

4.1.2.1 Wet-season survey

For the wet-season survey (between 26/05/2010 and 27/06/2010) photographs and point descriptions were taken either side of the road at five-kilometre intervals along all the major public access roads in the catchment. A total of 680 points were visited during this survey. Particular attention was paid to areas where natural vegetation occurred and notes were made on its diversity and the presence of alien vegetation.

4.1.2.2 Dry-season survey

For the summer survey (17/03/2011 and 11/04/2011) a systematic sampling approach was taken. A set of predefined points were designated in different land-cover classes. These points were loaded into a GPS and located. The points were photographed and subsequently described paying particular attention to the variation in natural and alien vegetation between the seasons. A total of 139 points were collected in this manner.

4.1.3 Ancillary data

Higher-resolution SPOT-5 imagery was used to distinguish features that were indistinct on the Landsat images. NLC data for 2000 and 2009 were employed as points of reference and to compare the way different areas were classified (Van Den Berg *et al*. 2008; SANBI 2009). In addition, two C.A.P.E data sets (CAPE Untransformed Areas and CAPE Priorities) were used to enhance the classification, namely the untransformed areas data set which represents areas mapped as being free from anthropogenic interference and the priority remnants data set which maps remnants of critically endangered vegetation types in the Western Cape.

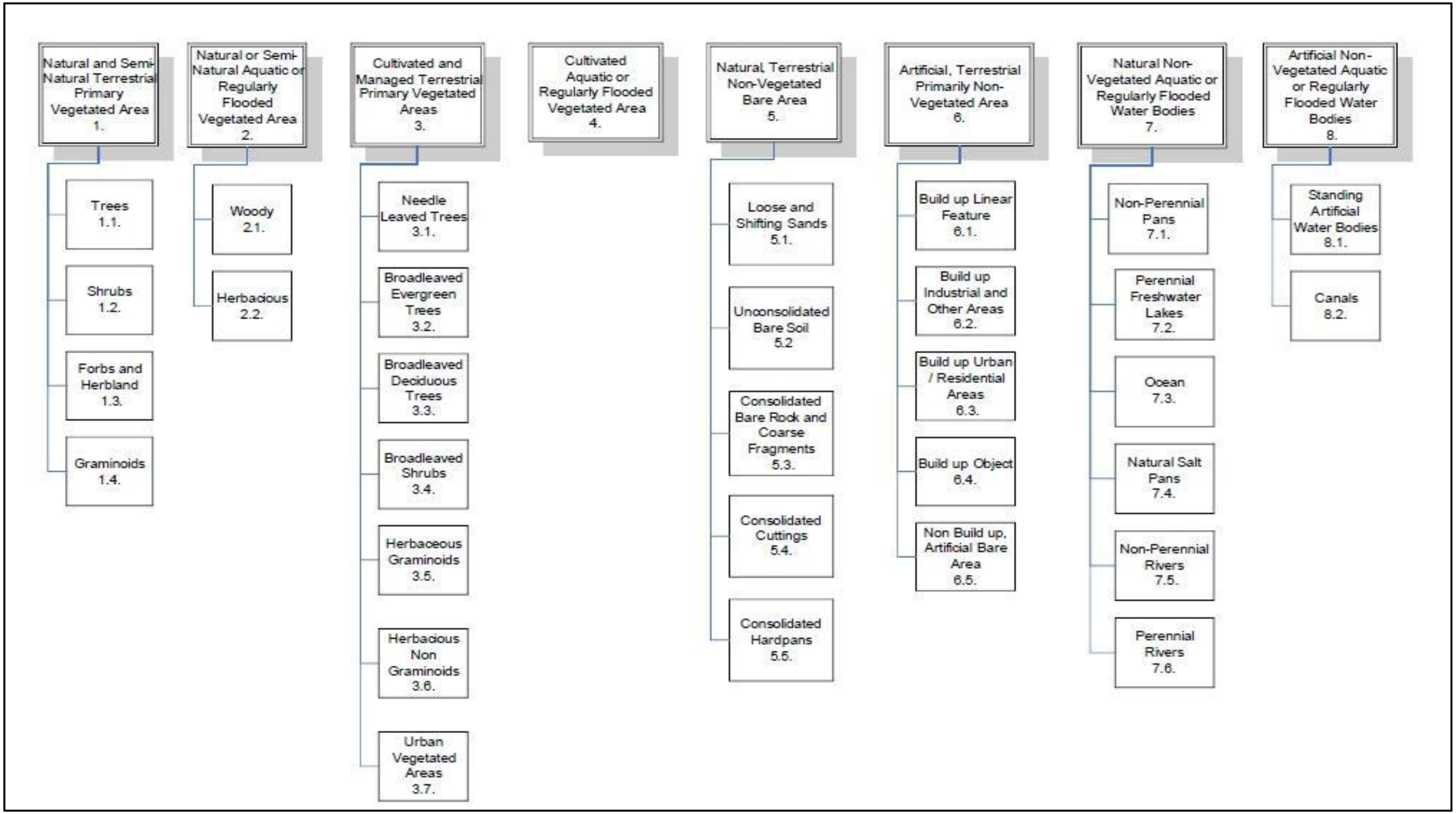
4.2 LAND-COVER CLASSIFICATION SYSTEM

A key consideration when generating land-cover maps is the legend used as this determines the information contained in the map and the uses to which it can be put. It is crucial to strike a balance between detail and functionality. Because this study was based in South Africa, a legend which is compatible with other land-cover maps for this area was chosen. A modified version of the Chief Directorate: National Geo-spatial Information’s (CD: NGI) new land-cover legend was selected to ensure that the derived land-cover classification is comparable with existing land-cover maps and could be easily edited within an established framework. This legend is loosely based on the amalgamation of several National Land-Cover (NLC) 2000 classes which are in turn based on the Food and Agricultural Organization's (FAO) land-cover classification system (LCCS) (Lück & Diemer 2008). The eight classes presented in Table 4.2 can be extracted with a high degree of accuracy but they are unlikely to meet all users' requirements (Lück 2006). However, these classes can be subdivided or refined fairly easily. An example is the division of the artificial terrestrial primarily non-vegetated areas class into classes such as residential areas, roads or quarries. The CSIR's new land-cover legend is divided into eight primary classes which can be further refined to meet user specifications.

Because this project was primarily concerned with the presence of indigenous natural vegetation some differentiation between pristine natural vegetation and degraded or otherwise altered vegetation was necessary. Subsequently, the first class (natural and semi-natural primarily vegetated areas) was split into indigenous natural vegetation and degraded or alien vegetation. Furthermore, differentiation between aquatic or regularly flooded areas in a natural or degraded state was required. For this purpose differentiation was made between predominately woody and herbaceous areas with the former assumed to be largely dominated by woody alien vegetation.

Cultivated and managed terrestrial primarily vegetated areas were subdivided into three classes with one representing commercial forestry, another as a generic cultivated class that would include most forms of agriculture practiced in the catchment. Urban vegetated areas were placed into a separate class because grouping features such as golf courses with cultivated land might be misleading in a change analysis. Finally, natural and artificial waterbodies were grouped in the same class owing to the difficultly of accurately discriminating between small dams and features such as vernal pools. The final legend used is presented in Table 4.3, followed by a detailed description of the individual classes.

Table 4.2: CD: NGI LCCS



Source: Lück & Diemer 2008 (2010: 13)

Table 4.3: Modified LCCS used to generate land-cover maps of the Berg River catchment

1. Natural and semi-natural primarily vegetated areas	**2.** Natural and semi-natural aquatic or regularly flooded areas	**3.** Cultivated and managed terrestrial primarily vegetated areas	**4.** Artificial terrestrial primarily non-vegetated areas	**5.** Natural terrestrial primarily non-vegetated areas	**6.** Natural or artificial primarily non-vegetated aquatic or regularly flooded water bodies 6
1.1 Primarily indigenous trees, shrubs, forbs, herbland and graminoids	2.1 Herbaceous	3.1 Needle-leaved broadleaved evergreen, broadleaved deciduous trees >3m	Built-up linear features, objects, urban, residential, industrial and other areas non-built-up artificial bare areas	Loose/ shifting sands, unconsolidated soil, consolidated bare rock and coarse fragments, consolidated cuttings and consolidated hardpans	Perennial lakes, ocean, perennial rivers, standing artificial waterbodies , canals, natural salt pans, non-perennial pans
1.2 Primarily degraded or alien trees, shrubs, forbs, herbland and graminoids	2.2 Woody	3.2 Broadleaved shrubs, herbaceous graminoids and non-graminoids			
		3.3 Urban vegetated areas			

4.2.1 Natural and semi-natural primarily vegetated areas

Natural and semi-natural vegetated areas are defined as areas where the vegetation does not require human activity for its continuance in the long term. Such areas are subdivided into two classes (Lück 2006): (i) *primarily indigenous trees, shrubs, forbs, herbland and graminoids* where primarily indigenous vegetation is an area where the indigenous phytocenoses remain largely intact and free from anthropogenic influence and biotic processes such as invasion by alien species, and (ii) *primarily degraded or alien trees, shrubs, forbs, herbland and graminoids* where primarily degraded or alien vegetation is vegetation not planted by humans but influenced by human activities either directly or indirectly to the extent that it no longer resembles indigenous phytocenoses and associated ecological processes. This can result from activities such as overgrazing or logging. The second class includes previously cultivated areas in which vegetation is regenerating as well as secondary vegetation taking root during a lengthy fallow period.

4.2.2 Natural and semi-natural aquatic or regularly flooded areas

The natural and semi-natural aquatic or regularly flooded areas land-cover class designates areas which are transitional between terrestrial and aquatic areas where the water table is found at or near the earth’s surface (Lück 2006). The vegetation cover is significantly influenced by water, often dependent on flooding and generally constituted by hydrophytes. Wetlands, mangroves, marshes and riparian zones are common examples of the type of land cover this land-cover class represents. Marshes or salt pans where the intense fluctuations in water level or high salt content prevent the development of hydrophytes are also included in this category. Like the natural and semi-natural primarily vegetated areas land-cover class, natural aquatic or regularly flooded areas are areas where anthropogenic activities have neither directly nor indirectly altered the indigenous phytocenoses while semi-natural areas have witnessed significant anthropogenic influence. However, it is often difficult to differentiate between natural and semi-natural areas as aquatic or regularly flooded areas are acutely sensitive to distant human activities which can significantly disturb the vegetation cover (Lück 2006). Common examples of these processes are damming and the addition of fertilizers into watercourses which may alter species composition. In many cases influences such as these cause vegetation to develop a new biotope in balance with artificial environmental conditions.

Two subsets were identified for this primary land-cover class, namely (i) *herbaceous* and *(ii) woody*. The *herbaceous* subclass includes all graminoid and non-graminoid non-woody vegetation dependent on standing water or temporary flooding. For mapping purposes these areas were considered natural and incorporated into the analysis of vegetation-type change. The *woody* subclass constitutes woody aquatic or regularly flooded areas when the woody component exceeds 15% of the area being considered. In the Berg River catchment the general absence of large stands of indigenous trees led to the conclusion that this land-cover class primarily comprises alien trees so that it was not included in the analysis of vegetation-type change.

4.2.3 Cultivated and managed terrestrial primarily vegetated areas

The cultivated and managed terrestrial primarily vegetated areas class refers to areas where the natural vegetation cover has been removed or radically altered and subsequently replaced by vegetation of anthropogenic origin (Lück 2006). The vegetation is artificial in that it requires human intervention to maintain its current appearance (Lück 2006). The phenology of vegetation in this class can be considerably influenced by human activities such as irrigation and harvesting. Surfaces that are bare prior to crop cultivation or after tillage, are included in this class (Lück 2006). All vegetation that is planted or managed with the intention to harvest is also allocated to this class. Three subtypes were designated:

(i) *Needle-leaved, broadleaved evergreen and broadleaved deciduous trees*, represents all needle-leaf and deciduous and evergreen broadleaved trees with a total height of 3 m or greater and displaying a distinct canopy. Most of this class comprises commercial forestry plantations. In the Berg River catchment it is assumed that all of such trees are non-indigenous, with the majority being either *Pinus* or *Eucalyptus*. While many orchards and olive groves are located within this catchment, they were included in the broadleaved shrubs and herbaceous graminoids and non-graminoids category as many were not of a sufficient height and did not display a distinctive canopy. *(ii) Broadleaved shrubs, herbaceous graminoids and non-graminoids,* includes various cultivated vegetated areas such as managed grasses, maize, cereals, sunflower and potatoes and constitute the bulk of agricultural activity in the Berg River catchment (RHP 2004). This class includes pastures and grazing land provided it can be differentiated for semi-natural vegetation. *(iii)Urban vegetated areas* comprise all primarily vegetated areas in an urban environment of anthropogenic origin. The dominant examples are golf courses, sports fields and parks.

4.2.4 Artificial terrestrial primarily non-vegetated areas

Artificial terrestrial primarily non-vegetated areas comprise areas that exhibit a vegetation cover of less than 4% and an artificial cover resulting from human activities (Lück 2006). This includes various types of built-up areas such as commercial, residential or industrial areas as well as rural dwellings with a low settlement density. Linear features such as airport runways, landing strips, roads and utility lines are incorporated in this category where they are discernible. Opencast mines, quarries and refuse sites are also included as is detritus created during the extraction of minerals or other activities that result in the deposition of various materials. Additionally, construction sites or areas being cleared for construction are subsumed under this category.

4.2.5 Natural terrestrial primarily non-vegetated areas

Natural terrestrial primarily non-vegetated areas is made up of bare areas with vegetation cover not exceeding 4% of the total area that lack an artificial surface (Lück 2006). This includes unconsolidated bare soils such as erosion scars and areas of negligible vegetation cover due to low precipitation or poor growth medium. Loose and shifting sands such as coastal dunes and beaches are included as are bare rock areas and desert. Consolidated hard pans that do not harbour vegetation or carry water are included but may be difficult to identify. Finally, landslides, steep riverbed embankments and consolidated cuttings, made to accommodate roads, are included in this category.

4.2.6 Natural or artificial primarily non-vegetated aquatic or regularly flooded waterbodies

Natural or artificial primarily non-vegetated aquatic or regularly flooded waterbodies includes areas that are covered with water perennially or non-perennially either naturally or due to the construction of artefacts or other anthropogenic influences and do not support vegetation cover (Lück 2006). Examples of natural waterbodies are rivers, lakes, non-perennial or perennial pans and the ocean. Artificial waterbodies include dams and other standing waterbodies such as reservoirs and artificial lakes as well as canals.

The land-cover classes described in this section represent a technical description of the land-cover classes that were mapped. For simplicity the class names were abbreviated as shown in Table 4.3. Henceforth, the abbreviated legend will be used when discussing the results of the mapping exercise.

Table 4.4: Land-cover classes and corresponding map classes

Land-cover class	Abbreviated map legend entry
Primarily indigenous trees, shrubs, forbs, herbland and graminoids	Natural vegetation
Primarily degraded or alien trees, shrubs, forbs, herbland and graminoids	Semi-natural vegetation
Natural and semi-natural aquatic or regularly flooded areas (herbaceous)	Aquatic vegetation (herbaceous)
Natural and semi-natural aquatic or regularly flooded areas (woody)	Aquatic vegetation (woody)
Needle-leaved, broadleaved evergreen and broadleaved deciduous trees	Plantations
Broadleaved shrubs, herbaceous graminoids and non-graminoids	Cultivation
Urban vegetated areas	Urban vegetated areas
Artificial terrestrial primarily non-vegetated areas	Artificial bare areas
Natural terrestrial primarily non-vegetated areas	Natural bare areas
Natural or artificial primarily non-vegetated aquatic or regularly flooded water bodies	Water

The land-cover classification was devised with the intention of ensuring the scientific integrity of the derived land-cover data, the practical orientation of the final products and to facilitate the comparison between classes derived from different classification systems (Lück 2006). As the classes are fairly broad they are intended to be unambiguous and promote clear boundaries between different classes. The move to present a simplified classification is motivated by an acknowledgement that land-cover should be easily and accurately extracted from remotely sensed data and represent actual land cover as opposed to land use (Lück & Diemer 2008). Such a system facilitates the further refinement of land-cover classes by future users with different needs and preferences.

4.3 IMAGE CLASSIFICATION AND CHANGE DETECTION

This section describes the processes whereby a selection of Landsat-5 and Landsat-7 imagery was used to develop three land-cover maps for the Berg River catchment. First it was necessary to devise an appropriate legend with which to inform the classification. An overview of the classification and change detection procedures employed in this research is presented in Figure 4.1.

4.3.1 Land-cover classification

An object-orientated nearest neighbour supervised classification was performed in eCognition Developer 8 to classify the Landsat-5 and Landsat-7 images. A bottom-up, region-growing segmentation approach was used to produce consistent results across the relatively large and heterogeneous study area. A multi-resolution segmentation scale parameter of 30, a shape parameter of 0.1 and a compactness parameter of 0.5 was found to produce highly homogenous image objects. Once the segmentation process had been completed a supervised classification algorithm was used to classify the segments in accordance with the legend presented in Section 4.2.
In excess of 20 training sites were used for each of the mapped classes. The classification was repeated, refining both the segmentation parameters and the training data, until a satisfactory classification was reached. The results of this process were exported as shapefiles and visually assessed against the raw images, national land-cover (NLC) maps and higher-resolution 2008 SPOT-5 images as reference.

While the analytically-generated land-cover maps were found to broadly represent true patterns of land cover, significant discrepancies were noticed. The maps were consequently manually edited to improve the overall accuracy of the land-cover maps and to differentiate between natural vegetation in a pristine and degraded state.

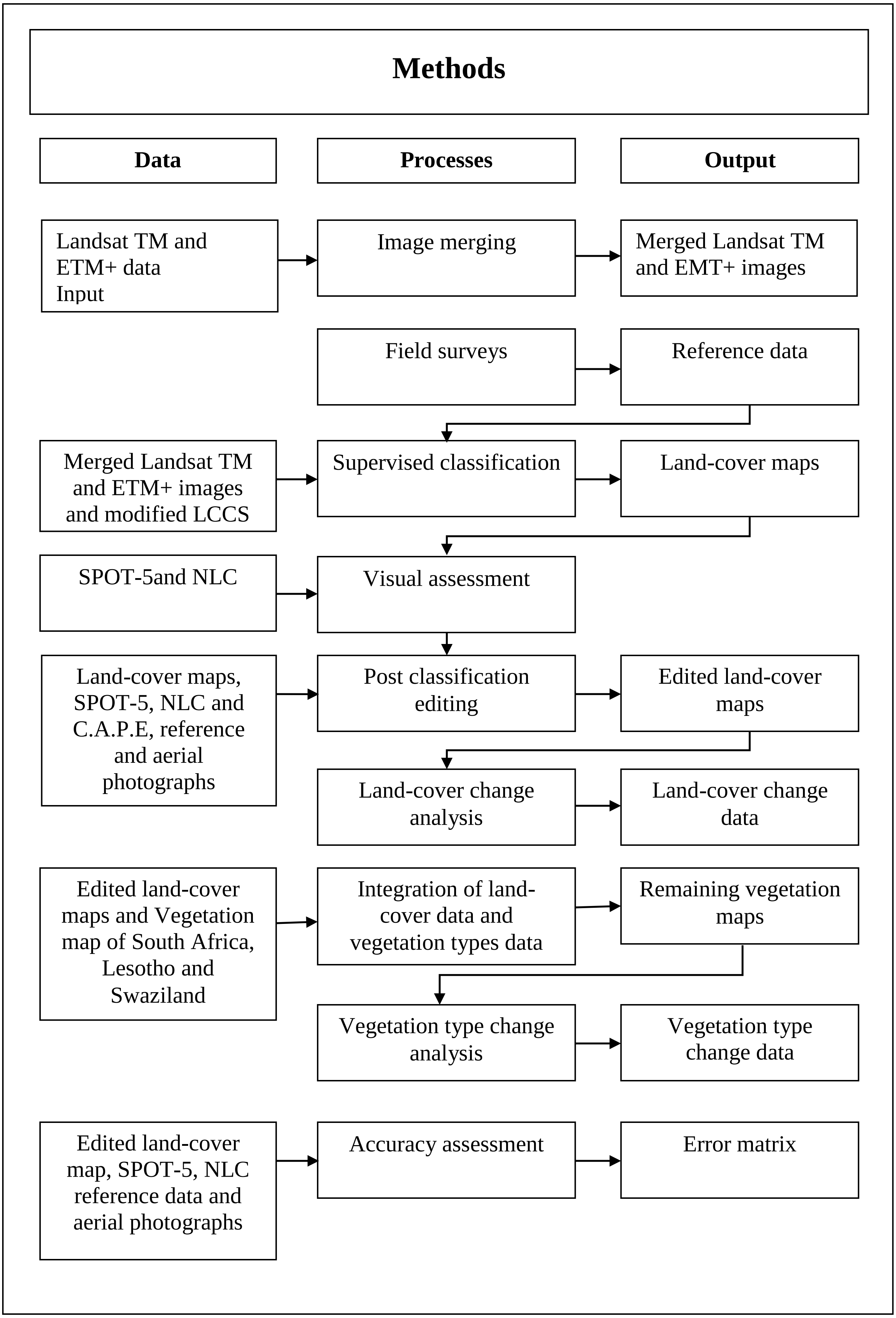


Figure 4.1: Image classification and change detection

4.3.2 Discrepancies between maps

Roads were inconsistently identified by the spectral classification as they do not usually constitute the primary land cover in any given pixel. Shadows formed by mountains and hills were occasionally incorrectly identified as water owing to the comparable spectral properties of these areas. Forestry plantations and water were sometimes confused by the spectral classification.

4.3.3 Post-classification editing

Following the spectral classification, the resultant land-cover maps were manually edited to reduce discrepancies, weed out classification errors, counter seasonal variations evident in the maps and differentiate between natural indigenous vegetation and degraded or alien-dominated areas. Higher-resolution SPOT-5 imagery and various GIS data sets were used in this process.

Most of the polygons generated through the nearest-neighbour classification were checked against the original Landsat imagery and against a set of ancillary data sets. Visual interpretation was used to reclassify certain polygons and the boundaries defined by the segmentation process were modified when they were found to inaccurately represent the boundaries of land-cover features. Roads were merged into the surrounding land-cover class as they could not be delineated consistently.

These methods were applied systematically to all the land-cover maps. Once this was done the maps were compared with one another to further ensure that the land cover was classified in a uniform manner. Following these steps it is necessary to provide an assessment of the accuracy of the final land-cover maps.

4.3.4 Land-cover change

Once all land the cover-maps had been generated and edited, they were exported to IDRISI and analysed in the Land Change Modeler (LCM) for Ecological Sustainability. The LCM is an application, available in IDRISI and several other GIS, developed to enhance the capacity of GIS to analyse and predict land-cover change and subsequently put forward recommendations on habitat and biodiversity management. It was developed by Clark Labs in conjunction with the Internal Union for the Conservation of Nature (IUCN) to meet the specific needs of conservation planning (Clark Labs 2009).

All of the land-cover maps were projected to Hartebeesthoek94/Lo19. To remove smaller detected changes that often resulted from slight incongruencies between polygons of the same class and to facilitate the analysis, transformed areas of less than 9 ha were eliminated using a sliver removal technique. The minimum mapping size for features was informed by McDonald *et al*. (1984).

4.3.5 Integration with vegetation type data

The resultant land-cover change matrices were analysed using ESRI's ArcMap software to display the results in an appropriate and aesthetically appealing format. Remnants of natural vegetation identified as natural vegetation from each of the land-cover maps were then integrated with Mucina & Rutherfords vegetation map data to create a new layer for each year: 1986/1987, 1999/2000 and 2007. Change analysis was performed both on the completed land-cover datasets, as well as on the integrated layers using IDRISI land change modeller (LCM).

4.4 ACCURACY ASSESSMENT

An accuracy assessment could only be carried out on the 2007 land-cover map, because of the lack of available field data for the historical images. Appropriate aerial photography and higher-resolution satellite imagery of the study area were also not available to verify the 1986/1987 and 1999/2000 maps. As all the maps were generated using the same techniques it is assumed that they are comparably accurate. The next subsections describe the procedure used to assess the accuracy of the 2007 land-cover map.

4.4.1 Sampling scheme

A combination of field data and aerial photography was used for the reference data. These were manually classified according to the legend was used in this study and then compared with the 2007 land-cover map. Equation 4.1, taken from Congalton & Green (2009), was used to determine an appropriate sample size.

$$n = B\Pi_i(1 - \Pi i)/b_i^2 \tag{4.1}$$

Where n : simple random sample size

B : upper (α/k) x 100th percentile of the χ^2 distribution with 1 d.f. where k is the number of mutually exclusive and exhaustive categories (land-cover classes)

Π_i : proportion of the population in the ith category/land-cover class

b_i : absolute precision of the sample.

This was found to be 2070 samples at a confidence level of 95% and a precision level of 0.3. A minimum sample size of 75 to 100 points is recommended for mapped areas exceeding 1 000 000 acres (approximately 4 047 km²). As the study area has an area of nearly 9 000 km², a minimum of 100 samples per class was used for each of the 10 classes. The remaining 1070 samples were selected from different land-cover classes based on the proportion of the total area which each class constituted.

Of the 819 sample points that were collected during the two field surveys, 664 were considered appropriate for the accuracy assessment. Points that fell outside of the catchment were excluded as were points that fell on areas in which change had been registered between the three derived land-cover maps. Points not considered to fall under the same land-cover classification in the 2010 and 2011 surveys as on aerial photographs and SPOT-5 imagery from 2007 or 2008 were also excluded. Further, several points that were considered too close together to cover different pixels were excluded from the final selection.

The remaining 1406 points were selected at random from within various land-cover classes using the Hawth's tools extension in ArcMap 9.3. The sample points were taken at least 30 m apart and not less than 30 m from the edge of any given feature. These points were compared against aerial photographs, captured in 2007, in order to gauge their accuracy.

4.4.2 Mapping accuracy

The error matrix is presented in Table 4.4 sets out the producer's, user's and overall accuracy. The assessment revealed that errors developed as a result of the segmentation process which assigned multiple land-cover features into single polygons which were classified according to the majority of pixels each contained. Another source of error was the boundaries between features where overlap was witnessed. It follows that many of the boundaries between land-cover classes are approximations of the actual boundaries with accuracies within a certain margin of error.

Artificial bare areas were sometimes confused with natural bare areas and semi-natural vegetation. The confusion between artificial and natural bare areas quite likely occurred during the spectral classification due to the similar reflectance properties of certain features in these classes. Notably dams smaller than 3600 m^2 were often incorporated into surrounding agricultural land during the segmentation process. The urban vegetated areas class was confused with artificial bare areas. This occurred primarily due to the proximity of these two classes as well as the development of golf estates which typically contain multiple land-cover features such as cultivated and managed greens and fairways, semi-natural vegetated areas, sand, water and houses within a small area and in a manner that is unlikely to be replicated in any other areas.

The two aquatic vegetation classes were mapped with relative accuracy, but in 16 instances they were confused with one another. This probably occurred as a result of the difficultly of defining an appropriate threshold to consider these areas as woody. Semi-natural vegetation was classified as natural due to the inability of analytic and visual classification to discriminate between alien and indigenous vegetation. The low producer’s accuracy for semi-natural vegetation suggests that significant swathes of land in the Berg River catchment should be classified as such. During the course of the accuracy assessment it became apparent that much natural vegetation should be classified as semi-natural, largely due to the presence of alien plant species which were discernible in aerial photographs and those taken during the course of the field survey.

The accuracy with which most land-cover classes were recorded was fairly good considering the small scale at which they were mapped. Based on an assessment of 2000 and 2009 NLC maps for this area there appears to be a general agreement between the generated land-cover maps that and the NLC maps. Most major features such as towns, mines and quarries and large dams are clearly visible on all the land-cover maps so that the maps are reasonable representations of the land cover of the study area.

Table 4.5: Error matrix for the 2007 land-cover map

	Reference data										
Map data	**Artificial bare areas**	**Cultivated trees**	**Cultivation**	**Natural bare areas**	**Natural vegetation**	**Aquatic vegetation (herbaceous)**	**Aquatic vegetation (woody)**	**Semi-natural vegetation**	**Urban vegetated areas**	**Water**	**Row total**	**Producer's accuracy**
Artificial bare areas	89	0	2	9	3	0	0	12	3	0	118	77.4
Plantations	0	103	2	0	0	0	0	6	0	0	111	92.8
Cultivation	2	4	696	0	2	2	2	51	0	0	759	93.8
Natural bare areas	6	0	9	73	15	0	0	0	0	1	104	88
Natural vegetation	1	3	16	1	336	2	2	75	0	0	436	91.6
Aquatic vegetation (herbaceous)	0	0	4	0	4	82	7	2	0	6	105	85.4
Aquatic vegetation (woody)	0	0	4	0	0	9	87	2	0	0	102	87.9
Semi-natural vegetation	0	0	4	0	4	1	0	119	0	0	128	43.8
Urban vegetated areas	15	0	4	0	2	0	0	5	75	0	101	96.2
Water	2	1	1	0	1	0	1	0	0	100	106	93.5
Column total	115	111	742	83	367	96	99	272	78	107	2070	
User's accuracy	75.4	92.8	91.7	70.2	77	78.1	85.3	93	74.3	94.3		
Overall accuracy	85%											

Great difficulty was encountered in effectively delineating cultivated areas in the much of the north-western portion of the Berg River catchment. This was due to a variety of factors. For example, the patchwork distribution of agriculture in this area made it difficult to identify spatially aggregated land-cover patterns and in the analytical and manual classification differentiation between natural vegetation, semi-natural vegetation was hampered by the patchy appearance of sand fynbos.

4.4.3 Change accuracy

Assessing the accuracy of changes between land-cover maps presents many challenges and most studies dealing with land-cover change neither employ nor provide quantitative measures of change accuracy (Congalton & Green 2009). In this study sufficient reference data were not available for the 1986/1987 and 1999/2000 land-cover maps to conduct an assessment of change accuracy. This is a considerable concern as even high mapping accuracies can produce very low change accuracies.

The change accuracy between the maps was assessed qualitatively by inspecting areas in which change was recorded for various classes and by comparing the land-cover maps to raw Landsat images to visually interpret change registered between land-cover maps. Although this approach is susceptible to bias and inaccuracy it presented the most appropriate solution available. Most of the areas of change that were examined in this manner were verified as having experienced land-cover change. However, variation was observed between different classes. Change in classes such as artificial bare areas and water were mapped with a high degree of accuracy whereas others, particularly conversions between natural vegetation, semi-natural vegetation and cultivation were often indeterminate. This is largely due to the ambiguity between certain portions of these land-cover classes on the Landsat imagery, especially in the north-western areas of the catchment.

4.5 CONCLUSION

This chapter described the methods used to generate land-cover maps for the Berg River catchment for three time periods. The chapter has also described by which the resultant land-cover maps were analysed to establish land-cover change and how remnants of natural vegetation were classified according to Mucina, Rutherford & Powrie's (2007) vegetation map of South Africa, Lesotho and Swaziland. The results of the analyses of land-cover changes and changes in vegetation types are presented in the following chapter.

CHAPTER 5: RESULTS

This chapter reports the results of the land-cover mapping and change analysis, describes the general pattern of registered land cover and explores trends in change witnessed at a catchment scale. The discussion focuses on selected areas in the catchment that exhibited marked land-cover change over the study period. The recorded changes are situated within the socio-economic and historical contexts of the area. Changes in the different vegetation types of the Berg River catchment are described and the significance of these changes is explained. Finally, recommendations are put forward on the optimal future management of biodiversity in this area.

5.1 LAND-COVER PATTERNS

As this study aims to assess the spatial extent of land-cover change and explores the impacts that these changes have had on biodiversity in the Berg River catchment, land-cover maps generated from historical Landsat imagery were used as basis for the change analysis. The following subsections present the result of the land-cover change analysis in the Berg River catchment over a 20-year period. The first subsection will describe the general patterns of land cover while the second will highlight the dominant changes that have been observed over the study period. The final subsection presents some of the dominant factors driving land-cover change in the Berg River catchment and comments on likely land-cover changes in the future.

5.1.1 Land-cover maps

The land-cover-maps generated for 1986/1987, 1999/2000 and 2007 are presented in Figures 5.1, 5.2 and 5.3 respectively. The dominance of cultivation in the catchment is evident on all three maps, constituting approximately 65%, 64% and 62% of the catchment's total area in 1986/1987, 1999/2000 and 2007 respectively. Cultivation is largely confined to low-lying areas and concentrated in the middle and upper reaches of the catchment (Figure 1.1). Little cultivation is found in the mountainous areas that flank the catchment. Areas of cultivation exhibit a patchy distribution between Hopefield and Aroura indicative of extensive grazing, the practice of crop rotation and lengthy fallow periods. This pattern is further visible on the Saldanha peninsula where the agricultural landscape mingles with alternative land-cover classes such as natural and semi-natural vegetation. The patchwork distribution of cultivation in this area is a response to the poor quality of soils and limited precipitation which characterize the lower catchment.

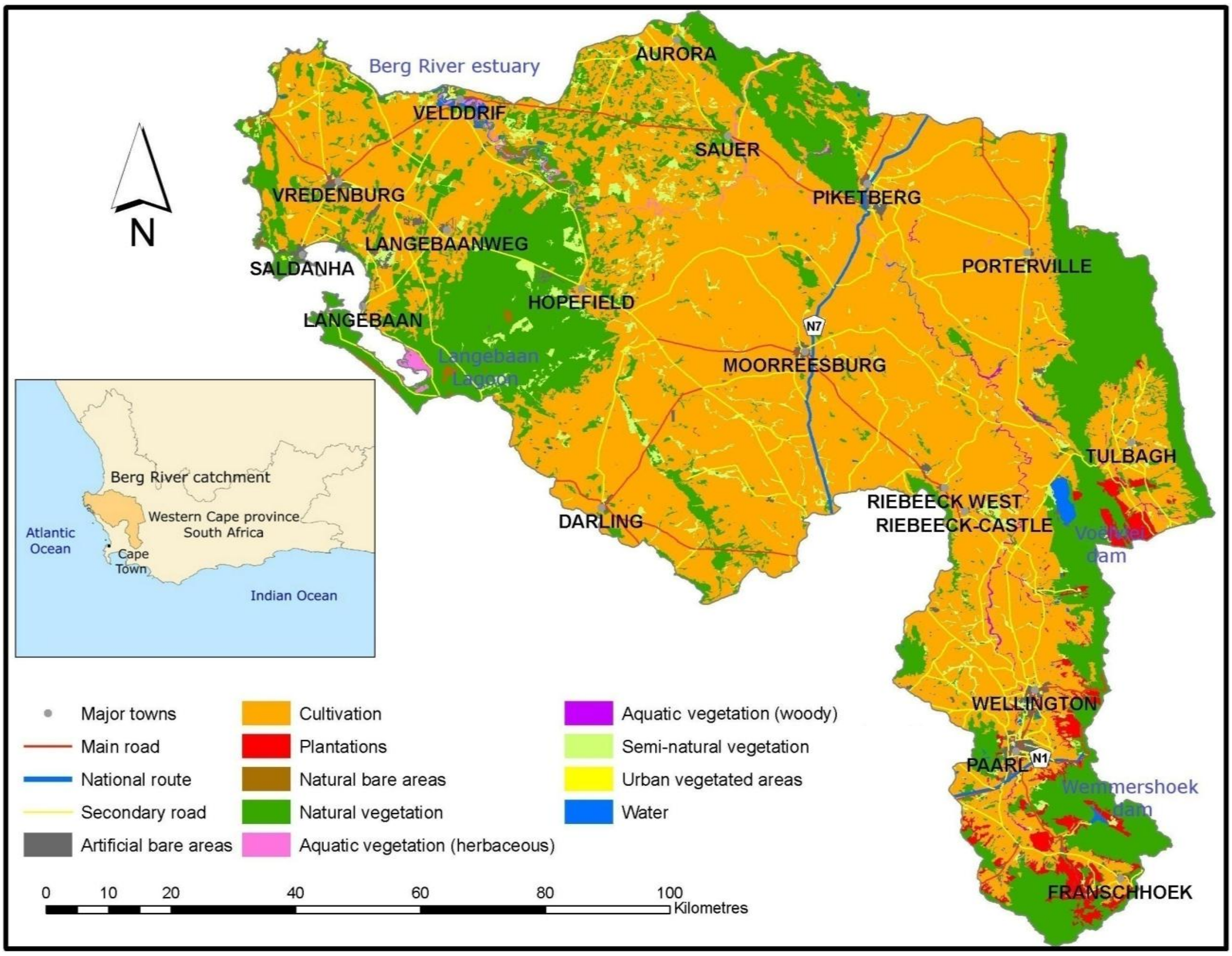


Figure 5.1: Land cover of the Berg River catchment, 1986/1987

Artificial bare areas are found throughout the catchment and are prominent around the major urban areas especially Paarl, Wellington, Moorreesburg and Vredenburg. Other noticeable artificial bare areas are found around Piketberg and Saldanha. Plantations are confined to the upper reaches of the catchment around Paarl, Wellington and Franschhoek. Large plantations are also located in the mountainous area south of Tulbagh. Traditionally afforestation has occurred mainly on steep slopes or areas otherwise unsuitable for cultivation and with large plantations often found lining mountain bases. Major waterbodies are the Voëlvlei and Wemmershoek dams and the Berg River estuary. Natural bare areas include beaches and areas of exposed earth throughout the catchment. These areas are visible along the coast, along lower reaches of the Berg River and irregularly between Hopefield and Langebaan but constitute a very small portion of the land cover of the Catchment. Aquatic vegetation is concentrated along the Berg River and around the wetlands at Langebaan lagoon and the Berg River estuary. Aquatic vegetation tends to display a prominent woody component in the upper catchment which gives way to herbaceous aquatic vegetation in the lower catchment.

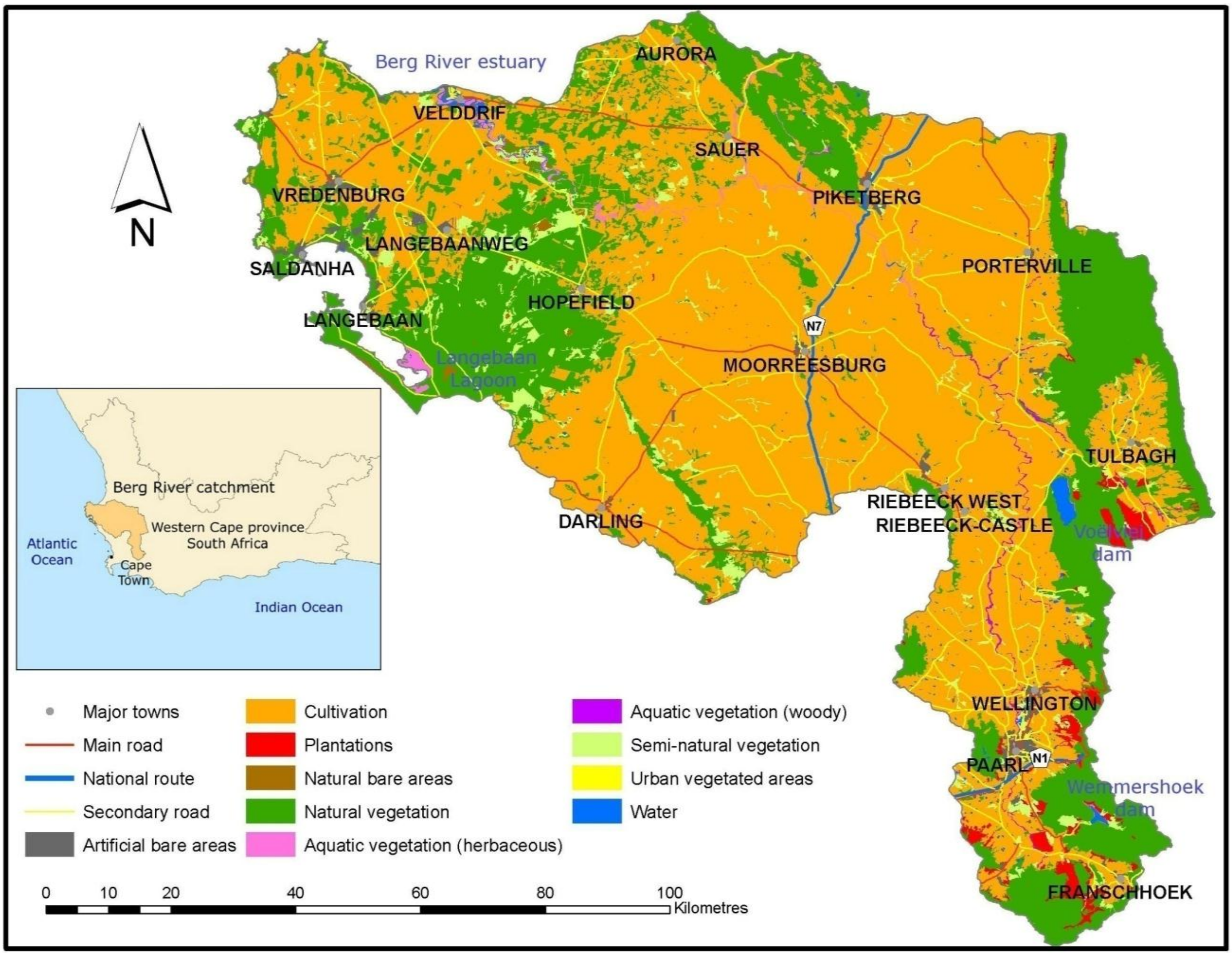


Figure 5.2: Land cover of the Berg River catchment, 1999/2000

Semi-natural vegetation displays a sporadic distribution within the Berg River catchment. Large expanses of semi-natural vegetation have been found in the southern catchment between Paarl and Franschhoek. The presence of semi-natural vegetation in this area is often the result of plantation clearing where denuded areas are left to develop a semi-natural vegetation cover. Semi-natural vegetation is also conspicuous between Darling and Aroura as a result of fires, invasion by alien vegetation and various agricultural practices. Urban vegetated areas tend to be located in or around the larger urban centres in the catchment and predominantly comprise golf courses and sports fields. Several large golf courses are located between Paarl and Franschhoek which constitute a large portion of the area occupied by this land-cover class. The development of golf courses on the Saldanha peninsula and along the coast around Velddrif has been largely responsible for the increased in spatial extent in this land-cover class which is discussed in the proceeding section.

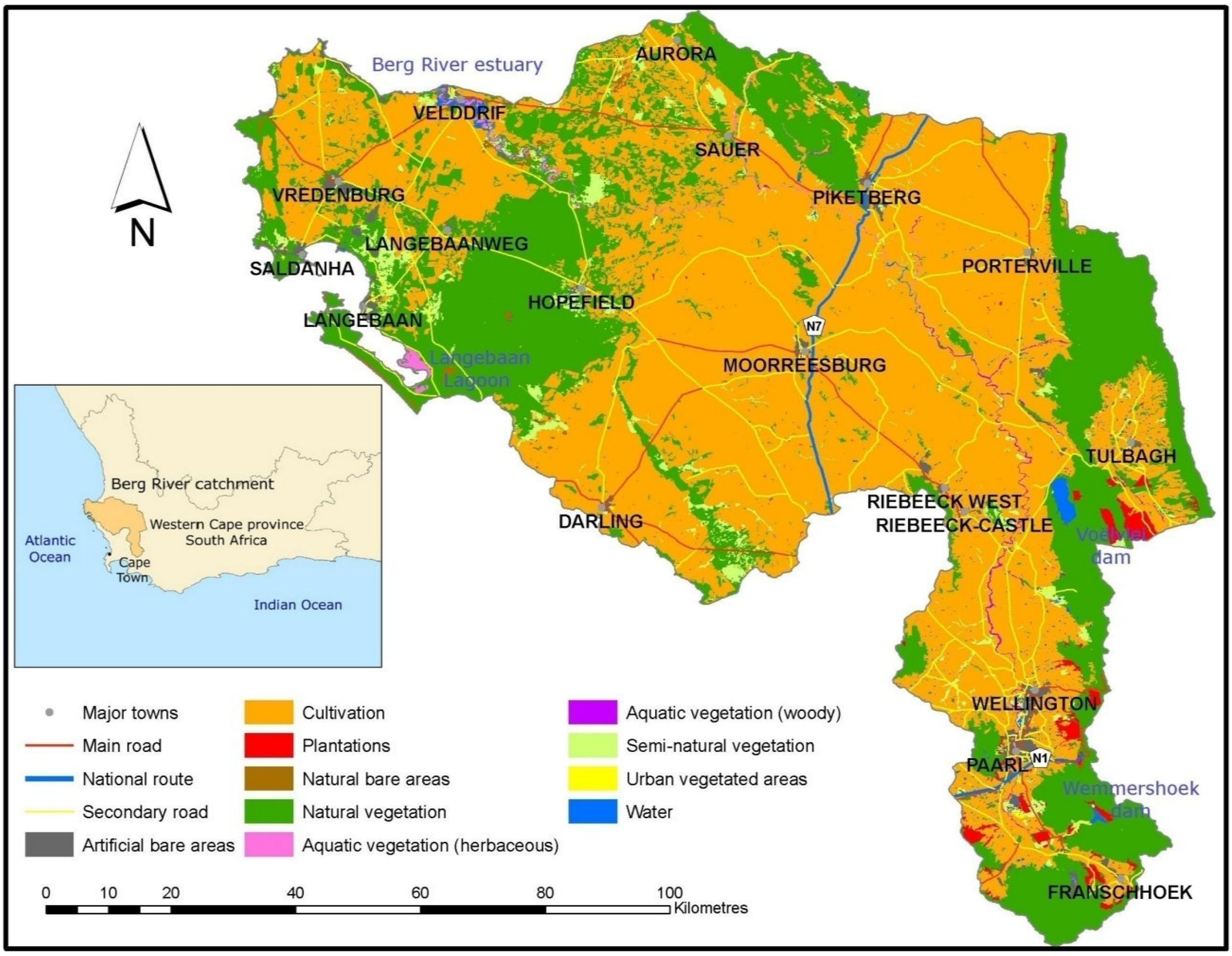


Figure 5.3: Land cover of the Berg River catchment, 2007

5.1.2 Land-cover change

While much of the land cover of the catchment has remained static (97% between 1986/1987 and 1999/2000 and 93% between 1999/2000 and 2007) over the study years, significant changes have manifested in several classes and areas. Salient changes are the expansion of artificial bare areas around Paarl and Wellington, reductions in the extent of commercial forestry in the upper reaches of the catchment and much of the vegetation between Langebaanweg and Hopefield which was classified as semi-natural in 1986/1987 and 1999/2000, but as natural in 2007. Other prominent changes are the conspicuous diminution of cultivation and a sharp increase in the area covered by natural vegetation. Also noteworthy is the sharp increase in extent witnessed in the urban vegetated areas land-cover class. The extent of land-cover change is recorded in Table 5.1 and it is discussed in greater detail in the succeeding sections.

Table 5.1: Area of land-cover classes in the Berg River catchment and percentage changes over time

Land-cover class	1986/1987		1999/2000		2007		1986/1987 - 1999/2000	1999/2000 - 2007	1986/1987 - 2007
	Area (km²)	Percentage	Area (km²)	Percentage	Area (km²)	Percentage	Change (%)	Change (%)	Change (%)
Artificial bare areas	117.2	1.3	127.4	1.4	146.9	1.7	8.7	15.3	25.3
Plantations	150.5	1.7	112.1	1.3	88.4	1.0	-25.5	-21.1	-41.2
Cultivation	5779.7	64.9	5682.3	63.8	5489.3	61.6	-1.7	-3.4	-5.0
Natural bare areas	32.7	0.4	28.6	0.3	34.7	0.4	-12.6	21.4	6.1
Natural vegetation	2445.2	27.5	2511.0	28.2	2797.1	31.4	2.7	11.4	14.4
Aquatic vegetation (herbaceous)	36.5	0.4	50.7	0.6	38.1	0.4	38.8	-24.9	4.3
Aquatic vegetation (woody)	21.0	0.2	26.2	0.3	20.7	0.2	24.7	-20.8	-1.2
Semi-natural vegetation	275.7	3.1	310.0	3.5	232.4	2.6	12.4	-25.0	-15.7
Urban vegetated areas	2.0	0.02	3.6	0.04	5.9	0.07	83.5	65.1	202.9
Water	46.8	0.5	55.5	0.6	53.6	0.6	18.6	-3.3	14.7

5.1.2.1 Artificial bare areas

Artificial bare areas in the catchment have increased in extent by over 25% between the 1986/1987 and 2007 land-cover maps. The expansion of artificial bare areas in the catchment was primarily the result of urban growth. Urban expansion accelerated between 1999/2000 and 2007 from 9% to 15%. Much of the registered change was concentrated around Paarl and Wellington with Franschhoek, Vredenburg and Langebaan also exhibiting noteworthy changes. Although constituting a relatively small portion of the total increase in this class, significant developments have occurred along the coast around Velddrif and Paternoster.

5.1.2.2 Plantations

The area occupied by plantations has decreased steadily and significantly by 41%, nearly halving the total area of this class over the whole study period. This development largely transpired in the upper catchment around Franschhoek, to the east of Paarl and Wellington and to the south of Tulbagh as shown in Figure 5.4 and is discussed in greater detail in succeeding sections.

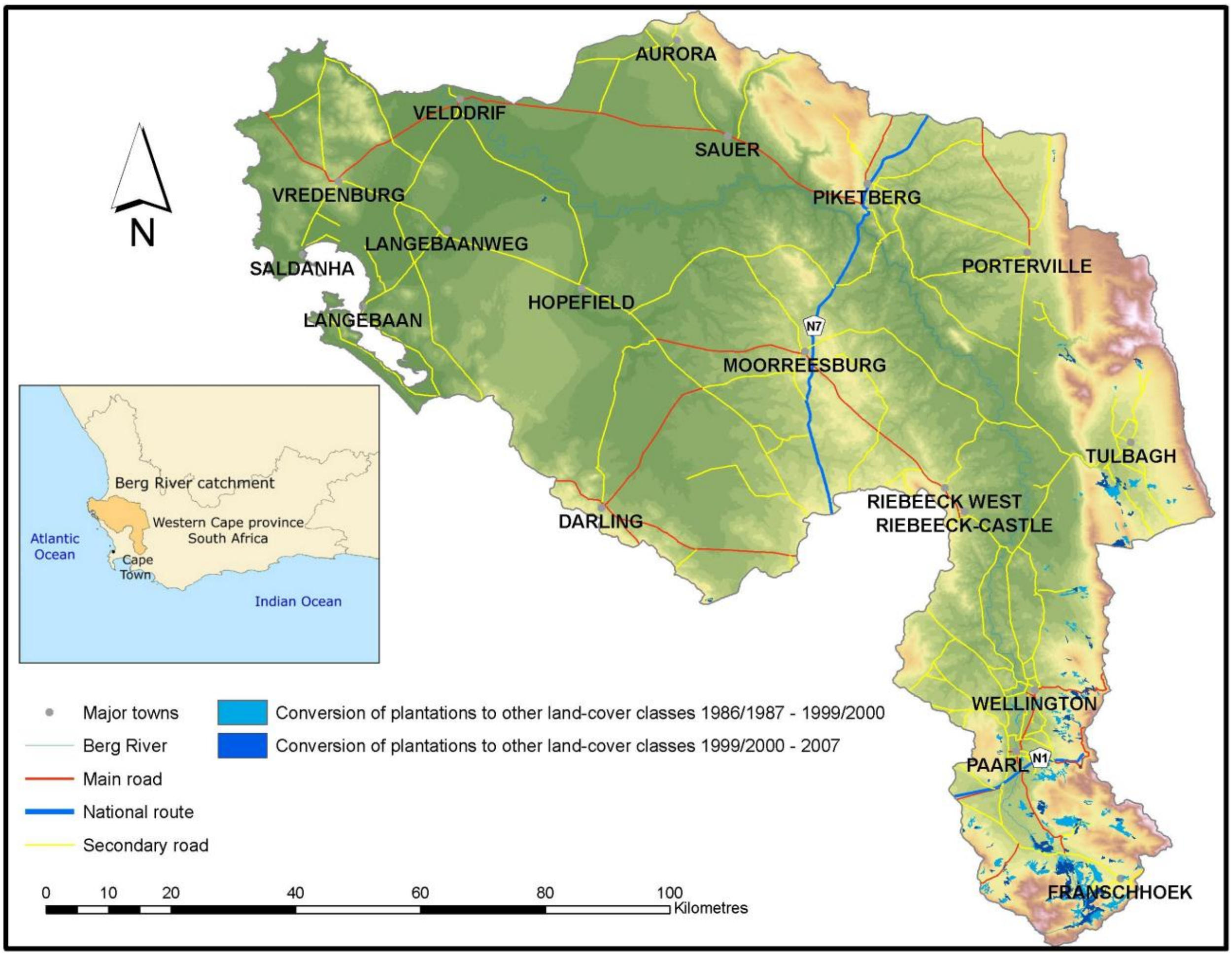


Figure 5.4: Conversions of plantations to other land-cover classes

5.1.2.3 Cultivation

Perhaps the most striking land-cover change is the sustained decline of agricultural areas in the catchment as shown in Figure 5.5. A total decrease of 5% was recorded between 1986/1987 and 2007. Due to the dominance of cultivated land, this translates to a reduction of about 290 km².

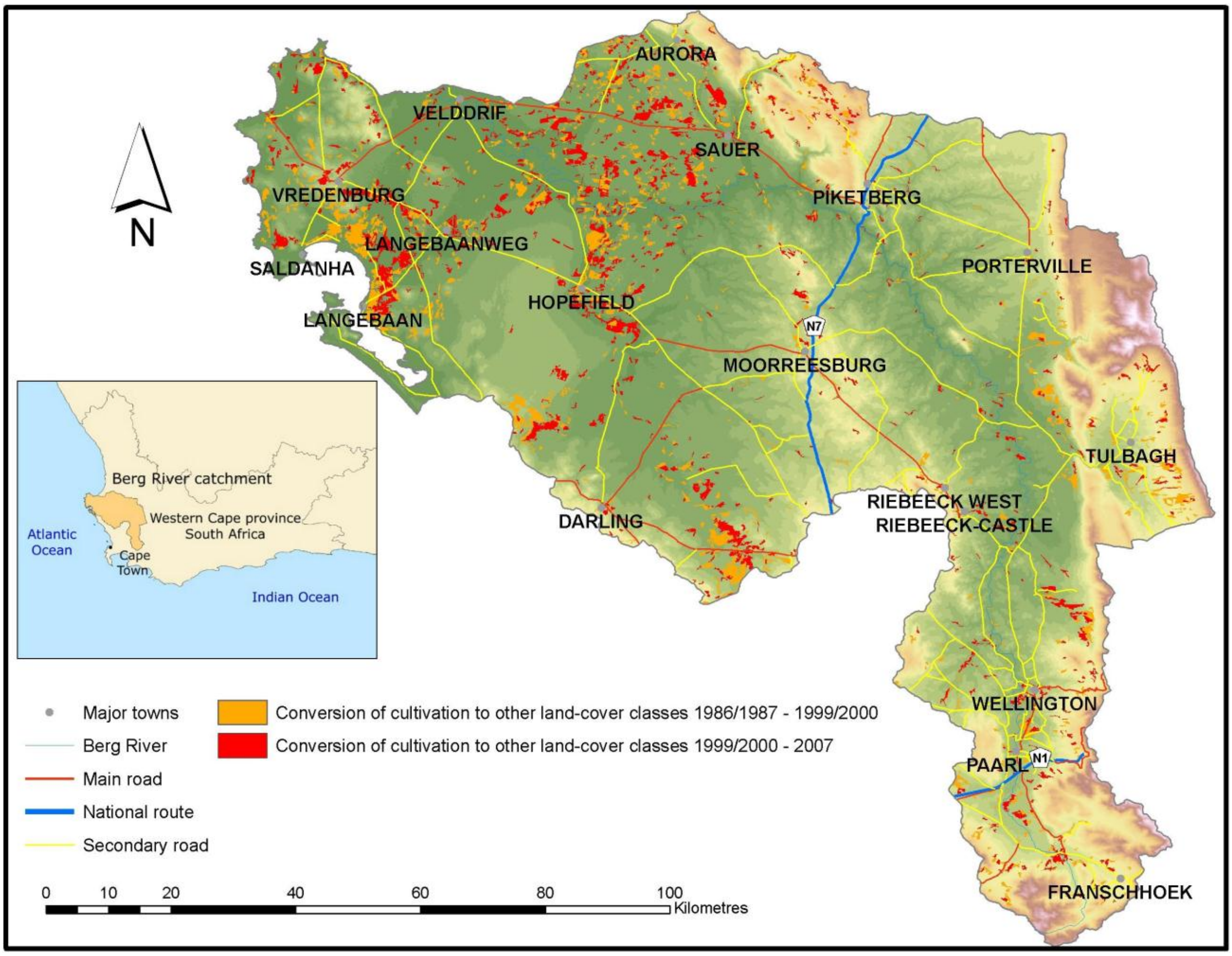


Figure 5.5: Conversions of cultivation to other land-cover classes

The bulk of this change is concentrated in three primary areas. The first is a large area between Hopefield, Sauer, Aurora and Velddrif. The second area to the east of Darling was also the site of significant reductions due to the establishment of the Riverlands Nature Reserve in 1985 (Holmes 2008). The reserve was established, in part, on disused fields with large stands of dense alien vegetation. The third is the area along the coast between Langebaan and Saldanha. Land-type maps obtained from the Agricultural Research Council (ARC), although of a very coarse scale, indicate that all of these areas have marginal agricultural potential. Caution should be used when interpreting these results as lengthy fallow periods are practiced in the lower catchment. Furthermore, areas that experience fire or have been brush cut may have been mistakenly identified as cultivation. Little change was witnessed in much of the rest of the catchment though the conversion of cultivation to artificial bare areas and urban vegetated areas occurred in the upper catchment as a result of the expansion of urban areas.

5.1.2.4 Natural bare areas

Natural bare areas originally increased between 1986/1987 and 1999/2000 then subsequently decreased between 1999/2000 and 2007. Much of this change can be attributed to seasonal variations in the satellite images used rather than any actual change in land cover. The 1986/1987 and 2007 Landsat-5 images used were both dry season images while the 1999/2000 Landsat-7 images were taken shortly after the winter rains. In the 1986/1987 image dried-up dams, pans and river banks would most likely have been classified as natural bare areas, then shown as inundated areas in the 1999 image and again as dried up areas in the 2007 image. Some areas likely exhibited reduced vegetation cover and were consequently classified as natural bare areas instead of cultivation or natural vegetation.

5.1.2.5 Natural vegetation areas

Natural vegetation increased by a total of 14% over 20 years to constitute nearly a third of the catchment's total area. Conversion of cultivation in the lower catchment and plantation clearing in the upper catchment and to the south of Tulbagh were the primary contributories to this increase. The location of gains in natural vegetation is shown in Figure5.6. Large gains were also recorded between Hopefield and Langebaanweg and they are likely a result of recently burnt areas being classified as semi-natural vegetation and later as natural vegetation when the vegetation cover regenerated.

The accuracy assessment of the 2007 land-cover map determined that significant portions of areas classified as natural vegetation was actually alien infested or otherwise degraded land. Moreover, vegetation that reclaimed previously cultivated areas is unlikely to exhibit as rich a compositional diversity as do areas of pristine vegetation cover.

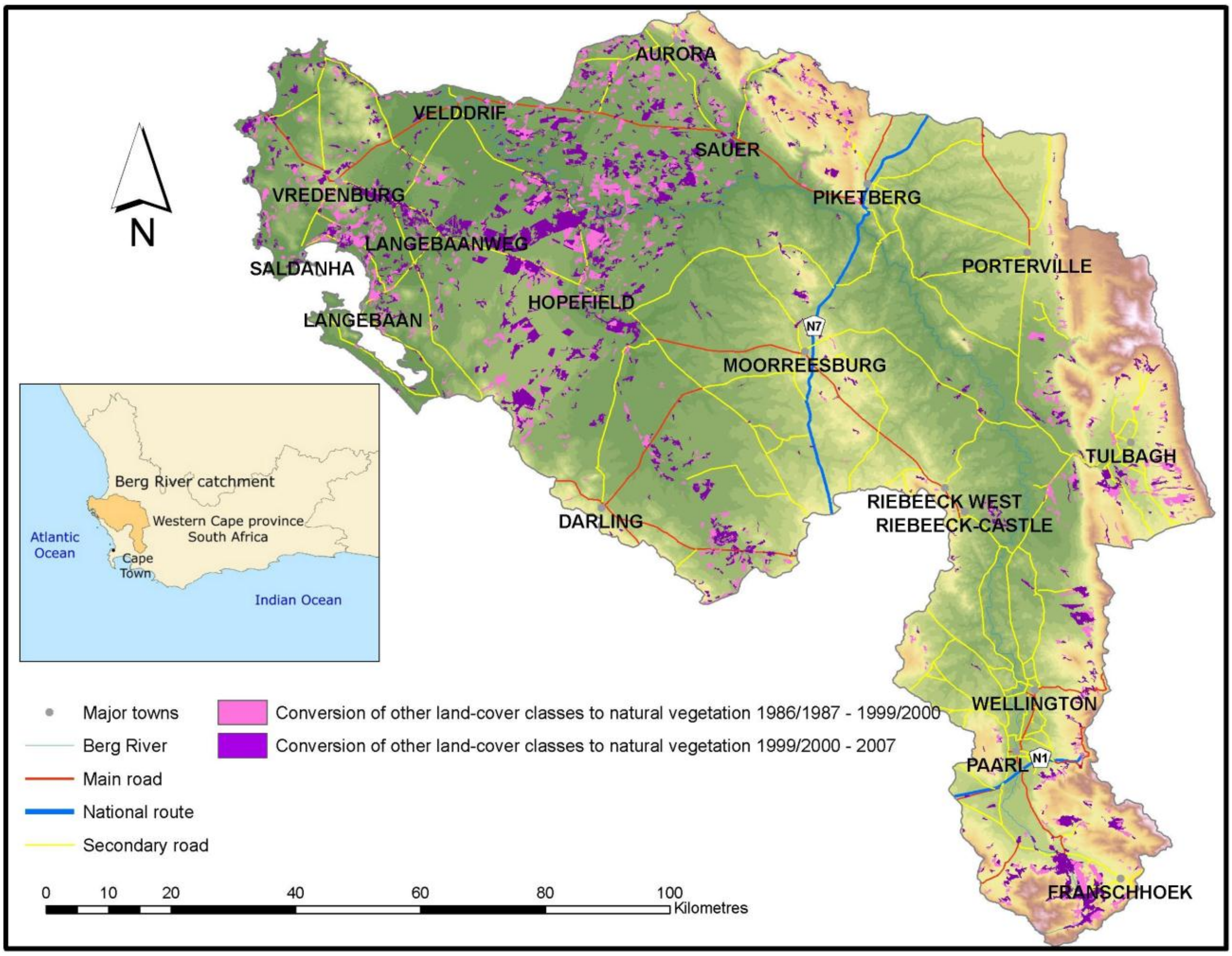


Figure 5.6: Conversions from other land-cover classes to natural vegetation

5.1.2.6 Aquatic vegetation (herbaceous)

As in the case with natural bare areas, fluctuations witnessed in herbaceous aquatic vegetation were most likely the result of seasonal variations between the various images used to generate the land-cover maps. The area covered by this class increased substantially between 1986/1987 and 1999/2000 only to recede by 2007. This is probably the result of increased flooding along river courses, higher water levels in dams and the inundation of vernal or other seasonal pools. The overall change in this class was, however, negligible considering its diminutive total area of less than 1% of the total study area.

5.1.2.7 Aquatic vegetation (woody)

Like the herbaceous vegetation class, this woody wetland and riparian vegetation manifested a marked increase in area of just under 25% between 1986/1987 and 1999/2000 and a subsequent decrease by 21% by 2007. Again, this is probably the result of seasonal variations in the images used. The effects of this phenomenon are less pronounced in the woody vegetation class as it is less sensitive to rapid fluctuations in water availability.

5.1.2.8 Semi-natural vegetation

Semi-natural vegetation initially increased in extent by 12% between 1986/1987 and 1999/2000 and subsequently significantly decreased by 25% between 1999/2000 and 2007. In the 1999/2000 land-cover map, large swathes of vegetation were classified as semi-natural only to be classified as natural in the 2007 land-cover map. Significant increases in the area covered by this class were evident around Langebaan in the 2007 land-cover map. However, considering the inaccuracy with which this class was mapped, caution should be employed when drawing inferences about the significance of the registered changes

5.1.2.9 Urban vegetated areas

Cultivated and managed urban vegetated areas in the catchment have expanded strikingly by 203% but still constitute a miniscule proportion (0.07% in 2007) of the total area of the catchment. The expansion of this class was primarily driven by golf course developments around Langebaan, Velddrif, Vredenburg and Paarl. Owing to the coarse resolution of the imagery used to develop the land-cover maps, urban vegetated areas such as smaller sports fields, parks and gardens were seldom registered in the land-cover maps.

5.1.2.10 Water

Waterbodies are characteristically sensitive to seasonal variation. Figure 5.7 shows a dam near Porterville in 1986/1987 (Figure 5.7a), 1999/2000 (Figure 5.7b) and 2007 (Figure 5.7c). In Figure 5.7a and Figure 5.7c the dam is noticeably smaller than in Figure 5.7b. The total area occupied by this class increased sharply by 19% between the 1986/1987 and 1999/2000 and decreased by 3% by 2007. This class increased in extent by 15% over the study period.

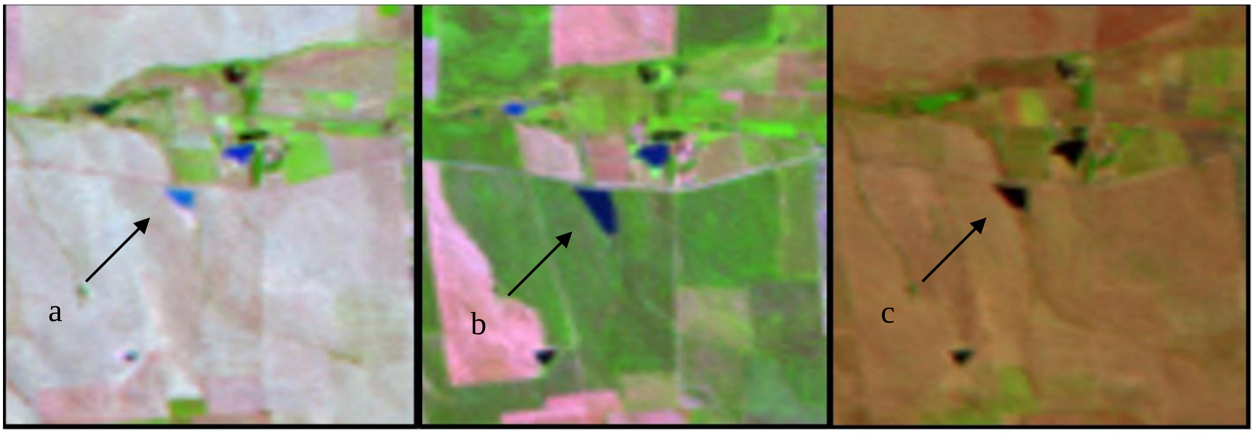


Figure 5.7: Seasonal variation of water for (a) 1986/1987, (b) 1999/2000 and (c) 2007

5.1.3 Factors driving land-cover change.

The conversion amounts (in km²) of land cover between 1986/1987 and 1999/2000 is expressed in Table 5.2. The most significant exchanges were between cultivation and natural vegetation with 75.7 km² of cultivated land being converted into natural vegetation. The expansion of Natural vegetation accounted for 78% of the decrease in cultivation. Factors driving the expansion of natural vegetation have been the removal of plantations and the conversion of semi-natural vegetation. Artificial bare areas have expanded by 13.4 km² into cultivated land as urban areas within the catchment have grown, fueled by a combination of in-migration and natural population growth. This has further contributed to the reduction of cultivation accounting for 14% of the decrease witnessed in cultivated areas.

Large areas (20 km²) of denuded plantations were classified as semi-natural vegetation accounting for just over 58% of the increase registered in the latter class between 1986/1987 and 1999/2000. Just less than 22 km² of natural vegetation on the 1986/1987 land-cover map was classified as semi-natural vegetation in the 1999/2000 land-cover map. This conversion occurred primarily in an area to the east of Langebaanweg. This occurrence is most likely the result of bush fires known to have occurred in this area early in 2000 (DWAF 2000). At this time extensive areas of natural vegetation were burnt giving the impression of a sparse, degraded vegetation cover. The expansion of water and aquatic vegetation between 1986/1987 and 1999/2000, mainly into cultivation, is no doubt attributable to seasonal variation between the images used to compile the land-cover maps.

Table 5.2: Net change in land-cover classes measured in km² between the 1986/1987 and 1999/2000 land-cover maps

Land-cover class	Artificial bare areas	Plantations	Cultivation	Natural bare areas	Natural vegetation	Aquatic vegetation (herbaceous)	Aquatic vegetation (woody)	Semi-natural vegetation	Urban vegetated areas	Water
Artificial bare areas		-1.3	13.4	0.9	-0.4	-0.5	0.02	-0.7	-0.4	-0.7
Plantations	1.3		-6.0	0	-20.0	-0.1	-1.2	-12.6	0	-0.3
Cultivation	-13.4	6.0		-1.2	-75.7	-7.3	-2.6	5.0	-0.4	-7.6
Natural bare areas	-0.9	0	1.2		3.4	-0.4	0	-6.3	0	-1.2
Natural vegetation	0.4	20.0	75.7	-3.4		-3.5	-0.8	-21.8	-0.4	-0.3
Aquatic vegetation (herbaceous)	0.5	0.1	7.3	0.4	3.5		0.2	0.4	0	1.8
Aquatic vegetation (woody)	-0.02	1.2	2.6	0	0.8	-0.2		0.5	0.01	0.3
Semi-natural vegetation	0.7	12.6	-5.02	6.3	21.8	-0.4	-0.5		-0.4	-0.7
Urban vegetated areas	0.4	0	0.4	0	0.4	0	-0.01	0.4		0.01
Water	0.7	0.3	7.6	1.2	0.3	-1.8	-0.3	0.7	-0.01	

The conversion between land-cover classes (in km²) from 1999/2000 to 2007 is recorded in Table 4.3. Most of the trends in land-cover change witnessed during the earlier 12-year period continued and some even accelerated in extent and rate. Foremost among these trends is the continued and accelerated decline of cultivation and plantations, and the expansion of artificial bare areas and natural vegetation.

The rapid expansion of natural vegetation in the later 7-year period has been primarily driven by the conversion of 142.9 km² of cultivation to natural vegetation. The conversion of cultivation to natural vegetation accounted for 50% of the increase in natural vegetation between 1999/2000 and 2007 and occurred primarily in an area between Velddrif, Hopefield, Sauer and Aurora. Another driving factor in the surge in natural vegetation was the reclassification of 107.1 km² of semi-natural vegetation as natural vegetation by 2007. The conversion of semi-natural vegetation to natural vegetation accounted for a further 37% of the increase in natural vegetation experienced between 1999/2000 and 2007 and was largely due to the re-establishment of vegetation cover in areas east of Langebaanweg. A total of 35.3 km² of cultivation was converted to semi-natural vegetation during the 1999/2000 to 2007 period. Exchanges between cultivation and semi-natural vegetation were concentrated in an area between Langebaan and Saldanha. The conversion of plantations to natural vegetation increased between 1999/2000 and 2007 with just under 20 km² of plantations being classified as natural vegetation by 2007. Artificial bare areas have continued to expand into cultivation and have done so at an increasing pace with 17.1 km² of cultivation being lost to artificial bare areas between 1999/2000 and 2007 compared to 13.4 km² between 1986/1987 and 1999/2000.

During the 20-year period the study data represented, considerable socio-economic and political transformations were experienced in South Africa. Although the relationships are complex and fraught with uncertainty, land cover as a manifestation of human activity has not been excluded from the effects of the aforementioned changes.

Table 5.3: Net change in land-cover classes measured in km² between the 1999/2000 and 2007 land-cover maps

Land-cover class	Artificial bare areas	Plantations	Cultivation	Natural bare areas	Natural vegetation	Aquatic vegetation (herbaceous)	Aquatic vegetation (woody)	Semi-natural vegetation	Urban vegetated areas	Water
Artificial bare areas		3.0	17.1	0.1	-5.9	0.1	-0.2	4.5	0.1	0.4
Plantations	-3.0		-0.9	0	-19.9	-0.1	0.1	0.6	0.02	-0.5
Cultivation	-17.1	0.9		-5.2	-142.9	4.0	3.6	-35.3	-1.3	-0.5
Natural bare areas	-0.1	0	5.2		-2.5	0.4	0.03	1.2	0	1.9
Natural vegetation	5.9	19.9	142.9	2.5		7.4	1.5	107.1	-0.9	1.0
Aquatic vegetation (herbaceous)	-0.1	0.1	-4.0	-0.4	-7.4		-0.3	-0.3	0	-0.3
Aquatic vegetation (woody)	0.2	-0.1	-3.6	-0.03	-1.5	0.3		-0.4	-0.03	-0.1
Semi-natural vegetation	-4.5	-0.6	35.3	-1.2	-107.1	0.3	0.4		-0.3	0.1
Urban vegetated areas	-0.1	-0.02	1.3	0	0.9	0	0.03	0.3		-0.03
Water	-0.4	0.5	0.5	-1.9	-1.0	0.3	0.1	-0.1	0.03	

Prior to the implementation of the Marketing of Agricultural Products Act of 1996 significant government intervention in the agricultural sector in the form of drought aid, production loans, interest subsidies and significant debt repayment coupled with good rains in the winter rainfall region served to boost agricultural production considerably (Kirsten, Van Zyl & Van Rooyen 1994; Borras 2003). The value of subsidies peaked in the mid-1980s and declined sharply over the course of the following two decades. This was coupled with rising producer costs, the increasing liberalization of agricultural markets, the abolition of price controls within the country and increased taxation rates (Kirsten, Van Zyl & Van Rooyen 1994). The effects of these reforms have been mixed, with some areas and sectors being adversely affected and others benefiting from decreased regulation and access to international markets. It has been claimed that the governmental self-sufficiency policies during this period contributed significantly to unsustainable agricultural production. In their absence cultivation, on marginal areas has ceased or decreased (Kirsten, Van Zyl & Van Rooyen 1994; Borras 2003). It has been noted earlier in this chapter that much of the decrease witnessed in cultivation occurred on land of marginal agricultural potential. It is, therefore, plausible that much of the decline witnessed in agricultural activity in the Berg River catchment may be attributed to shifts in governmental agricultural policies. While the trend may have been exaggerated, it is unlikely that it can be dismissed.

Over the 20-year study period, large areas of commercial forestry, particularly in the uppermost reaches of the catchment, have been removed. This process has been driven by the restructuring of state forest assets which began in earnest in 1998 (Ruiz 2003). The restructuring of state forest assets in the Western Cape included the privatization of significant portions of state-owned forest assets and the removal of 57 000 ha of forestry in favour alternative land uses (Ruiz 2003). About 15 000 ha of commercial forestry was removed from the Berg River catchment due to the lack of long-term profitability and high rate of water consumption. Most of this land was earmarked for conservation (Ruiz 2003). In addition, forest fires destroyed about 8 000 hectares of commercial forest plantations around Franschhoek (Currie, Milton & Steenkamp 2009). Currie, Milton & Steenkamp (2009) state that the Working for Water (WfW) programme has attempted to remove remaining *pinus* specimens as well as other invaders, particularly in riparian areas. Consequently these areas were classified as natural or semi-natural vegetation.

5.1.4 Trajectory of land-cover change

In order to elucidate the trajectory of land-cover change one needs to examine the economic, social, political as well as environmental incentives and constraints which have driven the process of land-cover change in the Berg River catchment. The trajectory of land-cover change in the study area will likely hinge on a variety of factors and cannot be predicted reliably based on the directions of change revealed in this study. Unquestionably, the artificial bare areas will continue to expand at an increasing rate due to population growth and urban development. Owing to the contribution of the river catchment to the City of Cape Town's water supply, further plantation clearing can be expected in the upper portions of the catchment. Changes in agriculture and natural vegetation cannot be dependably predicted until the factors underlying the changes have been uncovered and explored in greater detail.

5.2 CHANGES IN VEGETATION TYPES

The analysis of change in vegetation types within natural vegetation areas, mapped on the land-cover maps, has revealed marked changes in the spatial extent of several vegetation types over the 20-year period that was examined in this study. This section is concerned with documenting and describing these changes. Changes in each of the following vegetation types will be discussed: fynbos, renosterveld, sand fynbos, strandveld and azonal vegetation.

5.2.1 General trends

Contrary to expectations, significant increases in most vegetation types are observable with 23 of 28 listed vegetation types undergoing a net increase in area over the study period. Table 5.4 lists 28 of the 31 vegetation types found in the Berg River catchment. Cape Coastal Lagoons, Cape Vernal Pools and Cape Inland Salt Pans were excluded from the analysis but will be discussed in Section 5.2.5. Table 5.4 sets out the changes recorded in different vegetation types between 1986/1987, 1999/2000 and 2007. Table 5.4 records the remaining extent in km² of the vegetation types found within the study area. The remaining extent is also expressed as a percentage of their potential extent (hypothetical extent prior to anthropogenic disturbance) within the catchment. The change in area between 1986/1987, 1999/2000 and 2007 is also shown as a percentage.

The contemporary distribution of natural vegetation is clearly linked to terrain (compare Figure 1.1) in most of the catchment with large swathes of relatively undisturbed vegetation found on the mountains that line the eastern and southern-most portions of the catchment, as well as on the

Piketberg in the northern-most portion of the catchment. Vegetation types found in these areas are Hawequas Sandstone Fynbos, Kogelberg Sandstone Fynbos, Northern Inland Shale Band Vegetation, Western Altimontane Sandstone Fynbos, Western Coastal Shale Band Vegetation, Winterhoek Sandstone Fynbos, Olifants Sandstone Fynbos and Piketberg Sandstone Fynbos. The extent of all of these vegetation types has remained fairly static over the study period. The resistance of these vegetation types to alteration is likely due to the difficulties associated with cultivation on steep or rocky slopes. Furthermore, many of these naturally vegetated areas are located in mountain catchment areas and are thus not susceptible to anthropogenic transformation. The extent of differing vegetation types contained within protected areas will be discussed in detail in Section 5.4.1. Moderate changes were observed in several strandveld vegetation types located around Saldanha, Langebaan and Vredenburg, these will be discussed in Section 5.3.1

Another notable area of natural vegetation is a large swathe between Langebaan and Hopefield. The dominant vegetation types in this area are Atlantis Sand Fynbos and Hopefield Sand Fynbos. Substantial expansions in both Atlantis Sand Fynbos and Hopefield Sand Fynbos, have been witnessed over the study period and will be discussed in greater detail in Section 5.3.2. Remnants of various West Coast renosterveld vegetation types are visible on hilly outcrops throughout the middle catchment with smaller fragments confined to gullies that exist between large cultivated fields. Many of these are known to be heavily infested with alien vegetation and their ability to represent the exceptional diversity of natural vegetation is questionable (Mucina & Rutherford 2006). Several alluvium fynbos types are found in the upper reaches of the catchment. With the exception of Breede Alluvium Fynbos, Breede Shale Renosterveld and Swartland Silcrete Renosterveld, the extent of most renosterveld and alluvium vegetation types has remained moderately stable over the study period.

Riparian vegetation, classified as aquatic vegetation, is clearly visible along the course of the Berg River and some of its tributaries (Figures 5.1, 5.2 and 5.3). This vegetation exhibits a large woody component in the upper catchment which is lacking in its lower reaches. Much of the woody riparian vegetation is dominated by alien trees observed during the field surveys so that this vegetation is not representative of a natural and indigenous vegetation cover. The herbaceous aquatic vegetation of the catchment is assumed to occur in a relatively natural state along the course of the Berg River but it has quite likely been affected by altered flow regimes, deteriorating water quality and runoff from adjacent agricultural areas (RHP 2004).Despite fluctuations in the area covered by aquatic vegetation over the study period no pertinent changes in these areas have been recorded.

Table 5.4: Changes in vegetation types in the Berg River catchment

Vegetation type	1986/1987		1999/2000		2007		1986/1987 - 1999/2000	1999/2000 - 2007	1986/1987 - 2007
	Area (km²)	Percentage of potential extent	Area (km²)	Percentage of potential extent	Area (km²)	Percentage of potential extent	Change (%)	Change (%)	Change (%)
Atlantis Sand Fynbos	35.0	18.4	38.2	20.1	58.1	30.6	9.2	52.0	66.0
Boland Granite Fynbos	104.8	41.3	112.1	44.2	130.9	51.6	7.0	16.7	24.9
Breede Alluvium Fynbos	8.1	25.7	13.7	43.4	13.6	43.0	69.1	-0.9	67.6
Breede Shale Fynbos	51.8	44.5	48.9	42.0	50.3	43.2	-5.6	2.8	-2.9
Breede Shale Renosterveld	12.0	8.3	16.1	11.2	18.0	12.5	34.9	11.7	50.8
Cape Estuarine Salt Marshes	27.8	69.9	28.2	70.9	28.6	71.9	1.4	1.4	2.8
Cape Seashore Vegetation	0.3	9.1	0.5	11.9	0.7	19.2	31.3	60.9	111.3
Cape Winelands Shale Fynbos	10.4	60.8	11.7	68.4	11.6	68.0	12.4	-0.6	11.8
Hawequas Sandstone Fynbos	214.9	87.9	218.5	89.4	222.7	91.1	1.7	2.0	3.6
Hopefield Sand Fynbos	606.1	41.4	630.4	43.0	761.3	51.9	4.0	20.8	25.6
Kogelberg Sandstone Fynbos	87.9	83.9	93.0	88.8	99.8	95.3	5.8	7.3	13.6
Langebaan Dune Strandveld	126.1	47.0	133.4	49.7	135.1	50.4	5.8	1.3	7.2
Leipoldtville Sand Fynbos	2.5	11.3	1.4	6.4	1.4	6.3	-43.1	-1.8	-44.2
Northern Inland Shale Band Vegetation	22.1	92.1	21.8	90.9	22.5	93.5	-1.4	2.9	1.5
Olifants Sandstone Fynbos	38.8	97.6	38.7	97.4	38.7	97.4	-0.2	0.04	-0.2
Piketberg Sandstone Fynbos	244.5	79.9	246.4	80.5	256.6	83.9	0.8	4.2	5.0
Saldanha Flats Strandveld	235.6	33.8	254.4	36.5	291.4	41.8	8.0	14.6	23.7
Saldanha Granite Strandveld	64.6	27.9	70.5	30.4	75.1	32.4	9.1	6.5	16.3
Saldanha Limestone Strandveld	16.3	45.8	19.6	55.0	19.9	55.9	20.1	1.6	22.0
Southern Afrotemperate Forest	0.3	95.0	0.3	99.9	0.3	100.0	5.1	0.1	5.2
Swartland Alluvium Fynbos	91.0	21.9	88.9	21.4	97.4	23.5	-2.3	9.6	7.1
Swartland Alluvium Renosterveld	16.9	32.8	18.8	36.5	22.7	44.0	11.3	20.7	34.4
Swartland Granite Renosterveld	30.5	7.7	28.9	7.3	26.1	6.6	-5.5	-9.5	-14.4
Swartland Shale Renosterveld	120.8	3.5	116.9	3.4	139.9	4.1	-3.3	19.7	15.8
Swartland Silcrete Renosterveld	1.6	2.3	1.0	1.4	1.7	2.4	-38.3	73.4	7.0
Western Altimontane Sandstone Fynbos	2.0	100.0	2.0	100.0	2.0	100.0	0.0	0.0	0.0
Western Coastal Shale Band Vegetation	10.4	81.7	10.9	85.8	11.9	93.7	5.1	9.1	14.6
Winterhoek Sandstone Fynbos	298.8	99.3	296.7	98.6	296.6	98.6	-0.7	-0.03	-0.7

It is important to note that vegetation types occupying relatively small areas of the catchment are more prone to drastic changes as small alterations to their extent contributed to seemingly significant overall changes. This is evident in Table 5.4 where large percentage changes tend to be concentrated in vegetation types with a confined spatial extent while vegetation types occupying a greater total area recorded more moderate changes in extent (compare Table 1.1).

5.2.2 Fynbos

Hawequas Sandstone Fynbos, Kogelberg Sandstone Fynbos, Western Altimontane Sandstone Fynbos, Band Vegetation, Winterhoek Sandstone Fynbos, Olifants Sandstone Fynbos and Piketberg Sandstone Fynbos vegetation types are located in mountainous areas which have limited potential for agriculture or settlement so that they have been spared extensive transformation. Further, much of the fynbos-type vegetation located within the catchment is presently located within mountain catchment reserves and provincial reserves making significant transformations unlikely in the near future. This state of affairs has prevailed over the study period with most fynbos vegetation remaining fairly static and not exhibiting significant changes, the exception being Breede Alluvium Fynbos and Boland Granite Fynbos. Both vegetation types are located in low-lying areas and have been susceptible to anthropogenic transformation for cultivation.

Breede Alluvium Fynbos increased by 68% over the 20-year period. Owing to the small total area occupied by this vegetation type the net gain was an area of only 4.5 km^2. Breede Alluvium Fynbos is located within the Tulbagh valley and its increase in area is attributable to the abandonment of several agricultural fields over time. However, compositional diversity of reclaimed areas has not been assessed in this study and the re-establishment of this vegetation type is not documented in the literature. Boland Granite Fynbos is located in the southernmost portions of the catchment and has, in the past been cleared for cultivation and plantations. This vegetation witnessed a net increase of 25% between 1987/1986 and 2007. This translates to an area of 26.1 km^2. Gains recorded in Boland Granite Fynbos were mainly driven by the removal of plantations around the Wemmershoek dam and on the mountains to the west of Franschhoek. The restoration potential of fynbos vegetation following the clearing of plantations is discussed further in Section 5.3.3 and it is unlikely that denuded areas will exhibit species diversity comparable to pristine vegetation.

5.2.3 Renosterveld

Renosterveld is a lowland vegetation type endemic to the fynbos biome and is typically found on flat clayey soils that are well suited for several types of cultivation (Mucina & Rutherford 2006). Consequently, much renosterveld had been transformed prior to 1986. Successive gains in the area occupied by Breede Shale Renosterveld and Swartland Alluvium Renosterveld vegetation types were recorded between 1986/1987, 1999/2000 and 2007. Swartland Shale Renosterveld and Swartland Silcrete Renosterveld decreased in area between 1986/1987 and 1999/2000 and subsequently increased in extent between 1999/2000. Swartland Granite Renosterveld decreased in extent by 14% over the study period.

Significant (4.1 km² or 35%) expansion of Breede Shale Renosterveld was registered between 1986/1987 and 1999/2000 as a result of the clear felling and subsequent abandonment of a forest plantation to the south of Tulbagh. The trend continued from 1999/2000 to 2007 with a nearly 12% increase representing an area of 1.9 km². Figure 5.8 shows a reclaimed area of Breede Shale Renosterveld. Despite the seemingly natural appearance of this area, young pine and eucalyptus trees were noted during the field surveys. The presence of invasive alien species and relatively low species diversity was observed (Figure 5.8) indicating that restoration is required before this area can constitute an indigenous vegetation cover.

Swartland Alluvium Renosterveld registered an increase of 1.9 km² between 1999/2000 and 2007 and 3.9 km². This can be attributed to the expansion of renosterveld into formerly cultivated land as well as the reclassification of semi-natural vegetation to natural vegetation. Swartland Alluvium Renosterveld is found to the east of Darling where it occupies sandy riverine areas that have traditionally held little prospect for cultivation. The establishment of the Riverlands Nature Reserve and clearing of alien vegetation around it in 1985 has also contributed to the expansion witnessed in this vegetation type. A field visit to the area revealed that invasive alien species afflict much of this vegetation type.

Figure 5.8: Breede Shale Renosterveld recovering following the removal of plantation near Tulbagh

Swartland Shale Renosterveld is a critically endangered vegetation type with in excess of 95% of its original extent cleared primarily to make way for cultivation (Mucina & Rutherford 2006). This vegetation type was found to have decreased in area by 0.1 km² between 1986/1987 and 1999/2000 only to increase by 0.7 km² from 1999/2000 to 2007. Much of the Swartland Shale Renosterveld located in the catchment on the land-cover maps was in isolated fragments scattered throughout the catchment. The edge of a formerly cultivated area, reclaimed by Swartland Shale Renosterveld, is shown in Figure 5.9. Despite the presence of typical Renosterveld species, the area does not display a high level of diversity. Heelemann's (2010) findings note that renosterveld will spontaneously re-establish to some degree but will seldom display extensive species diversity. *Elytropappus rhinocerotis* (Renosterbos) is known to re-establish fairly quickly giving the superficial impression of an indigenous vegetation cover.

Figure 5.9: Swartland Shale Renosterveld reclaiming an agricultural area

Swartland Silcrete Renosterveld follows a sporadic distribution in the Berg River catchment with a large patch located to the east of Darling. This vegetation type was found to have decreased sharply by 38% between 1986/1987 and 1999/2000 and then increased by 73% from 1999/2000 to 2007. However, as the changes in this vegetation type constitute areas of only 0.6 km² and 0.7 km² respectively, it is unlikely that this finding is indicative of large changes in this vegetation type. Swartland Granite Renosterveld decreased by 14% over the study period denoting a loss of 4.5 km². This loss was caused by the conversion of small remnants of natural vegetation to cultivation.

5.2.4 Sand fynbos and strandveld

Sand fynbos and strandveld are patchy vegetation types that have developed on sandy soils. Much of these vegetation types are located in areas of marginal agricultural potential and as a result are only partially transformed (Holmes 2008). Atlantis Sand Fynbos increased by a total of 66% or an area of 33.1 km², with most (52%) of this change registered between 1999/2000 and 2007. This increase is largely attributable to the establishment of the Riverlands Nature Reserve where considerable effort was put into restoring indigenous vegetation. It was noted by Holmes

(2008) and also in the accuracy assessment that alien vegetation is prolific in this area and that much of the gains made by this vegetation type understate the presence of alien vegetation.

Hopefield Sand Fynbos increased by 26% during the course of the study period adding 155.2 km² to its extent in 1986/1987. This was largely in response to cultivation being classified as natural vegetation between 1999/2000 and 2007. While this vegetation type is known to re-establish itself, it is unlikely that it will exhibit an appropriate degree of compositional diversity in the absence of extensive restoration efforts (Helme *pers com* 2011). In an environmental impact assessment (EIA) conducted just north of the catchment boundary, a patch of re-established Hopefield Sand Fynbos was found to have a compositional diversity of only 10% of a pristine area (Helme *pers com* 2011). This is largely ascribable to the destruction of bulbous plants during ploughing. Many species are adversely affected by changes in soil quality often associated with cultivation. The relatively large increase in the areal extent of Hopefield Sand Fynbos has been greeted with skepticism by those who have worked extensively on this vegetation type as it is suspected that large swathes of vegetation classified as natural is infested with alien vegetation or is otherwise degraded. Leipoldtville Sand Fynbos registered a decline of 44% from 1986/1987 to 2007 losing an area of 1.1 km² largely due to the transformation of a single swathe of natural vegetation to cultivation.

Langebaan Dune Strandveld, Saldanha Flats Strandveld, Saldanha Granite Strandveld and Saldanha Limestone Strandveld all increased in area over the study period. Large gains were evidenced in Saldanha Flats Strandveld which increased by 23% (55.5 km²). Saldanha Granite Strandveld noted a 16% increase following the conversion of 10.5 km² of semi-natural vegetation and cultivation to natural vegetation. Saldanha Limestone Strandveld registered an increase of 22% (3.6 km²). Modest gains were experienced in Langebaan Dune Strandveld which grew by 9 km², an expansion of 7%. The increase in strandveld vegetation types can be attributed to the cessation of cultivation around Saldanha and Langebaan and the reclassification of semi-natural vegetation as natural vegetation. The factors driving this development are described in Sections 5.3.1 and 5.3.2.

5.2.5 Azonal vegetation

Cape Inland Salt pans and Cape Vernal Pools are not included in Table 5.4. Even though these areas do not typically exhibit vegetation cover, they still constitute important features for local biodiversity by harbouring unique ecosystems and highly confined habitats. Cape vernal pools are important breeding grounds for amphibians and other animal species. The vegetation map used in this study records very few of these pans and pools and during the field survey and post classification editing several such areas, not included in the vegetation map, were noticed. Figures 5.10 and 5.11 show two such areas where it appears that numerous salt pans, not included in Mucina & Rutherford (2006), occur. In addition to these areas, vernal pools have been located throughout the lower and middle catchment suggesting that these areas were not mapped in sufficient detail in Mucina & Rutherford's (2006) vegetation map to warrant consideration in this assessment. The failure of Mucina & Rutherford (2006) to delineate these areas is likely the result of the 1:1 000 000 scale at which it was published and because the salt pans and vernal pools identified on Figures 5.10 and 5.11 are smaller than 3 ha.

Neither Northern Inland Shale Band Vegetation nor Western Coastal Shale Band vegetation types exhibited significant change as they are disposed to locations within mountain catchment reserves in terrain unsuitable for agriculture and urban development. Despite the large increase of more than 110% of Cape Seashore Vegetation, it is unlikely that the real extent of this vegetation type has changed as it is located primarily within the West Coast National Park. The total area occupied by this vegetation type is minimal being 0.3 km^2 in 1986/1987, 0.5 km^2 in 1999/2000 and 0.7 km^2 in 2007. Registered changes can be attributed to shifting natural bare areas, beaches and coastal dunes within which Cape Seashore Vegetation is found.

Mucina, Rutherford & Powrie's (2007) vegetation map shows two small patches of Afromontane Forest in the Berg River catchment. Both areas are in a mountain catchment reserve and they were identified as natural vegetation on all three land-cover maps. However, it seems that these areas have long since been denuded and replaced by surrounding fynbos species. Incorporation of Afromontane Forest into the change assessment is therefore not justifiable.

Figure 5.10: Discrepancies between Landsat-identified salt pans and those mapped by Mucina, Rutherford & Powrie (2007) in an area west of Aurora

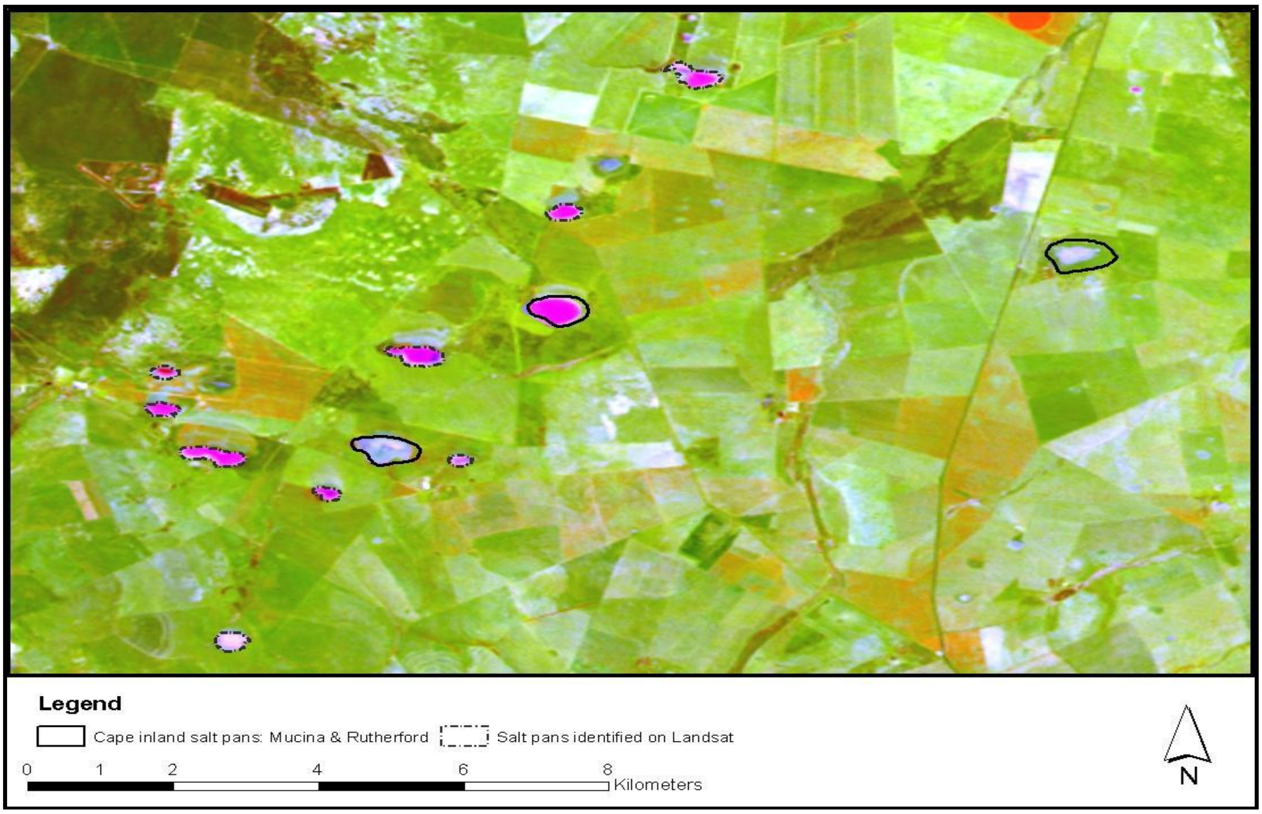


Figure 5.11: Discrepancies between Landsat-identified salt pans and those mapped by Mucina, Rutherford & Powrie (2007) in an area north of Darling

5.3 AREAS EXHIBITING EXCEPTIONAL LAND-COVER CHANGE

The analysis of land-cover change in the Berg River catchment revealed three areas exhibiting marked levels of land-cover change during the 20-year study period. These are the Saldanha peninsula, the agricultural areas between Hopefield, Velddrif, Sauer and Aurora and the south eastern portion of the Berg River catchment. The changes evident in these areas are described in greater detail in the following sections.

5.3.1 Saldanha peninsula

Many land-cover changes were registered on the Saldanha peninsula. The most significant of these are the urban expansion of Langebaan and the development of a golf course close to the town. Urban expansion is also evident around Saldanha. Large areas previously identified as cultivation in 1986/1987 have now been classified as semi-natural vegetation. During the field survey it was noted that some of these areas still appear to be grazed. Over the period between 1986/1987 and 2007 several areas that were previously classified as semi-natural vegetation or cultivation have been reclassified as natural vegetation suggesting that these areas have recovered at least a superficially natural vegetation cover. During the field survey furrows were noted on much semi-natural vegetation indicating that the area is either fallow, abandoned or used for agricultural practices other than cultivation. Mittal Steel South Africa's plant at Saldanha, was brought online in 1998. The development of this facility is clearly visible in the far north in Figure 5.12. It is assumed that cultivated areas surrounding the plant have been left to develop a natural or superficially natural vegetation cover.

During the 20 year period covered by this research, large areas of cultivation and plantations have given way to natural and semi-natural vegetation. The ability of natural vegetation to reclaim previously cultivated areas is dependent on a host of factors. These include the length of time the area was cultivated, the inputs that were used, especially nitrogen-based fertilizers, and the availability of an adjacent seed source (Holmes *pers com* 2011). Ploughing has a particularly adverse effect on species such as slow-growing bulbous plants (Holmes *pers com* 2011). In other areas that have been examined various vegetation types in the CFR have been found to re-establish but their typical species diversity is markedly lower than in a pristine area they and are dominated by a few resilient species with most of the rare species disappearing completely (Helme *pers com* 2011). A more extensive study into the capacity of indigenous vegetation to reclaim areas previously occupied by another land-cover class in this area would provide a further level of understanding of the impacts of land-cover change on biodiversity.

Figure 5.12: Land-cover changes in the Langebaan area

5.3.2 Decline of agriculture between Hopefield, Velddrif, Sauer and Aurora

Contrary to initial expectations, considerable reclamation of natural areas at the expense of cultivation was identified. A significant portion of this change was concentrated in a small, relatively well-defined geographical area between Aurora and Velddrif. Figure 5.13 illustrates the spatial distribution of the gains in natural and semi-natural vegetation between 1986/1987 and 2007. Most of these changes occurred within the Hopefield Sand Fynbos vegetation type.

The area has marginal agricultural potential, characterized by nutrient-poor, acidic soils with a low capacity to retain water and nutrients as well as a notably low cation exchange capacity (Holmes 2008). The area is subject to lengthy periods of crop rotation during which large areas are known to develop semi-natural vegetation cover (Helme *pers com* 2011). Strip cultivation is widely practiced in this area where strips of land are cultivated leaving areas of relatively natural vegetation cover between the strips which these can expand into the cultivated areas once abandoned.

Figure 5.14 shows a previously cultivated area reclaimed by Hopefield Sand Fynbos. Alien plants are clearly visible and the indigenous vegetation is dominated by *Willdenowia incurvata* while few other indigenous species. *Willdenowia incurvata* is naturally a dominant species in pristine Hopefield Sand Fynbos and is known to reclaim transformed land quickly due to the longevity and resilience of its seeds (Helme *pers com* 2011). This is easily misleading as reclaimed areas may appear to be in good condition but lack the diversity of pristine Hopefield Sand Fynbos.

It appears that re-establishment of Hopefield Sand Fynbos has occurred in the north-western portion of the Berg River catchment although the land-cover maps may exaggerate the trend. In all likelihood the diversity of the re-established areas has been considerably reduced and many invasive alien species have taken root in formerly cultivated areas. Some fire scars and areas that had been brush cut were incorrectly classified as cultivation, and have since been over grown by various plant species.

Figure 5.13: Gains in natural and semi-natural vegetation between between Hopefield, Velddrif, Sauer and Aurora

Figure 5.14: An area of reclaimed Hopefield Sand Fynbos

5.3.3 Plantation clearing in the upper catchment

Figure 5.16 shows the conversion of plantations to natural vegetation, semi-natural vegetation and other land-cover types in the southern portion of the Berg River catchment. The conversion of plantations to natural and semi-natural vegetation occurred mostly around the Wemmershoek dam and in an area to the east of Franschhoek. Much of the plantation clearly in this area was related to the construction of the Berg River dam. The establishment of forest plantations, especially *pinus* plantations, brings about severe changes to an area and the capacity of indigenous vegetation to reestablish following their removal has not been well established.

Holmes *et al.* (2000) assessed the recovery of fynbos vegetation in a denuded portion of the Wemmershoek valley 13 years after the removal of forest plantations. They found that species richness was lower in re-established fynbos than in pristine areas. The ability of species to re-establish varied, with certain species re-establishing effectively due to the resilience of their seeds or method of propagation. Variation also hinged on the length of time for which a given area was subject to forestry with old plantations being less prone to spontaneous recovery owing to seed attrition (Holmes *et al.* 2000). The ways in which the plantations were cleared also had marked bearing on the capacity of natural vegetation to re-establish. Plantations in which trees were felled and then burnt exhibited the lowest capacity for re-establishment as a result of the destruction of seed banks by heat penetrating the soil (Holmes *et al.* 2000).

Figure 5.15 shows an area's vegetation following the removal of commercial forestry. It remains heavily infested with alien vegetation and species diversity appears to be poor. While many of the identified denuded areas exhibit superficially-natural vegetation cover, most are unlikely to display an appropriate level of compositional diversity when compared to pristine areas. In the absence of extensive restoration efforts, areas subject to commercial forestry or other forms of cultivation, especially where ploughing or fertilizers are used, are unlikely to spontaneously recover an adequate degree of species diversity (Holmes *pers com* 2011). Giliomee (2006) cites that after more than a century, abandoned vineyards above Coetzenburg in Stellenbosch have not regained a natural vegetation cover.

Figure 5.15: Breede Shale Renosterveld recovering following the removal of commercial forestry near Tulbagh

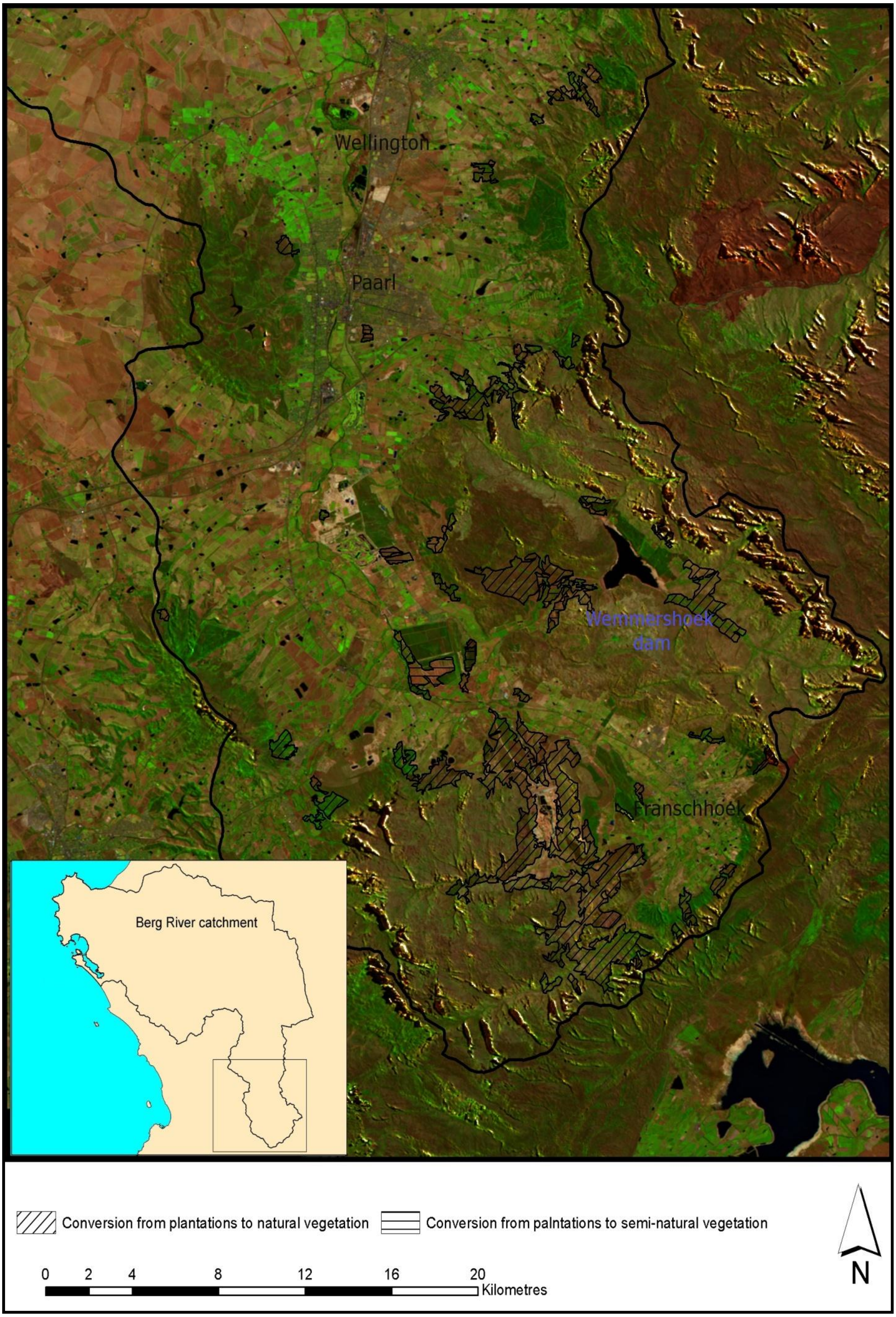


Figure 5.16: Land-cover changes in the upper Berg River catchment

5.4 IMPLICATIONS FOR BIODIVERSITY CONSERVATION

During the 20 year period covered by this research, large areas of cultivation and plantations have given way to natural and semi-natural vegetation. The ability of natural vegetation to reclaim previously cultivated areas is dependent on a host of factors. These include the length of time the area was cultivated, the inputs that were used, especially nitrogen-based fertilizers, and the availability of an adjacent seed source (Holmes *pers com* 2011). Ploughing has a particularly adverse effect on species such as slow-growing bulbous plants (Holmes *pers com* 2011). In other areas that have been examined various vegetation types in the CFR have been found to re-establish but their typical species diversity is markedly lower than in a pristine area they and are dominated by a few resilient species with most of the rare species disappearing completely (Helme *pers com* 2011).

Due to the plethora of views on biodiversity as a concept and its significance, conducting research or applying managerial principles based on this concept are contentious undertakings. Despite the registered expansion of natural vegetation in the study area these land-cover changes do not necessarily imply that biodiversity is stable or increasing. Intricate relationships between species and ecological communities and influences such as changing water availability and pollinator relationships may also have had notable bearing on biodiversity in the catchment but they were not considered in the adopted approach (Haines-Young 2009).

An area of contention surrounding the use of surrogate measures of biodiversity is the conservation value which is assigned to different indicators. This debate is essentially about whether species or the spatial extents of differing habitats are given equal weight or whether certain species or areas should be given priority status (Duelli & Obrist 2003). This means that while one area may display markedly lower levels of species diversity than another, the former may contain a higher number of endemic or biogeographically important species. Rare species or ecosystems are often considered to have a greater conservation value at a national or regional scale than a common species (Haines-Young 2009). Basing policy decisions on measures of total biodiversity is not possible and using narrow approaches that focus on a particular aspect of biodiversity has been heavily criticized.

The approach adopted by this study has been to correlate conservation value with the remaining extent of different vegetation types. While this approach is simple it can be used effectively to reduce the risk of extinction faced by species confined to particular vegetation types. On the grounds of this assumption, this study recommends prioritizing conservation efforts first in vegetation types exhibiting the greatest decline in total extent and, second where they are most feasible.

The underlying assumption concerning the use of vegetation types as a surrogate for biodiversity is that it represents a hypothetical pre-disturbance model of biodiversity in a given area. As such, its appeal in conservation planning is obvious. However, in the absence of large-scale changes in vegetation cover the results of this type of analysis are ambiguous. For this approach to be effective the degree to which areas of natural vegetation, identified on land-cover maps, is comparable to the vegetation type to which they are assigned. This approach could be accurately applied in areas exhibiting a highly homogenous vegetation cover where changes in composition can be easily established. In areas where biodiversity is threatened by alien invasion, compositional and structural changes, a potential-vegetation approach should be complemented by a more detailed survey of high-priority areas such as remaining renosterveld fragments.

This section examines the current extent of the reserve system in the Berg River catchment and provides recommendation of the optimal future management of land resources in catchment and more generally in the CFR.

5.4.1 The extent of the current reserve system

Of the 31 vegetation types located in the catchment 23 (75%) are represented in the formal reserve system. The locations of formal and informal reserves are shown in Figure 5.17. Here formal reserves refers to mountain catchment reserves, provincial reserves, local authority reserves and national parks while informal reserves are private nature reserves, conservancies and national heritage sites. The extent of the different vegetation types in the formal reserve system of the catchment is detailed in Table 5.5 while their extent within private nature reserves, conservancies and national heritage sites is listed in Table 5.6 (SANBI 2011). Tables 5.5 and 5.6 show the area in km² of each vegetation type located in different types of reserves. The total protected area of these vegetation types is also shown and expressed as a percentage of their remaining area (as of 2007) within the catchment and as a percentage of their potential extent within the catchment. Most formal reserves in the study area are mountain catchment reserves and consequently they protect vegetation types with a large portion of their potential extent remaining in an untransformed state thanks to the difficulties associated with cultivation or other activities in these mountainous areas. Critically the most endangered vegetation types, particularly renosterveld, are scarcely represented in the reserve system.

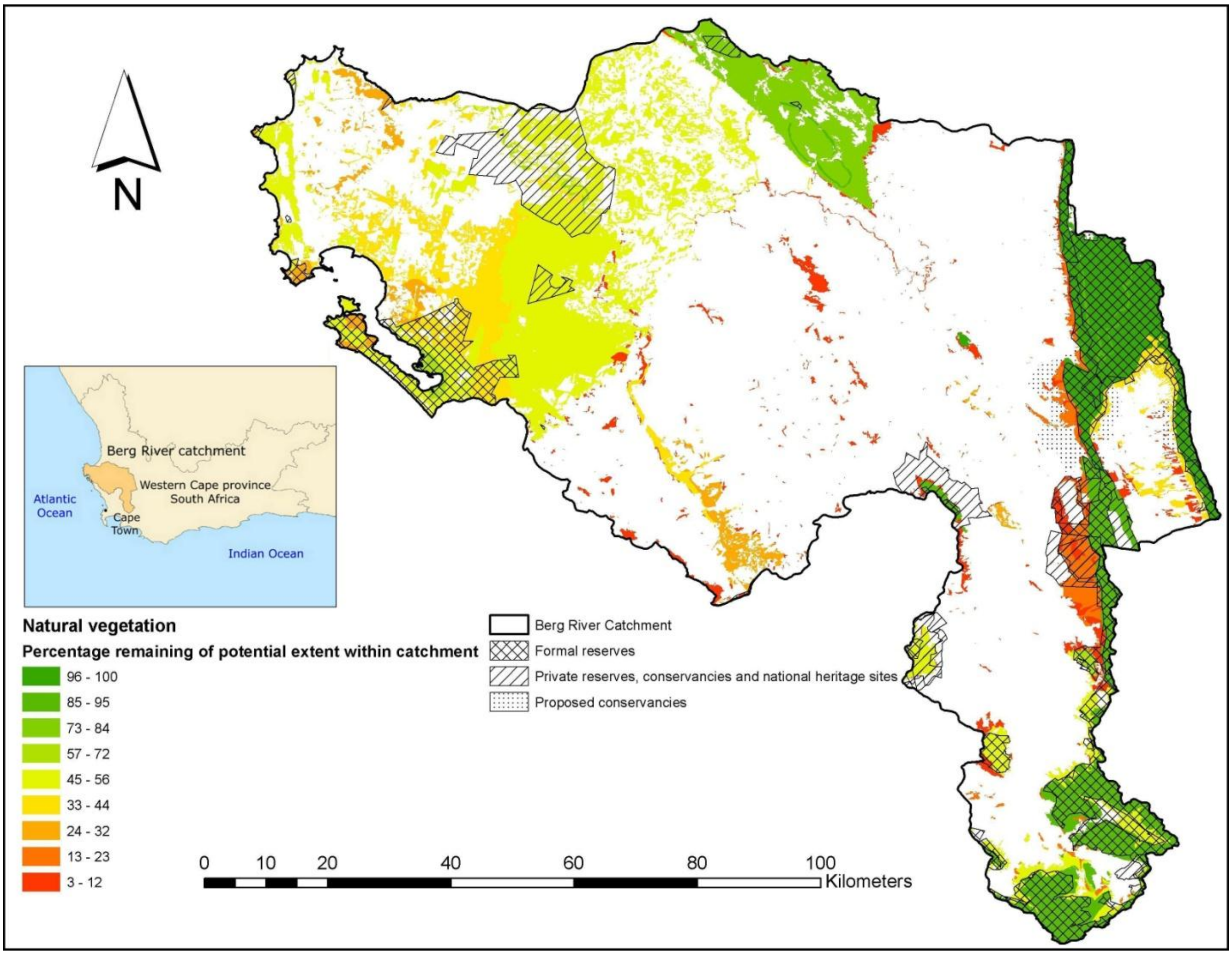


Figure 5.17: Current extent of the reserve system in the Berg River catchment

The endemic vegetation type Saldanha Limestone Sandveld is not represented in any formal reserve system. Only 0.1 km² of this vegetation type is afforded protection within a private nature reserve, in spite of it being regarded as endangered with over 40% of its potential extent lost to anthropogenic transformation. This vegetation type harbours 10 endemic and four biogeographically important taxa (Mucina & Rutherford 2006). Despite moderate gains exhibited by this vegetation it displays a patchy distribution within an agricultural matrix offering little prospect for incorporation into the reserve network.

The near endemic vegetation types, Saldanha Granite Strandveld, and Saldanha Flats Strandveld both lack noteworthy representation in the formal or informal reserve system and together contain 17 endemic and 13 biogeographically important taxa (Mucina & Rutherford 2006). Encouragingly the area occupied by both vegetation types increased during the study period. Saldanha Granite Strandveld experienced a 16% increase in extent while Saldanha Flats

Strandveld saw gains of nearly 24%. These developments, coupled with the possible establishment of the West Coast Biosphere Reserve, bode well for these vegetation types' future.

As much as 82% of Hopefield Sand Fynbos is found within the Berg River catchment. This vegetation type is considered endangered with over 40% of its extent lost to land-cover transformation and harbours five endemic taxa. Only 0.5% of the vegetation type's potential extent is protected within the catchment and 0.4% outside of the catchment. While 125.4 km² of Hopefield Sand Fynbos is located within informal reserves the state of these areas has not been established and, from personal observation, large tracks are known to be infested with invasive alien plants. Despite the substantial gains evidenced in this vegetation type over the course of this study of nearly 26%, further study is needed to establish whether reclaimed areas are comparable to pristine Hopefield Sand Fynbos.

Swartland Alluvium Fynbos is 88% confined to the Berg river catchment and is regarded as critically endangered with just under 75% of its original extent cleared, largely for cultivation. Only around 2% of this vegetation type's total potential extent is protected. However, 33.1 km² of Swartland Alluvium Fynbos is found within, private nature reserves, conservancies and national heritage sites and a further 17.4 km² will be incorporated in proposed conservancies. This vegetation type is home to 12 endemic plant species. The extent of Swartland Alluvium Fynbos was found to have decreased and then increased in this study. However, the fluctuation in area was minor and remaining patches of Swartland Alluvium Fynbos tend to be found in well-established agricultural areas leading to the conclusion that changes in the extent of this vegetation type are not significant for biodiversity conservation.

Swartland Alluvium Renosterveld as displays a large portion (83%) of its potential extent within the Berg River catchment. In total around 60% of this vegetation type's area remains intact. However, over 60% of its extent in the Berg River catchment has been transformed. This vegetation type is not protected at all. While minor gains were recorded by Swartland Alluvium Fynbos it is unlikely that these will do much to alter the precarious position of this vegetation type. Conservation efforts should be directed toward identifying potential areas in which to conserve this vegetation type.

Table 5.5: Vegetation types in the formal reserve system

Vegetation type	Mountain catchment reserves (km²)	Provincial reserves (km²)	Local authority reserves (km²)	National parks (km²)	Total formal reserves (km²)	Percentage of remaining area in catchment (%)	Percentage of potential area within catchment (%)
Atlantis Sand Fynbos	0.0	0.6	0.0	0.0	0.6	1.0	0.3
Boland Granite Fynbos	30.8	20.1	21.2	0.0	72.1	55.1	28.4
Breede Shale Fynbos	14.9	9.6	0.0	0.0	24.5	48.7	21.1
Breede Shale Renosterveld	0.3	0.03	0.0	0.0	0.3	1.9	0.2
Cape Estuarine Salt Marshes	0.0	0.0	0.0	15.0	15.0	52.6	37.8
Cape Seashore Vegetation	0.0	0.0	0.0	0.4	0.4	61.1	11.6
Cape Winelands Shale Fynbos	6.9	1.5	0.6	0.0	9.0	77.4	52.6
Hawequas Sandstone Fynbos	22.1	158.5	1.6	0.0	182.1	81.8	74.5
Hopefield Sand Fynbos	0.0	0.0	0.0	7.8	7.8	1.0	0.5
Kogelberg Sandstone Fynbos	3.0	79.3	1.1	0.0	83.3	83.5	79.6
Langebaan Dune Strandveld	0.0	1.4	1.1	69.8	72.3	53.5	27.0
Northern Inland Shale Band Vegetation	1.8	11.7	0.0	0.0	13.5	59.9	56.0
Olifants Sandstone Fynbos	30.3	3.1	0.0	0.0	33.4	86.1	83.9
Saldanha Flats Strandveld	0.0	0.0	0.0	56.3	56.3	19.3	8.1
Saldanha Granite Strandveld	0.0	6.5	0.8	11.7	19.0	25.3	8.2
Southern Afrotemperate Forest	0.0	0.3	0.0	0.0	0.3	100.0	100.0
Swartland Alluvium Fynbos	5.4	7.7	0.02	0.0	13.1	13.4	3.2
Swartland Granite Renosterveld	0.0	0.5	1.8	0.0	2.3	8.9	0.6
Swartland Shale Renosterveld	9.8	6.9	0.0	0.0	16.6	11.9	0.5
Swartland Silcrete Renosterveld	0.0	0.1	0.0	0.0	0.1	4.2	0.1
Western Altimontane Sandstone Fynbos	0.0	2.0	0.0	0.0	2.0	99.0	98.9
Western Coastal Shale Band Vegetation	0.6	8.1	0.0	0.0	8.7	73.2	68.5
Winterhoek Sandstone Fynbos	70.4	219.6	0.2	0.0	290.2	97.9	96.5

Table 5.6: Vegetation types in private reserves, conservancies and national heritage sites

Vegetation type	Private nature reserves (km²)	Conservancies (km²)	National heritage sites (km²)	Total informal reserves (km²)	Percentage of remaining area in catchment (%)	Percentage of potential area within catchment (%)	Proposed conservancies (km²)
Atlantis Sand Fynbos	0.0	0.1	0.0	0.07	0.1	0.04	0.0
Boland Granite Fynbos	0.0	15.9	2.3	18.2	13.9	7.2	0.0
Breede Shale Fynbos	0.4	0.0	0.0	0.4	0.8	0.3	6.4
Breede Shale Renosterveld	0.04	0.01	0.0	0.1	0.3	0.03	1.9
Cape Estuarine Salt Marshes	0.0	10.8	0.0	10.8	37.7	27.1	0.0
Cape Seashore Vegetation	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cape Winelands Shale Fynbos	0.0	0.6	0.0	0.6	5.4	3.7	0.0
Hawequas Sandstone Fynbos	2.2	23.4	0.0	25.6	11.5	10.5	0.4
Hopefield Sand Fynbos	17.1	108.3	0.0	125.4	16.5	8.6	0.0
Kogelberg Sandstone Fynbos	0.0	0.1	0.0	0.1	0.1	0.1	0.0
Langebaan Dune Strandveld	1.6	0.0	0.0	1.6	1.2	0.6	0.0
Northern Inland Shale Band Vegetation	0.05	0.0	0.0	0.1	0.2	0.2	0.0
Piketberg Sandstone Fynbos	0.7	0.0	10.4	11.1	4.3	3.6	0.0
Saldanha Flats Strandveld	0.0	6.4	0.0	6.4	2.2	0.9	0.0
Saldanha Granite Strandveld	1.1	0.0	0.0	1.1	1.5	0.5	0.0
Saldanha Limestone Strandveld	0.1	0.0	0.0	0.1	0.3	0.1	0.0
Swartland Alluvium Fynbos	20.8	12.1	0.2	33.1	34.0	8.0	17.4
Swartland Granite Renosterveld	0.0	0.3	0.01	0.31	1.2	0.1	0.0
Swartland Shale Renosterveld	5.0	10.2	0.0	15.2	10.8	0.4	4.0
Western Coastal Shale Band Vegetation	0.0	1.9	0.0	1.9	16.0	15.0	0.0
Winterhoek Sandstone Fynbos	0.0	0.01	0.0	0.01	0.0	0.0	3.7

Swartland Granite Renosterveld is a critically endangered vegetation type with just over 21% of its potential coverage intact. More than 41% of the potential area of this vegetation type is found in the Berg River catchment, while only 0.5% is protected. Remnants of this vegetation type displays a sporadic distribution surrounded by cultivation and it is unlikely that these could effectively be linked together to form a reserve. Remarkably, 27 endemic taxa are found in this vegetation type necessitating its preservation for the maintenance of biodiversity in the Berg River catchment and the CFR (Mucina & Rutherford 2006). Furthermore, this study revealed a 14% reduction in Swartland Granite Renosterveld obliging swift action to identify potential conservation areas and initiatives.

Swartland Shale Renosterveld is the most threatened vegetation type in the Berg River catchment with under 10% of its original extent intact. Nearly 70% of its potential extent is found in the Berg river catchment and of that only 4% (or only 4.1 km^2) was found to be left by 2007. Given the small total area occupied by this vegetation types it has experienced little change over the 20 year period of study with a small decrease between 1986/1987 and 1999/2000 and then a small increase between 1999/2000 and 2007. A total of only 0.1% of this vegetation type's potential extent is protected. Much of what remains of the vegetation type in the study area is found in isolated fragments scattered throughout the upper and middle catchment. At least 34 endemic plant species are found solely within this vegetation type strongly motivating additional efforts directed toward the preservation of remaining fragments as well as means of linking remaining fragments.

Swartland Silcrete Renosterveld is critically endangered with just over 10% of its potential extent remaining, and only 0.3% protected. Nearly 70% of the potential extent of this vegetation types is found in the Berg River catchment. Only 2% of Swartland Silcrete Renosterveld's potential extent in the catchment remains and just 4% of that which remains is under formal protection. No discernible trend was witnessed between 1986/1987 and 2007. However, it must be noted that with between 1 and 2 km^2 of Swartland Silcrete Renosterveld remaining the conservation and study of that which remains be of the utmost urgency.

Despite not suffering large scale anthropogenic disturbance Piketberg Sandstone Fynbos is not formally protected and around two thirds is found in the Berg river catchment. 39 endemic plant species are found in this vegetation type and large undisturbed areas are found on the Picketberg providing incentive to conserve this vegetation type before significant transformations are witnessed.

5.4.2 Recommendations for the optimal management of biodiversity in the Berg River catchment

In Figure 5.17 the configuration of the current reserve system in the catchment is overlayed onto the remaining areas of natural vegetation as registered by the 2007 land-cover map. The shapefile used to compile this map was obtained from SANBI but appears to be out of date as several private reserves are not located on this map. These are the Buffelsfontein Game Reserve to the northwest of Darling, Waterval Private Nature Reserve near Portersville, Welbedacht Nature Reserve north of Tulbagh and the Riebeekriviers Private Nature Reserve located between Riebeek West and Riebeek Castle. As much of the most endangered vegetation occurs on private land in small fragments there appears little potential to expand the current reserve system to cater for these vegetation types.

Given the limited capacity for the considerable expansion of the reserve system in the catchment, it is recommended that an effort be made to promote and preserve biodiversity within transformed areas and to curtail the transformation of remaining indigenous vegetation to other land-cover types. A more detailed study is required to identify remaining patches of renosterveld types and suggest ways of restoring connectivity between these fragments. Kemper, Cowling & Richardson (1999) found that species diversity in even very small renosterveld fragments located in agricultural areas to be high. This would strengthen the case for the preservation of these areas and their linkages to create some kind of a renosterveld reserve. This would largely consist of collaborating with local farmers and landowners to establish paths of connectivity between these remaining patches. This could be done via gullies and areas between cultivated fields by clearing alien vegetation, and by promoting the re-establishment of indigenous vegetation through reseeding of these areas.

O’Connor & Kuyler (2009) suggest that an assessment of the impact of different land-use types on ecosystem integrity should be made and ways in which they could be made less harmful should be explored. Ideally ploughing, frequent burning and overgrazing should be limited wherever possible. Biological pest control should be applied and fertilizers and pesticides used sparingly, particularly near wetlands, watercourses and patches of indigenous vegetation. Appropriate fire regimes are pivotal for the optimal conservation of most vegetation types in the fynbos biome and should be used appropriately. Pollinators are vital and often overlooked in conservation efforts. Focus should shift toward management of semi-natural areas and means need to be developed for extracting resources for areas in a way that is not needlessly detrimental to indigenous biodiversity. This will be vital in this region as agriculture is unlikely to expand significantly in the immediate future but the population and its resource requirements most definitely will. In a more practical sense, the most effective way to promote biodiversity is to establish conservation as a viable and profitable alterative economic sector.

5.5 CONCLUSION

This study recorded noteworthy gains in most vegetation types in the Berg River catchment between 1986/1987 and 2007. While this is a positive finding, concern has been raised over the degree to which newly identified natural vegetation can be taken to represent the diversity in species composition and structure for which many of these vegetation types are so well known. Renosterveld and sand fynbos vegetation types are poorly represented in the reserve system in the catchment; many remnants are degraded and infested with invasive alien plant species. Little capacity exists to extend statutory protection to renosterveld and Sand Fynbos vegetation types. The most pressing biodiversity management objectives in the Berg River catchment are likely to be the control and removal of alien plants and the restoration of degraded natural vegetation.

CHAPTER 6: DISCUSSION AND CONCLUSION

The aim of this study was to assess the spatial extent and dominant trends in changes in land cover with the intention to assess their impact on biodiversity in the Berg River catchment by using vegetation types as a surrogate for biodiversity. This chapter begins by providing an overview and critical evaluation of the study focussing on salient issues encountered. In light of these the discussion shifts to the capacity of medium-resolution historical satellite imagery, used in conjunction with potential vegetation, to assess and monitor biodiversity in the CFR, after which recommendations for further research are provided.

6.1 REVIEW AND EVALUATION

The first objective was to justify the use of vegetation types as a biodiversity surrogate in the study area. Vegetation was selected as a biodiversity surrogate due to various concerns associated with species-distribution data and environmental-type surrogates. Mucina, Rutherford & Powrie’s (2007) vegetation map, currently the most detailed vegetation map available for this area, was selected as the baseline document. The use of vegetation types was further justified by the high rates of plant-species diversity in the CFR and the ease with which vegetation-type data can be integrated with land-cover data.

The second and third objectives were to acquire or compile adequate land-cover maps for the study area over an appropriate time period. Landsat imagery was selected as an appropriate data source from which to generate land-cover maps owing to the temporal depth of the imagery, the versatility of the data, its low cost and wide availability. A supervised nearest neighbour classification was employed in an object-orientated environment to automatically classify the imagery into 10 classes based on a modified version of the CD: NGI’s land-cover legend. The resultant land-cover maps were manually edited to remove classification errors and discriminate between natural indigenous vegetation and vegetation in altered state.

Change reflected in the land-cover maps was analysed using IDRISI’s LCM. Contrary to initial expectations natural vegetation increased, largely at the expense of cultivation. Much of this change was concentrated in the lower reaches of the catchment and in the area surrounding Langebaan. Areas of natural vegetation were matched with vegetation types and the change analysis was repeated to assess changes in the extent of vegetation types. Most vegetation types within the natural land-cover class were found to have increased in extent over the study period. An assessment of reclaimed areas in the catchment revealed mixed responses with most reclaimed areas being unable to represent the species diversity of pristine areas. This finding, coupled with uncertainties in the classification of semi-natural vegetation and an inability to accurately map alien vegetation, led to the conclusion that the results of the study be interpreted with discretion as they are unlikely to translate into a clear-cut increase in indigenous biodiversity.

An error matrix was developed to assess the accuracy of the 2007 land-cover map using a combination of field data and aerial photography. While the overall accuracy of the 2007 land-cover map was high (in some cases exceeding 85%), semi-natural vegetation was confused with natural vegetation and cultivation. This led to the conclusion that semi-natural vegetation was underrepresented and that it is likely that a significant portion of what was classified as natural vegetation has been influenced by human activity or is otherwise not representative of a pristine vegetation cover.

The next objective sought to identify the dominant drivers of land-cover change in this area. Strong growth evident in urban areas was likely the result of in-migration and natural population growth. The reduction of plantations was the result of plantation clearing by the South African National Civics Organization (SANCO). A reduction in the extent of cultivation was deemed to be a result of decreased governmental support of agriculture over the duration of the 20-year study period which led to areas of marginal agricultural potential being abandoned.

An overarching aim of the study was to make recommendations about the optimal management of the catchment concerning the maintenance and promotion of indigenous biodiversity. The reserve system in the catchment is disposed to protecting the least threatened vegetation types confined to mountainous areas. Unfortunately, little capacity exists to extend the system to lowland vegetation types. An alternative is to engage with local landowners and attempt to identify remaining pockets of natural indigenous vegetation and link them, remove invasive alien vegetation and ideally reseed degraded patches.

The study attempts to measure the impacts of anthropogenic activity on the biodiversity of a particular area. In many instances the transformation or disruption of natural areas is assumed to translate into a direct loss of indigenous biodiversity. As demonstrated by this study, some ambiguity exists where it is not possible to translate changes in land cover directly into losses or gains in indigenous biodiversity.

6.2 THE POTENTIAL OF REMOTE SENSING AND VEGETATION-TYPE DATA TO MONITOR BIODIVERSITY IN THE CFR

According to Biggs, Reyers & Scholes (2006) it is widely acknowledged that land cover generated from satellite imagery is inclined to underrepresent land degradation. Unfortunately, the confusion between natural and alien or otherwise degraded vegetation has implications for this project which has sought to accurately map natural indigenous vegetation to function as a surrogate for overall biodiversity in the region. The CFR is noted for its extraordinary diversity of vascular plants. Identifying and maintaining this diversity is an undertaking of global importance and requires accurate information on the distribution and state of vegetation communities. From the results of this study it appears that the capacity of potential vegetation and remotely-derived land-cover maps offer capacity to assess and monitor biodiversity in the region.

6.2.1 Vegetation, biodiversity and remote sensing

It is unlikely that differences in species composition and structure in otherwise spectrally similar vegetation types can be registered by satellite imagery, particularly at a medium or coarse resolution such as used in this study. This is a major obstacle for this type of approach where the potentially significant changes in indigenous biodiversity cannot be adequately measured. This concern is compounded by the myriad ecological changes that can occur in response to increases or decreases in the spatial extent of a vegetation type. Critically, the threshold at which the species composition of a vegetation type will change in response to fragmentation or reductions in overall extent has not been established. It is uncertain at what point ecological processes will be affected by compositional and structural changes. The extent or change in occupancy of different vegetation types should be seen as a first step in a holistic assessment of biodiversity.

A useful aspect of the study is its ability to identify areas where biodiversity changes are likely to be taking place. In this way an approach that assesses changes in potential vegetation with land-cover maps can direct more focused research in a way that is more efficient than direct methods. To adequately assess the impact of land-cover changes on biodiversity in the Berg River catchment it is necessary to conduct further field research focused on establishing the health of patches of natural vegetation experiencing change. In this way land cover derived from satellite imagery can be used as the first stage in a comprehensive biodiversity assessment, using the breadth and expedience of remote sensing to hone more detailed and time consuming field surveys. The methodology employed by this study offers a scoping mechanism where areas experiencing significant changes can be examined at a fine-scale. This approach would serve to prioritise and concentrate scarce research and conservation resources on a scale not otherwise attainable.

6.2.2 The way forward

Nicholson, Keith & Wilcove (2009) propose the development of a series of subcategories of identified natural vegetation based on the degree of alteration witnessed and the likely effects of anthropogenic activity. This is done in order to provide an assessment of impacts of changes in ecological functioning associated with land-cover change. Such as system categorizes natural vegetation based on composition and structural considerations measured against a potential or ideal level. Assessment would be based on the assumed structural integrity of different vegetation types based on proximity to transformed land and visible signs of alteration. In this approach the degree to which they represent an indigenous vegetation cover and their capacity for restoration could be assessed.

Using higher-resolution satellite imagery may address some of shortcomings associated with the project, such as the inability to identify alien plants and to differentiate between pristine and degraded indigenous vegetation. Rouget *et al*. (2003) concluded that automated spectral classification or visual classification of vegetation in the CFR is unlikely to effectively illuminate the composition of vegetation in this area owing to its heterogeneity, aesthetic similarities between communities and the superficial similarity with degraded or invaded areas. Monitoring vegetation with remotely sensed imagery has found useful applications in many parts of the world where the difference between indigenous and altered vegetation cover is clear. In such areas, pronouncements on the impact of land-cover change on biodiversity can be made easily and with confidence. In the CFR a means of extracting a measure of the compositional diversity of different vegetation communities from remotely sensed data would greatly enhance the capacity of remotely-sensed data to monitor biodiversity.

6.3 SUGGESTIONS FOR FURTHER RESEARCH

The analysis of land-cover change in the Berg River catchment has revealed notable changes occurring in various portions of the catchment. However, many questions are posed by the result presented in the preceding chapter. This section suggests avenues for further study.

As noted in preceding sections, significant areas of re-established Hopefield Sand Fynbos were registered in the change analysis. A detailed assessment of this vegetation type that measures species diversity between different areas of natural vegetation would contribute significantly toward an assessment of biodiversity in this area. Pivotally, the status of reclaimed areas of Hopefield Sand Fynbos in terms of species composition and diversity should be established as this would provide a good indication of the prospects for species associated with this vegetation type. Such research could contribute significantly toward the sustainable management of this area.

Significant reductions in cultivation were evident over the study period. While the degree to which this is indicative of genuine large-scale changes in agricultural practices is questionable, it is clear that undocumented changes are taking place. Various explanations for the changes have been offered ranging from a decreasing watertable to the intensification of centre-point irrigated potato cultivation (Helme *pers com* 2011). However, no satisfactory explanations have yet been found. As shifts in agricultural practice have far-reaching consequences for biodiversity as well as economic activity in this area, assessment of agricultural changes should be undertaken to determine whether the trends described in this study are authentic, what is driving them and their potential implications for the sustainable management of the Berg River catchment.

Large-scale removal of commercial forestry in and around the headwaters of the Berg River was revealed by this study. Some work has been done on the capacity of indigenous vegetation to re-establish following the removal of commercial forestry, but little has focused on the current state of the areas which have been cleared. This study advocates a detailed assessment of the compositional diversity of re-established fynbos and renosterveld vegetation types in denuded plantations. As the WfW programme has prioritized the removal of alien trees in riparian areas of the Berg River catchment, the status of these areas following the removal of alien vegetation should be assessed as well.

Invasive alien vegetation poses one of the single greatest threats to indigenous biodiversity in the CFR. It is hence a pivotal and pressing concern that the invaded areas be identified and contained. Direct measures of mapping alien vegetation are time-consuming and, given the scale of the problem, are unlikely to offer an effective solution. Rouget *et al*. (2004a) have modeled the potential range of invasive alien plant species in accordance with environmental characteristics to streamline further research and clearing efforts. However, this approach fails to give an indication of actual invasions and does not adequately address the problem. Data concerning the distribution of alien vegetation in South Africa are at a very coarse scale and impractical for catchment-scale studies. Identifying alien vegetation through remote-sensing techniques and technologies is most likely the only means by which alien vegetation can be identified over large areas. The capacity to monitor invasions with this approach provides added incentive to develop means of identifying alien vegetation through a remote sensing approach. This study was unable to establish reliable spectral approaches to identify alien vegetation. Moreover, visual identification is time-consuming and ineffective with many of the species because of their aesthetic similarity to indigenous vegetation when considered at a coarse resolution. A means of automatically delineating areas of alien vegetation would be of tremendous import to future biodiversity assessment in this area.

Cape inland salt pans and Cape vernal pools represent important ecosystems, particularly for amphibians and certain bird species. The study showed these features to be substantially underrepresented on Mucina, Rutherford & Powrie’s (2007) vegetation map. It is recommended that an effort be made to accurately map these features. Further, as these features are often surrounded by agricultural land it is urged that the impacts of agricultural practices on these areas be assessed.

While not described in detail in this report, rapid urban development is taking place along the coastal zone to the east of Velddrif. This coastal zone accommodates diverse and acutely sensitive ecosystems and owing to their aesthetic appeal the zone is often subjected to intensive development (Palmer 2008). Little has been done to assess the impacts that development in this area have and will have on coastal ecosystems and the diverse assemblage of species that they support. It is consequently recommended that detailed research be done on the optimal management of this coastal area.

6.4 CONCLUSION

Biodiversity is a diverse concept that covers a range of entities over multiple spatial and temporal scales. Because of this it can be measured in many ways, each subject to its own unique set of advantages, shortcomings and assumptions. The relationship between biodiversity and land cover is complex and no doubt varies considerably depending on the context and the area in which it is assessed. This research has clearly shown the difficulty of positing simple cause-and-effect relationships between biodiversity and land-cover change. However, land-cover change provides an ideal platform from which to initiate more intensive analyses of biodiversity changes.

The most insightful conclusion that can be drawn from this study is that there is a need to develop more accurate and comprehensive monitoring systems for biodiversity in South Africa and especially in the CFR. Large gaps exist in our capacity to translate changes in land cover into changes in biodiversity at all levels of biological organization and to devise conservation and management plans in the light of these impacts. If the myriad challenges that face human societies at the dawn of the 21st century, such as increased pressure on a finite natural resource base associated with population growth and socio-economic development, anthropogenic climate change and rapidly escalating biodiversity loss, are to be adequately mitigated, a robust understanding of the relationship between land cover and biodiversity is not only essential but will likely prove crucial to our future well-being.

REFERENCES

Baatz M and Schäpe A 1999. Multiresolution segmentation – an optimization approach for high quality multi-scale image segmentation. In Strobl J, Blaschke T and Griesebner G (eds) 2000, *Angewandte Geographische Informations-Verarbeitung XII*, 12–23. Karlsruhe: Wichmann Verlag.

Baatz M, Hofmann P & Willhauck G 2008. Progressing from object-based to object-oriented image analysis. In Blaschke T, Lang S & Hay GJ (eds) *Object-based image analysis: Spatial concepts for knowledge-driven remote sensing applications*, 29-42. Berlin: Springer.

Bailey RG 1996. *Ecosystem geography*. Berlin: Springer.

Biggs R & Scholes RJ 2002. Land-cover changes in South Africa 1911-1993. *South African Journal of Science* 98: 420-424.

Biggs R, Reyers B & Scholes RJ 2006. A biodiversity intactness score for South Africa. *South African Journal of Science* 102: 277-283.

Biggs H Simons H, Bakkenes M, Scholes RJ, Eickhout B, VanVuuren D & Alkemade R 2008. Scenarios of biodiversity loss in southern Africa in the 21st century. *Global Environmental Change* 18: 296-309.

Blaschke T 2010. Object based image analysis for remote sensing. *Journal of Photogrammetry and Remote Sensing* 65: 2-16.

Borras SM 2003. Questioning market-led agrarian reforms: Experiences from Brazil, Columbia and South Africa. *Journal of Agrarian Change* 3: 367-394.

Botes A, McGeoch MA, Robertson HG, van Niekerk A, Davids HP & Chown SL 2006. Ants, altitude and change in the northern Cape Floristic Region. *Journal of Biogeography* 33: 71-90.

Bunnell FL & Huggard DJ 1999. Biodiversity across spatial and temporal scales: Problems and opportunities. *Forest Ecology and Management* 115: 113-126.

CapeNature 2007. Western Cape Province State of Biodiversity 2007. [Online]. Available from: http://www.capenature.co.za/docs/1016/Biodiversity%20Review.pdf [Accessed: 13 April 2010]

Chen X 2002. Using remote sensing and GIS to analyse land cover change and its impacts on regional sustainable development. *International Journal of Remote Sensing* 23: 107-124.

Clark B & Ratcliffe G (eds) 2007. *Berg River baseline monitoring programme. Final report volume 5: Synthesis*. Pretoria: Department of Water Affairs and Forestry.

Clark Labs 2009. IDRISI focus paper: The Land Change Modeler for Ecological Sustainability. [Online]. Available from: http://www.clarklabs.org/applications/upload/Land-Change-Modeler-IDRISI-Focus-Paper.pdf. [Accessed 30 January 2012].

Cloete TE, Nel LH & Theron J 2006. Biotechnology in South Africa. *Trends in Biotechnology* 24: 557-562.

Congalton RG & Green K 2009. *Assessing the accuracy of remotely-sensed data: Principles and applications*. 2nd ed. Boca Raton: CRC Press.

Cowling RM & Heijnis CE 2001. The identification of broad habitat units as biodiversity entities for systematic conservation planning in the Cape Floristic Region. *South African Journal of Botany* 67: 15-38.

Cowling RM, Pressey RL, Rouget M & Lombard AT 2003. A conservation plan for a global biodiversity hotspot—the Cape Floristic Region, South Africa. *Biological Conservation* 112: 191-216.

Crist PJ, Kohley TW & Oakleaf J 2000. Assessing land-use impacts on biodiversity using an expert systems tool. *Landscape Ecology* 15: 47-62.

Currie B, Milton SJ & Steenkamp JC 2009. Cost–benefit analysis of alien vegetation clearing for water yield and tourism in a mountain catchment in the Western Cape of South Africa. *Ecological Economics* 68: 2574-2579.

De Long DC 1996. Defining biodiversity. *Wildlife Society Bulletin* 24: 738-749.

DEAT (Department of Environmental Affairs and Tourism) 2005. *South Africa's national biodiversity strategy and action plan*. Pretoria: Department of Environmental Affairs and Tourism.

De Villiers CC, Driver A, Clark B, Euston-Brown DIW, Day EG, Job N, Helme NA, Holmes PM, Brownlie S & Rebelo AB 2005. *Fynbos forum: Ecosystem guidelines for environmental assessment in the Western Cape*. Kirstenbosch: Fynbos Forum and Botanical Society of South Africa.

De Villiers S 2007. The deteriorating nutrient status of the Berg River, South Africa. *Water SA*: 33: 659-664.

Didham RK, Tylianakis JM, Gemmell NJ, Rand TA & Ewers RM 2007. Interactive effects of habitat modification and species invasion on native species decline. *Trends in Ecology and Evolution* 22: 489-496.

Duelli P & Obrist M 2003. Biodiversity indicators: The choice of values and measures. *Agriculture, Ecosystems and Environment* 98: 87-98.

DWAF (Department of Water Affairs and Forestry) 2000. A review of the veld fires in the Western Cape during 15 January to 25 January 2000. Report to the minister of water affairs and forestry and premier of the Western Cape by the task team: Toward improved veldfire management in South Africa. [Online]. Available from: http://www2.dwaf.gov.za/dwaf/cmsdocs/Elsa/Docs/Fire/Review-veld%20fires%20WCape%20Jan%202000.pdf. [Accessed 2 February 2012].

Fahrig L 2001. How much habitat is enough? *Biological Conservation* 100: 65-74.

Fahrig L 2003. The effects of habitat fragmentation on biodiversity. *Annual Review of Ecology, Evolution and Systematics* 34: 487-515.

Faith DP 1996. Environmental diversity as the best-possible use of surrogate data for assessing the relative biodiversity of sets of areas. *Biodiversity and Conservation* 5: 399-415.

Faith DP 2003. Environmental diversity (ED) as surrogate information for species-level biodiversity. *Ecography* 23: 374-379.

Faith DP, Ferrier S & Walker PA 2004. The ED strategy: how species-level surrogates indicate general biodiversity patterns through an ‘environmental diversity’ perspective. *Journal of biogeography* 31: 1207-1217.

Falcucci A, Maiorano L & Boitani L 2007. Changes in land-use/land-cover patterns in Italy and their implications for biodiversity conservation. *Landscape Ecology* 22:617–631.

Falkner E & Morgan D 2002. *Aerial mapping: Methods and applications*. London: CRC Press LLC.

Ferrier S 2002. Mapping spatial pattern in biodiversity for regional conservation planning: Where to from here? *Systematic Biology* 51: 331-363.

Ferrier S & Guisan A 2006. Spatial modelling of biodiversity at the community level. *Journal of Applied Ecology* 43: 393-404.

Fischer & Lindenmayer 2007. Landscape modification and habitat fragmentation: A synthesis. *Global Ecology and Biogeography* 16: 265-280.

Flügel WA 1995. River salination due to dryland agriculture in the Western Cape province, Republic of South Africa. *Environment International* 21: 679-686.

Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, Chapin FS, Coe MT, Daily GC, Gibbs HK, Helkowski JH, Holloway T, Howard EA, Kucharik CJ, Monfreda C, Patz JA, Prentice C, Ramankutty N & Snyder PK 2005. Global consequences of land use. *Science* 309: 570-574.

Foody GM 2002. Status of land cover classification accuracy assessment. *Remote Sensing of the Environment* 80: 185-201.

Foody GM 2008. GIS: Biodiversity applications. *Progress in Physical Geography* 32: 223-235.

Fourie CE, Van Niekerk A & Mucina L 2011. Optimising a one-class SVM for geographic object-based novelty. Paper presented to AfricaGeo.

Galley C & Linder HP 2005. Geographical affinities of the Cape flora, South Africa. *Journal of Biogeography* 33: 236-250.

Gaston KJ (ed) 1996. *Biodiversity – a biology of numbers and difference*. Cambridge: Blackwell Science Ltd.

GCOS (Global Climate Observing System) 2006. Systematic observation requirements for satellite-based products for climate-supplemental details to the satellite-based component of the implementation 17 plan for global observing system for climate support [Online]. Available from: http://www.wmo.int/pages/prog/gcos/Publications/gcos-107.pdf [Accessed 16 August 2011].

Gelderblom CM, Van Wilgen BW, Nel JL, Sandwith T, Botha M & Hauck M 2003. Turning stratergy into action: Implmenting a conservation actiopn plan in the Cape Floristic Region. *Biological Conservation* 112: 291-297.

Gerber L 2005. Biodiversity risk assessment of South Africa's municipalities. Master's thesis. Stellenbosch: Stellenbosch University.

Gibson PJ 2000. *Introductory remote sensing: Principles and concepts*. London: Routledge.

Giliomee JH 2006. Conserving and increasing biodiversity in the large-scale, intensive farming systems of the Western Cape, South Africa. *South African Journal of Science*. 102: 375-378.

Gould W 2000. Remote sensing of vegetation, plant species richness and regional biodiversity hotspots. *Ecological Applications* 10: 1861-1870.

Grantham HS, Pressey RL, Wells JA, Beattie AJ 2010. Effectiveness of biodiversity surrogates for conservation planning: Different measures of effectiveness generate a kaleidoscope of variation [Online]. Available from: http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0011430 [Accessed 9 July 2010].

Griscom HR, Miller SN, Gyedu-Ababio T & Sivanpillai R 2009. Mapping land cover change of the Luvuvhu catchment, South Africa for environmental modelling. *GeoJournal* 75:163-173.

Haines-Young R 2009. Land use and biodiversity relationships. *Land Use Policy* 26: 178-186.

Hamilton AJ 2005. Species diversity or biodiversity? *Journal of Environmental Management* 75: 89-92.

Hay GJ & Baschke T 2010. Geographic Object-Based Image Interpretation (GEOBIA). Photogrammetric Engineering and Remote Sensing 76: 121-123.

Heelemann S 2010. Renosterveld remnants: Current ecological situation and restoration perspectives. Doctoral dissertation. Regensburg: University of Regensburg.

Helmer EH, Brown S & Cohen WB 2000 Mapping montane tropical forest successional stage and land use with multi-date Landsat imagery. *International Journal of Remote Sensing* 21: 2163-2183.

Helmer EH, Ramos O, López TDM, Quiñones M & Diaz W 2002. Mapping the forest type and land cover of Puerto Rico, a component of the Caribbean biodiversity hotspot. *Caribbean Journal of Science* 38: 165-183.

Hill D, Fasham M, Tucker G, Sherwry M and Shaw P 2005. *Handbook of biodiversity methods: Survey, evaluation and monitoring*. Cambridge: Cambridge University Press.

Holmes PM & Richardson DM 1999. Protocols for restoration based on dynamics, community structure and ecosystem function: Perspectives from South African Fynbos. *Restoration Ecology* 7: 215-230.

Holmes PM, Richardson DM, Van Wilgen BW & Gelderblom C 2000. Recovery of South African fynbos vegetation following alien woody plant clearing and fire: Implications for restoration. *Austral Ecology* 25: 631-639.

Holmes PM 2008. Optimal ground preparation treatments for restoring lowland sand fynbos vegetation on old fields. *South African Journal of Botany* 74: 33-40.

Houghton RA 1994. The worldwide extent of land-use change. *Bioscience* 44: 305-313.

Johnston CA 1998. *Geographic information systems in ecology*. Ames: Blackwell Science.

Kamish W 2008. Hydrosalinity modelling of the Berg River using ACRU Salinity. Master's thesis. Stellenbosch: Stellenbosch University.

Kartikeyan B, Sarkar A & Majumder KL 1998. A segmentation approach to classification of remote sensing imagery. *International Journal of Remote Sensing* 19: 1695-1709.

Keith DA, Orscheg C, Simpson CC, Clarke PJ, Hughes L, Kennelly SJ, Major RE, Soderquist TR, Wilson AL & Bedward M 2009. A new approach and case study for estimating extent and rates of habitat loss for ecological communities. *Biological Conservation* 142: 1469-1479.

Kemper J, Cowling RM & Richardson DM 1999. Fragmentation of South African renosterveld shrublands: Effects on plant community structure and conservation implications. *Biological Conservation* 90: 103-111.

Kim M, Madden M & Bo X 2010. GEOBIA vegetation mapping in Great Smoky Mountains National Park with spectral and non-spectral ancillary information. *Photogrammetric Engineering and Remote Sensing* 76: 137-149.

Kirsten JF, Van Zyl J & Van Rooyen J 1994. South African agriculture during the 1980s. *South African Journal of Economic History* 9: 19-48.

Kontula T & Raunio A 2009. New method and criteria for national assessments of threatened habitat types. *Biodiversity Conservation* 18: 3861-3876.

Lambin EF & Ehrlich D 1997. Land-cover changes in sub-Saharan Africa (1982–1991): Application of a change index based on remotely sensed surface temperature and vegetation indices at a continental scale. *Remote Sensing of Environment* 61: 181-200.

Lawler JJ & White D 2008. Assessing the mechanism behind successful surrogates for biodiversity in conservation planning. *Animal Conservation* 11: 270-280.

Lepers E, Lambin EF, Janetos AC, Defries R, Achard F, Ramankutty N & Scholes RJ 2005. A synthesis of information on rapid land-cover change for the period 1981–2000. *BioScience* 55: 115-124.

Lillesand TM, Kiefer RW & Chipman JW 2004. *Remote sensing and image interpretation*. 5th ed. New York: John Wiley and Sons.

Lochner P, Weaver A, Gelderblom C, Peart R, Sandwith T & Fowkes S 2003. Aligning the diverse: The development of a biodiversity conservation strategy for the Cape Floristic Region. *Biological Conservation* 112: 29-43.

Lombard AT, Cowling RM, Pressey RL & Rebelo AG 2003. Effectiveness of land classes as surrogates for species in conservation planning for the Cape Floristic Region. *Biological Conservation* 112: 45-62.

Loreau M, Naeem S & Inchausti P (eds) 2002. *Biodiversity and ecosystem functioning*. New York: Oxford University Press.

Low AB & Rebelo AG (eds) 1996. *Vegetation of South Africa, Lesotho and Swaziland*. Pretoria: Department of Environmental Affairs and Tourism.

Lück W 2006. *CSIR Satellite Application Centre: Earth Observation Service Centre landcover class definition report*. Pretoria: CSIR.

Lück W & Diemer N 2008. *Land cover class definition report*. Observation Service Centre Report, Version 1.1. Pretoria: CSIR Satellite Applications Centre: Earth.

McDonald RC, Isbell RF, Speight JG, Walker J & Hopkins MS 1984. *Australian soil and land survey: Field handbook*. Melbourne: Inkata Press Propriety Limited.

McCoy 2005. *Field Methods in Remote Sensing*. New York: Guilford Press.

Machange RW, Jenkins AR & Navarro RA 2005. Eagles as indicators of ecosystem health: Is the distribution of Martial Eagle nests in the Karoo, South Africa, influenced by variations in land-use and rangeland quality? *Journal of Arid Environments* 63: 223-243.

Mac Nally R, Bennett AF, Brown GW, Lumsden LF, Yen A, Hinkley S, Lillywhite P & Ward D 2002. How well do ecosystem-based planning units represent different components of biodiversity? *Ecological Applications* 12: 900-912.

Mannion AM 2002. *Dynamic world: Land-cover and land-use change*. London, Edward Arnold.

Margules CR, Pressey RL & Williams PH 2002. Representing biodiversity: Data and procedures for identifying priority areas for conservation. *Journal of Biosicence* 27: 309-326.

Memiaghe HR 2008. Old field restoration: Vegetation response to soil changes and restoration efforts in the Western Cape lowlands. Masters thesis. Stellenbosch: Stellenbosch University.

Meyer WB & Turner BL 1992. Human population growth and global land-use/cover change. *Annual Review of Ecology and Systematics* 23: 39-61.

Morain S 1999. *GIS solutions in natural resource management: Balancing the technical-political equation*. Santa Fe: OnWord press Press.

Morgan D & Falkner E 2002. *Aerial Mapping: Methods and Applications*. London: CRC Press.

Mountain A 2003. *The first people of the Cape*. Cape Town: David Philip Publishers.

Mucina L & Rutherford MC (eds) 2006. *The vegetation of South Africa, Lesotho and Swaziland*. Pretoria: South African National Biodiversity Institute.

Mucina L, Rutherford MC & Powrie LW (eds) 2007. *Vegetation map of South Africa, Lesotho and Swaziland*. 1:1000000 scale sheet maps. Pretoria: South African National Biodiversity Institute.

Müller R, Nowicki C, Barthlott W & Ibisch PL 2003. Biodiversity and endemism mapping as a tool for regional conservation planning – case study of the *Pleurothallidinae (Orchidaceae)* of the Andean rain forests in Bolivia. *Biodiversity and Conservation* 12: 2005-2024.

Muradian R 2001: Ecological thresholds: A survey. *Ecological Economics* 38: 7-24.

Nagendra H, Munroe DK & Southworth J 2004. From pattern to process: Landscape fragmentation and the analysis of land use/land cover change. *Agriculture, Ecosystems and Environment* 101: 111-115.

NASA (National Space and Aeronautics Agency) 2011. The Landsat Program [Online]. Available from: http://landsat.gsfc.nasa.gov. [Accessed: 20 September 2011].

Nicholson E, Keith DA & Wilcove DS 2009. Assessing the threat status of ecological communities. *Conservation Biology* 23: 259-274.

Noss RF 1990. Indicators for monitoring biodiversity: A hierarchical approach. *Conservation Biology* 4: 355-364.

O'Connor TG & Kuyler P 2009. Impact of land use on the biodiversity integrity of the moist sub-biome of the grassland biome, South Africa. *Journal of Environmental Management* 90: 384-395.

Oliver I, Holmes A, Dangerfield MJ, Gillings M, Pik AJ, Britton DR, Holley M, Montgomery ME, Raison M, Logan V, Pressey RL & Beattie AJ 2004. Land systems as surrogates for biodiversity in conservation planning. *Ecological Applications* 14: 485-503.

Palmer BJ 2008. A GIS assessment of development and land use change in the coastal zone of the Ndlambe and Ngqushwa local municipalities, Eastern Cape, South Africa. Masters thesis. Grahamstown: Rhodes University.

Panzer R & Schwartz MW 1998. Effectiveness of a vegetation-based approach to insect conservation. *Conservation Biology* 12: 693-702.

Pauleit S, Ennos R & Golding Y 2005. Modelling the environmental impacts of urban land use and land cover change - a study in Merseyside, UK. *Landscape and Urban Planning* 71: 295-310.

Parker M & Mac Nally R 2002. Habitat loss and the habitat fragmentation threshold: An experimental evaluation of impacts on richness and total abundances using grassland invertebrates. *Biological Conservation*. 105: 217-229.

Payet K 2007: The effect of spatial scale on the use of biodiversity surrogates and scoio-economic criteria in systematic conservation assessments. Masters thesis. Stellenbosch: Stellenbosch University.

Pence GQK 2008. *C.A.P.E. Fine-scale systematic conservation planning assessment: Technical report*. Produced for CapeNature as part of the GEF-funded C.A.P.E. Fine-Scale Biodiversity Planning Project. Cape Town: South Africa.

Petit CC & Lambin EF 2002. Impact of data integration technique on historical land-use/land-cover change: Comparing historical maps with remote sensing data in the Belgian Ardennes. *Landscape Ecology* 17: 117-132.

Pickles J 2004: *A History of Spaces: Cartographic Reason, Mapping and the Geo-Coded World*. London, Routledge

Poiani KA, Richter BD, Anderson MG & Richter HE 2000. Biodiversity conservation at multiple scales: Functional sites, landscapes, and networks. *Bioscience* 50: 133-146.

Pressey RL, Cowling RM & Rouget M 2003. Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation* 112: 99–127.

Pressey RL 2004. Conservation planning and biodiversity: Assembling the best data for the job. *Conservation Biology* 18: 1677-1681.

Reyers B & Van Jaarsveld AS 2000. Assessment techniques for biodiversity surrogates. *South African Journal of Science* 96: 406-408.

Reyers B, Fairbanks DHK, Van Jaarsveld AS & Thompson M 2001: Priority areas for the conservation of South African vegetation: A coarse-filter approach, *Diversity and Distributions* 7: 79-95.

Reyers 2004. Incorporating anthropogenic threats into evaluations of regional biodiversity and prioritisation of conservation areas in the Limpopo Province, South Africa. *Biological Conservation* 118: 521-531.

Reyers B, Rouget M, Jonas Z, Cowling RM, Driver A, Maze K & Desmet P 2007. Developing products for conservation decision-making: Lessons from a spatial biodiversity assessment for South Africa. *Diversity and Distributions* 13: 608-619.

RHP (River Health Programme) 2004. *State-of-rivers report: Berg River system*. Pretoria: Department of Water Affairs and Forestry.

Rodriguez ASL & Brooks TM 2007. Shortcuts for biodiversity conservation planning: The effectiveness of surrogates. *The Annual Review of Ecology, Evolution, and Systematics* 38: 713-737.

Rodriguez JP, Balch JK & Rodrı´guez-Clark KM 2007. Assessing extinction risk in the absence of species-level data: Quantitative criteria for terrestrial ecosystems. *Biodiversity Conservation* 16: 183–209.

Rouget M, Richardson DM, Cowling RM, Lloyd JW & Lombard AT 2003. Current patterns of habitat transformation and future threats to biodiversity in terrestrial ecosystems of the Cape Floristic Region, South Africa. *Biological Conservation* 112: 63-85.

Rouget M, Richardson DM, Nel JL, Le Maitre DC, Egoh B & Mgidi T 2004a. Mapping the potential ranges of major plant invaders in South Africa, Lesotho and Swaziland using climatic suitability. *Diversity and Distributions* 10: 475-484.

Rouget M, Reyers B, Jonas Z, Desmet P, Driver A, Maze K, Egoh B & Cowling RM 2004b. *South African National Spatial Biodiversity Assessment 2004: Technical report. Volume 1: terrestrial component.* Pretoria, South African National Biodiversity Institute (SANBI). [Online]. Available from: http://www.sanbi.org/biodiversity/nsba.htm [Accessed 16 July 2010].

Rozenstein O & Karnieli A 2011. Comparison of methods for land-use classification incorporating remote sensing and GIS inputs. *Applied Geography* 31: 533-544.

Ruiz RF 2003. Alternative land uses to forestry in the Western Cape: A case study of La Motte plantation. Masters thesis. Stellenbosch: Stellenbosch University.

SADC (Southern African Development Community) 2008. *Southern African environment outlook*. Gaborone/Harare/Nairobi, SADC, SARDC, IUCN & UNEP.

Salem 2003. Application of GIS to biodiversity monitoring. *Journal of Arid Environments* 54: 91–114.

SANBI (South African National Biodiversity Institute) 2011. Protected areas shapefiles and NLC 2009. [Online]. Available from: http://bgis.sanbi.org/LUDS.asp. [Accessed 8 September 2011].

Santi E, Maccherini S, Rocchini D, Bonini I, Brunialti G, Favilli L, Perini C, Pezzo F, Piazzini S, Rota E, Salerni E & Chiarucci A 2009. Simple to sample: Vascular plants as surrogate group in a nature reserve. *Journal for Nature Conservation* 18: 2-11.

Sarkar S & Margules CR 2002. Operationalizing biodiversity for conservation planning. *Journal of Biosciences* 27: 299-308.

Sarkar S, Pressey RL, Faith DP, Margules CR, Fuller T, Stoms DM, Moffett A, Wilson KA, Williams KJ, Williams PH & Andelman S 2006. Biodiversity conservation planning tools: Present status and challenges for the future. *Annual Review of Environment and Resources* 31: 123-159.

Satellite Imaging Corporation 2012. SPOT-5 satellite imagery and sensor characteristics. [Online]. Available from: http://www.satimagingcorp.com/satellite-sensors/spot-5.html. [Accessed 31 January 2012].

Schulze RE, Maharaj M, Lynch SD, Howe BJ & Melvil-Thomson B 1997. *South African atlas of agrohydrology and climatology*. Pretoria: Water Research Commission.

Scott MJ, Davis F, Cusuti B, Noss R, Butterfield B, Groves C, Anderson H, Caicco S, D'Erchia F, Edwards TC, Ulliman J & Wright RG 1993. GAP analysis: A geographic approach to protection of biological diversity. *Wildlife Monographs* 123: 1-41.

Settele J 2005. ALARM: Assessing-large scale risks for biodiversity with tested methods. *GAIA* 14: 69-72.

Soulé ME 1991. Conservation tactics for a constant crisis. *Science* 253: 744-749.

Spangenberg JH 2007. Biodiversity pressure and the driving forces behind. *Ecological Economics* 61: 146-158.

SPOT Image 2005. SPOT Satellite technical data [online]. Available from: http://www.spotimage.com/web/en/229-the-spot-satellites.php [Accessed 30 January 2012].

Stephenson G & Van Niekerk A 2009. A cost-effective, rule-based technique to improve forestry inventory on a national scale. Paper presented to the Geoscience and Remote Sensing Symposium IEEE International, IGARSS 2009.

Stillwell J & Clarke G 2004. *Applied GIS and Spatial Analysis*. Chichester: John Wiley & Sons.

Stockwell D & Townsend-Peterson A 2003. Comparison of resolution of methods used in mapping biodiversity patterns from point-occurrence data. *Ecological Indicators* 3: 213-221.

Stoms DM, Comer PJ, Crist PJ & Grossman DH 2005. Choosing surrogates for biodiversity conservation in complex planning environments. *Journal of Conservation Planning* 1: 44-63.

Thuiller W, Midgley GF, Rouget M & Cowling RM 2006. Predicting patterns of plant species richness in megadiverse South Africa. *Ecography* 29: 733-744.

Tucker CJ, Townshend JRG & Goff TE 1985. African land-cover classification using satellite data. *Science* 227: 369-375.

Turpie JK 2003. The existence value of biodiversity in South Africa: How interest, experience, knowledge, income and perceived level of threat influence local willingness to pay. *Ecological Economics* 46: 199-216

UNEP (United Nations Environmental Program) 1992. *Convention on biological diversity. Text and annexes*. Geneva: UNEP.

Van Den Berg EC, Plarre C, Van Den Berg HM & Thompson MW 2008. *The South African National Land Cover 2000*. Agricultural Research Council-Institute for Soil, Climate and Water, Pretoria, Report GW/A/2008/86.

Van Jaarsveld AS, Freitag S, Chown SL, Muller C, Koch S, Hull H, Bellamy C, Krüger M, Endrödy-Younga S, Mansell MW & Scholtz CH 1998. Biodiversity assessment and conservation strategies. *Science* 279: 2106-2108.

Van Niekerk 2010. A comparison of land unit delineation techniques for land evaluation in the Western Cape, South Africa. *Land Use Policy* 27: 937-945.

Von Hase A, Rouget M, Maze K & Helme N 2003. *A fine-scale conservation plan for Cape lowlands Renosterveld: Technical report*. Claremont: Botanical Society of South Africa.

Velásquez A, Bocco G, Romero FJ, & Vega AP 2003. A landscape perspective on biodiversity: The case of central Mexico. *Mountain Research and Development* 23: 240-246.

Walter V 2004. Object-based classification of remote sensing data for change detection. *Journal of Photogrammetry & Remote Sensing* 58: 225-238.

Walther BA, Schäffer N, Van Niekerk A, Thuiller W, Rahbek C & Chown SL 2007. Modelling the winter distribution of a rare and endangered migrant, the Aquatic Warbler *Acrocephalus paludicola*. *Ibis* 149: 701-714.

Walther BA, Van Niekerk A & Rahbek C 2011. Long-term population declines of Palearctic passerine migrant birds: a signal from the Sahel? Paper presented at the British Ornithologists' Union's (BOU) 2011 Annual Conference. [Online] Available from: http://www.bou.org.uk/bouproc-net/migratory-birds/walther-etal.pdf. [Accessed: 27 January 2012].

Weng Q 2002. Land use change analysis in the Zhujiang Delta of China using satellite remote sensing, GIS and stochastic modelling. *Journal of Environmental Management* 64: 273-284.

Wessels KJ, Freitag S & Van Jaarsveld AS 1999. The use of land facets as biodiversity surrogates during reserve selection at a local scale. *Biological conservation* 89: 21-28.

Wessels KJ, Reyers B & Van Jaarsveld AS 2000. Incorporating land cover information into regional biodiversity assessments in South Africa. *Animal Conservation* 3: 67-79.

Wessels KJ, Reyers B, Van Jaarsveld AS & Rutherford MC 2003. Identification of potential conflict areas between land transformation and biodiversity conservation in north-eastern South Africa. *Agriculture, Ecosystems and Environment* 95: 157-178.

Willhauck G 2000. Comparison of object-oriented classification techniques and standard image analysis for the use of change detection between SPOT multispectral satellite images and aerial photos. *International Archives of Photogrammetry and Remote Sensing* 33: 35-42.

Williams P, Faith D, Manne L, Sechrest W & Preston C 2006. Complementarity analysis: Mapping the performance of surrogates for biodiversity. *Biological Conservation* 128: 253 -264.

Wishart MJ 2000. Catchments as conservation units for riverine biodiversity. *African Journal of Aquatic Science* 25: 169-174.

World Bank 2010. South Africa country data profile. [Online]. Available from: http://web.worldbank.org/WBSITE/EXTERNAL/COUNTRIES/AFRICAEXT/SOUTHAFRICAEXTN/0,,menuPK:368102~pagePK:141132~piPK:141109~theSitePK:368057,00.html. [Accessed: 26 January 2012].

Younge A & Fowkes S 2003. The Cape Action Plan for the Environment: Overview of an ecoregional planning process. *Biological Conservation*: 15-28.

Zhang J, Zhengjun L & Xiaoxia S 2009. Changing landscape in the Three Gorges Reservoir Area of Yangtze River from1977 to 2005: Land use/land cover, vegetation cover changes estimated using multi-source satellite data. *International Journal of Applied Earth Observation and Geoinformation* 11: 403-412.

PERSONAL COMMUNICATIONS

Helme N 2011: Botanical consultant at Nick Helme Botanical Surveys. E-mail correspondence and telephonic interviews on 28 June and 7 July. Re: Vegetation and agricultural changes in the lower Berg River catchment.

Holmes PM (Patricia.Holmes@capetown.gov.za) 2011: Biophysical specialist at Environmental Resource Management Department, City of Cape Town. Re: Regeneration potential of Hopefield Sand Fynbos. E-mail to TM Stuckenberg (tstuckenberg@gmail.com) (27 June).