AN ECOLOGICAL AND CONSERVATION PLANNING STUDY OF THE PLANT BIODIVERSITY ON NATURAL AND SEMI-NATURAL REMNANTS ON THE CAPE FLATS, SOUTH AFRICA

by Janice S. Golding

An ecological and conservation planning study of the plant biodiversity on natural and semi-natural remnants on the Cape Flats, South Africa

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Declaration

I, the undersigned, hereby declare that the work contained in this thesis is my own original work and has not previously, in its entirety or in part, been submitted at any university for a degree.

Signature *Date*

All things by immortal power
near or far
hiddenly
To each other linked are,
That though canst not stir a flower
Without troubling of a star

- Francis Thompson

Photo: National Botanical Institute

Moraea aristata (D.Delaroche) Asch. & Graebn. (Iridaceae)

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AN ECOLOGICAL AND CONSERVATION PLANNING STUDY OF THE PLANT BIODIVERSITY ON NATURAL AND SEMI-NATURAL REMNANTS ON THE CAPE FLATS, SOUTH AFRICA

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Chapter 1

INTRODUCTION: AN ECOLOGICAL AND CONSERVATION PLANNING STUDY OF THE PLANT BIODIVERSITY IN NATURAL AND SEMI-NATURAL REMNANTS ON THE CAPE FLATS, SOUTH AFRICA

Abstract

The Cape Flats (Western Cape Province, South Africa) is situated on the low-lying sandy plain immediately adjacent to the Cape Peninsula, and is part of the mediterranean-climate Cape Floristic Region. Fire-dependant Sand Plain Fynbos, Strandveld and West Coast Renosterveld exist on natural and semi-natural remnants interspersed in a hostile landscape. The Cape Flats, its botanical attributes and the threatening processes that prevail in its densely populated urban matrix are introduced. Broad overviews of aspects of the biophysical environment, the flora, land transformation, the socio-political and institutional environment, and conservation initiatives are presented. This chapter provides a framework for the general approach and the methodologies used in the studies contained in this thesis.

INTRODUCTION: AN ECOLOGICAL AND CONSERVATION PLANNING STUDY OF THE PLANT BIODIVERSITY ON NATURAL AND SEMI-NATURAL REMNANTS ON THE CAPE FLATS, SOUTH AFRICA

"The future of this paradise gives occasion for some anxiety. Some of the vleis and pools near civilisation have gone within the writer's memory, drained or filled in... There is a real need to save some of the particularly interesting and accessible areas for the use and enjoyment of future generations…" *Ms Edith L. Stevens, the first female lecturer in Botany at the University of Cape Town, Cape Town, South Africa (Stephens, 1929‡).*

1. INTRODUCTION

The Cape Flats is one of the oldest urban settlements in South Africa (Deacon, 1992), and is situated in the Western Cape Province. The landscape is a matrix of urban developments interspersed with fragments of natural and semi-natural habitat. Urbanisation is steadily escalating, principally in areas of high human density; and human density has been shown to be inversely related to land protection (Dewar & Watson, 1990; Wood *et al.*, 1994; van Wilgen, 1996). Given the unprecedented declines in plant species and habitats as a result of urbanisation processes, the need for strengthening conservation protection is imperative.

The overall objective of the thesis is to provide insights to the fragmentation of the Cape Flats flora and the selection of key areas that can best conserve the species and habitats. While there is a large body of literature on these issues, this thesis is aimed at contributing to current conservation initiatives. This chapter introduces the environment of the Cape Flats and serves as the basis for the general approach of the thesis. A broad overview is provided of the biophysical environment, the flora, land transformation, conservation initiatives, as well as the socio-political environment and institutional context that have, and continue, to influence conservation on the Cape Flats.

1.1. STUDY AREA

The political/administrative jurisdiction of the Cape Flats falls within the Cape Metropolitan Area (> 1 300 km^2) (Western Cape Province, South Africa) (Figure 1.1).

‡ This quotation is from a book intended as a reference guide for visiting botanists to the Cape participating in the British Association for the Advancement of Science.

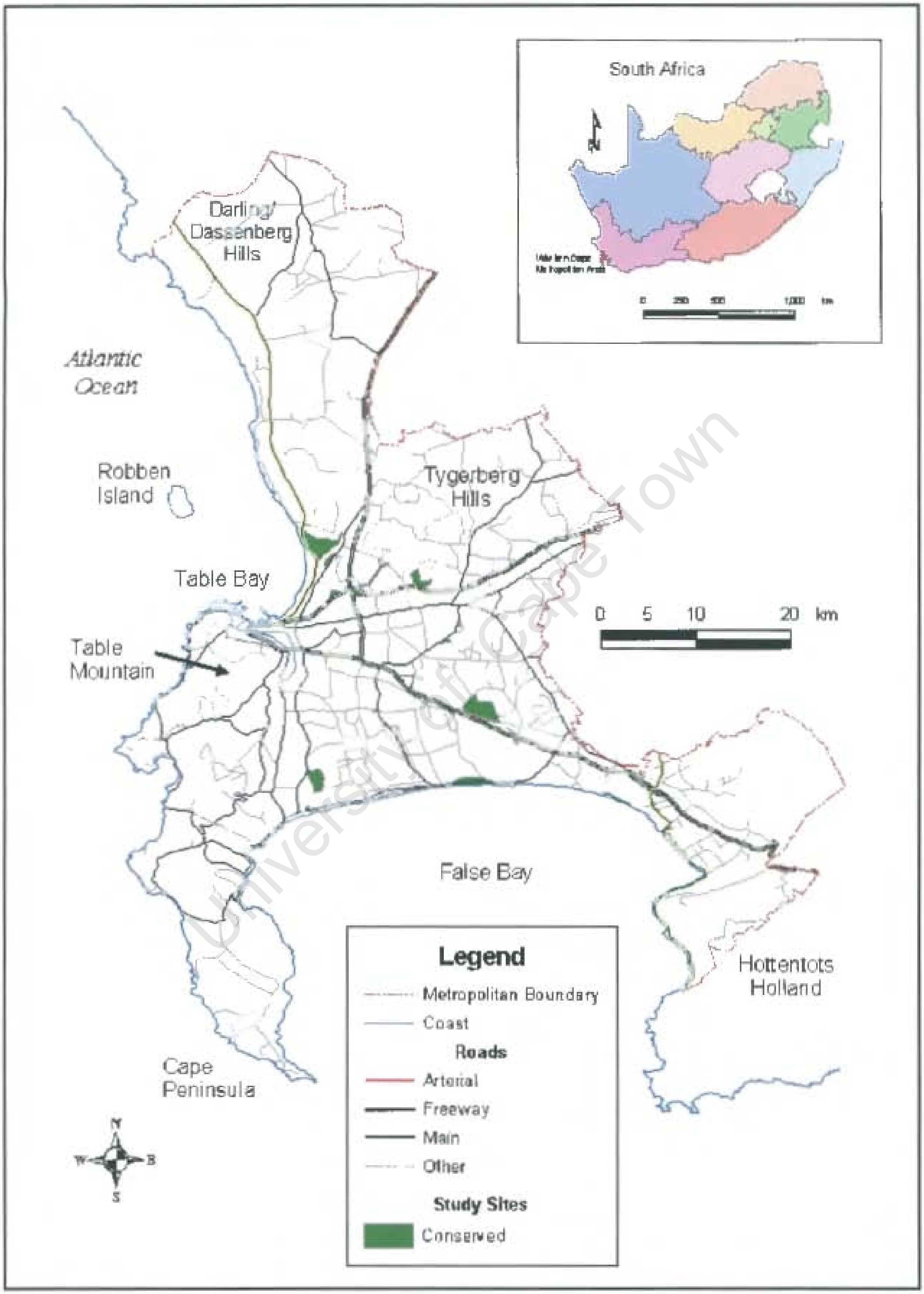

Figure 1.1 General topographic and geopolitical features of the Cape Metropolitan Area. The conserved sites are are germane to 1997/1998.

The Cape Metropolitan Area incorporates three broad physiographic features: the Cape Peninsula (mountain chain), the Cape Flats (a low-lying plain adjacent to the Cape Peninsula) and the Darling/Dassenberg area (a series of undulating plains to the north of the Cape Flats and Cape Peninsula) (Wood *et al.*, 1994), also nominally referred to as the West Coast Lowlands (Boucher, 1987; Boucher & Moll, 1980).

The major physiographic feature of the Cape Flats is Tygerberg Hill (most of which is a protected area), which at its highest altitude of over 400 m, forms the northern, most conspicuous boundary of the study area (Figure 1.1). Several coastal sites to the northwest (in close proximity to Tygerberg Hill) are also considered part of the Cape Flats. The False Bay coast forms the southern boundary, and was historically characterised by pockets of fixed parabolic and transverse mobile dunefields rising to over 80 m in places. Most of these dunefields have now made way for low-income housing and others are covered by dense invasive alien infestations (e.g. *Acacia cyclops*). To the east lies the Hottentots-Holland mountain range, and to the west, the world famous landmark of Table Mountain and the Table Bay shoreline (McDowell & Low, 1990; Wood *et al.*, 1994).

The general understanding of the extent and range of the Cape Flats has, over the years, changed considerably, mainly as a function of political and institutional jurisdictions to facilitate the implementation of conservation activities. Maze & Rebelo (1999) refer to more than 80% of the Cape Metropolitan Area as the Cape Flats, stating that the entire lowlands of the Cape Metropolitan Area as well as parts of the higher-lying Darling/Dassenberg area are now part of the Cape Flats (Cape Flats *sensu lato*) (Figure 1.1). The Cape Flats *sensu stricto* is generally less than 120 m above sea level (Schalke, 1973; McDowell & Low, 1990; Theron *et al.*, 1992), and covers an area of approximately 670 km^2. The Cape Flats *sensu stricto* has been subjected to more intense and far more complex urban impacts over a longer period of time and has for long been recognised as a unique entity (flora, socio-economic dynamics and the land ownership situation) (Low, 1979; Hall, 1984; Dewar & Watson, 1990; McDowell & Low, 1990; McDowell *et al.*, 1991; Behrens & Watson, 1992; Bridgeman *et al.*, 1992; Wood *et al.*, 1994; McKenzie & Rebelo, 1997) compared to its most recent delimitation by Maze & Rebelo (1999). Hereinafter, the Cape Flats is referred to in conventional terms (Cape Flats *sensu stricto*).

The human population of the Cape Peninsula and the adjacent Cape Flats far exceeds three million and is estimated to be increasing at 4 - 5% per annum (Dewar & Watson, 1990; van Wilgen, 1996). The pressure to develop these lowlands, which have been rated as amongst the most threatened ecosystems in South Africa (Hall, 1984; Jarman, 1986; Wood *et al.*, 1994; Trinder-Smith, 1996), is immense. Areas along the False Bay coastline are particularly susceptible to low-cost housing as part of an extension of housing projects designed for the economically and politically marginalised people under South Africa's previous political dispensation prior to 1994 (Wood *et al.*, 1994).

2. THE CAPE FLATS ENVIRONMENT

2.1 BIOPHYSICAL FEATURES

The Fynbos Biome forms part of the Cape Floristic Region (*ca* 90 000 km^2), and occupies the extreme south-west of South Africa (Taylor, 1978; Bond & Goldblatt, 1984; Cowling & Hilton-Taylor, 1994; 1997). The biome is known for its high level of diversity and endemism: seven families and 198 genera are restricted to this region (Cowling & Hilton-Taylor, 1997). More recent taxonomic work indicates that some 160 vascular genera are endemic to the Cape Floristic Region (16%, all dicotyledonous genera) and that a total of 988 genera, approximately half of which occur in southern Africa (Botswana, Lesotho, Namibia and Swaziland), are represented in the flora (Goldblatt & Manning, 2000). This outstanding plant diversity is attributed to a high turnover of species within and between habitats, coupled with steep habitat gradients (Simmons & Cowling, 1996). The extreme south-west of the biome is arguably the most biologically well studied and understood of all areas in Africa.

Fire-prone and fire-dependent shrublands dominate the Fynbos Biome, a feature of mediterranean-climate ecosystems (Taylor, 1978; Kruger 1979; Cowling, 1992; Cowling *et al.*, 1998). Some of the primary determinants of the vegetation include periodic fire, predictable seasonality (hot summer drought and cold winter rainfall), edaphic variability, and on the Cape Flats and adjacent Cape Peninsula, a predominantly strong, summer south-easterly wind pattern (Boucher, 1987; Cowling *et al.*, 1992; Fredericks, 1992; Simmons & Cowling, 1996; Cowling *et al.*, 1998; Privett, 1998). The environmental determinants of the Cape Flats have been well documented in the above-stated references and elsewhere, and are thus not repeated here.

The Cape Flats is essentially a sandy isthmus that separates Table Bay and False Bay (Deacon, 1992) (Figure 1.1). Its inclusion to the African mainland occurred *ca* 18 000 years ago during the Last Glacial Maximum when the sea level was lowered by *ca* 120 m (Schalke, 1973; Theron *et al.*, 1992). The surface materials of the Cape Flats lowlands are mainly beach deposits that have been subsequently reworked by wind action, and a few sporadic geological outcrops (limestone, calcrete, ferricrete, sandstone and granite). Due to the wind pattern, sands on the north and west of the Cape Flats are therefore older (Schalke, 1973).

Three distinct pedological bands extend from the coast inland (Schalke, 1973), each of which broadly support their own floras (Taylor, 1972; Kruger, 1979; Moll & Bossi, 1984; Boucher, 1987; Fredericks; 1992). Soils further inland are derived from ancient, Malmesbury Group shales and Cape Suite granites (Schalke, 1973). Sands nearer the coast are derived from wind blown sands and beach deposits (alkaline), in addition to marine sands consisting of leached carbonate compounds (acidic).

Before the onset of canalisation and filling in of wetlands, the Cape Flats landscape was a hydrologically dynamic environment (Stevens, 1929; Taylor, 1972; Boucher, 1987; McDowell & Low, 1990). In some areas this is still evident where the water table is near the surface. Depressions often fluctuate with seasonal water levels while other areas remain permanently waterlogged. Within coastal dune systems, wetlands originate from beaches that have become partially or completely cut off from the ocean. The duration and extent of winter flooding is thus regarded as a critical phase for the submergence of aquatic and semi-aquatic plants such as *Isoetes* species (Stevens, 1929). Depending on the groundwater regime, different inundation processes occur, thus leading to finely delimited edaphic gradients (Fredericks, 1992). However, these hydrological dynamics have now been disrupted by human activities.

2.2 FLORA

Based on structural, floristic and environmental elements, three vegetation types, Fynbos Shrublands (Lowland Fynbos or Sand Plain Fynbos), Cape Transitional Shrublands (Strandveld) and West Coast Renosterveld have been recognised on the Cape Flats (Moll *et al.*, 1984) (Figure 1.2). This phytosociological classification has become the accepted norm

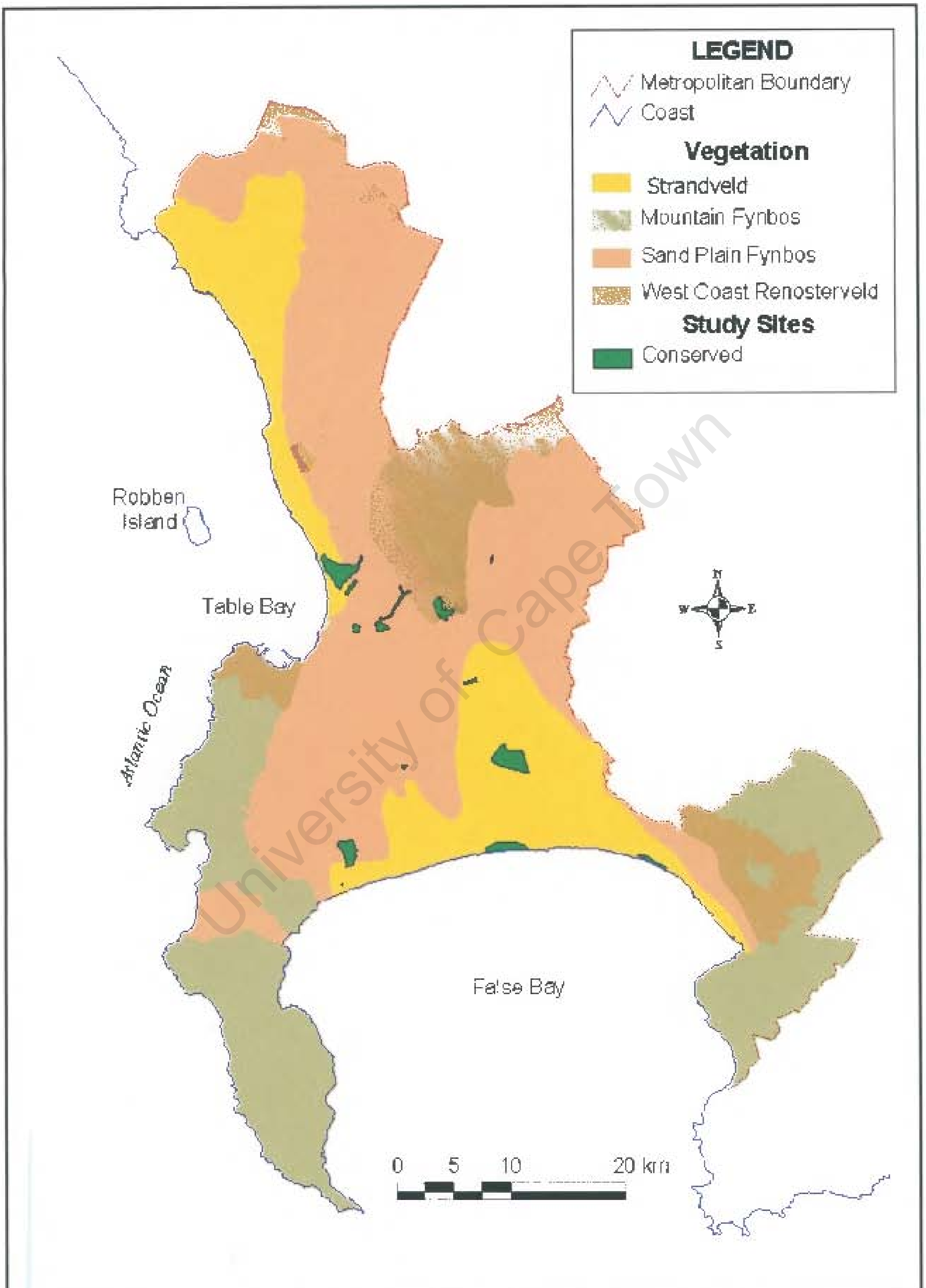

Figure 1.2 Vegetation types (Strandveld, Sand Plain Fynbos and West Coast Renosterveld) of the Cape Flats and Cape Metropolitan Area (after Low & Rebelo, 1996). The conserved sites are germane to 1997/1998.

amongst conservationists working on the Cape Flats (e.g. Low & Rebelo, 1996). The southernmost limit of West Coast Renosterveld (Low & Rebelo, 1996) occurs in the northern and north-easterly, higher-lying extremities of the study area. In terms of the historical distribution, the Cape Flats is dominated by Sand Plain Fynbos and Strandveld, and to a lesser extent, by West Coast Renosterveld (Figure 1.3). The adjacent Cape Peninsula is predominantly vegetated by Mountain Fynbos (Low & Rebelo, 1996).

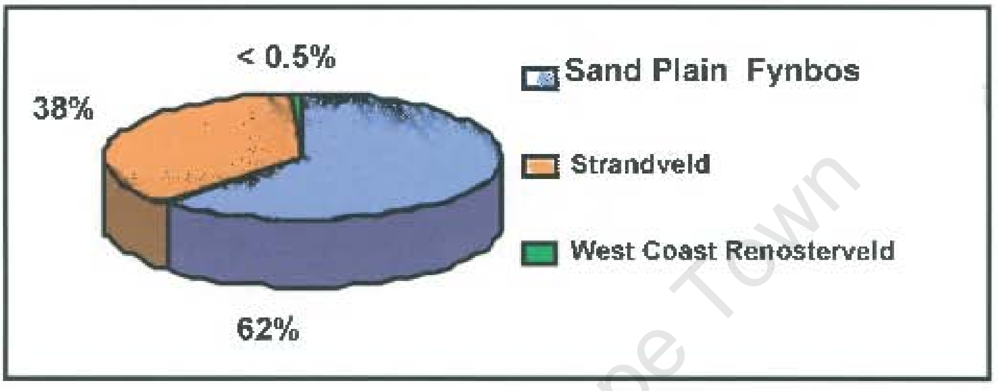

Figure 1.3 The original extent of the vegetation types on the Cape Flats (after Wood *et al.*, 1994).

Cowling & Heijnis (2001) developed a system of land classes (Broad Habitat Units, BHUs) using a Geographical Information System (GIS), for the purposes of conservation planning. These land classes were based on environmental (Homogenous Climate Zones, geology and topography) and associated floristic heterogeneity (Low & Rebelo, 1996), and are thus surrogates for vegetation types. In their study, Cape Flats Fynbos/Thicket Mosaic and Blackheath Sand Plain Fynbos correspond with Strandveld and Sand Plain Fynbos on the Cape Flats respectively.

2.2.1. Strandveld

Young sands which include mobile dunefields (but now largely stabilised) occur on the coastal belt of the Cape Flats (Figure 1.2). Strandveld is adapted to harsh windy conditions and fine sea sprays. These dunefields are extremely alkaline, poorly consolidated, and principally support Strandveld. This vegetation type is characterised by broad leaved and often weakly spinescent shrubs, with a comparatively high incidence of succulents (Boucher & Moll, 1979; Kruger, 1979; Boucher, 1987).

There is low diversity of understorey species. Few members of the Restionaceae and virtually no members of the Proteaceae and Ericaceae are represented in Strandveld (Low & Rebelo, 1996). Prominent Strandveld members include the genera *Euclea, Ischyrolepis, Passerina, Phylica, Rhus* and *Salvia* (Boucher, 1987). A sharp ecotone exists on the boundary of Strandveld and Sand Plain Fynbos. This ecotone is characterised by a steep gradient in soil pH. For example, this is evident in Rondevlei Nature Reserve where the pH ranges from 4.7 to 9.2 over a north-south gradient spanning less than 2 km (Wood & Gibbs, 1997).

2.2.2 Sand Plain Fynbos

Older, inland sands support Sand Plain Fynbos, and are typically acidic, shallow and grey (Figure 1.2; Low & Rebelo, 1996). The sands are floristically characterised by a diversity of overstorey dominants of the Proteaceae, Asteraceae, *Phylica cephalantha*, and to a lesser extent, by the Restionaceae (e.g. *Thamnochortus punctatus*) (Moll *et al.*, 1984). Mid-high shrubs with ericoid features (small leaves with rolled margins) are well-represented.

The various plant communities within Sand Plain Fynbos can generally be distinguished by the composition of Proteaceae (Low & Rebelo, 1996). Although Sand Plain Fynbos has similar structural features to adjacent Mountain Fynbos on the Cape Peninsula (Low & Rebelo, 1996), few species are shared between the two vegetation types.

2.2.3 West Coast Renosterveld

While Strandveld and Sand Plain Fynbos are associated with nutrient deficient sands, West Coast Renosterveld is associated with fine-grained, moderately fertile soils (neutral, heavy clays and loams) derived entirely from shale and granite (Figure 1.2). This vegetation type is characterised by ericoid-leaved, evergreen shrubs. The Restionaceae is particularly poorly represented and is instead replaced by grasses (Poaceae). Proteoid shrubs (Proteaceae) are virtually absent. The understorey is mainly annual and herbaceous interspersed with sparse, perennial C_3 grasses (Low & Rebelo, 1996).

West Coast Renosterveld is known for its rich geophytic flora with Iridaceae, Liliaceae and Oxalidaceae particularly well represented (Daines & Low, 1993; Low & Rebelo, 1996).

The typical overstorey dominant in West Coast Renosterveld is *Elytropappus rhinocerotis* (Renosterbos), while sub-dominants include *Anthospermum aethiopicum* and *Eriocephalus africanus* (Daines & Low, 1993).

2.3 LAND TRANSFORMATION

Prior to the onset of urbanisation (1915 - 1919), 595 plant species were collected on a 343 ha estate on the western fringe of the Cape Flats (Rourke *et al.*, 1981). This is almost 50% of the total number of species recorded on the Cape Flats in this study. In terms of the indigenous vegetation types, it has been estimated that less than 1% of Sand Plain Fynbos and less than 15% of the original extent of Strandveld of the Cape Flats are protected in nature reserves (Wood *et al.*, 1994).

More than 97% of the original extent of West Coast Renosterveld in South Africa has been transformed; 1.76% is formally conserved (Low & Rebelo, 1996). The rate of transformation of Sand Plain Fynbos and Strandveld habitats peaked in the 1980s due to massive housing demands (Figure 1.4) (Dewar & Watson, 1990; McDowell & Low, 1990; Wood *et al.*, 1994).

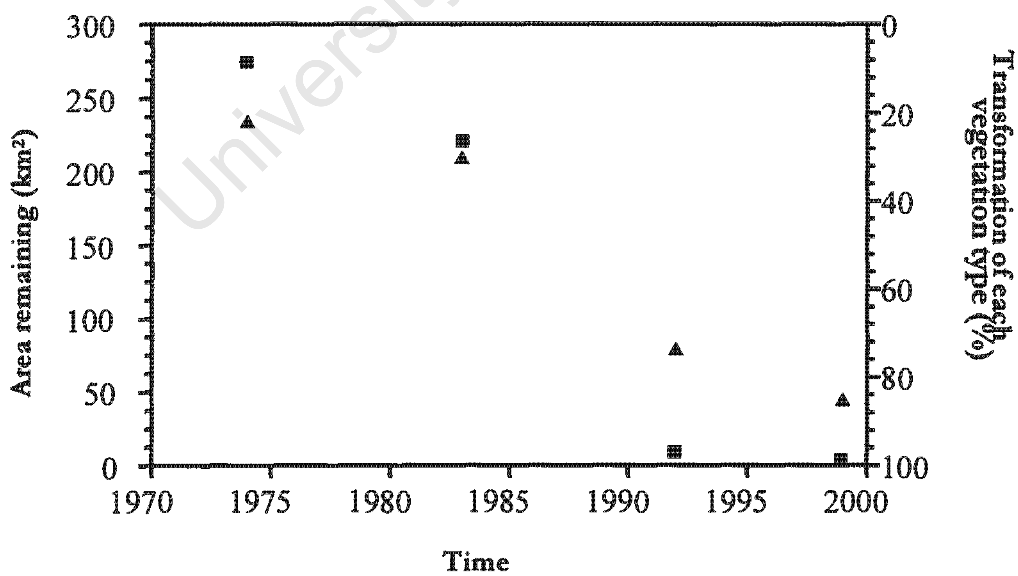

Figure 1.4 The rate of transformation over a 25 year period (1974 – 1999, by woody alien invasive encroachment, agriculture and urbanisation) of the two major vegetation types on the Cape Flats. Squares (■) represent remnants of Sand Plain Fynbos and triangles (▲) represent Strandveld (Data sources: McDowell & Low, 1990; McDowell *et al.*, 1991; Wood *et al.*, 1994).

Between 1983 and 1989, the Cape Flats experienced a 50% reduction in these habitats accompanied with an increase in threatened plant species (McDowell & Low, 1990). In the 1990's, the decline of transformation of Sand Plain Fynbos and Strandveld is evident only because large-scale developments are limited due to the availability of continuous tracts of land.

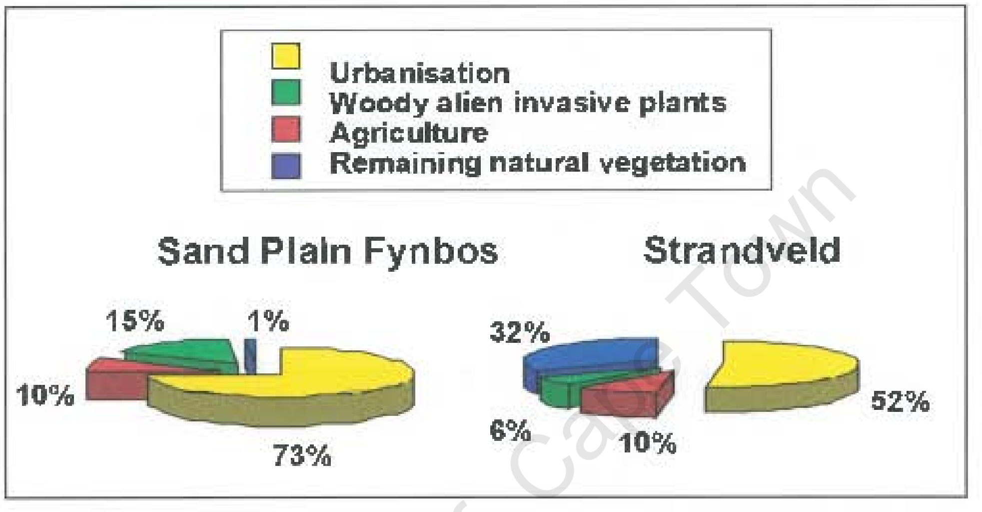

Figure 1.5 The extent of transformation by urbanisation, alien invasive encroachment and agriculture in the two main vegetation types on the Cape Flats (adapted from Wood *et al.*, 1994).

Habitat conversion on the Cape Flats is principally due to urbanisation, invasive alien encroachment and small-scale agricultural activities (Figure 1.5). Sand Plain Fynbos habitats are relatively easier to exploit for urbanisation since these inland sands are devoid of bedrock, have a lower water table than Strandveld and the relief is generally flatter than the dune plains (Wood *et al.*, 1994). As available land diminished in Sand Plain Fynbos habitats, land in Strandveld habitats became the subject of urbanisation (Figure 1.4). The switch co-incided with a rapid increase in urbanisation in the 1970s when, under the previous political dispensation of South Africa, forced removals and an influx of people from suburbs surrounding Cape Town brought about an additional pressure to develop Strandveld habitats (Low & McKenzie, 1988).

The earliest, and most notable disturbances on the Cape Flats, date back more than 150 years. An increase in driftsands brought about by increased landuse (Taylor, 1972; Low, 1979) led to the introduction of *Acacia saligna* and *A. cyclops* from Australia by the Colonial

Secretary of the Cape in an attempt to stabilise beach sand deposition (Low, 1979). Today, the situation is such that more than 85% of the unreserved remnant surface area on the Cape Flats has been degraded in some way by woody and grassy (*Bromus* sp. and *Lolium* sp.) invasive alien species (revised from Wood *et al.*, 1994). Alien plants present a significant threat to plant diversity (Figure 1.5). Higgins *et al.* (1999) estimate that as much as 89% of continuous tracts of land on the adjacent Cape Peninsula could be covered by woody invasives during the next 10 years if preventative measures are not implemented. Moreover, the probability of recruitment failure of the dominant functional types in lowland fynbos increases by 40 - 80% due to invasive Australian acacias (Musil, 1993), a finding corroborated by Holmes & Cowling (1997).

The remnant vegetation of the Cape Flats is subject to inappropriate fire regimes - either too frequent (in low-income areas) or too infrequent (in higher-income areas). The use of fire as a tool for land clearing is motivated by the fact that dense vegetation cover is associated with criminal activities (McDowell *et al.*, 1991; Behrens & Watson, 1992; Mohammed, 1998). In addition, people in informal settlements on the Cape Flats use fire as a form of energy for heating and cooking, and for burning refuse. Fires often spread from these sources (*pers. obs.*). In other areas, fire is totally excluded as it is considered a threat to more established properties and assets. The legal and financial ramifications of fire as a management tool (simulation of the natural fire regime) is cumbersome to implement (van Wilgen *et al.*, 1992; Wood *et al.*, 1994). Seedbank longevity, and hence species recruitment on the Cape lowlands, is poor with prolonged fire deferral (Musil, 1991). Soil-stored seedbanks of lowland fynbos have also been shown to be transient and thus extremely susceptible to threatening processes (Musil 1991). As such, the instatement of appropriate fire regimes is important.

Sand-mining permits which require local and national approval under the Minerals Act by the Department of Mineral and Energy Affairs have allowed extensive tracts of land, particularly coastal land, to be exploited for the building industry (Mohammed, 1998). Advances in dune-grading technology have exacerbated this (McDowell *et al.*, 1991). There are few records of successful habitat restoration efforts on the Cape Flats, mainly because legislation is not always enforced (e.g. Mohammed, 1998).

Areas of Sand Plain Fynbos with suitable soil characteristics (relating to soil texture) have been transformed and heavily fertilised for vegetable growing on the Cape Flats (Figure 1.5). The extent of West Coast Renosterveld falling outside the Cape Flats study area has been extensively reduced mainly by cereal- and pasture crops (Daines & Low, 1993). Within the study area, West Coast Renosterveld has been transformed by urbanisation and viticulture, and pristine and near-pristine remnants now lie adjacent to farmlands or are confined to higher-lying areas (e.g. Tygerberg Nature Reserve, Blouberg Renosterveld site and some sites on the eastern fringe of the Cape Flats).

2.4 SOCIO-POLITICAL CONTEXT

The Cape Flats was set aside primarily for formerly marginalised ethnic groups under the South African government's previous political dispensation. The area is beset by poverty, violent crime, and inadequate housing, health care and employment opportunities. In addition, there are a host of rapidly growing informal settlements where new urban migrants are illegally occupying previously vacant land and sites zoned as nature reserves (Behrens & Watson, 1992; Wood *et al.,* 1994). This has resulted in increased land pressure on the Cape Flats to provide more housing and better services, which impact on the flora (Mohammed, 1998; Khan, 1999). The formal and informal settlements on the Cape Flats constitute Cape Town's growing Third World character (Dewar & Watson, 1990).

In contrast, many areas surrounding the Cape Flats are characterised by all the amenities and opportunities found in First World cities. While these two extremes of urban inequality may not be unique to the developing world, the physical and spatial separation of these worlds resulting from the former legal codification of racial separation is uniquely South African (Low & McKenzie, 1988; Dewar & Watson, 1990; Khan, 1999). Socio-political aspects surrounding conservation on the Cape Flats run deep and are emotional issues that often have the ability to either hamper or promote conservation initiatives (e.g. Ngeleza, 1990; Nguta, 1992; Planning Partnership, 1993; Mohammed, 1998; Khan, 1999).

2.5 LEGISLATIVE AND INSTITUTIONAL CONTEXT

There is a range of important factors that contrive to frustrate conservation planning and implementation initiatives on the Cape Flats. Broadly, the factors are resource, institutional and space limitations (Behrens &Watson, 1992).

It is recognised, even in the uppermost planning echelons, that the increasing need for land acquisition for the delivery of primary services, such as housing and sanitation, and infrastructural development, often takes precedence over conservation issues (Dewar & Watson, 1990; Behrens & Watson, 1992). A few years ago, a pervasive notion existed amongst urban planners and local inhabitants that undeveloped land on the Cape Flats, was 'waste land' (Behrens & Watson, 1992; Mohammed, 1998), the legacy of which has had far-reaching ecological consequences. The historical and current social discourse, economic pressures and political agendas peculiar to the Cape Flats are realities that make conservation goals extremely challenging to achieve. (Khan, 1999).

The protection of species and the conservation and restoration of their core habitats within and outside protected areas, is largely mediated by national environmental legislation (Environment Conservation Act (1989), 1997; White Paper, 1997) and provincial policies. Policies and legislation pertaining to biodiversity-related issues on the Cape Flats are administered at the provincial level by the Cape Metropolitan Council (Environmental Department), and implemented by the Western Cape Nature Conservation Board and local/municipal authorities. The jurisdiction of the Cape Metropolitan Council is the Cape Metropolitan Area. The Environment Conservation Act (1989) (1997) is currently being revisited as per the drafting of the new National Biodiversity Bill (Department of Environmental Affairs and Tourism, Pretoria, June 2002).

The provincial policies espoused by the Cape Metropolitan Council generally reflect that the enhancement of overall human welfare and quality of life should be achieved through the dual benefits of economic development and environmental protection. Still, the central challenge is to entrench biodiversity conservation issues more firmly into development agendas.

3. CONSERVATION INITIATIVES

Many dedicated botanists have contributed to our primary botanical understanding of the Cape Flats (e.g. Stephens, 1929; Taylor, 1972; Kruger, 1979; Moll *et al.*, 1984; Boucher, 1987; Cowling, 1992). There have also been others who have forewarned a critical situation regarding the persistence of the Cape Flats flora (Low, 1979; Hall, 1984; McDowell & Low, 1990; McDowell *et al.*, 1991; Jarman, 1986; Wood *et al.*, 1994; Trinder-Smith, 1996; McKenzie & Rebelo, 1997; Wood & Gibbs, 1997; Maze & Rebelo, 1999). Much work has also taken place with regards to the identification of key sites for conservation on the Cape Flats (Jarman, 1986; McDowell & Low, 1990; McKenzie & Rebelo, 1997; Maze & Rebelo, 1999; Maddock & Benn, *in prep.*). These have taken place in tandem with the refinement of conservation planning methodologies.

Conservation planning has seen as shift over the last ten years to become more systematic and strategic, and the approach has been to consider socio-economic issues that influence the implementation of plans. The Cape Action Plan for the Environment (CAPE Project) (see Cowling *et al.*, 1999) and the Core Flora Conservation Areas on the Cape Flats (Maze & Rebelo, 1999) are significant conservation planning initiatives to this end.

A wide variety of research activities has therefore focussed on the Cape Flats, all primarily addressing the conservation plight of its flora. A positive outcome of this is that good data sets have been developed. Due to the dynamic changes taking place in the urban environment of the Cape Flats, these data sets are constantly changing. The implication of this is that as fast as conservation and management recommendations are being made, they are becoming 'outdated'. As such, planning processes need to be dynamic in that they should reflect these changes in conservation and management recommendations.

4. THESIS STRUCTURE AND OUTLINE OF CHAPTERS

This thesis studies the fragmented landscape of the Cape Flats, and attempts to prioritise fragments for conservation based on their biological and physical properties, and their socio-economic setting. The thesis first explores the biodiversity features by describing and quantifying the species and habitats; it also explores the threats to these features, and hence, covers the derivation of metrics that relate to disturbance and socio-economic factors. The degree of habitat loss and degradation on the Cape Flats has had obvious consequences in terms of the plant biota persisting within and outside the existing reserve estate. As such, the thesis also focuses on the ecological dynamics that are thought to be responsible for determining species richness, namely habitat heterogeneity, threatening processes and fragment size. Maintaining current levels of species richness requires a strategic and systematic approach to conservation plans and their implementation. The ensuing chapters address these issues.

Chapter 2 presents the study sites used in this thesis and their biodiversity features, namely species, habitats, threatened species (referred to as Red Data List species) and endemics. It deals specifically with the classification of the study sites in terms of habitats, and the similarity grouping (ordination) of species and habitat assemblages comprising the Cape Flats flora. It also evaluates the differences between reserved and unreserved sites based on the number of biodiversity features and the sizes of sites. *Chapter 3* uses a multivariate modeling approach to determine the extent to which fragment size, habitat heterogeneity and threatening processes (as per a vulnerability index) control levels of species richness, and puts forward practical recommendations for preventing or minimising further species loss. *Chapter 4* focuses on the selection of priority sites for conservation based on species and habitats in relation to threatening processes and public/institutional support (defense). Data sets for 1997/1998 and 2001 are analysed separately using a step-wise, richness-based algorithm. Conservation priorities and approaches to implementation are discussed. The final chapter summarises the overall findings of the previous chapters, and closes with some personal and very general perspectives on how urban conservation might be successful in an area such as the Cape Flats.

Chapter 2

AN ACCOUNT OF THE FLORA OF THE CAPE FLATS: STUDY SITES, SPECIES AND HABITATS

Abstract

The amount and distribution of biodiversity features across 59 sites on the Cape Flats, and differences between conserved and unconserved sites, were addressed. The biodiversity features examined were all species, habitats, Red Data List species and endemics. A total of 1 192 vascular plant species, 47 (*ca* 4%) of which have appeared in Red Data Lists, were used in this study. The number of endemics strictly confined to the Cape Flats was found to be lower than previous accounts (*Erica verticillata, Euphorbia marlothiana, Isoetes stellenbossiensis, Serruria aemula* var. *congesta*, *S. aemula* var. *foeniculacea* and *S. furcellata*). Although all the strict endemics were recorded from protected areas, further efforts are required to resolve taxonomic anomalies and to clarify the true extent of their distribution outside reserved sites. A habitat classification system was developed, and 62 natural habitat types were recognised across remnant sites collectively totaling 10% of the area of Cape Flats. Heterogeneity, as per the number of habitats, was determined on the basis of unique combinations of four primary habitat variables and their subdivisions (soil, geology, relief and water-bodies). Species and habitat assemblages of the Cape Flats were examined using ordination techniques. Lastly, a 1-way ANOVA was used to test for differences between reserved and unreserved sites in terms of the number of biodiversity features and size of the sites. The overall findings of this chapter infer that the highest level of biodiversity on the Cape Flats appears to exist in the reserve estate.

AN ACCOUNT OF THE FLORA OF THE CAPE FLATS: STUDY SITES, SPECIES AND HABITATS

1. INTRODUCTION

While *Chapter 1* introduced the Cape Flats environment, this chapter introduces the study sites used in this thesis. The objective of this chapter is to describe the compositional elements of the Cape Flats flora, in terms of its species, habitats, Red Data List species (RDLs) and endemics, in order to gain clarity on the amount and distribution of these biodiversity features across the study sites. A classification system was developed for habitats, and this habitat system was applied elsewhere in the thesis, namely analysing how fragment size, threats and habitat heterogeneity control species richness (*Chapter 3*), and conservation planning (*Chapter 4*). A classification grouping and ordination, of species and habitat assemblages, was conducted across the study sites to understand more about the spatial arrangement of the biodiversity features of the Cape Flats. Lastly, differences between reserved and unreserved sites were tested for in terms of the number of biodiversity features (species, habitats, RDLs and endemics) and size of the sites. This provided a coarse assessment of the extent to which the reserve estate of the Cape Flats contributes to conserving plant biodiversity using the number of biodiversity features and the size of sites as indicators.

The key questions presented in this chapter thus relate to the amount and distribution of biodiversity features across the study sites, and differences between conserved and unconserved sites.

2. METHODS

2.1 SITE SELECTION

Study sites were selected on the basis of available information (reports and management plans) and whether they were generally regarded as being conservation-worthy (Figure 2.1, Appendix 1). These sites cover a total area of approximately 64.8 km^2, roughly 10% of the Cape Flats. The sizes of the sites varied between 1.3 ha and 1 242.6 ha.

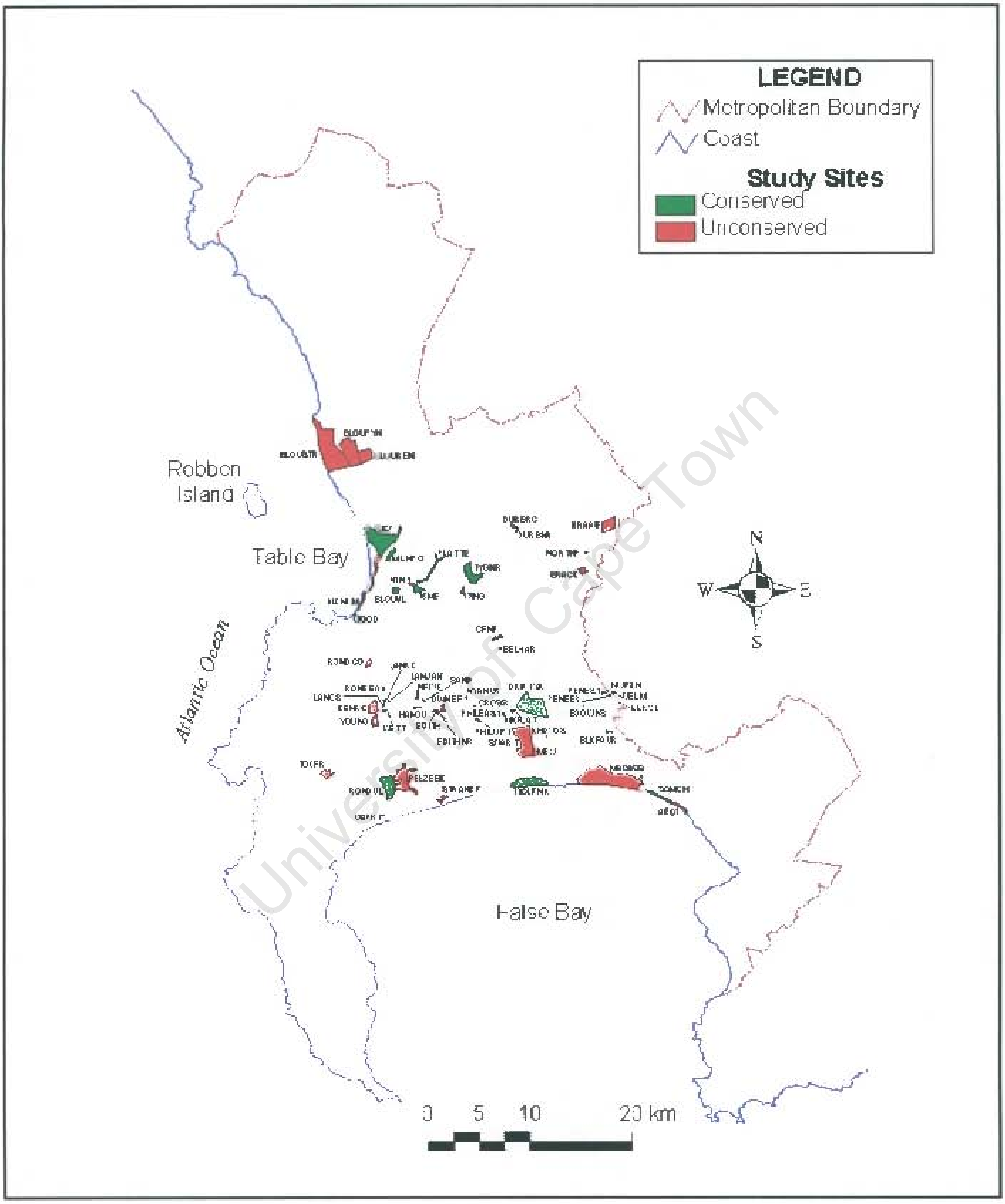

Figure 2.1 Study sites on the Cape Flats. Conserved sites are as at 1997/1998. Full names of the sites are given in Appendix 1.

Locally important open spaces, such as those of the Metropolitan Spatial Development Framework Corridor Project (Cape Metropolitan Council, 1996) and the City of Tygerberg Structure Plan (1997), that are earmarked for upgrading, and the 15 highly-recommended priority conservation areas of the Cape Flats (McKenzie & Rebelo, 1997) were included as study sites.

Sites that had not been previously studied or that were not part of any management or development plans, were not considered. Sites with formal and informal protection status, or pending, (Provincial and Local Reserve, Protected Natural Environment and Natural Heritage Site status) were referred to as conserved/reserved sites. Broadly, conserved areas were defined in terms of where a combination of legal and political mechanisms place restrictions on land thus preventing urban development indefinitely.

2.2 SPECIES DATABASE

Three species-level categories were used in the analyses as measures of richness: (i) all species; (ii) those referred to as target species, namely RDLs; and (iii) species endemic to the Cape Flats. According to guidelines established by the World Conservation Union (IUCN), species that are threatened or potentially threatened with extinction are referred to as RDLs (Mace & Lande, 1991; IUCN 2001).

Presence/absence data were collated from recent (1987-1998) reports, management plans, unpublished material and checklists from a variety of sources. Lists were generally compiled during favourable seasons when annuals and bulbs were flowering. In cases where more than one species list existed for a site, new records were noted.

The nomenclature of species lists was cross-checked using Arnold & De Wet (1993) and species localities were confirmed using PRECIS (Pretoria National Herbarium Computerised Information System, National Botanical Institute, 1998), a frequently updated national database of botanical collection records. Plants which were classified as invasive, exotic or naturalised (Wells *et al.*, 1986; Henderson, 1995; Bromilow, 1997) were rejected, and thus only indigenous species were used in this study.

Table 2.1 Red Data List species (after Hilton-Taylor, 1996a) recorded on the Cape Flats. V=Vulnerable, E=Endangered, I=Indeterminate and X=Extinct. * denotes endemic to the Cape Flats (Bond & Goldblatt, 1984; Goldblatt & Manning, 2000). CF=Cape Flats, CP=Cape Peninsula and WCLs=West Coast Lowlands.

SPECIES	RED DATA LIST STATUS	DISTRIBUTION
Agathosma corymbosa	E	extends to CP and WCLs
Aspalathus areneosa	R	extends to CP and WCLs
Aspalathus capensis	I	extends to CP
Cliffortia ericifolia	E	extends to CP and WCLs
Cliffortia hirta	I	extends to CP and WCLs
Cotula myriophylloides	I	extends to CP
Cynanchum zeyheri	V	extends to CP and beyond the eastern CF
Diosma dichotoma	V	extends to CP and WCLs
Elegia verreauxii	V	extends to CP and beyond the eastern CF
*Erepsia dunensis**	E	extends to CP
*Erica margaritacea**	E	extends to CP
Erica mauritanica	I	extends to CP and beyond the eastern CF
*Erica verticillata**	X	CF
*Euphorbia marlothiana**	E	CF
*Ficinia fastigiata**	I	extends to CP
Geissorhiza bonaspei	V	extends to CP
Geissorhiza pusilla	I	extends to CP and beyond the eastern CF
Haemanthus pumilio	E	extends beyond the eastern CF
Hermannia rugosa	V	extends to CP and WCLs
Ischyrolepis sabulosa	E	extends to CP and beyond the eastern CF
*Isoetes capensis var. stephensenii**	I	extends to CP and beyond the eastern CF
*Isoetes stellenbossiensis**	I	CF
Ixia monodelpha	I	extends to CP and WCLs
Lachenalia liliflora	I	extends beyond the eastern CF
Lampranthus altistylus	E	extends to CP and WCLs
Lampranthus sociorum	E	extends to WCLs
Lampranthus stenus	R	extends to CP
*Leucadendron floridum**	E	extends to CP and beyond the eastern CF
Leucadendron levisanus	E	extends to WCLs
Leucadendron thymifolium	E	extends to WCLs
Leucadendron verticillatum	E	extends beyond the eastern CF
Leucospermum grandiflorum	V	extends beyond the eastern CF
Lobelia limosa	I	extends to CP and beyond the eastern CF
Muraltia satureioides var salteri	E	extends to CP and beyond the eastern CF
*Otholobium fruticans**	V	extends to CP
*Passerina paludosa**	E	extends to CP
Protea odorata	E	extends to WCLs
Psoralea glaucina	E	extends to CP
Restio micans	E	extends to CP and WCLs
*Serruria aemula var. congesta**	E	CF
*Serruria aemula var. foeniculacea**	E	CF
*Serruria furcellata**	X?	CF
Serruria trilopha	E	extends to CP and WCLs
Trachyandra brachypoda	I	extends to CP and WCLs
*Trianoptiles solitaria**	I	extends to CP
Watsonia tabularis	I	extends to CP
*Wurmbea hiemalis**	I	extends to CP

There were several instances in the database where taxonomy and distribution ranges were spurious such as for *Isoetes* (J. Burrows, R. Glen, 2001, *pers. comm.*), *Serruria furcellata* (considered extinct by Hilton-Taylor (1996a)) and the *Serruria aemula* complex (J. Rourke, A.G. Rebelo, 1999, *pers. comm.*). *Erica verticillata* is suspected to be extinct in the wild and is currently known only from a reserve to which it was translocated (Hilton-Taylor, 1996a). *E. turgida*, known to be extinct (Hilton-Taylor, 1996a), is not considered in these analyses.

Thirty-two species considered globally threatened (categorised as Endangered, Rare, or Vulnerable) (in Hilton-Taylor, 1996a) occur on the Cape Flats. The remaining 17 species, categorised as Indeterminate on the Red Data List, are suspected to be threatened but current information is insufficient to decide which threatened category is appropriate (Hilton-Taylor, 1996a; 1996b, 1997).

Fifteen species confined to the (i) Cape Flats (*sensu stricto*) or (ii) found on the adjacent Cape Peninsula but with the greatest proportion of their range on the Cape Flats or (iii) primarily found on the Cape Flats but extending further north or east into adjacent areas, were identified as endemics (Bond & Goldblatt, 1984 but updated using Goldblatt & Manning, 2000) (Table 2.1). All these endemics are RDLs. Forty-seven species in the database appear on the Red Data List (Hilton-Taylor, 1996a). The exception is *Euphorbia marlothiana,* considered a Cape Flats endemic (Bond & Goldblatt, 1984) and a *bona fide* species (Arnold & de Wet, 1993), but which was not listed in Goldblatt & Manning (2000). Because *E. marlothiana* was recorded from several study sites, it was included in the analyses.

The species database was therefore considered reasonably complete (i.e. all available data were included). A total of 1 192 indigenous vascular species (terrestrial and aquatic) were enumerated.

2.3 HABITAT CLASSIFICATION

Natural habitat types were identified on the study sites on the basis of four primary variables (soil, geology, water-bodies and relief) (Figure 2.2). These variables and their subdivisions are well understood as the primary determinants of fine-scale species pattern on the lowlands of the Cape Floristic Region (Boucher, 1987; Cowling & Holmes, 1992; Fredericks, 1992; Simmons & Cowling, 1996; Privett, 1998; Cowling & Heijnis, 2001), and this is broadly explained in *Chapter 1* in terms of the biophysical environment of the Cape Flats.

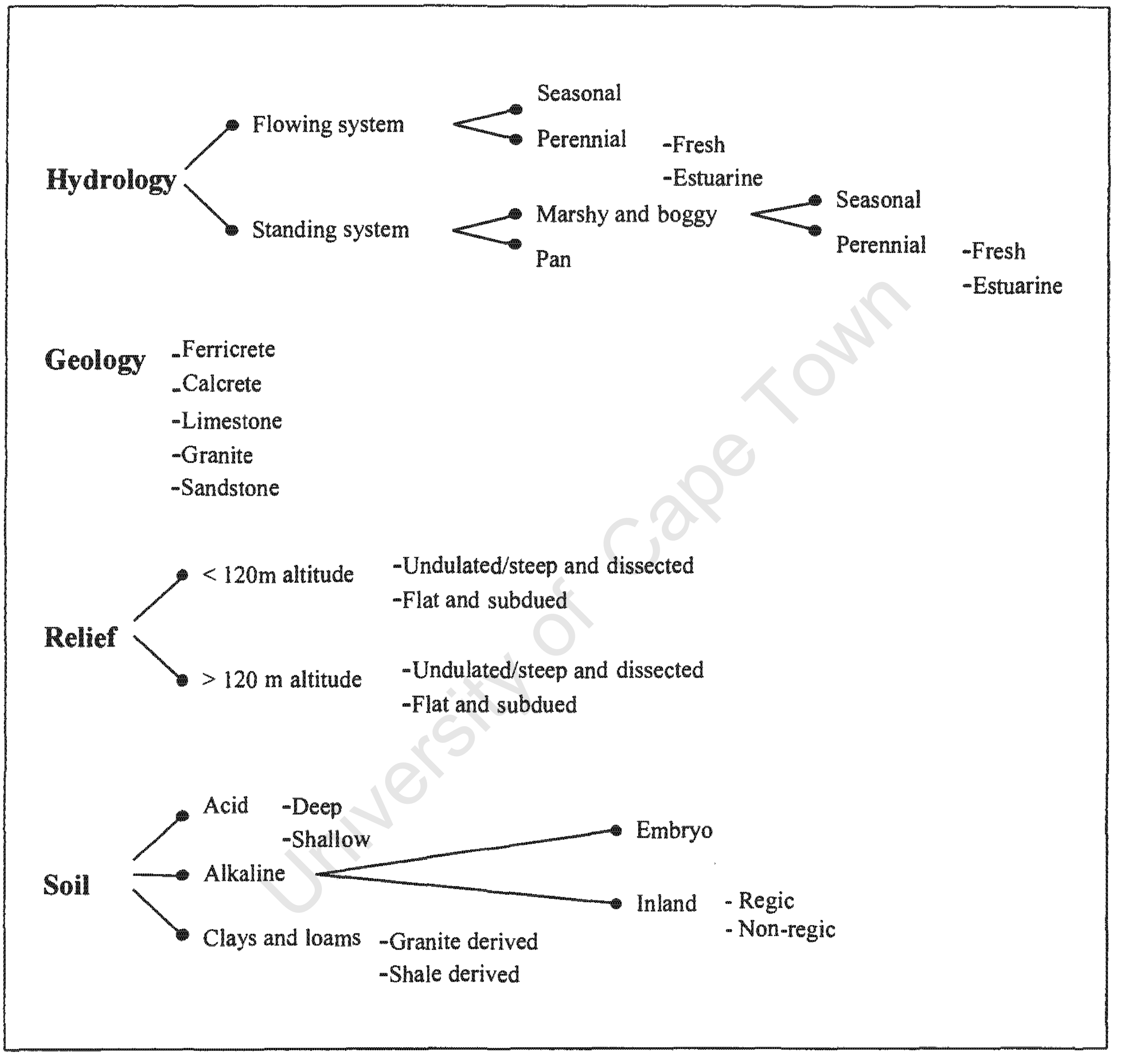

Figure 2.2 Categories of hydrology, geology, relief and soil, together with their sub-categories, were used to develop the 62 habitats. A unique combination of features as described by the categories (**bold script**) and their sub-categories defines a habitat.

The habitats were 'ground-truthed' during surveys of the study sites in 1997-1998 and their reliability confirmed with individuals who have extensive field experience of the study area (B. Low, D. Gibbs, C. McDowell & L.Raitt, *pers. comm.*, 1998-1999). Each habitat is thus defined as the unique combination of the variables and their subdivisions (Appendix 2).

A total of 62 unique combinations were recognised (Appendix 2). It is important to note that this habitat classification system is composed at a scale of finer than 1: 10 000 (see Cowling & Heijnis, 2001; Maddock & Benn, *in prep.*). It is applied elsewhere in this thesis (used a measure of or surrogate for heterogeneity), namely in the analysis of ecological processes (*Chapter 3*) and conservation planning (*Chapter 4*).

2.4 SPECIES AND HABITAT ASSEMBLAGES

Species and habitat richness were recorded for each of the 59 sites of the study (1 192 species and 62 habitats). The frequency distribution of species and habitats were first examined. Site similarity analyses using species and habitat presence/absence data were performed using the CLUSTER and MDS programs in the computer package, PRIMER (Plymouth Routines in Multivariate Ecological Research). Dendrograms were used to explore the presence of various species and habitat assemblages across the study area. Since dendrograms suffer certain limitations (outlined in Field *et al.*, 1982), the similarity analyses were also displayed in a two dimensional ordination plot using non-metric multidimensional scaling (MDS).

This method is considered the preferred ordination technique for assemblage data (Clarke & Warwick, 1994a). Where appropriate, data were transformed (Euclidean distance, complete/squared linkage) to moderate for the effects of species and habitats with a high presence (extremely common) and a low absence (extremely rare) so as not to skew the results. A stress function was calculated to assess how well the 2-dimensional plot represents the actual site relationships. The lower the value, the more accurate the representation of the sites in the ordination (Clarke & Warwick, 1994b).

2.5 COMPARISON BETWEEN RESERVED AND UNRESERVED SITES

I tested the null hypothesis that there is no significant difference in the size of sites (ha) that are reserved and sites that are not protected. I also tested the null hypothesis that there is no difference in the number of biodiversity features on reserved and unreserved sites; tests were done for the number of total species, endemics, RDLs and habitats. A 1-way analysis of variance (ANOVA) (Zar, 1984) was used for both these tests. In the latter, size was used as a covariable since there is likely to be a relationship between feature richness and size of the site (albeit weak across all sites, see *Chapter 3*).

The sample sizes used was n=14 for reserves and n=45 for unprotected sites. However, in preliminary tests for feature richness, Brackenfell Granite outcrop (BRACK) emerged as an outlier (size of 1.6 ha; total number of species = 88), and was excluded in the analysis. Because the sizes of sites vary between 1.3 ha and 1 242.6 ha and counts of biodiversity features are very different, this inherently violates the assumption of the 1-way ANOVA (Zar, 1984). This was clearly reflected in the result output where all the tests were significantly different but associated with high coefficients of variation (Table 2.2).

Table 2.2 Outputs of the 1-way ANOVA (with site size as a co-variable) of average log counts for biodiversity features. CV=Coefficient of variation for all sites. Means are back-transformed. Sites conserved (n=14); sites unconserved (n=44).

OUTPUT	ALL SPECIES	RDLS	ENDEMICS	HABITAT
Conserved	Mean=3.47	Mean=0.98	Mean=0.29	Mean=1.06
Unconserved	Mean=4.7[illegible]	Mean=1.77	Mean=0.91	Mean=1.52
F statistic	18.48 (p<0.001)	11.89 (p=0.001)	18.16 (p<0.001)	16.4 (p<0.001)
CV %	26.0	63.8	105.9	31.1

Furthermore, the results were exacerbated by the big difference in sample size between conserved and unconserved sites. This prompted further tests to shed more light on differences between reserved and unreserved sites. As such, differences in the density of features (number of biodiversity features per hectare) between sites were tested for, also using a 1-way ANOVA. This allowed for an analysis of biodiversity features between sites that excluded the consideration of the size of sites. Data were log-transformed in all the tests.

3. RESULTS

3.1 BIODIVERSITY FEATURES OF SITES

A total of 47 RDLs out of 1 192 species are known from the 59 study sites (*ca* 4%) (Table 2.1). Of these, six species are strictly confined to the Cape Flats, and are currently known not to occur further afield (*Erica verticillata, Euphorbia marlothiana, Isoetes stellenbossiensis, Serruria aemula* var. *congesta, S. aemula* var. *foeniculacea* and *S. furcellata*). All these endemics are represented in protected areas on the Cape Flats.

All the reserves have at least one record of the occurrence of an endemic species and a RDL except for Rietvlei Protected Area (RIET) that has neither one. The latter is important for it's high number of aquatic habitats (eight) (Appendix 2). Some 50% of the species recorded had a single occurrence (Figure 2.3); 36 habitats (58%) had a single occurrence across the study sites and the remainder had two to 15 occurrences (Figure 2.4). Habitat 6, Habitat 19 and Habitat 20, all which were not associated with geological substrates, were the most common habitats (i.e. found on more than 10 sites) (see Appendix 2).

3.2 SPECIES AND HABITAT ASSEMBLAGES

The cluster analysis for habitat similarity (Euclidean distance, squared linkage) showed that Cluster I (linkage distance 8) separates sites with alkaline sand elements (Figure 2.5). The outlier at the highest linkage distance (16) was Blouberg Renosterveld koppie site (BLOUREN), and the next highest was Tygerberg Nature Reserve (TYGNR), both Renosterveld sites with clay soils (Figure 1.2, 2.1) (see Wood & Low, 1993) (Appendix 2). Blouberg Fynbos (BLOUFYN), Durbanville Nature Reserve (DURBNR), Durbanville Racecourse (DURBRC), Milnerton Racecourse (MILNRC) and Bleak House-Faure road verge (BLKFAUR) are also sites bearing Renosterveld elements. The remainder of the tree (cluster II) appears to consist of sites with acid sands and/or clay and loams.

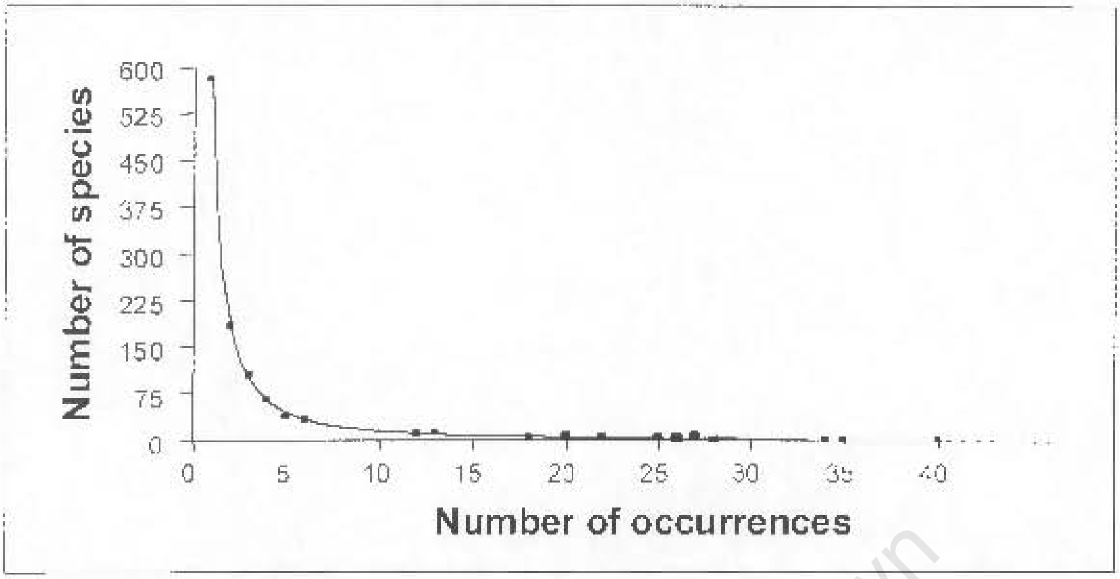

Figure 2.3 Frequency distribution of species occurrences across the study sites.

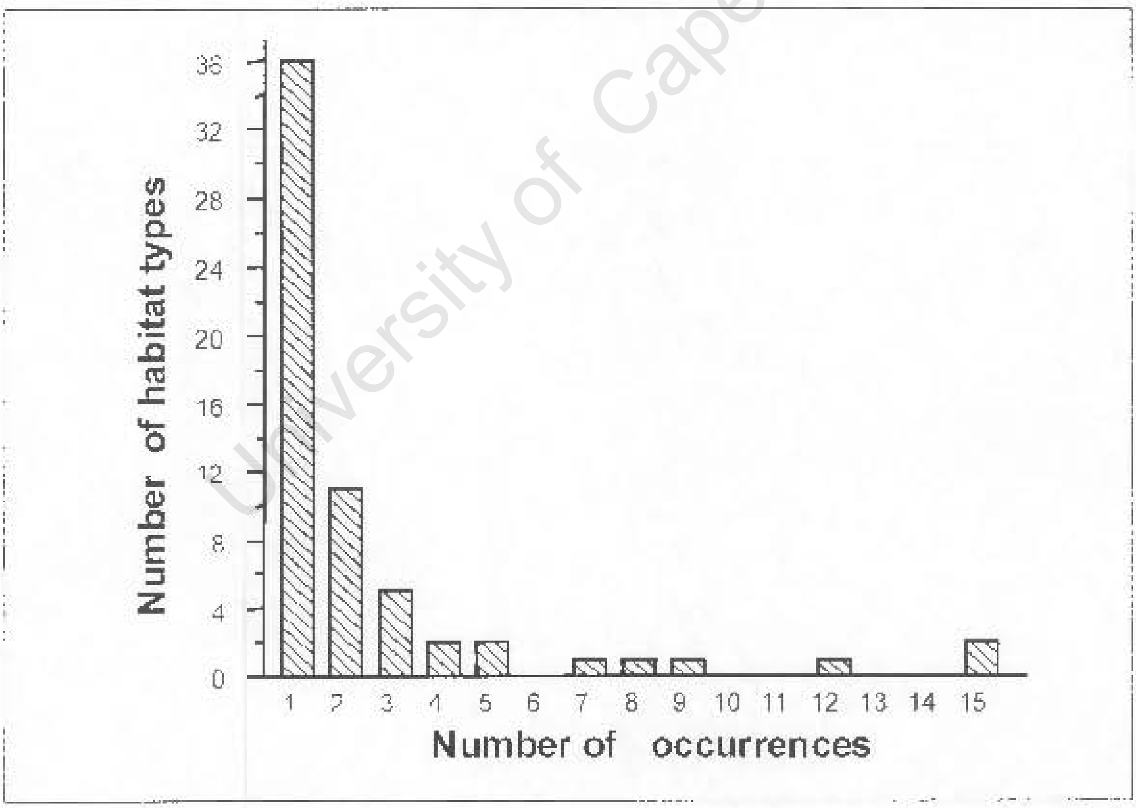

Figure 2.4 Frequency distribution of habitat occurrences across the study sites.

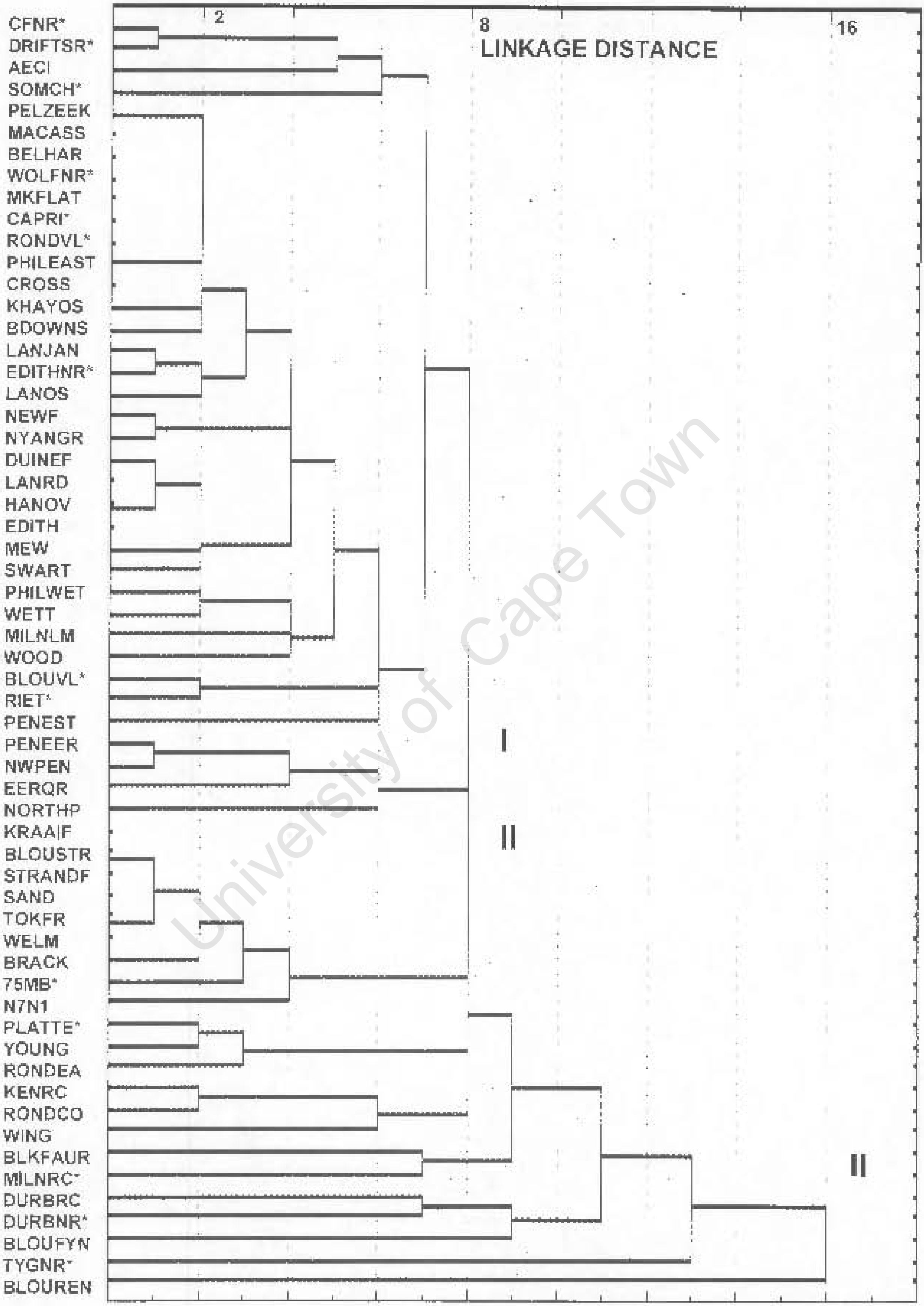

Figure 2.5 Tree diagram of habitat assemblages across the study sites (see Appendix 1 for the abbreviated site names). I and II represent cluster groups as discussed in the text. * denotes a protected area.

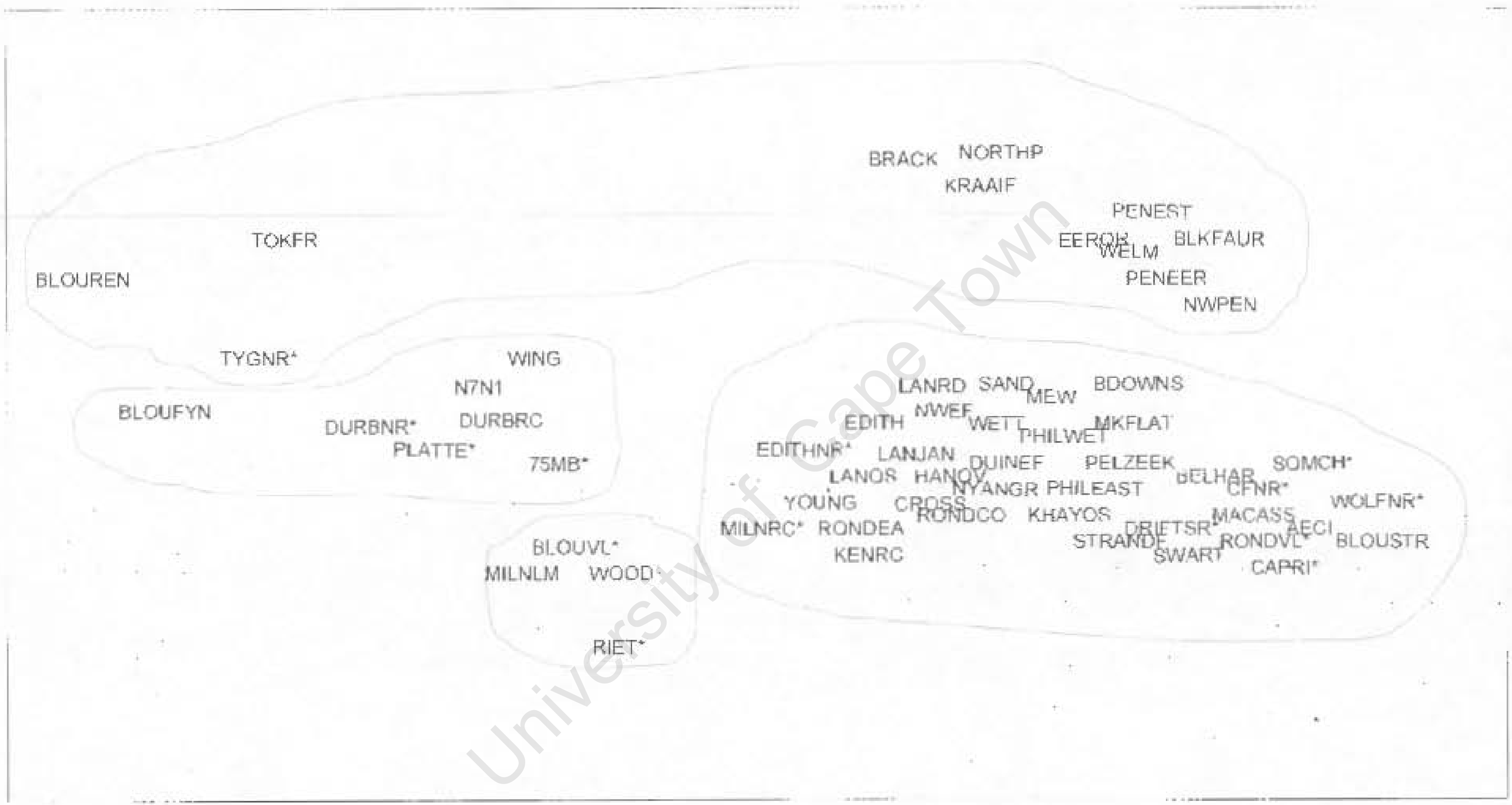

Figure 2.6 Natural groupings for a 2-dimensional ordination for habitats using multidimensional scaling (MDS). * denotes a protected area. Appendix 1 gives abbreviated site names.

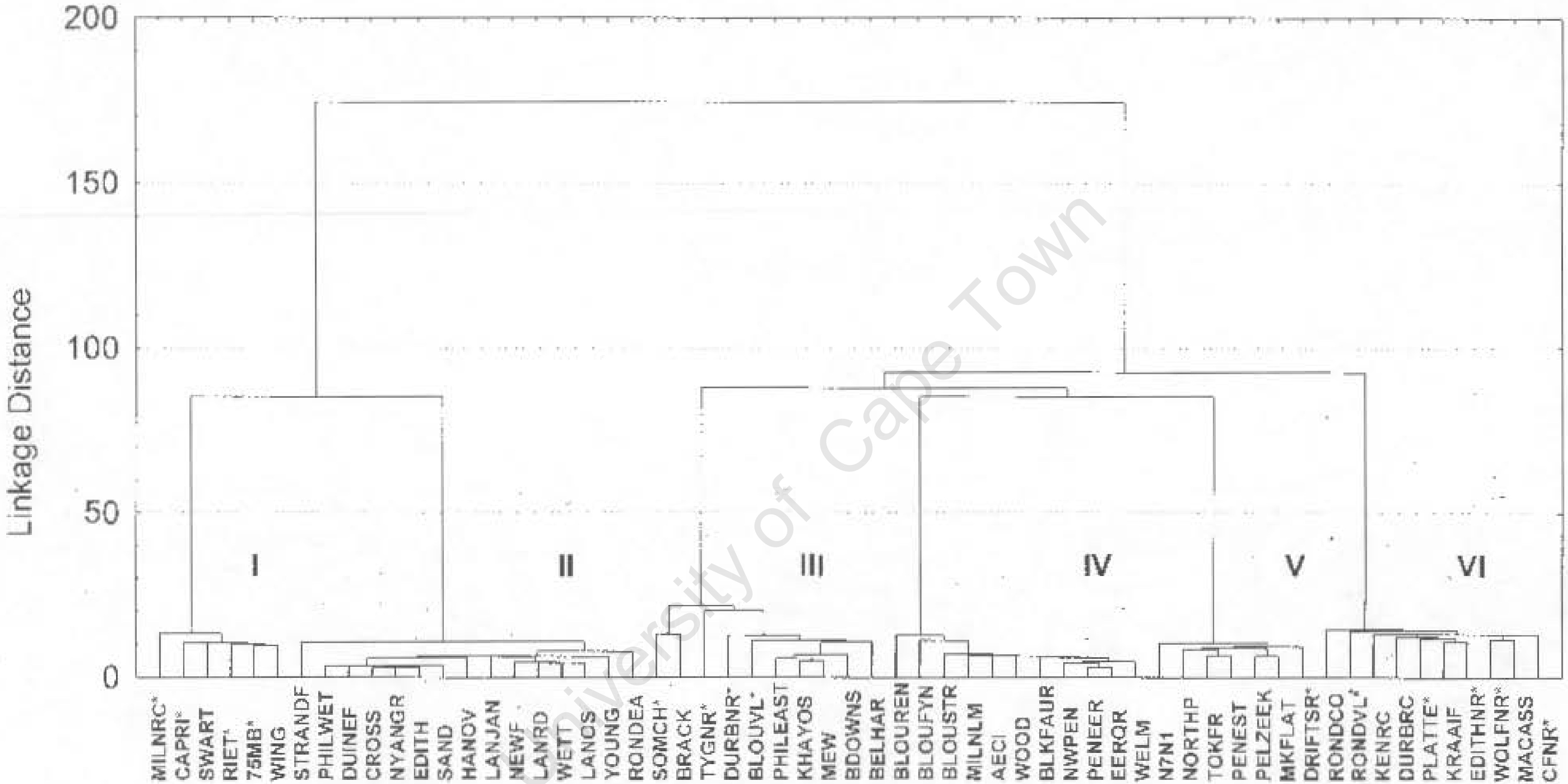

Figure 2.7 Tree diagram of species assemblages across the study sites (see Appendix 1 for the abbreviated site names). I-VI represent cluster groups as discussed in the text. * denotes a protected area.

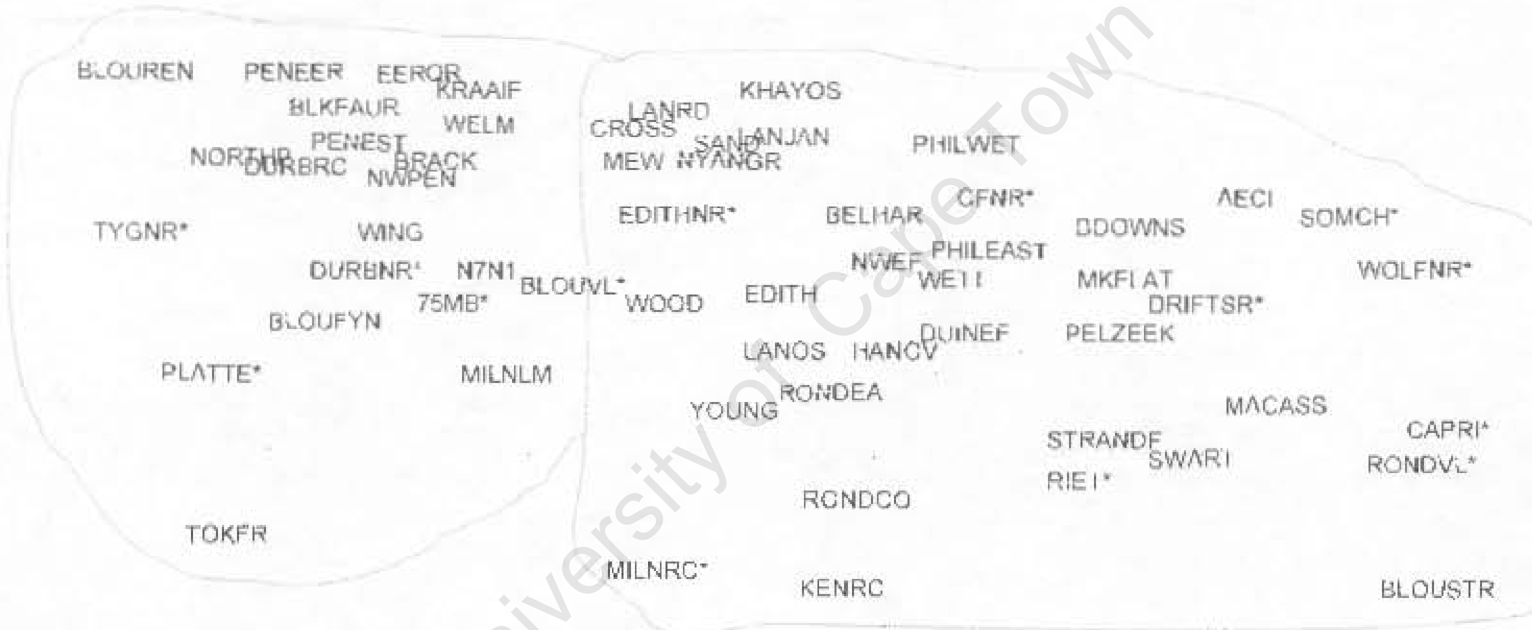

Figure 2.8 Natural groupings for a 2-dimensional ordination for species using multidimensional scaling (MDS). * denotes a protected area. Appendix 1 gives abbreviated site names.

These findings are consistent with the MDS (Figure 2.6) which delimited four groups of sites based on habitat assemblages: acid sands, alkaline sands, a super-group consisting of clays/loams and wetland habitats overlain by acid/alkaline sands. The stress-value (=0.05) of this MDS indicates that the ordination produced has no real chance of yielding a misleading interpretation (Clarke & Warwick, 1994b).

Two distinct groups are presented in the species dendrogram (Euclidean distance, complete linkage). At this high linkage level the results were not intuitively appealing, but were more interesting at relatively low linkage distances (*ca* distance 12-22, clusters I-VI) (Figure 2.7).

Sites belonging to clusters II and V are generally degraded areas (see McDowell & Low, 1990; Cape Metropolitan Council, 1996; City of Tygerberg Structure Plan, 1997; McKenzie & Rebelo, 1997). It appears that the analysis has delineated these degraded sites on the basis of the ubiquitous nature of ephemeral (common and widespread) species. Milnerton Racecourse (MILNRC) is a distinct entity in cluster I; the same can be said for Tygerberg Nature Reserve (TYGNR) (cluster III), whereas Rondebosch Common (RONDCO) and Rondevlei Nature Reserve (RONDVL) (cluster VI), and Blouberg Fynbos (BLOUFYN) and Blouberg Renosterveld koppie site (BLOUREN) (cluster IV) form distinct outliers in their group. No obvious pattern regarding the MDS for species assemblages except where outliers were discernable (Figure 2.8, stress-value=0.196).

3.3 COMPARISON BETWEEN RESERVED AND UNRESERVED SITES

No significant difference was found between the size of protected and unprotected areas ($F=1.33$, $p=0.254$, CV%=41.3). The average number of total species, endemics, RDLs and habitats was significantly higher on reserved sites than on unreserved sites, but the values of the coefficient of variation (CV%) were high in all cases (Table 2.2). However, the average density of habitats and RDLs respectively was not statistically significant between reserved and unreserved sites, but was highly significant for all species and endemics (Table 2.3). Again, the coefficient of variation was high in all cases (124.9% - 171.2%).

Table 2.3 Outputs of the 1-way ANOVA of average density of biodiversity features (log transformed). Means are back-transformed. CV=Coefficient of variation for all the sites. ns=not significant. Sites conserved (n=14), unconserved (n=45).

OUTPUT	ALL SPECIES	RDLS	ENDEMICS	HABITAT
Conserved	Mean=5.56	Mean=0.26	Mean=0.09	Mean=0.18
Unconserved	Mean=2.21	Mean=0.13	Mean=0.01	Mean=0.12
F-values	8.48 ($p<0.005$)	2.76 ($p=0.10^{ns}$)	22.6 ($p<0.001$)	0.92 ($p<0.34^{ns}$)
CV%	124.9	151.0	171.2	144.8

4. DISCUSSION

Some 1 800 plant species of the Cape Flora are known or suspected to be threatened with extinction (Rebelo, 2002). This chapter has found the presence of RDLs on the Cape Flats to be *ca* 4% of the total number of species recorded in this study (1 192 species), and *ca* 3% of the total number of species recorded in the Cape Flora that are known or thought to be threatened (47/1 800 species). The number of endemics strictly (six) confined to the Cape Flats are *Erica verticillata, Euphorbia marlothiana, Isoetes stellenbossiensis, Serruria aemula* var. *congesta, S. aemula* var. *foeniculacea* and *S. furcellata.* These species, which are also RDLs, are currently known only from reserves and are not known to occur outside the study area. Compared to the Durban Metropole (KwaZulu-Natal Province), a similar-sized urban centre in South Africa, the numbers of strict endemics and RDLs of the Cape Flats are at least two to three times lower (see Scott-Shaw, 1999).

Goldblatt & Manning (2000) enumerate a total of *ca* 9 000 plant species occurring in the Cape Flora. Some 2 250 plant species occur in the area their study delimits as the Cape Peninsula, with endemism at 7.5% (size is 4 700 km^2). As such, roughly half the number of species recorded on the Cape Peninsula is recorded on the Cape Flats, and *ca* 2% of the Cape Peninsula's flora consists of RDLs recorded from the Cape Flats. Of concern, therefore, is that the number of endemics strictly confined to the Cape Flats is much lower than previously reported (McDowell & Low, 1990; Wood *et al.*, 1994; McKenzie & Rebelo, 1997). This, together with the high number of species recorded on remnants of the Cape Flats (1 192) and the level of heterogeneity (number of habitats=62), does not provide great evidence that the Cape Flats is a 'centre of neoendemism' as has been alluded by several conservationists. Quoting from Wood *et al.* (1994) "The Cape Peninsula, the Cape Flats and the Darling/Dassenberg area have been identified as three distinct centres of diversity and endemism (Daines & Low, 1993; McDowell et al., 1991; McDowell & Low, 1990; Rebelo, 1992; Rebelo & Tansley, 1992)".

The figures obtained in this chapter are substantially lower than those calculated by Wood *et al.* (1994) who based their figures of endemicity (*ca* 30 endemics) and threat (82 RDLs) on estimates that were perpetuated from figures contained in McDowell & Low (1990). A first reason for the discrepancy in the number of threatened/endemic species lies in the taxonomic treatment of species that were listed in Red Data Lists of the 1980s (Hall *et al.*, 1980; Hall & Ashton, 1983; Hall & Veldhuis, 1985). There are a large number of species listed as threatened or extinct in these early Red Data Lists (these were used as information sources by McDowell & Low (1990) and Wood *et al.* (1994)), but in later Red Data Lists (Hilton-Taylor, 1996a, b; 1997), were not listed as threatened due to taxonomic revisions (Arnold & de Wet, 1993) (J. Golding, *unpublished data).* For example, *Urginea duthiae* Adamson and *U. ecklonii* Bak., simultaneously listed as extinct in the 1980s, were subsequently found to be synonyms of each other (see Arnold & De Wet, 1993), and categorised as not being threatened in Hilton-Taylor (1996a). Additional examples include *Staavia trichotoma* (Thunb.)Pillans, *Corycium vestitum* Sweet, *Satyrium guthriei* Bol., *Caralluma arenicola* N.E.Br., *Leptocarpus ramossisum* Pillans, amongst many others (see Arnold & de Wet, 1993). The figures in this chapter also differ to those quoted in Maze & Rebelo (1999). This brings about a second explanation for the discrepancy, namely that the study area of Maze & Rebelo (1999) is almost three times the size of the Cape Flats (i.e. they viewed endemicity of the Cape Flats in terms of endemicity to the entire lowlands of the Cape Metropolitan Area). Taxonomic changes and the geographic scope of the Cape Flats therefore appear to be significant factors for determining the threatened and endemic species.

There are deficiencies in knowledge on the abundance and distribution range of key species such as endemics and RDLs, including *Serruria furcellata*, a species last recorded from Northpine open space (NORTHP) but suspected of being extinct or known from only a few surviving individual plants. Another concern is unresolved taxonomic relationships (i.e. the identification of 'real' species) (see Goldblatt & Manning, 2000). One can start addressing all these concerns by consulting the relevant taxonomic authorities, databases, literature sources, undertaking field surveys, recording the findings of surveys, and most importantly, clearly defining the scale and extent (i.e. boundaries) at which the level of endemicity is being evaluated. The implication of failing to address these concerns can result in many important target species, and those as yet unknown to science, to 'fall through the net' either by being overlooked in conservation planning processes or as not being identified as candidates for *ex situ* conservation initiatives like propagation or relocation to safer sites.

4.1 SPECIES AND HABITAT ASSEMBLAGES

It is important to note that the habitat classification system developed in this chapter was specifically for the purposes of the objectives of this thesis. This system was used as surrogate for heterogeneity and also allowed for its quantification, and examples of its utility are presented in the ensuing chapters.

Site similarities based on habitat assemblages produced more robust results than species assemblages due to the fact that habitats, in particular edaphic gradients, are well-defined on the Cape Flats (Boucher, 1987; Cowling & Holmes, 1992; Fredericks, 1992; Simmons & Cowling, 1996; Privett, 1998). Sites occurring on Strandveld, Sand Plain Fynbos and West Coast Renosterveld are clearly delimited. The distinction of sites based on species assemblages are blurred, suggesting the presence of ecotonal plant communities that mostly consist of common and widespread species. Virtually all the sites of the two extremities in the species dendrogram (cluster I and VI, Figure 2.7) are protected areas that have been defined as core conservation sites by McKenzie & Rebelo (1997). This, together with the low stress-value associated with the ordination, may indicate that common, widespread or ephemeral species (e.g. *Metalasia muricata* or *Zantedeschia aethiopica*) are most prevalent on sites that are not protected, and by inference, more highly disturbed.

4.2 RESERVED AND UNRESERVED SITES

The statistical method used to establish a difference between the sizes of protected and unprotected areas found no significant difference. This result is unexpected because historically, recommendations for identifying conservation-worthy areas on the Cape Flats have been based on the size of a site, amongst other factors (e.g. Jarman, 1986; Wood *et al.*, 1994).

The average number of species, endemics, RDLs and habitats was shown to be significantly higher on reserve sites than on unreserved site, but the high values of the coefficient of variation produced a high degree of generality (Table 2.2). The reason for this is that the statistical method used (introduced site size as a co-variable) was unable to moderate for the copious effect of size (Zar, 1984). The density of species and endemics was significantly higher on reserves compared to sites that are not protected (Table 2.3). This may indicate that on average, communities of species (clusters or 'flocks' of species) are more prevalent on protected areas, and that assemblages on unprotected sites may have suffered losses of species, possibly as a result of threatening processes. An alternative explanation for the lack of congruence in species density between protected and unprotected land may be the result of reserves being established in areas with highest richness, historically also a key criterion for identifying conservation-worthy areas on the Cape Flats (Jarman, 1986; Wood *et al.*, 1994).

No statistical difference was found in the density of habitats and RDLs between protected and unprotected areas (Table 2.3). The implication of this finding is that unprotected areas, possibly face more intensive levels of threat than protected areas, and as a result, generally harbour equally high densities of RDLs and habitats. The results also suggest that the level of habitat loss and has been more severe than the loss of species richness (i.e. there is a difference for RDLs but no difference for endemics between protected and unprotected sites), and that this is most pronounced on unprotected areas. The overall findings thus infer that the highest level of biodiversity on the Cape Flats appears to exist in protected areas.

The following chapter addresses species decline, and focuses on which factors – habitat heterogeneity, remnant size and threats – are principally responsible for controlling species richness on the Cape Flats.

Chapter 3

RELATIONSHIPS BETWEEN SPECIES RICHNESS, FRAGMENT SIZE, HETEROGENEITY AND EXPOSURE TO THREATENING PROCESSES IN NATURAL AND SEMI-NATURAL HABITAT ON THE CAPE FLATS

Abstract

Loss of habitat heterogeneity, threatening processes and fragment size are inextricably linked factors that have been widely attributed to declines in species richness on the Cape Flats. Uncertainty remains as to the function of these variables in determining species richness, and the extent to which they operate in relation to each other. A scoring technique that integrated a range of threatening processes relating to biological, landuse and defense factors that currently prevail on 59 study sites on the Cape Flats, was used to quantify a vulnerability index for each site. A large proportion of the sites (48%) is subjected to fairly intense to extreme threatening processes (V-index >13). Heterogeneity, size and threats were found to be poor independent predictors of richness. However, an ecological model developed using multiple linear regression techniques suggests that heterogeneity and threats together are the critical limiting factors for maintaining species richness on the Cape Flats. This model loses explanatory power for species susceptible to extinction (Red Data List species) and range restricted species recorded on the Cape Flats. Stochastic factors associated with small population size on small-sized remnants appear to be far less important in relation to heterogeneity and threats. The sizes of sites on the Cape Flats varied between 1.3 ha and 1 242.6 ha. Conservation implications are that small-sized fragments should not be precluded protection status based on their size since they can make a valuable contribution to conservation. Management emphasis should be placed on retaining factors for maintaining diversity such as implementation of appropriate fire regimes, alien plant eradication and support from civil society. The use of abundance data will strengthen the predictive value of future extinction trajectories and provide more insight as to whether the Cape Flats flora is in a state of species decline.

RELATIONSHIPS BETWEEN SPECIES RICHNESS, FRAGMENT SIZE, HETEROGENEITY AND EXPOSURE TO THREATENING PROCESSES IN NATURAL AND SEMI-NATURAL HABITAT ON THE CAPE FLATS

1. INTRODUCTION

Richness is determined by a number of factors associated with natural and anthropogenic processes (e.g. Whittaker, 1977; Rosenzweig, 1995; Richardson *et al.*, 1996). Human-induced threats are difficult to measure and predict since they are driven by social, economic and political factors (Ehrlich, 1986; Myers, 1993; Matowanyika, 1994; Folke *et al.*, 1996). There is a large body of theory showing that fragmentation effects, through the erosion of heterogeneity and reduction in habitat size (MacArthur & Wilson, 1967; Connor & McCoy, 1979; Brown & Dinsmore, 1988; Lande, 1988; Rosenzweig, 1995), are consequences of human activities (Saunders *et al.*, 1991). Natural disturbances refer specifically to changes in ecological processes that are essential for maintaining richness, such as appropriate fire regimes in mediterranean-climate regions (Bond & van Wilgen, 1996; Bond, 1998). As threatening processes mount, they may become non-linear (synergistic) causing cascading losses (Richardson *et al.*, 1996). It is therefore important to gain a predictive understanding of these processes so that extinctions can be anticipated, and therefore minimised or prevented.

In conservation ecology, habitat size has been traditionally regarded as a primary variable in influencing the persistence of species (Saunders *et al.*, 1991; Reid, 1992). The literature is replete on studies that suggest, but less commonly show, that small fragments, especially when embedded in a matrix of transformed land, are vulnerable to losses as a result of threatening processes (Ehrlich, 1986; Shafer, 1994; Walker, 1995). Island theory applied to these landscapes, predicts that through fragmentation processes, the role of size *per se* is influential in that smaller fragments experience greater population declines than larger ones (Saunders *et al.*, 1991).

There are mainly three hypotheses that predict species loss on isolated habitat fragments, namely those associated with (i) stochastic factors, (ii) deterministic factors and (iii) heterogeneity. Island theory predicts that species loss is due to stochastic small-population effects like inbreeding depression and loss of heterozygosity. It invokes that minimum viable populations (MVPs) and minimum critical areas are important factors which determine extinction thresholds (Lande, 1988; Caughley, 1994). As such, size is often considered a non-deterministic factor, since it is assumed that species are lost as a result of stochastic factors. Loss rates are reflected in the classic species-area relationship described by MacArthur & Wilson (1967). Alternatively, smaller-sized remnants may be more vulnerable to changes in disturbance regimes that negatively affect richness (i.e. deterministic factors that cause species decline). For example, they may experience different fire regimes (Bond *et al.*, 1988) or be more exposed to grazing impacts (Kemper *et al.*, 1999). Of critical importance is the life-history strategies of species which confer an ability to persist (e.g. sprouting ability) in relation to altered fire regimes (Bond & van Wilgen, 1996). Finally, the size effects on diversity in fragmented landscapes may actually be a consequence of heterogeneity (Cowling *et al.*, 1992). Since habitat area and habitat heterogeneity are usually strongly and positively correlated, (Rosenzweig, 1995), small fragments may support fewer species as a result of lower heterogeneity (Walker, 1995; Venier & Fahrieg, 1996).

The Cape Flats has exceptional plant diversity confined to largely isolated remnants (>1 ha) in a generally low-income urban landscape. (Low, 1979; Hall, 1984; Low & McKenzie, 1988; McDowell & Low, 1990; McDowell *et al.*, 1991; Wood *et al.*, 1994). There is uncertainty whether size, habitat heterogeneity and human-induced threats play a role in determining levels of richness on the Cape Flats, and the extent to which these variables operate in relation to each other. The debate stems mainly from three assumptions: i) Conservation practitioners perceive that habitat losses and conversions lead to species losses, which in turn give rise to increasing numbers of threatened plant species (McDowell *et al.*, 1991; Rebelo & Tansley, 1993; Hilton-Taylor, 1996a; Trinder-Smith *et al.*, 1996a); ii) The size of sites is an indicator or an approximation of species richness; (Jarman, 1986; McDowell & Low, 1990) and iii) Disturbance-impacts associated with the urban environment, have detrimental effects on richness (Wood *et al.*, 1994). However, none of these assumptions have been scrutinised. It is important to gain clarification as they may influence approaches and methodologies in addressing conservation priorities on the Cape Flats.

A great deal of research has been conducted on the Cape Flats where the conditions leading to fragmentation effects differ extensively from 'mainland' (large, undisturbed and contiguous tracts of land regarded as a source area) and rural (where fragments are surrounded by agricultural landscapes) areas. In general, urban landscapes present a unique and interesting opportunity to monitor the effects of a myriad disturbance impacts on remnant patches, ranging from ecological to the practices espoused from policy and legislation (McDonnell & Pickett, 1990; Garrod & Willis, 1994; Niemelä, 1999).

The objective of this chapter was to test each of the three hypotheses raised above ((i) habitat loss leads to species loss, (ii) size is indicative of species richness and (iii) disturbance-impacts have a detrimental effect on richness, using three categories of species richness (total species, Red Data List species (RDLs) and species restricted (endemic) to the Cape Flats). I assessed the collective impact of size, habitat heterogeneity and threats, and examine how these variables influence the number of species by answering the following: to what extent is plant species richness on the Cape Flats a function of fragment size, heterogeneity and human-induced threats? Answering this question is very important for understanding the causes and effects of species decline, and therefore important for preventing species loss on the Cape Flats.

2. METHODS

The 59 study sites and their associated habitat and species data (total species, RDLs and endemics) were used in these analyses (*Chapter 2*). The number of habitats per site is used as a measure of heterogeneity.

An index (V-index) that measures the vulnerability of a site to a variety of threatening processes was developed. Ten categories, each with sub-categories, were identified as factors that contribute to species and habitat loss on the Cape Flats (Table 3.1). Justified scores were assigned to each of the categories and sub-categories (Table 3.2), giving a V-index for each site (Table 3.3).

Table 3.1 List of factors (categories and sub-categories) that contribute to species and habitat loss on the Cape Flats. Reasons justifying their scores and importance are given in Table 3.2.

CATEGORY	SUB-CATEGORY
• **Surrounding landuse**	*- Formal residential* *- Informal residential* *- Informal settlements* *- Public amenity* *- Transport features* *- Industrial* *- Commercial* *- Farming*
• **Zonation of site**	*- Residential* *- Amenity/ open space* *- Commercial* *- Industrial* *- Military area* *- Privately-owned* *Unknown/ other*
• Conserved	*- Yes* *- Uncertain* *- No*
• **Defence**	*- None* *- Large-scale support* *- Organised support* *- Controversial* *- Informal support*
• **Threats**	*- Sand mining* *- Agriculture* *- Inappropriate fire regime* *- Informal settlements* *- None/ none listed here*
• **% Alien cover**	*- 0 - 20%* *- 21 - 60%* *- 61 - 100%*
• **% Dumping cover**	*- 0 - 20%* *- 21 - 60%* *- 61-100%*

Table 3.2 Categories from Table 3.1 were used to determine the measure of threat (V-index). 1 = encourages or is conducive to conservation; 2 = 'middle-of–the–road friendly' or uncertain; 3 = hostile to conservation. References supporting the assigned scores can be found in *Chapter 1*, and additional references are listed below the table.

Categories	Sub-categories	Score	Justification for scoring
Surrounding landuse	Formal residential	1	Established people have the financial means, and often support conserved areas.[1]
	Informal residential	2	Less established people may derive benefits from conserved areas, thereby enhancing their quality of life.[1, 2, 3]
	Informal settlements	3	Pressing matters such as housing shortages take precedence over conservation issues, and land invasions by shack dwellers on sites bordering reserves have intensified.[2]
	Public amenity	1	These areas have a potential and much-desired function as both a recreational and conservation area.[2, 4]
	Transport features	1	The boundaries of these structures are fixed, resulting in clearly delineated surrounding properties.[2]
	Industrial	2	Law stipulates that future and existing establishments are obliged to confer some protection to areas.[5]
	Commercial	2	Law stipulates that future and existing establishments are obliged to confer some protection to areas.[5]
	Farming	3	Small-scale farming and conservation on the Cape Flats are incompatible.[6]
Zonation of site	Residential	3	The provision of housing is a national concern and is regarded as a priority.[2, 7]
	Amenity/open space	2	The establishment of a conservation area can be achieved if the area's multi function potential is realised.[3, 4]
	Commercial	3	Ventures of this nature are associated with employment opportunities and financial spin-offs.[8]
	Industrial	3	Industrial ventures are always encouraged since the country as a whole directly benefits from it.[8]
	Military area	2	These areas are generally not intensely impacted; activities are highest near building localities.[9]
	Privately-owned	2	It is not easy to convince landowners to alter their landuse practises.[6]
	Unknown/Other	1	This applies to land zoned as reserves or areas that are remote from threats.[3, 10]
Conserved	Yes	1	This is the desired state.
	Uncertain	2	The site may or may not be a proclaimed protected area.
	No	3	This is not the desired state.
Defence	None	3	This is the least acceptable state.[4, 7]
	Large-scale support	1	Public support in the form community organisations reflect public attitude and care for the environment.[3]
	Organised support	1	Support from decision-makers, and formal action/management plans indicate commitment to conservation.
	Controversial	2	Areas engaged in social, political or legal ramifications receive media coverage, thereby raising awareness.
	Informal support	2	Depending on their agenda, agencies may or may not take the plight of areas seriously.[2]

Continues overleaf

Categories	Sub-categories	Score	Justification for scoring
Threats	Sand mining	3	Mining operations remove whole biotas, thereby robbing the area of its restoration potential.[1, 5, 7]
	Agriculture	3	The nutrient and topsoil profile is altered, thereby rendering agricultural lands difficult to rehabilitate.[6]
	Inappropriate fire regime	2	Inappropriate fire frequencies negatively impact recruitment, regeneration and successional processes.[8, 10]
	Informal settlements	3	The widespread phenomenon of informal settlements of this nature degrades the conservation worthiness.[4, 7]
	None/none listed here	0	Other threatening processes not listed here are in operation.[2, 3]
% Alien cover	0 – 20%	1	Alien invasive infestations are reversible processes particularly if the area is of prime biological interest.[8, 10]
	21 – 60%	2	Depending on major resource availability and incentives, the area could probably be rehabilitated.
	61 – 100%	3	This extent of infestation would make clearing operations very expensive.
% Dumping cover	0 – 20%	1	Rubble dumping is reversible in that the biological integrity of the area is salvageable.[2, 3]
	21 – 60%	2	Rubble removal is expensive but probably possible.
	61 – 100%	3	Areas like this are usually regarded as a dumping area by the public who will continue to dump there.[4, 7]

[1] Hall (1984); [2] Behrens & Watson (1992); [3] Wood *et al.* (1994); [4] McDowell & Low (1990); [5] Environment Conservation Act (1997); [6] McDowell (1988); [7] Low & McKenzie (1988); [8] van Wilgen *et al.* (1996); [9] van Blerk (1998); [10] Richardson *et al.* (1996a)

Table 3.3 V index scores for the study sites. V index for each site was calculated by summing the score of each category (developed from Table 3.2). Open cells score zero. * denotes a protected area. Names of the sites are given in Appendix 1.

SITE	Landuse	Zonation	Conserved	Defense	Sand mining	Informal settlements	Agriculture	Inappropriate fire	%Alien cover	%Dumping cover	V-INDEX
75MB*	1		3	2					1	[illegible]	9
AECI	2	3	3	3				2	2	1	16
BELHAR	2	2	3	3		3			3	3	19
BLKFAUR	1	2	3	3				2	3	1	15
BLOUFYN	3	2	3	2			3		1	1	15
BLOUREN	3	2	1	1			3		1	1	12
BLOUSTR	1	1	2	2	3				1	1	11
BLOUVL*	2	1	1	1					1	1	7
BDOWNS	1	2	3	3		3		2	3	1	18
BRACK	1	1	1	2	3				2	2	12
CFNR*	1	1	1	1					1	1	6
CAPRI*	3	1	1	1					1	1	8
CROSS	3	2	3	3		3			3	1	18
DRIFTSR*	2	1	3	3		3			3	2	17
DUINEF	1	2	3	3		3		2	3	2	19
DURBRC	1	2	3	2					1	1	10
DURBNR*	1	1	1	1					1	1	6
EDITH	1	1	1	2					1	1	7
EDITHNR*	2	2	3	3		3		2	3	1	19
EERQR	1	3	3	3	3			2	3	1	19
WELM	2	2	3	3				2	3	1	16
HANOV	1	2	3	3					3	2	14
KENRC	3	2	3	1					1	1	11
KHAYOS	3	2	3	3		3			2	2	18
KRAAIF	3	3	3	3		3			3	2	20
LANOS	2	2	3	3					3	3	16
LANJAN	1	2	3	3					3	3	15
LANRD	2	2	3	3					3	3	16
MACASS	2	2	3	2	3	3			2	1	18
MEW	1	3	3	3		3			3	2	18
MILNLM	2	3	3	3					1	1	13
MILNRC*	2	1	1	2					1	1	8
MKFLAT	3	3	3	3		3			3	3	21
N7N1	1	2	3	2		3			1	1	13
NEWF	1	2	3	3					2	2	13
NORTHP	1	3	3	3					2	1	13
NWPEN	1	2	3	3				2	2	2	15
NYANGR	3	2	3	3		3			2	1	17
PELZEEK	2	2	3	2					3	3	15
PENEER	1	2	3	3					3	1	13
PENEST	1	2	3	3					3	1	13
PHILEAST	3	3	3	3		3			3	3	21
PHILWET	3	2	3	2		3			3	3	19
PLATTE*	1	1	1	2					1	1	7

Continues overleaf

SITE	Landuse	Zonation	Conserved	Defense	Sand mining	Informal settlements	Agriculture	Inappropriate fire	%Alien cover	%Dumping cover	V-INDEX
RIET*	2	1	1	1					2	1	8
RONDCO	1	2	2	1					2	1	9
RONDEA	1	2	3	1					2	1	10
RONDVL*	1	1	1	1					1	1	6
SAND	2	2	3	3					2	1	13
SOMCH*	2	1	1	2					1	1	8
STRANDF	2	2	3	3					2	3	15
SWART	2	3	3	2					2	1	13
TOKFR	1	2	3	3				2	3	1	15
TYGNR*	1	1	1	1					1	1	6
WETT	2	2	3	3					2	2	14
WING	1	1	3	2					3	1	11
WOLFNR*	3	1	1	2		3			2	1	13
WOOD	1	2	3	3					1	1	11
YOUNG	2	1	3	3					2	1	12

2.1 RATIONALE SURROUNDING THE V-INDEX

The conceptual approach to the V-index is premised on the following: the susceptibility of a site to biodiversity loss is a combination of the extent to which a site has already been/will be transformed by disturbance, and the extent to which social, economic and political issues are conducive to prevent or delay the extirpation of species and habitats present on a site. As such, all the categories and their sub-categories that were used to determine the V-index are recognised as disturbance factors that contribute to the loss of diversity (Table 3.1; *Chapter 1*; D. Gibbs, B. Low, L. Raitt, J. Wood, *pers. comm.*, 1997-1999). Table 3.1 thus presented a comprehensive list of the factors that lead to species and habitat loss, and Table 3.2 attempted to provide justification that these are indeed contributing factors. However, there are several conceptual difficulties in developing the V-index.

Firstly, there are inherent problems associated with quantifying factors relating to the qualitative nature of policy and legal issues, society-/insitutional-based support and the capacity/willingness of institutions to support and enforce conservation actions (see Hall, 1993). Failing to integrate these factors in a calculation of a vulnerability index, particularly in relation to an urban context, ignores the role of these factors in maintaining diversity. Secondly, the complexity and intensity of elements comprising the V-index are not ubiquitous on all the sites, and this is cumbersome to reflect in terms of the limitations relating to quantifiable measurements. Moreover, the synergistic effects or the co-variation between these different influences are not well understood (Hall, 1983). Thirdly, it is recognised that multi-criteria scoring approaches are generally difficult in terms of subjectively assigning scores (Gaston, 1996). Fourthly, because the relative importance of the factors listed in Table 3.1 are not known (principally due to measurement limitations), the effects of each factor were considered mutually exclusive and equal in impact, and as such, the categories were not weighted in relation to each other. In this light, the V-index presented for each site (Table 3.3) reflects an objective attempt to capture all the complexities associated with factors that lead to habitat and species loss on the Cape Flats.

The categories of *Surrounding Landuse*, *Zonation of site* and *Conservation Status* are associated with issues of policy and legality, and any changes in either of these categories require considerable debate and substantial resources, and usually is a lengthy process (Dewar & Watson, 1990; Behrens & Watson, 1992).

The category of *Defense* captures society-/insitutional-based support and the capacity/willingness of institutions to support and enforce conservation actions (Matowanyika, 1994).

Alien encroachment (*% Alien Cover*) and refuse dumping ('*% Dumping Cover*) were categorised as separate components on the assumption that they are regarded as potentially reversible and manageable factors, this being dependent on the extent of encroachment and dumping (McDowell & Low, 1990; Wood *et al.*, 1994). Sand mining, agricultural practices, inappropriate fire regimes (too frequent or too infrequent) and informal settlements were assumed to be irreversible factors because they cost expensive resources to monitor and regulate, particularly in low income, high-density suburbs (Low & McKenzie, 1988; McDowell, 1988; Wood *et al.*, 1994).

2.2 CALCULATION OF THE V-INDEX

A score (1, 2 or 3) was assigned to one of the sub-categories in each of the 10 categories (Table 3.2, 3.3). Factors that encourage or are conducive to conservation score 1; site attributes which are not supportive of conservation are 'unfriendly' and score 3, whilst site attributes which are indeterminate or uncertain score 2. V-index for each site was calculated by summing the scores across all categories (Table 3.3). There were only two exceptions to these rules. The first was that for the category of *Threats*, each sub-category was scored individually because more than one of these threats can be in operation on a site. The second exception was that unreserved sites (see Appendix 1) were discriminated against because reserves were assigned a score of 1 under the categories of *Zonation of site* and *Conserved.*

Thus, sites with a low V-index are less threatened than sites with a higher value. A maximum V-index of 29 was assigned to the most highly threatened unconserved sites experiencing sand mining, agriculture, an unsuitable burning regime and informal settlements, but there were no sites with these characteristics (Table 3.3). Sites scored as the least threatened had a minimum V-index of 6.

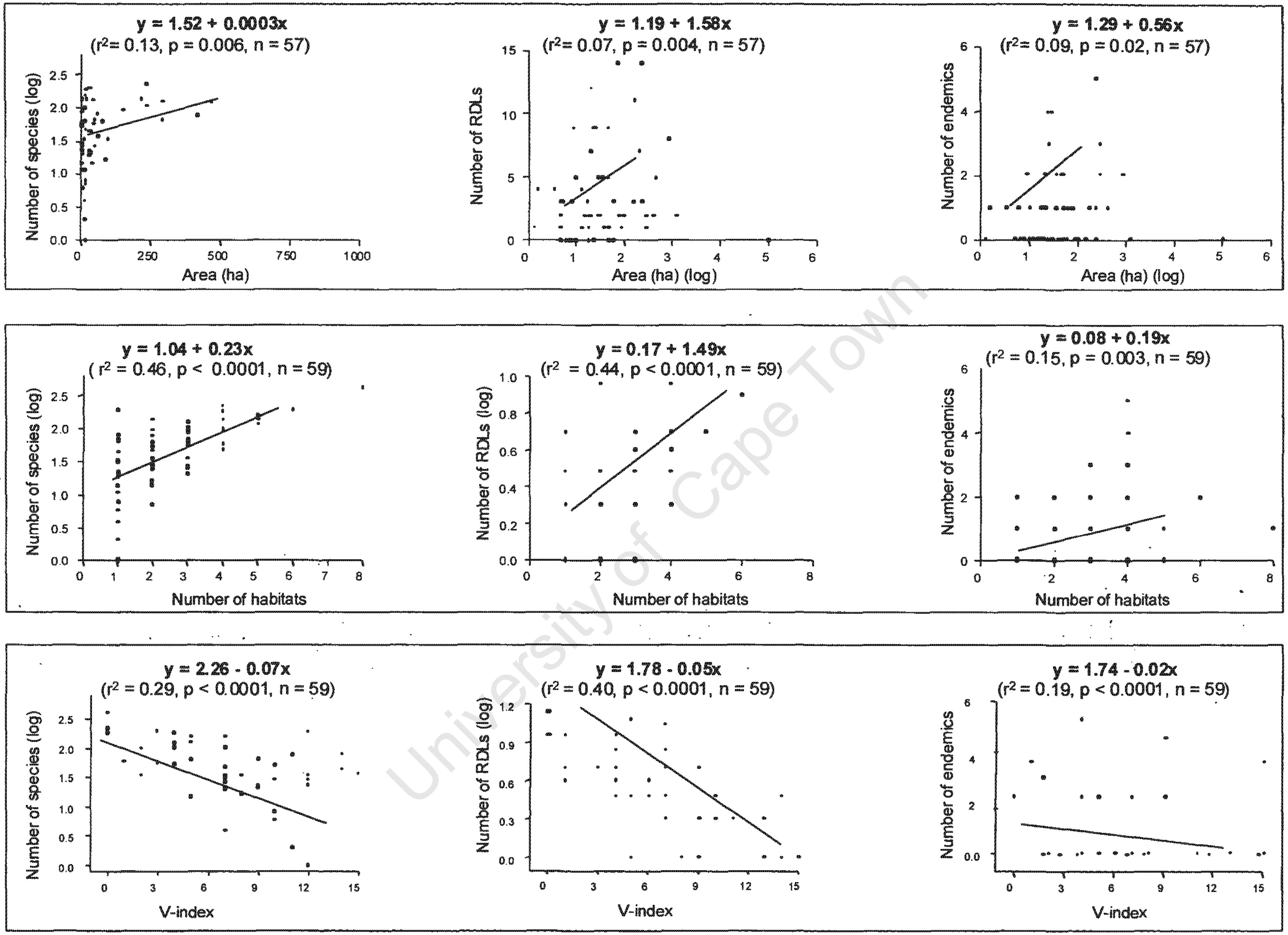

Figure 3.1 Relationships between explanatory variables (size, heterogeneity (number of habitats) and threats (V-index)) for fragments on the Cape Flats.

2.3 ANALYSES

Simple and multiple linear regressions were used to test the hypotheses regarding the role of size, heterogeneity and threats in explaining patterns of plant richness on fragments (n = 59) of natural habitat on the Cape Flats.

The AECI industrial site (AECI, 1 242 ha) and Rietvlei Protected Area (RIET, 547 ha) sites were excluded because they have a disproportionately low species richness. The AECI is an industrial site where most of the pockets of indigenous vegetation are surrounded by transformed land. The size of the total area occupied by indigenous vegetation is unknown. A large proportion (> 70%) of RIET comprises a waterbody. The sample size used for species-area regressions (all species, RDLs and endemic species) was 57.

A least squares regression was performed to explore bivariate relationships between size, heterogeneity and threat. The set of independent explanatory variables was then incorporated into a Multiple Linear Regression Model (MLRM) using a step-wise, backward approach (Zar, 1984) for all species, habitats, RDLs and endemics. The use of a paired t-test validated confidence in the model. Where appropriate, data were log-transformed to achieve normality (Zar, 1984).

3. RESULTS

3.1 BIVARIATE LINEAR RELATIONSHIPS

The regression analysis shows that there is a significant, but relatively weak relationship between area and richness for all species categories (total species, RDLs and endemics) (Figure 3.1). Values of r^2 ranged from 7 - 13% for species-area relationships.

Species richness has a positive relationship with heterogeneity for all species categories, especially total species and RDLs. In contrast, there was a significant, albeit weak positive relationships between heterogeneity and the number of endemics. No site was simultaneously subjected to agriculture, inappropriate burning, informal settlements and sand mining. V-index scores range from 6 to 21 (Table 3.3), with 48% of the sites having a V-index value >13. This suggests that a large proportion of the sites is subjected to fairly intense to extreme threatening processes. Sites with the lowest V-index (=6) are all protected areas, namely the Cape Flats Nature Reserve (CFNR), Durbanville Nature

Reserve (DURBNR), Rondevlei Nature Reserve (RONDVL) and Tygerberg Nature Reserve (TYGNR). Mitchells Plain-Khayeltisha Flats (MKFLAT) and Phillipi East (PHILEAST) had the highest V-index (=21).

The V-index, hereinafter referred to as 'threat', was found to be negatively related to total species, RDLs and endemics. All these results were statistically significant but the low r^2 values indicate that heterogeneity, size and threat are not strong independent predictors of richness.

3.2 COLLECTIVE EFFECTS OF VARIABLES

The independent contribution of each of the explanatory variables (heterogeneity, size and threat) to explain richness patterns for each species category was used as a basis for building the Multiple Linear Regression Models (MLRMs). The form of the MLRM equation is $Y = \alpha + \beta_1X_1 + \beta_2X_2 + \beta_3X_3$ where Y=species richness, α is a constant (y-intercept), β is the regression coefficient of the variable and where X_1 = heterogeneity, X_2= size and X_3= threats (V-index) (Zar, 1984).

The statistical output of the multiple linear regression is given for total species (Table 3.4a), RDLs (Table 3.4b) and endemics (Table 3.4c). In each case, size was found to be a weak variable in the regression because the coefficient (β) had a low value and was statistically non-significant. Overall, the regression statistics were not found to be particularly robust as indicated by the r^2-value and the analysis of variance (F). The regression results were used to build a model that accounts for richness.

3.2.1 BUILDING THE MLRM

The statistical results of the multiple linear regressions were total species, RDLs and endemics incorporated into MLRM equation (Step 1 of Table 3.5). In Step 2, X_2 was excluded from the MLRM in all cases, giving *Equation A*. However, in a separate analysis (Spearman's Rank correlation), size was not co-linear with heterogeneity ($r_s = 0.276$; $p > 0.50$; $n = 59$) and threat ($r_s = 0.008$; $p > 0.50$; $n = 59$). Variables should not be co-linear for them to be incorporated into an MLRM although MLRMs acknowledge that variables can bear some relationship to each other (Zar, 1984).

Table 3.4 Statistical MLRM output for (a) all species, (b) RDLs and (c) endemics. X_1=heterogeneity, X_2=size and X_3=V-index. T is from the t-test. ns = non-significant.

(a) Total species.

$r^2 = 63.8\%$; $T = 1.93$ ($p<0.05$)

Estimates of regression coefficients:	
X_1	$T = 6.90$ ($p < 0.001$)
X_2	$T = -0.76$ ($p = 0.45^{ns}$)
X_3	$T = -2.92$ ($p < 0.005$)

(b) Red Data List species.

$r^2 = 58.2\%$; $T = 1.68$ ($p<0.05$)

Estimates of regression coefficients:	
X_1	$T = 6.37$ ($p < 0.001$)
X_2	$T = -1.76$ ($p = 0.084^{ns}$)
X_3	$T = -0.01$ ($p = 0.014$)

(c) Endemics.

$r^2 = 19.9\%$; $T = 2.37$ ($p<0.02$)

Estimates of regression coefficients:	
X_1	$T = 1.69$ ($p < 0.005$)
X_2	$T = -0.16$ ($p = 0.087^{ns}$)
X_3	$T = 2.47$ ($p < 0.017$)

Table 3.5 The steps for building and validating a predictive model (multiple linear) for different categories of species richness (total species, RDLs and endemic species) using heterogeneity (number of habitats per fragment) (X_1), size (X_2) and V-index (X_3) as explanatory variables. Step 1 was derived from Table 3.4a-c.

	TOTAL SPECIES	RDLs	ENDEMICS
STEP 1	$Y = 54.4 + 32.32(X_1) - 0.022(X_2) - 4.57(X_3)$	$Y = 2.55 + 1.065(X_1) - 0.003(X_2) - 0.213(X_3)$	$Y = 1.60 + 0.19(X_1) - 0.0001(X_2) - 0.092(X_3)$
STEP 2	$Y = 54.4 + 32.32(X_1) - 4.57(X_3)$ *Equation A*	$Y = 2.55 + 1.065(X_1) - 0.213(X_3)$ *Equation A*	$Y = 1.60 + 0.19(X_1) - 0.092(X_3)$ *Equation A*
STEP 3	$X3 = 4.647 - (0.158) X_1$ *Equation B*	$X3 = 4.647 - (0.158) X_1$ *Equation B*	$X3 = 4.647 - (0.158) X_1$ *Equation B*
STEP 4	$Y = 33.163 + 33.042X_1$ *Equation C*	$Y = 1.560 + 1.099X_1$ *Equation C*	$Y = 1.189 + 0.205X_1$ *Equation C*
STEP 5	$t = 17.11$, $p < 0.0001$	$t = 17.12$, $p < 0.0001$	$t = 17.48$, $p < 0.0001$
MODEL	*$Y = 54.4 + 32.32(X_1) - 4.57(X_3)$*	*$Y = 2.55 + 1.065(X_1) - 0.213(X_3)$*	*$Y = 1.60 + 0.19(X_1) - 0.092(X_3)$*

Because heterogeneity (X_1) and threat (X_3) were found to be weakly co-linear in the Spearman's Rank correlation (r_s = -0.402; p = 0.002; n = 59), this implies a violation of the assumption of the MLRM. However, this was resolved by interpreting one of these variables as a function of the other, and a simple linear regression on untransformed data with heterogeneity as the response variable and threat as the explanatory variable was significant ($r^2 = 0.194$, $p < 0.001$) and thus used. The variables of the defining equation lie in the 95% confidence interval, and is of the standard form Y = C + MX (Zar, 1984) where Y=threat, C is a constant, M is the gradient and X=heterogeneity (Step 3, *Equation B* in Table 3.5).

Equation B was substituted into *Equation A*, and *Equation C* was derived. Thus, the effect of threatening processes was still maintained, and was represented in terms of its linear relationship to heterogeneity.

3.2.2 Validating the MLRM

Paired t-tests (Step 5 from Table 3.5) were used to validate *Equation C* by comparing the species richness ($Y_{predicted}$) in it, to species richness (Y_{actual}) in *Equation A* (Zar, 1984). The t-value was highly significant for total species, RDLs and endemics. *Equation C* was more unreliable than *Equation A* because the weak co-linearity between heterogeneity and threat is negligible. Therefore, *Equation C* was rejected and *Equation A* was accepted as the final model. The final models for total species RDLs and endemics are presented in Table 3.5.

The model suggests that habitat heterogeneity and, to a lesser extent, threatening processes, are the principle factors controlling the richness of all species categories on fragments on the Cape Flats. In conjunction with heterogeneity and threats, remnant size was found to be a poor predictor of species richness. The model was much weaker for endemics and RDLs compared to total species.

4. DISCUSSION

It has been shown that neither size, heterogeneity (number of habitats) or threats (V-index) singly account for plant species richness in isolated fragments on the urbanised Cape Flats. Rather, it is their collective contribution that accounts for patterns of total richness, numbers of RDLs and endemic species. *Chapter 2* has also showed that there is no difference in the size of protected and unprotected areas. The model presented here indicates that irrespective of fragment size, the total number of species, RDLs and endemic species are highest on sites with a high number of habitats (high heterogeneity) and where threats are lowest. These results suggest that deterministic factors - loss of habitat and anthropogenic impacts – are principally responsible for controlling the richness of all species categories on fragments of natural habitat on the Cape Flats (see Bond, 1998). Stochastic factors associated with small population size on small-sized remnants (Lande, 1988; Rosenzweig, 1995) appear to play less of a role. Species are thus capable of persisting in very small populations on very small areas provided they are protected from these impacts (Bond *et al.*, 1988).

These results are consistent with other work in the Cape Floristic Region. Bond *et al.* (1988) found that the chief cause of species loss on small (< 600 ha) fynbos fragments in a temperate forest matrix in the southern Cape was a change in the disturbance regime - fire exclusion. A similar study of limestone-associated fynbos surrounded by leached acid sands, suggests that species are able persist on small-sized areas (4 – 15 ha) providing their necessary ecological regimes such as pollinator-disperser services and appropriate fire regimes are maintained in the system (Cowling & Bond, 1991). Only those species with highly vulnerable life-histories such as non-sprouting shrubs with small and weakly persistent seed banks (Bond, 1998) disappeared on small fragments (Cowling & Bond, 1991). Processes that degrade habitats were found to result in selection for ruderals, sprouters and annuals (Richardson *et al.*, 1996; Trinder-Smith *et al.*, 1996a). Finally, Kemper *et al.* (1999) were unable to determine an overall, significant fragmentation effect on community structure and composition in patches of coastal Renosterveld (< 1 ha) in a matrix of agriculturally transformed land. The effects that were evident were apparently the result of deterministic factors such as alien plant invasions and grazing.

4.1 SIZE

The results show that large-sized remnants generally never have few species, and that small-sized remnants can have many or few. The results also show that many habitats never support few species, but one habitat can support many or few. A possible cause for this may be that 'older' remnants that have been fragmented or isolated for a longer period of time, have lost more species (after MacArthur & Wilson, 1967). However, the MacArthur & Wilson (1967) model predicts that there would be 'relaxation' on each remnant until a new equilibrium between species and area would be reached. The lack of a strong relationship between species and area might therefore be considered an artefact of the very recent fragmentation of these remnants with insufficient time for species losses to take their course.

Small-sized remnants that have a greater proportion of their area subject to edge effects, bear a high probability of reduction in population sizes and lowered species numbers, and therefore are more prone to local extinctions compared to larger-sized remnants (MacArthur & Wilson, 1967; Boecklen & Gotelli, 1984). In conjunction with heterogeneity and threats, size was found to be a poor predictor of richness on the Cape Flats. The survival of populations on the Cape Flats remnants is not size-dependent, at least not until a critical minimum area is reached (Lande, 1988; Caughley, 1994; Shafer, 1994). It is probably more the case that the importance of heterogeneity and threats with respect to species numbers far outweigh the impacts of size. This means that small-sized remnants are not necessarily the loci of species loss and extinctions (Brown & Dinsmore, 1988; Cowling & Bond, 1991).

For undisturbed large-sized areas within the Cape Floristic Region, size has been shown to be the primary variable in explaining richness patterns (e.g. Cowling *et al.*, 1992). For example, species-area regressions at an order of magnitude $r^2 > 90\%$ have been found by both Cowling *et al.* (1992) in the south-western Cape and for Proteaceae RDLs throughout the Cape Floristic Kingdom (Rebelo & Tansley, 1993).

4.2 HABITAT AND THREAT

High spatial heterogeneity means a wider range of habitats, and hence a higher number of species, and the corollary is that the loss of heterogeneity, either through habitat conversion or degradation, brings about species losses (Ehrlich, 1986). Heterogeneity is important in maintaining levels of biodiversity through species-species interactions (Danielson, 1991), sustaining species abundance, distribution patterns and persistence (Thompson & Brown, 1992; Venier & Fahrig, 1994; Fahrig, 2001), and buffering threatening processes (Shafer, 1994; Walker, 1995). In addition, there is a large body of literature demonstrating that discrete species assemblages (in habitats) facilitate the persistence of rare species by providing refuge niches (e.g. Wood, 2001) and that sites supporting a high species richness support rare species (e.g. Debinski & Brussard, 1994) or that high numbers of rare species contribute to richness (e.g. Lombard *et al.*, 1999).

It was found that heterogeneity, in conjunction with threatening processes, is a good predictor of species richness on the Cape Flats, and that threatening processes account for less variation in richness than heterogeneity. This is probably because heterogeneity offers more niche opportunities to rare species in that they provide refuges against threatening processes (Cowling *et al.,* 1992). Fredericks (1992) found significant differences between discrete species assemblages on the Cape Flats with respect to heterogeneity. For this, the soil mineral content (calcium and magnesium) emerged as the singlemost significant explanatory factor. This suggests that heterogeneity, and in particular, edaphic gradients, are plausible determinants for richness and possibly a high degree of habitat specialisation (Cowling, 1990; Cowling *et al.,* 1992).

4.3 WHY THE HIGH LEVELS OF RICHNESS?

The fact that such a large number of species (1 192) has managed to survive within the biologically hostile Cape Flats environment implies the presence of resilient species (Bond *et al.,* 1988; Bond, 1998; L. Raitt, *unpublished data*). It would seem that generally, plant species in the study area display unique adaptations that confer fitness, thereby enabling their persistence. The composition on these remnants may consist mainly of species that are not easily prone to extinction because of resistance to inbreeding depression and heterozygosity vigour (Bond, 1998; A. Pauw, *unpublished data*). Alternatively, species may have accrued important reproductive traits that enable persistence. In a study that centred on seed : ovule ratios (a measure of reproductive effort) amongst closely related congeners of Proteaceae species from the Cape Flora, Cowling & Eggenberg (2000) showed that rare/range-restricted congeners are more likely to persist due to their increased relative contribution to the pool of propagules available for post-fire recruitment. Thus, locally rare species may have reproductive attributes that enable persistence (over the long term), and it is in fact the fragmented populations of common and widespread species that may be at a greater risk of extinction (in the short term).

Severe species losses (associated with small populations), reflected in the high number of species having Red Data List status, must have occurred in the recent past when urban development began escalating on the Cape Flats (based on a weak species-area relationship, after MacArthur & Wilson (1967)). Urbanisation, which embodies threatening processes (McDonnell & Pickett, 1990; Myers, 1993), has burgeoned on the Cape Flats over the past 40 years (Anon., 1993). For example, over a twenty-year monitoring period, L. Raitt (*unpublished data*) observed a marked reduction in the species richness and distribution of the Proteaceae and Ericaceae, both dominants, along the eastern fringe of the Cape Flats that was frequently subjected to destructive fire regimes and bush-cutting. McDowell & Low (1990) also calculated a 50% reduction in habitats on the Cape Flats (1983 – 1989), with an accompanied increase in RDLs. They also observed that a substantial number of indigenous species had managed to persist on sites where human-induced threatening processes were intense, namely along road verges, near railway lines and even racecourses.

How would increased adaptive capacity have come about? Archaeological records indicate that the Cape Flats was inhabited by indigenous people (Khoi Khoi) from as early as *ca* 120 000 years ago, who at that time, practised fire-stick farming and stock-farming (Deacon, 1992). The arrival of the first Europeans in 1652 brought the advent of intensified landuse. The flora of the Cape Flats thus has a long history of human-induced disturbance. These selection pressures may have conferred resilience to threat and may have predisposed the flora to withstand contemporary threatening processes. Interestingly, conservation biology theory alludes that areas with a shorter history of anthropogenic disturbance are regarded as more important conservation targets than areas with a longer disturbance history (see Balmford, 1996). The Cape Flats may therefore be considered as important in this context.

4.4 CONSERVATION IMPLICATIONS

There are two reasons for managers and decision-makers on the Cape Flats to abandon the idea that larger areas support more plant species. Firstly, species-area relationships have been used to predict species losses (Reid, 1992). However, this prediction of a change in species number cannot assign extinction probabilities to certain species because some are more susceptible to threats than others. Secondly, species-area relationships provide managers with a rough estimation of how many species occur on a site of a given area, but it provides no information on the practical issue of what controls richness. Heterogeneity and human-induced threats, and not size alone, were found to be the major determinants of the richness. Hence, small-sized sites on the Cape Flats can make meaningful contributions towards conserving plant diversity (Shafer, 1994), over the short term.

4.4.1 Management of biodiversity

In practice, how can species losses be minimised on the Cape Flats? Managers should be concerned to perpetuate all species if their remit is the preservation of biodiversity. The loss of any one part of the spectrum of habitats would inevitably cause some species to disappear. Managing at the scale of heterogeneity should be given due consideration (Kirkpatrick & Gilfedder, 1995; Noss, 1996), and in practical terms, this would mean targeting areas of high heterogeneity for conservation and restoration efforts.

Broadly, habitat restoration and selecting additional reserves with high habitat heterogeneity and minimising deleterious anthropogenic impacts present one way of prolonging the persistence the Cape Flats flora. For example, physical processes that alter the soil profile, such as sand-mining (Wood *et al.,* 1994), inappropriate burning (McDowell *et al.,* 1991), rubble dumping (*pers. obs.*) and alien encroachment (McDowell & Low, 1990) should be curbed as these have direct bearing on habitat degradation and depress species richness. Despite the Cape Flats having the apparent ability to absorb threats, Higgins *et al.* (1999) explain that the indigenous flora of the Cape Peninsula and surrounds (also characterised by high alpha and beta diversity) have a low resistance to alien invasions. Every effort should therefore be made to mitigate the effects of encroachment.

Inappropriate fire regimes hold serious ecological impediments. Musil (1991) showed that seedbank longevity, and hence species recruitment on the Cape lowlands is poor with prolonged fire deferral. Musil (1991) also presented an alternative hypothesis stating that soil-stored seedbanks of lowland fynbos are transient and thus extremely vulnerable to threatening processes. Random, frequent fire events are a common occurrence in some parts of the Cape Flats (L. Raitt, *pers. comm.,* 1998), and in other areas, fire has been totally excluded (*pers. obs.*). As such, fragments should be appropriately burnt.

Habitat destruction disrupts vital ecosystem services like pollination and dispersal mechanisms (Johnson, 1992; Bond 1994); the converse has also been shown that reducing factors that cause mortality of pollination and dispersal agents assist in sustaining habitats, and consequently, reduce the risk of species extinction (Fahrieg, 2001). Restoring habitats can provide structural and trophic requirements for these mechanisms to function. Monitoring of the biggest sub-population of *Serruria aemula* (< 200 individuals), a critically endangered species confined to the Cape Flats, revealed the absence of a pollinator. Similar findings exist for *Ixia maculata* and *Satyrium carneum* that experience infrequent pollinator visits (A. Pauw, *pers. comm.*, 1998). Managing at the level of heterogeneity can further be achieved by intensive management which simulates the necessary ecological driving forces for these systems (Cowling & Bond, 1991), and by buffering deleterious disturbances like edge effects (associated with high edge-to-area ratios) for example, as have been demonstrated in gallery forest studies (Kellman, 1996; Brokaw, 1998).

4.4.2. Management of social and institutional issues

It is more complex to alleviate threats in the guise of defense (inadequate/inappropriate support of institutions and civil society towards conservation). It is assumed that defense (support emanating from civil society and institutions) is able to facilitate the implementation of suitable management systems. As the Cape Flats becomes more densely populated, threatening processes will inevitably escalate. If we are to practice biodiversity conservation, a renewed emphasis on the positive aspects of a green urban environment will have to found.

Firstly, there should be consensus amongst all stakeholders that historically, the Cape Flats has always been designated as a residential area for the urban poor, but that this is inherently unreflected in present conservation interests. Equally important, is convincing residents that having a reasonable quality of life is integral to preserving natural areas, and that biodiversity indeed has a societal function - the maximum benefits to residents should be communicated (Garrod & Willis, 1994). If these issues can be accepted, then it should be possible to achieve a more unified vision to implement management plans.

Secondly, stronger citizen involvement should be seen as central to conservation decision-support systems and decision-making processes. Institutional support for the preservation of the Cape Flats flora should encompass corrective management in the form of public involvement (drawing on 'lessons learnt'), that hitherto, has systematically remained unaddressed.

Resolving these institutional and social challenges will result in more successful coordinated efforts for maintaining levels of plant species richness. Existing and planned reserves on the Cape Flats can be immediately accessible to more than 4 million residents on the Cape Flats, and thus has the potential to address ecological and human values.

5. CONCLUSIONS

A shortcoming of this study is that the models reflect on the number of species lost is a function of how much habitat is lost (as a result of threats), not on which sites were degraded. A follow-up study incorporating range extent requirements and abundance will inculcate more confidence in future extinction trajectory models for the Cape Flats flora.

Levels of insular effects are time-dependent in that sites that have been isolated for longer through fragmentation processes experience greater species losses (MacArthur & Wilson, 1967; Saunders *et al.*, 1991). Since the main ecological driving force (fire) for the persistence of fynbos species is virtually absent, aberrant and difficult to manage in urban landscapes (van Wilgen *et al.*, 1992; Wood *et al.*, 1994), small-sized areas may well be in a non-equilibrium state of species loss. This is evident in the model that shows that the collective power of heterogeneity and threatening processes in accounting for and predicting richness, has limitations. One way of testing whether the Cape Flats flora is becoming 'more urbanised' in character would be to undertake census and monitoring of RDLs and endemics.

In an ecological study of remnant bushlands, Kirkpatrick & Gilfedder (1995) state that "the maintenance of good condition are not necessarily the causes of survival of rare and threatened species"... More positively, targeting areas of high diversity for conservation and restoration efforts, buffering extreme threatening processes (particularly alien invasions), and simulating natural disturbance regimes (fire) should be given major consideration in management systems and in conservation planning directed at new and existing reserves on the Cape Flats. Most importantly, the public support issues should be treated as the overarching determinant for the persistence of the Cape Flats flora and should necessarily become integrated with refined conservation strategies. Lastly, within the urban context of the Cape Flats, we should strongly reject the notion of 'too small for conservation' and call for the efficient deployment of legal, political, societal and institutional capacities to help preserve this botanical heritage for future generations.

Chapter 4

IDEAL- AND REAL-WORLD CONSERVATION PLANNING SOLUTIONS FOR THE CAPE FLATS FLORA USING THREAT AND SUPPORT INDICATORS

Abstract

Conservation planning work on the Cape Flats has concentrated on species representation and has not addressed the retention of ecological processes to prolong persistence. Similarly, real-world constraints pertaining to biological threats (e.g. alien plant cover, removal of rubble, surrounding landuse and so forth) and threats relating to institutional/public support (protection status and levels of support/defense) have not been incorporated into reserve selection studies. These previously ignored issues were addressed and integrated into reservation targets. Representation and retention issues were dealt with by using presence/absence data for species and a surrogate for ecological processes (habitat heterogeneity); real-world constraints were integrated as components of vulnerability indices. A step-wise, richness-based algorithm was used to address targets for conserving each biodiversity unit at least once, for two chronological sequences (1997/1998 and 2001). The first scenario was aimed at an ideal-world ('constraint-free') minimum set. The second scenario avoided sites that are highly threatened and the third avoided sites that enjoy little public/institutional support. The most efficient solution required twenty additional sites (totalling *ca* 36 km^2) to achieve the conservation target, thereby protecting *ca* 7% of the land area of the Cape Flats. In pristine landscapes, real-world and ideal-world scenarios are likely to be the same, but the results produced here show that in highly transformed landscapes such as the Cape Flats, this appears to be similar. Reserved areas contribute > 70% of the complimentary biodiversity features towards all minimum sets generated for 1997/1998 and 2001 indicating that reserves are vestiges of biodiversity. These results suggest that criticisms leveled at existing Cape Flats reserves may have been unduly pessimistic. Attention should instead be directed at prospects for acquiring land for additional reserves and at enforcing the protection of the existing reserve estate. Every site considered for the minimum set analyses in 2001 is irreplaceable, a function of limited land availability for conservation. Simple linear and exponential regressions predict that all additional land for conservation will be used up within 10 to fifteen years assuming that threats continue unabated.

IDEAL- AND REAL-WORLD CONSERVATION PLANNING SOLUTIONS FOR THE CAPE FLATS FLORA USING THREAT AND SUPPORT INDICATORS

"A useful point of entry to a debate is that on the other hand, you have different fingers..." (Anon.)

1. INTRODUCTION

Worldwide, there are only a few examples where the urban environment has formed the basis of detailed ecological studies (McDonnell & Pickett, 1993; Niemelä, 1999), and few that have concentrated on conservation planning (McKenzie Rebelo, 1997; Maze & Rebelo, 1999; Maddock & Benn, *in prep.*). Urban ecological studies have generally focussed on invertebrates, and have been primarily undertaken in industrialised cities of North America and Europe. Yet, an important reason for setting aside reserves for maintaining plant diversity in urban areas is to increase property prices (Tyrväinen, 1997). Remnants of natural habitat in urban areas, particularly those which have protection status, also represent a huge potential for secondary conservation options via educational and recreational amenities in urban areas (Dewar & Watson, 1990; McDowell & Low, 1990; Ngeleza, 1990; McDowell *et al.*, 1991; Nguta, 1992). More importantly, roadsides, spoilheaps and quarries in urban landscapes are nodes that are often associated with a high diversity of species that are threatened with extinction (Shephard, 1994; Eversham *et al.*, 1996). Urban areas serve as source areas for extinctions ('extinction sinks') that can have implications for cascading species losses along gradients of urban activity (e.g. McDonnell *et al.*, 1997; Duhme & Pauleit, 1998).

To many, the conservation of the Cape Flats biota is regarded as a 'lost cause' or 'not viable' because the alleviation of the variety, complexity and extent of the threatening processes incur expensive and unaffordable resources (Behrens & Watson, 1992; Dewar & Watson, 1990). While this may be true, it also reflects the general world-wide view that urban environments are less worthy of conservation investments than non-urban environments (McDonnell & Pickett, 1993; McDonnell *et al.*, 1997; Niemelä, 1999).

The Cape Flats is regarded by many as one of Africa's major 'hotspots' of plant biodiversity - the flora is extraordinarily rich, and has been reported as highly endemic and highly threatened (Myers, 1990; Wood *et al.*, 1994; Maze & Rebelo, 1999) (see *Chapter 2*). The area is a highly fragmented and densely populated (Wood *et al.*, 1994; van Wilgen, 1996), and the numbers of remnant-bound plant species that are locally threatened with extinction are expected to increase (Bond & Goldblatt, 1984; Hilton-Taylor, 1996a; Goldblatt & Manning, 2000). A systematic approach to conservation planning provides a defensible way of identifying priorities (Pressey, 1994; Pressey & Tully, 1994; Margules & Pressey, 2000). The opportunity for setting aside additional land to secure the protection of plant species is bleak due to the difficulties of implementing the outcomes of systematic conservation plans.

Ad hoc conservation has biological, economic and social implications; Pressey & Tulley (1994) describe the shortcomings and impracticalities of this approach. The concept of designing or identifying a system of nature reserves is in itself an attempt to strategically plan towards (i) targeting areas most in need of preservation, (ii) increasing the representation of features or ecological pattern, (iii) increasing the likelihood of persistence of ecological processes, and hence ecological pattern, over time, and (iv) reducing costs to future land acquisitions for conservation purposes. It has also been recognised that representation targets in conservation planning will only temporarily delay the rate of extinction, hence the need to preserve ecological and evolutionary processes that sustain and generate biodiversity (see Cowling *et al.*, 1999; Cowling & Pressey, 2001).

Any attempt to adopt a strategic conservation approach should seek to integrate the following real-world constraints on the Cape Flats biota with conservation interests: (i) burgeoning demands for housing (ii) limited human, financial and infrastructural resources for the implementation of conservation action and (iii) ongoing threatening processes.

1.1 CONCEPTUAL FRAMEWORK

The basis of reserve selection is to select land fragments from within a landscape to ensure the future survival of its biota, under the remit of reservation targets (Margules *et al.*, 1988). 'Minimum sets' identify the smallest number of fragments with complementary features that satisfies a conservation goal. A complimentarity analysis is an iterative selection procedure (algorithm-driven), and may be applied as a tool for selecting biodiversity features, such as species, habitats, landuse systems, communities (Bedward *et al.*, 1992) and spatial surrogates of ecological processes (Cowling & Pressey, 2001). Selection procedures carry out a step-wise incorporation of sites into a system by applying a sequence of rules that do not unnecessarily duplicate biodiversity features.

Algorithms may be modified to redefine conservation targets, such as considering constraints relating to land suitability or land availability. However, targets should not be unnecessarily influenced by threatening processes; threats influence priorities and hence, the implementation of conservation plans.

Targets should be established to accommodate biodiversity features, and should be boosted to accommodate possibilities of the future loss of these biodiversity features. Target setting should also not be confounded by the implementation of conservation plans, in order to address priorities before the implementation of conservation plans. There are least two general tenets that govern implementation issues. The first is that surrounding land should not be hostile to a reserve system such that it interferes with the target, and secondly, reserves should be drawn from the least disturbed sites that enjoy the greatest support (or potential support) from civil society and conservation institutions.

The concepts of 'efficiency', 'flexibility' and 'irreplaceability' fundamentally underpin spatially explicit reserve selection using iterative procedures (Pressey, 1994). These are explained below:

1. A design is most 'efficient' when the 'smallest' minimum set represents all the biodiversity features in an area, and this depends on the extent to which the features complement each other. The most cost-effective reserve design in terms of the number of sites or the collective size of the sites has a high measure of efficiency (Pressey, 1994).
2. Conservation targets are established within the remit of conservation goals. Goals reflect an objective towards which targets are directed. Targets are generally measurable. Goals and targets can be set under a scenario (a postulated event) or a strategy.
3. Within the context of a conservation goal, 'flexibility' presents alternative combinations of sites (a larger number of sites than the minimum set) to, for example, accommodate post-selection conservation goals. This is particularly useful to maximise representation, but more importantly, it allows for negotiation amongst parties when a site belonging to the minimum set becomes unavailable or unsuitable for conservation. Usually, the more flexible sites there are, the greater the negotiation options, and therefore the more likely the achievement of the conservation goal (Pressey *et al.*, 1994a).

4. 'Irreplaceability' is an objective measure of the conservation value of a site, in relation to other sites, in terms of its potential contribution to a conservation goal. Iterative procedures select sites ranging from most to least irreplaceable. While there are various methods for determining measures of irreplaceability, the measure is dependent on biodiversity features present on other sites being considered (Pressey *et al.*, 1994b).

The prioritisation and scheduling of conservation action to implement a reserve system lies beyond systematic reserve selection. The conservation of sites are prioritised in terms of their irreplaceability, explained above, in addition to the degree of threat faced by a site, referred to as 'vulnerability' (Pressey, 1996). Vulnerability serves as measure of urgency for conservation. Conservation action should ideally first be directed to sites that require the most urgent conservation action and which are simultaneously highly irreplaceable (Pressey *et al.*, 1996).

1.2 CONSERVATION PLANNING HISTORY OF THE CAPE FLATS

The Cape Flats has been a conservation concern for many years, during which ideas for its conservation have evolved immensely (Low, 1979; Jarman, 1986; McDowell & Low, 1990; McDowell *et al.*, 1990; McKenzie & Rebelo, 1997; Maze & Rebelo, 1999; Maddock & Benn, *in prep.*).

Jarman (1986) laid the foundation for conservation planning on the Cape Flats and its outskirts. Only large sites (> 4 415 ha) were included in the survey, thereby deeming small natural remnants in the urban core as unworthy of consideration. The use of a multi-scoring approach in this study masked potentially important sites that were outstanding for one or a few criteria. The seven highest scoring sites were assigned priority. This study was the first to attempt the identification of key conservation areas on the Cape Flats.

Sites (> 1 ha) of major botanical importance were identified on the Cape Flats by McDowell & Low (1990). These sites were described on a conservation priority basis, and took into account their potential benefits to the surrounding public. Twenty-five natural remnants and corridors received priority status on the basis of indigenous species richness and the presence of Red Data List species (RDLs) (Hall *et al.*, 1980), in relation to minimising the threat by woody invasive alien plants (e.g. *Acacia saligna* and *A. cyclops*).

In an assessment of 20 threatened plant species of the Cape Peninsula, Hall (1993) used the concepts of 'need' (number of individuals, subpopulations and distribution density) and 'ease' (principally relating to the difficulties associated with alleviating threats) to set conservation priorities. This study represented an important shift in the thinking behind conservation planning since it acknowledged that securing protection of species depends on strategic approaches. Pressey (1994) formalised these concepts into quantitative measures of 'irreplaceability' and vulnerability'.

McKenzie & Rebelo (1997) identified the smallest set of sites that capture the greatest number of species using an iterative algorithm (Rebelo, 1994). Only the top 15 irreplaceable sites (sites that contributed to the conservation goal) were flagshipped for conservation and the other irreplaceable sites were omitted from the working agenda. Despite the significant contribution of their work, many of the recommendations that emerged in the report were not analysed in conjunction with the conflicts and challenges unique to the Cape Flats. For example, threatening processes that result in the loss of habitat heterogeneity and species richness, were not addressed. Maze & Rebelo (1999) further expanded this analysis by incorporating a far larger suite of sites (118) over a much broader area spanning the lowlands of the Cape Metropolitan Area (Cape Flats *sensu lato*) (see *Chapter 1*). Their conservation targets considered various minimum sets of species based on the inference that conservation zoning of a site (Provincial, National, Local, World Heritage Site) was indicative that the site was securely protected and safe from threats. Their study pointed out that conservation planning for the Cape Flats required a more comprehensive integration of issues around preserving ecological processes and incorporating real-world constraints into selection procedures. Maddock & Benn (*in prep.*) used a rank order of criteria to select sites (irreplaceability, large areas chosen first, connectivity, select areas with greatest vegetation type diversity and restorability) across the Cape Metropolitan Area. Graduated targets were set to conserve the percentage of the historic distribution for coarse grain vegetation surrogates (representation), and remnant size and connectivity were used to set targets in an attempt to maintain ecological processes (persistence). Some 242 sites were required to attain all the targets.

Historically, conservation planning work on the Cape Flats has tended to concentrate on remnant size (e.g. Jarman, 1986; McDowell & Low, 1990) and later, on species-based representation (e.g. McKenzie & Rebelo, 1997; Maze & Rebelo, 1999). To date, no studies have addressed the issue of retention. Retention relates to contemporary threats to biodiversity whereas persistence involves sustaining biotic processes associated with the generation of biodiversity. Persistence is typically associated with large spatial and temporal scales (Cowling *et al.*, 1999). It is inevitable that species will be lost as urbanisation developments continue. In this light, it would be judicious to give attention to ecological processes that are responsible for sustaining species patterns (Cowling & Pressey, 2001). Furthermore, the failure to address realistic constraints relating to issues of land suitability and availability in conservation planning, will result in setting unrealistic conservation targets and even underachieving these targets (Pressey, 1996). In this chapter, I attempted to address both these previously ignored issues for the Cape Flats flora. Retention issues are addressed by trying to avoid sites at risk of biologically threatening processes or that do not currently benefit from the support of the public/institutions (defense) (see *Chapter 3*). Species data are combined with fine resolution habitat variables defined in *Chapter 2*, which, for the purposes of this study, are referred to as Fine Habitat Units (FHUs) (see Cowling & Heijnis, 2001 for Broad Habitat Units (BHUs) of the Cape Floristic Region).

1.3 OBJECTIVES

The key objective of this chapter was to identify minimum sets of sites incorporating the real-world constraints of threatening processes and society/institutional support respectively as reserve selection factors. The target was to conserve each species and FHU at least once under three different scenarios. Both representation and retention goals were thus addressed. A richness-based algorithm was used to determine a minimum set for 1997/1998 and 2001 respectively (i) in an ideal world (Scenario 1), (ii) by considering threatening processes (Scenario 2) and lastly, (iii) by taking into account support or defense (Scenario 3). All the scenarios assumed that all the reserves are mandatory. A second objective took the form of an audit, by assessing the biological costs to conservation in relation to the sites that existed five years ago but that have since been extirpated due to ongoing urban development, as well as making broad future predictions.

As such, the objectives of this chapter allowed for (i) a broad investigation of whether social considerations produce a more efficient solution than considerations pertaining to biologically threatening processes in the highly transformed Cape Flats landscape, in addition to (ii) evaluating the extent to which conservation options change over a relatively short time-frame.

2. METHODS

2.1 DATABASES, AND THREAT AND SUPPORT INDICATORS

Fifty-nine candidate sites were identified as having conservation potential (*Chapter 2*, Figure 2.1). Presence-absence databases for species and habitats for these sites were merged and these biodiversity features were treated as being of equal importance. Each habitat was considered a unit of heterogeneity (*Chapter 3*) and was referred to as a Fine Habitat Unit (FHU). Each record of a species presence represented one subpopulation; similarly, each record of a FHU may comprise only a very small proportion of a habitat. The database consisted of 4 410 records for 1 192 species and 62 FHUs (1 254 biodiversity features). Fourteen of the 59 sites were reserves (*Chapter 2*). These databases are germane to the years 1997/1998.

The Vulnerability Index, V-index (from *Chapter 3*), was modified for the purposes of this chapter. A sub-criterion score under the category of *Threats* was changed (see Table 3.2). The score for the sub-criterion of *None/none listed here* was changed from 0 to 1. The reason for this modification was to enable non- or weakly-threatened sites to be more readily selected in the reserve selection algorithms described below. A new index was then calculated for society/institutional support i.e. defense (Vs-index) by summing all the scores for the categories *Conserved* and *Defense*, thus assigning a maximum score of 6 and a minimum of 2 (Table 4.1). Similarly, an index was calculated for biological threats (Vt-index) using the categories, *Surrounding Landuse, Zonation, % Dumping Cover, % Alien Cover* and all the *Threats* sub-criteria, thus assigning a maximum score of 23 and a minimum of 5. Thus, V-index = Vs-index + Vt-index. (i.e. V-index in *Chapter 4* is a composite of biologically threatening processes (Vt-index) and support (Vs-index)). Vs-index and Vt-index (Table 4.1) were then categorised in 10% sub-classes in order for minimum sets to be drawn from high or low measures of threat and support.

Twenty-eight sites have since been lost to urban development and are no longer viable for conservation purposes at present (2001) including two reserves, namely BLOUVL and DRIFTSR (Figure 2.1) (see Maze & Rebelo, 1999). The extirpated sites (Table 4.1) and their associated biodiversity features were thus removed from a second database in order to make comparisons between 1997/1998 and 2001. The databases were queried to determine the extent of change of species and FHUs.

Table 4.1 Threat (Vt-index) and support (Vs-index) scores of the 28 sites that were extirpated between 1997/1998 and 2001. V-index is the sum of threat and support scores. * denotes a reserve. Full site names are given in Appendix 1.

SITE CODE	VT-INDEX	VS-INDEX	V-INDEX
BELHAR	13	6	19
BLKFAUR	9	6	15
BLOUVL*	6	2	8
BDOWNS	12	6	18
CROSS	12	6	18
DRIFTSR*	11	6	17
DUINEF	13	6	19
HANOV	9	6	15
KHAYOS	12	6	18
KRAAIF	14	6	20
LANOS	11	6	17
LANJAN	10	6	16
LANRD	11	6	17
MEW	12	6	18
MILNLM	8	6	14
MKFLAT	15	6	21
NEWF	8	6	14
NORTHP	8	6	14
NWPEN	9	6	15
NYANGR	11	6	17
PENEER	8	6	14
PHILEAST	15	6	21
PENEST	9	4	14
PHILWET	14	5	19
SAND	8	6	14
STRANDF	10	6	16
WETT	9	6	15
WOOD	6	6	12

A rudimentary prediction was made for estimating the number of years for all the sites to be extirpated excluding the 12 protected areas (1 664 records of biodiversity features; total of 514 features known only from these sites). The data points used were for 1997/1998, 2001 and the projection that only the 12 protected sites would remain. A linear decrease was based on the assumption that species loss is not synergistic whereas the converse was true for the exponential rate. In order to determine the rate of unidirectional loss, a linear ($Y=MX+C$) loss rate and an exponential ($Y=M_1X^2+M_2X+C$ i.e. a polynomial curve) loss rate were assumed [Y=number of unique biodiversity features/number of sites, M is the slope of the curve, C is a constant and X=number of records of biodiversity features/time (Zar, 1984)]. Once the relationships were established, the equations were solved for time. The key assumption for this prediction was that existing reserves retain all the biodiversity features recorded on them.

2.2 RESERVE SELECTION ALGORITHMS

The target of identifying complementary sites for conserving each species and FHU at least once (minimum set) was achieved by using a richness-based iterative algorithm modified from Wessels *et al.* (2000). The algorithm was modified to exclude adjacency rules for the purposes of this study. The basis for this decision was that it is not feasible to consider connectivity if sites are in close proximity to each other and the surrounding matrix is hostile in terms of threats (Shafer, 2001). Site flexibility options (alternatives to achieving the conservation goal) were not considered because it was felt that scarce resources (availability of viable land for conservation) on the Cape Flats disallow for using flexible sites as alternative 'bargaining chips' for maximising conservation (Dewar & Watson, 1990; Behrens & Watson, 1992) (see *Chapter 1*). An additional justification is that flexibility options are greatly reduced in areas of high beta and gamma diversity (see Lombard *et al.*, 1999), a situation akin to the Cape Flats.

The first conservation scenario set out to identify minimum sets for both the 1997/1998 and 2001 situation and assumed an ideal world where no constraints were present. An ideal-world scenario refers to a situation that does not take existing resources into consideration. The second scenario identified minimum sets in a systematic, step-wise fashion by selecting areas that were 0-10%, 11-20%, 21-30%...91-100% affected by biological threats (a low Vt-index) for both 1997/1998 and 2001. The third scenario was aimed at selecting sites in the same step-wise fashion but on the basis of support (a high Vs-index). Once chosen, a site was considered nominally conserved as were its biodiversity features, and both were removed from the subsequent analyses so that selected sites complemented each other. The target of all these scenarios was to minimise the number of sites required to represent all the biodiversity features at least once (species and FHUs). The algorithm for the second and third scenario is thus based on real-world trade-offs between the primary goals of (i) discriminating against biological threatening processes (high Vt-index) and (ii) capitalising on actual and potential support/defense (high Vs-index).

The size of sites was not a consideration in the analyses as size does not control species richness on the Cape Flats but does determine the number of habitats (*Chapter 3*) (but see Cowling & Bond, 1991). It has been demonstrated that maintaining habitat heterogeneity on the Cape Flats, and minimising human-induced threats and increasing support (the V-index from *Chapter 3*), are pivotal in reducing species extinction risks. By introducing FHUs into the reserve selection procedures, attention is given to species representation, in addition to incorporating a surrogate for the retention of ecological processes and so to prolong species persistence.

This study utilised a measure of irreplaceability and vulnerability to clarify the scheduling of conservation action (priorities). Irreplaceability was taken as the inverse rank order of the most efficient minimum set and vulnerability is the V-index (Table 4.1).

3. RESULTS

3.1 CONSERVATION OPTIONS IN 1997/1998

Minimum sets produced under the ideal-world scenario (absence of constraints) and the two real-world constraints of threat and support were congruent and produced very similar rankings (Table 4.2). All the 14 conserved areas were included in all these minimum sets. In all cases, more than 75% of the sites were selected to achieve the targets (≥ 45/59 sites).

In an ideal world, 45 sites were required for the target of representing all the biodiversity features at least once (Table 4.2). The minimum set incorporating the support constraint produced an exact duplicate of the minimum set in an ideal world. However, by considering threat constraints, only a single additional site to the sets above, Wingfield (WING), was required to fulfill the conservation target (minimum set of 46 sites). Wingfield contributed two complementary species that are common outside the Cape Flats (*Lampranthus aureus* (L.) N.E.Br., associated with granite outcrops, and *Trachyandra brachypoda* (Baker) Oberm., associated with seasonally marshy flats) (*Chapter 2*, Table 2.1) and one habitat (Habitat 30, Sand Plain Fynbos: standing seasonal wetland in deep acid sand on a flat and subdued plain at a low altitude) (Appendix 2) all of which were not represented on the other sites of the minimum set that were developed from an ideal- world situation. The top two sites in each of these three minimum sets were Tygerberg (TYGNR) and Rondevlei Nature Reserves (RONDVL), which collectively contributed 46% (572/1254) unique biodiversity features to the minimum sets. Biodiversity features were thus rapidly added during the first tenth of the algorithm curves, after which progress decelerated (Figure 4.1).

Table 4.2 Conservation options in 1997/1998. Algorithm results of the minimum sets of sites to conserve 1 254 biodiversity features (1 192 species and 62 FHUs). % features accumulated at two decimal places. *denotes conserved sites. The full site names are given in Appendix 1.

	UNCONSTRAINED			THREAT CONSTRAINT			SUPPORT CONSTRAINT		
Order of selection	*Site*	*Feature contribution*	*% features accumulated*	*Site*	*Feature contribution*	*% features accumulated*	*Site*	*Feature contribution*	*% features accumulated*
1	TYGNR*	417	66.74	TYGNR*	417	66.74	TYGNR*	417	66.74
2	RONDVL*	155	87.63	RONDVL*	155	87.63	RONDVL*	155	87.63
3	RONDCO	107	91.46	DURBNR*	83	93.38	DURBNR*	83	93.38
4	DURBNR*	76	93.93	CFNR*	65	94.81	CFNR*	65	94.81
5	KENRC	64	94.89	PLATTE*	48	96.17	BLOUVL*	47	96.25
6	MACASS	61	95.13	BLOUVL*	40	96.81	BLOUREN	41	96.73
7	CFNR*	41	96.73	DURBRC	22	98.24	WOLFNR*	33	97.20
8	BLOUFYN	37	97.04	75MB*	14	98.88	RIET*	15	98.80
9	BLOUVL*	32	97.44	EDITHNR*	12	99.04	CAPRI*	10	99.20
10	BLOUREN	30	97.60	WOOD*	8	99.36	RONDCO	82	93.46
11	MILNRC*	25	98.00	RONDCO	77	93.85	PLATTE*	31	97.52
12	PLATTE*	20	98.40	BLOUSTR	26	97.92	MILNRC*	30	97.60
13	SWART	17	98.64	RIET*	13	98.96	SOMCH*	20	98.40
14	BLOUSTR	16	98.72	CAPRI*	11	99.12	BRACK	13	98.96
15	N7N1	14	98.88	YOUNG	4	99.68	EDITHNR*	10	99.20
16	WOLFNR*	14	98.88	RONDEA	3	99.76	KENRC	47	96.25
17	BRACK	12	99.04	WING	3	99.76	BLOUSTR	18	98.56
18	75MB*	11	99.12	KENRC	42	96.65	RONDEA	1	99.92
19	SOMCH*	10	99.20	BLOUREN	31	97.52	BLOUFYN	17	98.64
20	DURBRC	9	99.28	MACASS	25	98.00	SWART	15	98.80
21	EDITHNR*	9	99.28	MILNRC*	22	98.24	N7N1	13	98.96
22	WELM	8	99.36	SWART	17	98.64	75MB*	11	99.12
23	RIET*	8	99.36	BLOUFYN	13	98.96	DURBRC	9	99.28
24	CAPRI*	7	99.44	WOLFNR*	13	98.96	MACASS	8	99.36

Continues overleaf

25	NORTHP	6	99.52	BRACK	12	99.04	PELZEEK	[illegible]	99.92
26	MILNLM	6	99.52	NTNT	11	99.12	WELM	8	99.36
27	KRAAIF	6	99.52	SOMCHP	10	99.20	NORTHP	6	99.52
28	BDOWNS	4	99.68	WELM	8	99.36	MILNLM	6	99.52
29	BELHAR	4	99.68	NORTHP	6	99.52	KRAAIF	6	99.52
30	LANJAN	4	99.68	MILNLM	6	99.52	BDOWNS	4	99.68
31	DRIFTSR*	3	99.76	KRAAIF	6	99.52	BELHAR	4	99.68
32	NWPEN	3	99.76	BDOWNS	4	99.68	LANJAN	4	99.68
33	BLKFAUR	3	99.76	BELHAR	4	99.68	DRIFTSR*	3	99.76
34	WOOD	3	99.76	LANJAN	4	99.68	NWPEN	3	99.76
35	KHAYOS	2	99.84	DRIFTSR*	3	99.76	BLKFAUR	3	99.76
36	MKFLAT	1	99.92	NWPEN	3	99.76	WOOD	3	99.76
37	PELZEEK	1	99.92	BLKFAUR	3	99.76	KHAYOS	2	99.84
38	TOKER	1	99.92	KHAYOS	2	99.84	MKFLAT	1	99.92
39	EERQR	1	99.92	MKFLAT	1	99.92	TOKER	1	99.92
40	AECI	1	99.92	PELZEEK	1	99.92	EERQR	1	99.92
41	PHILEAST	1	99.92	TOKER	1	99.92	AECI	1	99.92
42	RONDEA	1	99.92	EERQR	1	99.92	PHILEAST	1	99.92
43	STRANDF	1	99.92	AECI	1	99.92	STRANDF	1	99.92
44	YOUNG	1	99.92	PHILEAST	1	99.92	YOUNG	1	99.92
45	HANOV	1	100	STRANDF	1	99.92	HANOV	1	100
46				HANOV	1	100			

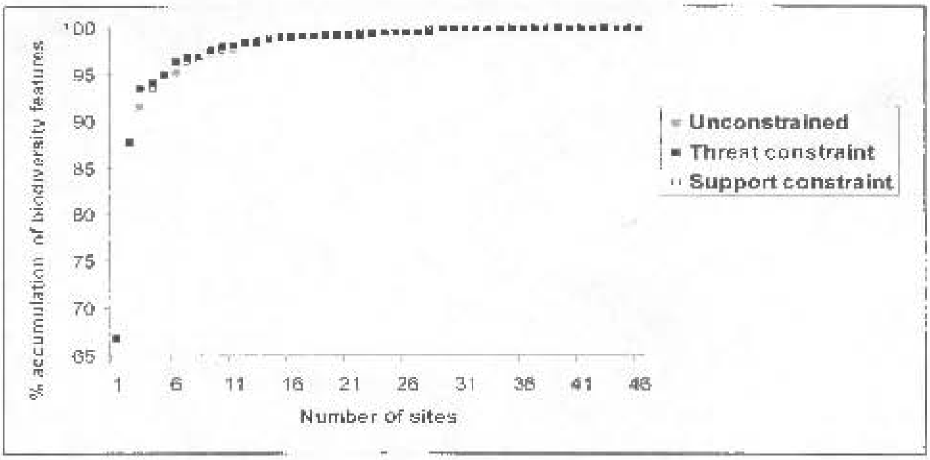

Figure 4.1 Minimum sets in 1997/1998: Efficiency of ideal-world (unconstrained) and real-world (constrained) algorithms.

The algorithm constrained site selection to those with a high Vs-index and a low Vt-index (Figure 4.1). All the sites of the minimum set selected under the threat constraint were extracted from sites that were weakly to moderately threatened (Vt-index >30% and ≤ 60%), but with 63% of the sites from the >51 - ≤60% sub-class (Figure 4.2). On the other hand, all the sites of the minimum set selected under the support constraint were drawn from sites scattered throughout the entire range of sub-classes (Vs-index >30% and ≤ 100%).

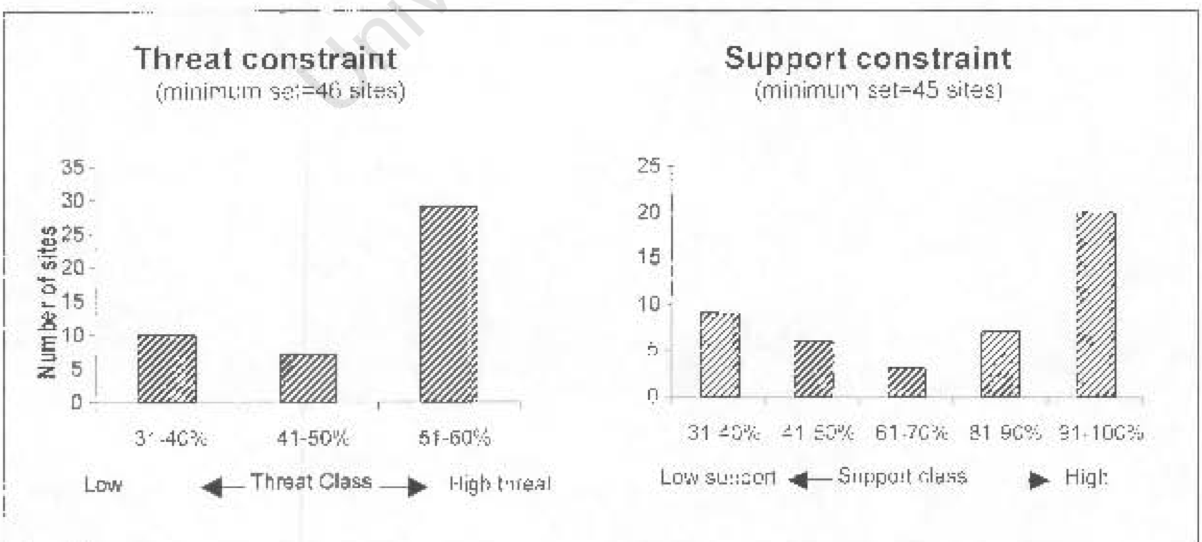

Figure 4.2 1997/1998: Constraint classes from which minimum sets were drawn.

The minimum set generated under the support constraint appeared to be the most feasible option since it required 45 sites and there were only eight instances where sites contained single complementary features (Table 4.2). This is in contrast to the minimum set generated without constraints which required 45 sites (10 sites each with a single complementary feature) and the minimum set that considered threats (46 sites; 8 sites each contributing a single complementary feature).

3.2 CONSERVATION OPTIONS IN 2001

The most discernible difference between conservation options in 1997/1998 and 2001, is that the minimum sets generated in 2001 required virtually all the sites from the selection pool (Table 4.3). The minimum sets produced under the threat constraint required all the available sites (31/31) whereas the minimum set under the support constraint requires 30 sites. All the 12 remaining conservation areas were represented in all the minimum sets. Wingfield (WING) only appeared in the minimum set generated under the threat constraint, contributing the same complementary biodiversity features as in 1997/1998. In all cases, the top two sites in the minimum sets (Tygerberg (TYGNR) and Rondevlei Nature Reserves (RONDVL)) conserve 572 out of 1186 features (48%) (Table 4.3).

The efficiency of the constrained algorithms was clearly compromised to avoid highly threatened sites and sites that enjoyed little or no public/institutional support, and this can be seen as a cost. The algorithms decelerated periodically as they attempted to represent the maximum number of biodiversity features within successive sets of sites associated with low Vt-index (biological threats) and Vs-index (support) measures (Figure 4.3). This is in contrast to the algorithm results in 1997/1998 that selected sites in a smooth manner (Figure 4.1).

The minimum set generated under the support constraint appeared to be the most feasible option since the minimum set comprised 30 sites and there were only five instances where sites contributed a single complementary feature (Table 4.3). This is in contrast to the minimum set generated under the threat constraint (31 sites; 5 sites contribute a single complementary feature) and the 'constraint-free' situation (30 sites; 7 sites contribute single complementary features).

Table 4.3 Conservation options in 2001. Algorithm results of the minimum sets of sites to conserve 1 186 biodiversity features (1 129 species and 57 FHUs). *denotes conserved sites. % features accumulated at two decimal places. The full site names are given in Appendix 1.

	UNCONSTRAINED			THREAT CONSTRAINT			SUPPORT CONSTRAINT		
Order of selection	*Site*	*Feature contribution*	*% features accumulated*	*Site*	*Feature contribution*	*% features accumulated*	*Site*	*Feature contribution*	*% features accumulated*
1	TYGNR*	417	64.84	TYGNR*	417	64.83	TYGNR*	417	64.83
2	RONDVL*	155	86.93	RONDVL*	155	86.93	RONDVL*	155	86.93
3	RONDCO	107	90.98	DURBNR*	83	93.00	DURBNR*	83	93.00
4	DURBNR*	[illegible]	93.59	CFNR*	65	94.52	CFNR*	65	94.51
5	KENRC	64	94.60	PLATTE*	48	95.95	BLOUREN	42	96.45
6	MACASS	51	94.86	DURBRC	24	97.97	WOLFNR*	37	96.88
7	CFNR*	41	96.54	75MB*	15	98.73	RIET*	17	98.56
8	BLOUFYN	37	96.88	EDITHNR*	14	98.81	CAPRI*	10	99.15
9	MILNRC*	32	97.30	RONDCO	79	93.33	RONDCO	89	92.49
10	BLOUREN	27	97.72	BLOUSTR	32	97.30	MILNRC*	37	96.88
11	PLATTE*	21	98.23	RIET*	14	98.81	PLATTE*	32	97.30
12	BLOUSTR	18	98.48	CAPRI*	12	98.98	SOMCH*	20	98.31
13	SWART	17	98.57	YOUNG	4	99.66	BRACK	13	98.90
14	N7N1	15	98.74	RONDEA	3	99.74	EDITHNR*	11	99.07
15	WOLFNR*	15	98.74	WING	3	99.74	KENRC	48	95.95
16	BRACK	12	98.99	KENRC	43	96.37	BLOUSTR	20	98.31
17	75MB*	11	99.07	BLOUREN	32	97.30	RONDEA	1	99.91
18	EDITHNR*	10	99.16	MILNRC*	26	97.80	BLOUFYN	18	98.48
19	SOMCH*	10	99.16	MACASS	25	97.89	N7N1	15	98.73
20	DURBRC	9	99.24	SWART	17	98.56	SWART	14	98.81
21	RIET*	9	99.24	BLOUFYN	14	98.81	75MB*	11	99.07
22	WELM	8	99.33	WOLFNR*	[illegible]	98.81	DURBRC	9	99.24
23	CAPRI*	7	99.41	N7N1	12	98.98	MACASS	8	99.32
24	PELZEEK	1	99.92	BRACK	12	98.98	PELZEEK	1	99.91
25	TOKER	1	99.92	SOMCH*	10	99.15	WELM	8	99.32
26	EERQR	1	99.92	WELM	8	99.32	TOKER	1	99.91
27	AECI	1	99.92	PELZEEK	1	99.92	EERQR	1	99.92

Continues overleaf

28	DEA	1	99.92	TOKER	1	99.92	AECI	1	99.92
29	[illegible]	1	99.92	EERQR	1	99.92	YOUNG	1	99.92
30	EDITH	1	100.00	AECI	1	99.92	EDITH		100
31				EDITH	1	100			

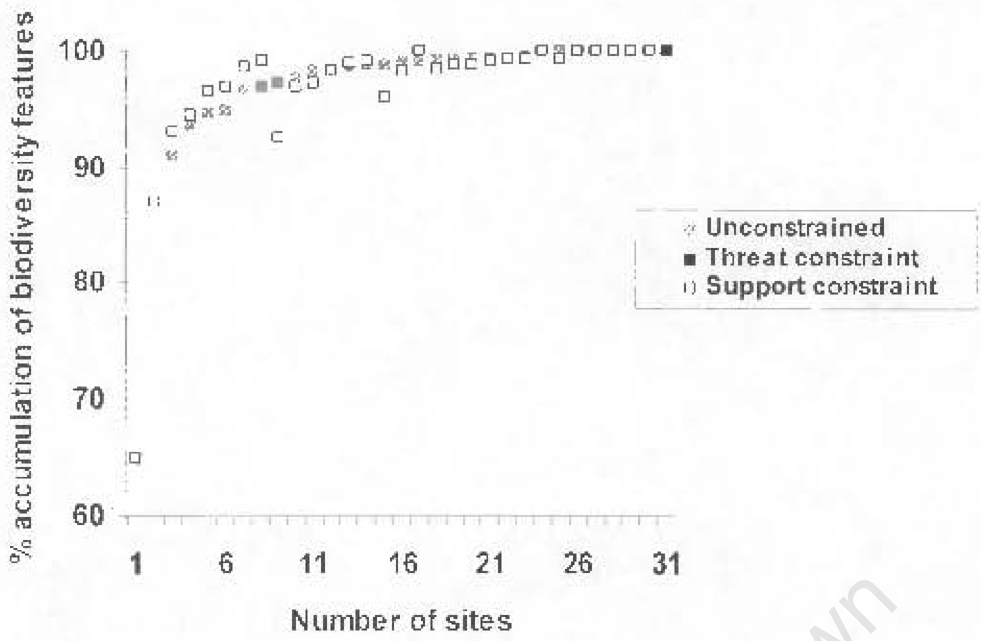

Figure 4.3 Minimum sets in 2001: Efficiency of ideal-world (unconstrained) and real-world (constrained) algorithms.

All the sites of the minimum set produced under the threats constraint were selected from sites that were weakly to moderately threatened (Vt-index >30% and ≤ 60%) (Figure 4.4), whereas all the sites of the minimum set selected under the support constraint were sourced from sites dispersed throughout the entire range of sub-classes (Vs-index >30% and ≤100%). This is a very similar to the trend in 1997/1998 (Figure 4.2).

In 1997/1998 and in terms of land acquisition, the most economical minimum set solution (support constraint) required a total of 6 080.6 ha land area to achieve the reservation target (Figure 4.5). At the time, 2 165.0 ha (36%) was reserved (14 protected areas). Thus, 31 additional sites of total size 3 915.6 ha needed to be secured to achieve the conservation goal. The situation looks very different five years later. By 2001, the available land surface area for conservation had been reduced by *ca* 15%. The cheapest minimum set solution (support constraint) in terms of land area, was that a total of 5 187.9 ha was required. Some 29% (1 495.2 ha) is currently being conserved (12 protected areas). Twenty additional sites of total size 3 692.6 ha require protection status to conserve all species and FHUs at least once i.e. to achieve the target.

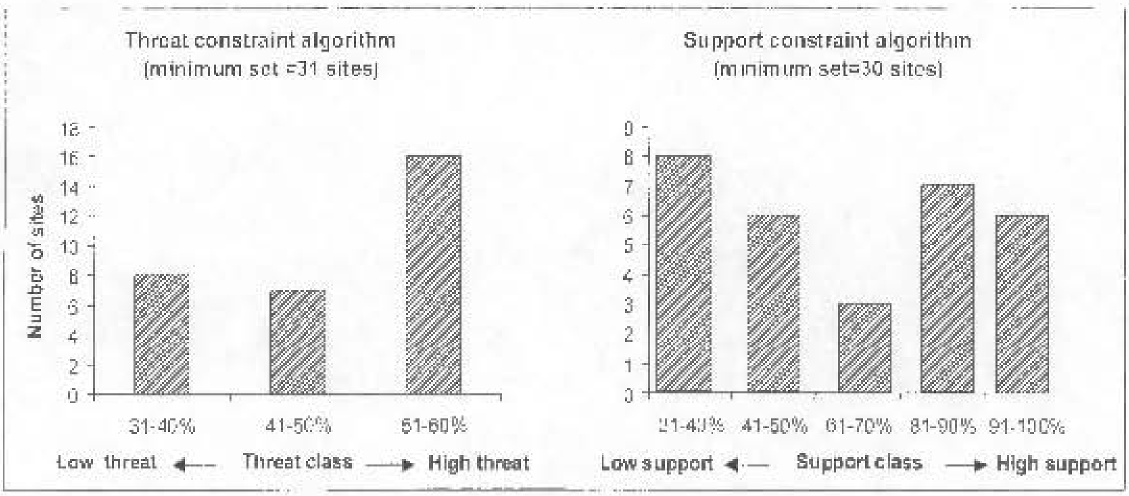

Figure 4.4 2001: Constraint classes from which minimum sets were drawn.

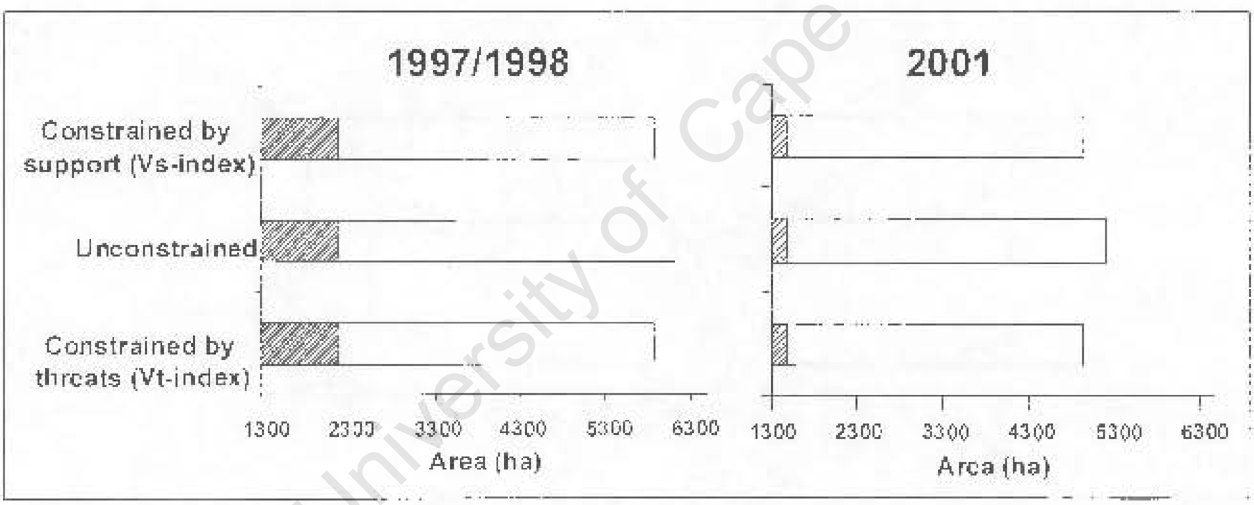

Figure 4.5 Land area of minimum sets. Filled = Reserved, Unfilled = Unreserved.

3.3 MAKING WORST-CASE PREDICTIONS BASED ON A LOSS AUDIT

In 1997/1998, 59 sites were available for conservation, and they contained 1254 biodiversity features (4 410 records of features). By 2001, 28 sites had been extirpated (including two of the 14 reserves), leaving only 31 sites available for reserve selection. These remaining sites consist of 3 494 records of biodiversity features of which 1186 are unique (1 129 species and 5 FHUs). The loss over this five-year time-frame represents 68 biodiversity features (63 species and 5 FHUs) which were unique to the extirpated sites.

There is the possibility that all the sites identified in the smallest minimum will not be secured, and that only the 12 existing protected areas will remain. This possibility presents one future scenario (best of the worst) for the Cape Flats flora if threatening processes continue unabated and if civil society/institutional support is not sufficiently mobilised. It is predicted that if the decline of species and FHUs is linear, then it will take 14.7 years from now for all the available remnants to become extirpated, leaving only the existing reserves with its complement of species and FHUs. Should the loss be exponential, then it will take 9.6 years from now (Figure 4.6).

The importance of this prediction, albeit rudimentary (based on three data points), is that the conservation of existing sites over the next 10 to 15 years is crucial to prevent the likelihood of this worst-case scenario from becoming a reality.

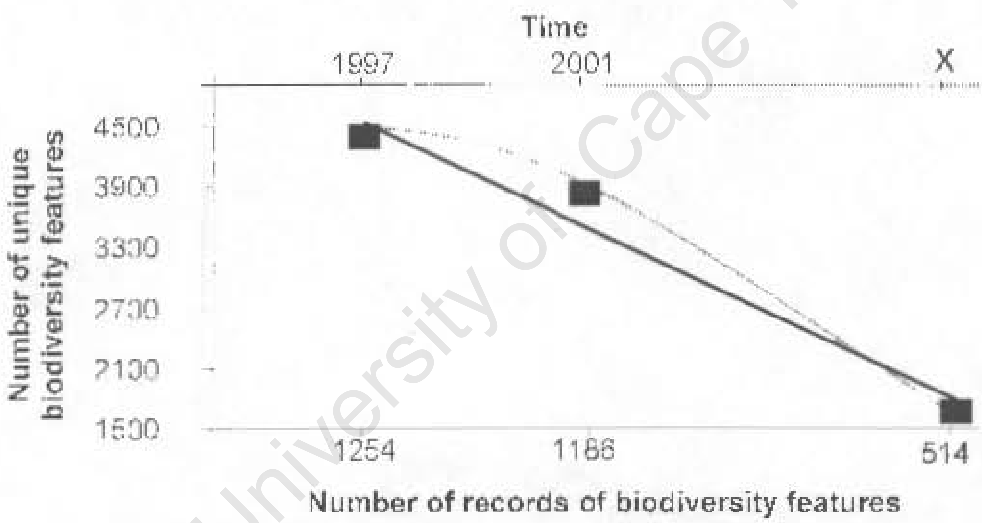

Figure 4.6 Predicted rate of the loss of biodiversity features for the worst case scenario of only the 12 reserves remaining over time. Exponential rate (broken line): $y=-457x^2+455x+4412$, $r^2 = 0.998$, $p<0.0001$; Linear rate (unbroken line): $y=-1373x+5935.3$, $r^2 = 0.964$, $p<0.001$. X is the time taken for all the protected areas to become extirpated.

4. DISCUSSION

To date, iterative studies on the selection of priority sites on the Cape Flats have been characterised by species-based approaches incorporating larger data sets than before (more sites and more species) (McKenzie & Rebelo, 1997; Maze & Rebelo, 1999). These have contributed relatively little to the understanding of how urban processes should be combined with identifying remnants bearing conservation potential with the aim of prolonging the retention of species and the persistence of ecological processes.

This study has attempted to integrate the realm of conflict between support and threatening processes, with conservation interests. In the highly transformed landscape of the Cape Flats, real- and ideal-world algorithm outputs have been shown to be almost convergent for two chronological comparisons: 1997/1998 and 2001. There is also a high degree of congruency between the minimum set outputs of constrained algorithms for 1997/1998 and 2001 respectively, although support-constrained solutions are only marginally better options.

The altered landscape mosaic has resulted in the extirpation of 28 sites, including two reserved areas, and this has significantly reshuffled the available conservation options. In 2001, all the available sites exhibiting conservation potential have been shown to be irreplaceable in terms of featuring all species and all FHUs at least once in the minimum set. The situation on the Cape Flats has been shown to have drastically worsened over the last five years in terms of reductions in species numbers and heterogeneity. If unattended, this downward spiral may result in all the species and FHUs present on remaining fragments that are currently suitable for conservation being lost within the next 10 to 15 years.

The paucity of land availability for conservation on the Cape Flats has placed restrictions on a number of issues relating to the approach and methodology of this study (see Pressey *et al.*, 1993). Firstly, it prohibited flexibility analyses that would have resulted in a broader spectrum for land negotiation. Secondly, it precluded selection procedures run using adjacency rules (selecting areas in closest proximity to each other). The recent work of Schafer (2001) suggests the alternative, that selecting sites furthest apart may contribute to lessening the likelihood of extinctions particularly when threats are common to remnants across a study area. This challenge to existing conservation dogma that it is more feasible to select sites that are in closest proximity to each other, requires empirical research for it to have been integrated into reserve selection procedures for the Cape Flats. Lastly, the land restriction forced a minimum set approach of protecting biodiversity features at least once. The approach of selecting species (i.e. one subpopulation of each species) and FHUs at least once in the identification of priority sites in this study was the most feasible option.

4.1 Adequacy of the existing reserve network

Compared with other biodiversity assessments, where only 8% (Bega Valley, New South Wales, Australia: Keith, 1995) or 7% of natural vegetation remains (Western Australia wheatbelt: Saunders *et al.*, 1993), the biodiversity of the Cape Flats appears to be in a more dire situation than is prevailing elsewhere in the world. The high number of Red Data List species (RDLs, 47) and the fact that the two vegetation types (Blackheath Fynbos and Cape Flats Fynbos/Thicket Mosaic) are centered on the Cape Flats is cause for concern (Cowling & Heijnis, 2001) (*Chapter 1*).

In Amazonia, some 25% of the known plant species are currently represented in reserves (Peres & Terborgh, 1995); in Québec (Canada), 65% of threatened species exist outside protected areas (Sarakinos *et al.*, 2001); Trinder-Smith *et al.* (1996b) show that 6% (119/2159 species) of the species fall outside the reserve network of the Cape Peninsula and that existing reserves conserve 29% of its total area. In 1997/1998, the reserve estate of the Cape Flats conferred protection on some 76% of its species and 62% of its habitats. The extirpation of two reserves (BLOUVL and DRIFTSR) since then has resulted in a loss of 68 biodiversity features not recorded elsewhere in the study area; the extirpated sites that were unconserved resulted in a loss of 17 biodiversity features. Eleven extirpated sites that did not feature in the minimum set analyses for 1997/1998 had V-index values ranging from 14 to 19 (i.e. highly to extremely threatened by biological threats and lack of support) (Table 4.1). The other 17 extirpated sites had V-index values ranging from 12 to 21, with the exception of BLOUVL that had V-index value of 8. This is a reflection that the criteria used for calculating V-index were not weighted, and this represents shortcoming in the scoring technique (Gaston, 1996). Moreover, this highlights the inherent difficulties in objectively quantifying important non-biological threatening factors such as society-/instutional-based support and the capacity/willingness to support and enforce conservation actions (see *Chapter 3*).

The existing reserve network (12 conserved areas) contributed *ca* 72% of complementary biodiversity features (858-859) in the minimum set analyses (Table 4.3). However, an additional minimum of 3692.6 ha (20 sites) is required to protect the unreserved species and habitats at least once. Thus, by securing these additional sites, this system would protect 100% of the plant biodiversity of the Cape Flats (all species and FHUs protected at least once) in *ca* 7% of the total area of the Cape Flats.

Studies have shown that costs of correcting *ad hoc* conservation characteristically increase with time (higher land acquisition prices and more competing elements) (e.g. Pressey, 1994; Pressey & Tully, 1994). This study has shown that land availability on the Cape Flats has dwindled to such an extent that every fragment with conservation potential is required to achieve the conservation target, a situation different to 5 years ago.

The reserves on the Cape Flats then (1997/1998) and now (2001) generally do a satisfactory job of conserving biodiversity, and previous studies attempting to quantify the adequacy of the existing reserve network on the Cape Flats may have been unduly pessimistic (e.g. Jarman, 1986; Mckenzie & Rebelo, 1997; Maze & Rebelo, 1999). Attention should instead be directed at prospects for acquiring additional land for conservation and for enforcing policies and legislation that confer protection status. The strict endemics of the Cape Flats (*Erica verticillata, Euphorbia marlothiana, Isoetes stellenbossiensis, Serruria aemula* var. *congesta* and *S. aemula* var. *foeniculacea*) all exist in protected areas, with the exception of *Serruria furcellata* (presumed extinct) that was known from Northpine open space (extirpated site) (*Chapter 2*). The remaining RDLs and endemics occur on other sites beyond the Cape Flats. All the reserves were present in all the minimum sets for 1997/1998 and 2001. Because the reserves possess many unique species and FHUs, they generally contributed more complementary features to the minimum sets than unreserved sites. In all cases, contributions of single complementary features were drawn from unreserved land, thus indicating that protected areas on the Cape Flats are extraordinary reservoirs of biodiversity. There are several plausible reasons for this phenomenon. Firstly, assuming that all the sites were similarly rich and diverse against some historical benchmark, the loss of habitat heterogeneity coupled with the impact of threatening processes resulted in subsequent species declines on unreserved land. Reserved sites might have faired better at retaining levels of heterogeneity and buffering threatening processes compared to unreserved sites. *Chapter 2* has shown that, generally, reserves have a higher richness of all biodiversity compared to unreserved sites, but there was no statistical difference in the density of habitats and RDLs between reserved and unreserved sites. Alternatively, protected areas may originally have been established on the basis of high levels of biodiversity compared to other sites (i.e. historically, unreserved sites were depauperate). Lastly, the paradigm of species-based conservation is still heavily entrenched in conservation planning on the Cape Flats (e.g. Jarman, 1986; McDowell & Low, 1990; McKenzie & Rebelo, 1997; Maze & Rebelo, 1999), and this bias is evident in that species have a higher representation in reserves than habitats.

4.2 Additional reserved areas

In 1997/1998, 17 - 22% of sites in the minimum sets contributed a single complementary biodiversity feature (Table 4.2), whereas the order of magnitude of single feature contributions in 2001 is not much more efficient with regards to meeting the conservation target (*ca* 16%) (Table 4.3). In 1997/1998, the rejection of all sites contributing only a single complementary feature to the minimum set meant that > 99% of the reservation target of conserving each biodiversity feature at least once would still have been achieved; this generally still holds today. Attaining > 99% of a reservation target is more than satisfactory compared to modest achievements that are usually set at the order of 50 - 90% such as for biosphere reserve partitioning on the West Coast Lowlands, South Africa (Heijnis *et al.*, 1999), representation of threatened species in a reserve network for the Succulent Karoo in the Northern Cape, South Africa (Lombard *et al.*, 1999) and conserving special species in Addo Elephant Park in the Eastern Cape, South Africa (Lombard *et al.*, 2001).

There is an overarching consideration to be addressed when irreplaceable sites are designated on the basis of a single biodiversity feature. Firstly, that unique biodiversity feature (species or FHU) may be a 'false positive' in the presence/absence database. Many species have not been recollected in many years on the Cape Flats (many species are known from herbarium specimens or from taxonomic accounts such as monographs/conspecti). Without confirming the presence of species and their population viability through field work, it may not be desirable to accept *de facto* that the site is irreplaceable on the basis of a single species. FHUs can be similarly ground-truthed in the field. Further records may well reduce the number of sites contributing single complementary biodiversity feature to the minimum sets. Alternatively, single complementary features delimiting irreplaceable sites may be attributed to biodiversity features at the edge of their distribution range (i.e. possibly have a wide occurrence outside the study area). Complementary analyses are typically sensitive to 'false negatives' and 'false positives', particularly when abundance data are not used (Lombard, 1995). However, an advantage of this sensitivity is that the protection of cryptic species in marginal habitats and the protection of small and fragmented populations are not discriminated against compared to abundance data which biases against naturally rare species and habitats.

The possibility of affording protection status on unconserved sites on the Cape Flats that contribute only a single biodiversity feature is unlikely (for a species and even less likely for a habitat). It is also unlikely that all the irreplaceable sites identified in these analyses can be formally protected, but then this should be reconciled with the worst case scenario of only the existing reserves surviving the next decade. The forfeiture of delivering on the conservation target of protecting each species at least once comes at a high price – the local extinction of species and habitat assemblages on the Cape Flats.

In this study, Macassar (MACASS), Rietvlei Protected Area (RIET), Swartklip open space (SWART) and Wolfgat Nature Reserve (WOLFNR) were identified as fairly high priority sites, contributing between eight and 37 complementary biodiversity features to the minimum sets produced under the threat and support constraints (Table 4.3). In terms of representation alone, these sites featured as exceptionally high priority sites in the work of McKenzie & Rebelo (1997) and the earlier work of McDowell & Low (1990). However, these sites are not recognised as being important in Maze & Rebelo (1999), principally because of issues relating to scale. Maze & Rebelo (1999) set species representation targets for 118 remnants over a much larger area (Cape Flats *sensu lato*) (*Chapter 1*), and thus the effects of representing complementary species of these sites in a minimum set were diluted (there were other/better options). McKenzie & Rebelo (1997) included MACASS, RIET, SWART and WOLFNR as 'core sites', high priority sites designated as lobbying and advocacy focal points for the conservation of the Cape Flats flora. Ironically, their recommendations for these core sites include … "*We challenge the responsible authorities and landowners to provide adequate conservation status to the 15 core sites. We believe ensuring the conservation status of these sites is an appropriate measure and the least that should be done to fulfill the country's obligations under the International Convention on Biological Diversity*". The Cape Flats *sensu stricto* has been subjected to more intense and far more complex urbanisation ramifications over a longer period of time and has for long been recognised as a unit apart (flora, socio-economic dynamics and the land ownership climate) (Low, 1979; Hall, 1984; Dewar & Watson, 1990; McDowell & Low, 1990; McDowell *et al.*, 1991; Behrens & Watson, 1992; Bridgeman *et al.*, 1992; Wood *et al.*, 1994; McKenzie & Rebelo, 1997) (*Chapter 1*). Priority shifts (e.g. extending the boundaries of the study area) are a natural and inevitable progression in response to changing extraneous events (Pressey *et al.*, 1996). However, this is inherently a double-edged sword, particularly when the regional importance masks the local importance of high priority sites. This may subsequently lead to a halt in the implementation processes for the conservation of these sites due to what may be seen as ambiguous demands and conflicting efforts.

4.3 Congruent solutions and diminishing options

Real-world outlooks (Pressey, 1996) concede that only some, or at worse, no unconserved sites are available for future conservation purposes. At the city planning level of the Cape Flats, land acquisition is contentious (*Chapter 1*), and is based on the number of sites associated with budgetary cost of management, maintenance and staffing of reserves (Behrens & Watson, 1992). Thus, acknowledging real-world limitations (i.e. limited site availability/suitability for conservation and capitalising on the existing reserve situation) presents one realistic way of achieving a sustainable conservation solution. The larger the minimum set of sites (and the greater the total size (area) of the minimum set), the greater the problem of implementation as resources need to be spread (Pressey, 1994; Pressey & Tulley, 1994).

In pristine landscapes, real-world and ideal-world scenarios are likely to be the same (see Lombard *et al.*, 1999). But, it appears that in highly transformed landscapes, such as the Cape Flats, this is the same. The major reason for this is offset by two extremes: high species and habitat richness (many unique biodiversity features) occur in a critical mass of sites (reserves), and the remainder of the sites is relatively depauperate (very few unique biodiversity features). Coupled with limited land availability (small sample sizes, n=59 in 1997/1998 and n=31 in 2001), the options for identifying sites to reach conservation targets are greatly reduced because the sites (reserves) bearing the highest richness and contributing the most complementary features are consistently selected as top priority (see *Chapter 2*).

Many years of land protection without coordination and a broad-scale plan (e.g. Jarman, 1986; McDowell & Low, 1990) has created threats in the form of biologically destructive processes and a lack of public/institutional support. Two alternative realistic approaches for protecting the biodiversity of the Cape Flats have been examined against a scenario that assumed the absence of constraints. The disadvantage of incorporating constraints carries the cost of selecting a larger total number of sites, and hence larger human and fiscal efforts for acquiring additional land surface area for conservation. It was expected that the minimum set produced under the support constraint (Vs-index) would have a more marked influence on the reserve selection outcomes (at least in 1997/1998). The rationale for this expectation is that the Cape Flats is a densely populated urban area inferred to be governed by social pressures with respect to lobbying and advocacy for conservation (Folke *et al.*, 1996). However, the consideration of threat constraints (Vt-index) did not appear very different (high degree of overlap of the minimum sets) to the support constraint consideration in the past (1997/1998) nor in the present (2001). In the one instance, priority sites were weakly to moderately threatened by destructive urban processes (Vt-index). In the other, priority sites ranged from weakly to highly threatened by the absence, inadequacy and inappropriateness of support emanating from the public and institutional sectors (Vs-index). Interestingly, the minimum set produced under the support constraint directly overlapped (a direct duplicate) with the ideal-world output (for both 1997/1998 and 2001). In order to implement the most feasible minimum set solution, increasing and restructuring support (obtaining protection status for sites and raising levels of societal support) in addition to minimising threats will have to be carried out. Sustaining habitat heterogeneity by alleviating threatening processes (principally the amelioration of alien plant infestations, removal of rubble and instatement of appropriate fire regimes) greatly reduces the risk of species loss (*Chapter 3*), and this can duly facilitate the implementation of reservation targets.

4.4 Prolonging persistence

Pressey (1994) pointed out that assumed relationships between species distribution environmental classes are often reflected in discrepancies in protected area partitioning. In addition, assumptions of these relationships, together with using coarse selection surrogates may detrimentally result in rare species confined to cryptic habitats 'falling through the net'. Despite the obvious shortcoming of species-based reserve selection approaches in conservation planning, higher order surrogates may well fail to identify the composition, configuration and quantity of elements necessary for retention and persistence, making species data an essential component of the conservation planning processes (Lambeck, 1997). The shortcomings associated with species distribution data and the limitations of surrogates in the selection of priority conservation sites suggest that a combination of the two may yield more optimal solutions. The effect of integrating species and FHUs in this study sought to minimise these limitations.

Various traits of species and ecological processes (e.g. life-history, reproductive strategies, growth form and so forth) which increase extinction risk represent important, long-term (evolutionary) value to conservation (Cowling *et al.*, 1999; see Cowling & Pressey, 2001 for an overview). For example, 'dispersal-limited' species (e.g. specialists and mutualists) may define characteristics of linking remnants; 'process-limited' species (e.g. short-lived reseeders) may define the rates and intensities at which threatening processes should be managed (Bond, 1994; Bond, 1998). Levels of habitat heterogeneity and the deferral of threatening processes largely determine species richness on the Cape Flats (*Chapter 3*). Lombard *et al.* (1997) attempted to accommodate ongoing evolutionary processes in reserve design by using contiguous edaphic gradients (habitats) for lowland fynbos, since edaphic (ecological) speciation appears to be the primary force driving diversification. The concept of FHUs in this study is applied in the same manner as Lombard *et al.* (1997). To this end, the retention of parcels of heterogeneity (FHUs) in reserve networks provides one way to prolong the persistence of the various special and vulnerable traits of the Cape Flats biota. The persistence of evolutionary processes on the Cape Flats should make for an interesting study, but it may not be feasible in the light of ongoing destructive urban activities that have disrupted long ecological gradients that are prerequisites for persistence.

There is much uncertainty regarding the persistence of the Cape Flats flora over longer time periods. Global Climate Models predict increased temperatures and precipitation variability in the years to come, and fire regime systems, ecological diversity, hydrology and atmospheric feed-back systems in mediterranean-climate urban areas will be most affected (Lavorel *et al.*, 1998). In South Africa, it has been predicted that climatic conditions will dry out and that temperatures will rise, and this will see an eastward shift in range extensions of species (the mediterranean climate zone of western South Africa will be most affected) (Rutherford *et al.*, 1999). Since we do not yet understand the ecological implications of these dynamics on the persistence of the Cape Flats flora and how and whether species will be able to track climate-induced range shifts, the use of conservation planning goals that cater exclusively for the promotion of evolutionary change may be regarded with great skepticism amidst scarce conservation resources. The integration of representation and retention goals in this study presents a unique opportunity to maintain current conservation interest on the Cape Flats.

4.5 Approaches to implementation

The optimal rate of accumulation of species as more sites are added to a notional network, does not necessarily indicate the best sequence of protection (Pressey *et al.*, 1996). In other words, the selection order of sites sheds no light on the extent to which the sites require immediate protection. The most appropriate order of protection (implementation of priorities) is one that takes into account some measure of irreplaceability and vulnerability so that maximum options for achieving delivery on reservation targets are open five, ten or fifteen years hence. The literature is replete on this issue, and in South Africa, priorities for action have been established using irreplaceability/vulnerability analyses for the implementation of reserve systems (e.g. Heijnis *et al.*, 1999; Lombard *et al.*, 2001).

The scheduling of conservation action used measures of irreplaceability and vulnerability. Irreplaceability was taken as the inverse of the rank order of the minimum set produced under the support constraint (the most feasible solution) (Table 4.3). Figure 4.7 shows that sites requiring first priority are reserves, namely Wolfgat (WOLFNR) and Edith Stevens (EDITHNR), as they are highly irreplaceable and highly vulnerable. The vast majority of the Cape Flats reserves are highly irreplaceable and have a low vulnerability compared to the other sites, which is the desired state. Pressey *et al.* (1996) recommends that if threatening processes can be readily reversed or when conservation resources are sufficient such that the rate of implementation into a reserve system is expected to be rapid, then the next level of priority should probably be sites with a lower irreplaceability and a high vulnerability. Thus, Blouberg Fynbos site (BLOUFYN), Macassar Dunes (MACASS), Swartklip open space (SWART), the Pelican Park-Zeekoeivlei Flats (PELZEEK) and so forth should form the next highest priorities.

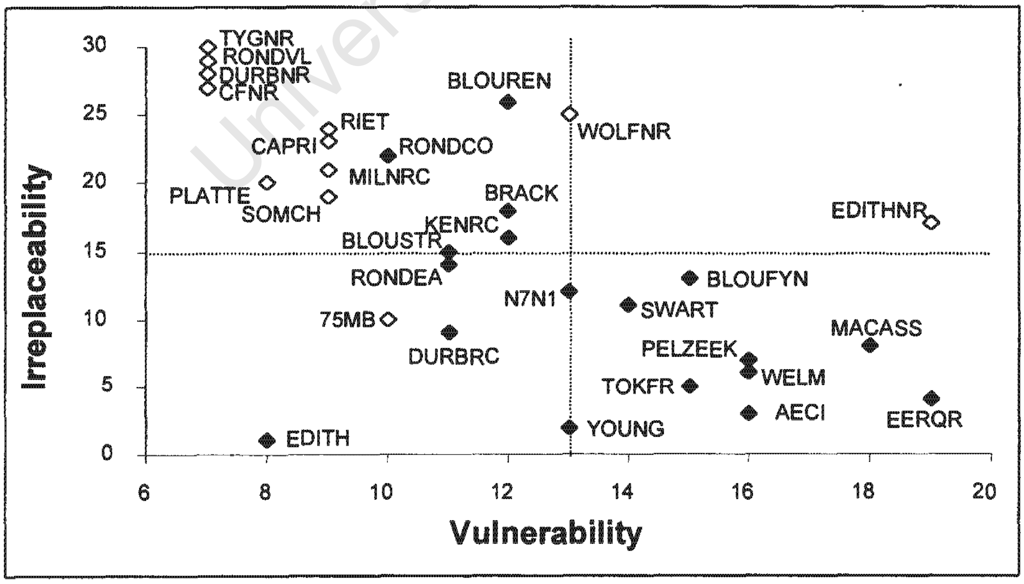

Figure 4.7 Conservation priorities for 2001 based on the minimum set developed under the support constraint. Open cells represent reserved sites; closed cells represent unprotected sites. Full site names and protection status are given in Appendix 1.

If insufficient resources are available, it may be wiser to shift priorities to Blouberg Renosterveld (BLOUREN)/Strandveld (BLOUSTR), Kenilworth Racecourse (KENRC) and Rondebosch Common (RONDCO). With a database characterised by many unreserved sites contributing only single complementary features, substantial changes to this implementation sequence for the Cape Flats is not envisaged (cf. high local endemism in Lombard *et al.*, 1999) unless sites are extirpated, and then this requires a new process of re-analysis.

Intervention strategies need to be identified and implemented immediately to prevent the likelihood of worst-case scenarios, for example, that all sites will be extirpated with the exception of the 12 protected areas. The conservation status of the existing reserves needs enforcement at the legal, political and social level (see Maze & Rebelo, 1999). In order to achieve 100% of the conservation target, competing elements of land prices, provision of resources and negotiations amongst stakeholders should take place in a multi-disciplinary climate to reconcile these conflicting issues. Designating additional sites with the aim of protecting each biodiversity feature at least once is unlikely to be socially and economically feasible.

Alternative adaptive plans will have to be forged to reduce biological (Vt-index) and support (Vs-index) threats, and the benefits of protecting the Cape Flats flora should be socio-economically justifiable. For example, nearly all the irreplaceable sites identified in this study surround the development periphery (the urban core) of the Cape Flats (e.g. Tygerberg Nature Reserve, the Blouberg sites and Rondevlei Nature Reserve)(Figure 2.1). These sites are adjacent to the more pristine, upland areas (which are better managed) of the Cape Peninsula (Trinder-Smith *et al.*, 1996b) and the area proposed as the West Coast Biosphere that lies to the north of the Cape Flats (Heijnis *et al.*, 1999) that were not included in this study. High priority sites of the Cape Flats can serve as significant buffer sites against deleterious biological threats (Vt-index) to these ecotourist destinations, and this could well form the basis for exploiting other opportunities to preserve the remnants of the Cape Flats flora. Conservation planning work in the Cape Metropolitan Area should treat the Cape Flats as a unique entity on the basis of its unique threats. This protocol should be integrated into general conservation planning frameworks of developing reserve systems at appropriate scales in relation to the prevailing real-world issues rather than attempting to develop reserve systems across very different areas.

This greatly restricts the choices of reservation targets and implementation of conservation plans that may not be sensitive to local scale threats.

4. 6 General predictions

Indicators of vulnerability, such as the V-index, will shift increasingly in magnitude as threatening processes become more intense over time on the Cape Flats. Ultimately, vulnerability dimensions may probably be discarded in priority analyses when each natural remnant becomes irreplaceable (valuable) purely in terms of its status as an undeveloped open space (Behrens & Watson, 1992; Wood *et al.*, 1994). Even so, relative priorities measured against vulnerability should still be developed. For example, in the Western Australian wheatbelt (Hobbs, 1993), all the sites under selection are both highly vulnerable and highly irreplaceable sites (Pressey, *pers. comm.*, 1998) – and this is the same for the Renosterveld wheatlands that lie to the north of the Cape Flats (Low & Rebelo, 1996; Kemper *et al.*, 1999; Cowling *et al.*, 1999).

Pressey (1996) forewarns that if priority setting ceases in heavily transformed landscapes, it may result in conservation practitioners undertaking haphazard land acquisition procedures for conservation purposes and that this would be coupled with the tendency to conserve any land fragment that becomes available for that purpose. Intense land pressures and high land ('erf') prices will motivate conservation in scattered, small and remote areas (beyond the urban core) where benefits to the residents of the Cape Flats may be questionable (see Pressey, 1994). At this point then, it becomes crucial to refine measures of conservation value (irreplaceability) and to state even more explicit and realistic reservation targets. Lastly, it becomes imperative that reservation targets should be pitched at appropriate scales that sufficiently capture the detail of biological threats support criteria unique to the Cape Flats. Should the Cape Flats be integrated into regional scale reserve selection analyses (e.g. Cape Flats *sensu lato* or at the scale of the Cape Metropolitan Area), important fine-scale operatives may be overlooked, which in turn may ultimately hamper implementation of targets.

5. CONCLUSIONS

This study has not tried to prescribe a definitive reserve system. Rather, it has tried to develop more clarity on identifying potential reserves on the urban Cape Flats. The use of threat and support constraints produced results only marginally different to a 'constraint-free' situation. More conservation options were available in 1997/1998 compared to five years later where every remnant exhibiting conservation potential was shown to be irreplaceable (i.e. more is needed but less is available). The current reserve network is satisfactory but the likelihood of assigning additional reserves is bleak. The implementation of the minimum set generated under the support constraint may provide a window of opportunity to enable the retention of species over the short- to medium term.

Land reform initiatives currently taking place in many regions of South Africa have seen a shift of land ownership from government control to private control. However, the vast majority of the priority remnants identified in this study is State controlled land (Appendix 3, *Chapter 3*, Table 3.1: sites zoned as Residential, Amenity/open space, Military Area or Protected Area). Thus, the responsibility of protecting and managing these remnants falls squarely under the presiding jurisdictions of government bodies at the level of local, provincial and national authorities.

The most powerful tool for conservation is not the 'park fence in isolation, but policies and reforms that make conservation a matter of private as well as social interest' (Folke *et al.*, 1996). In other words, the factors underpinning society – norms and values that drive the loss of biodiversity – should be addressed through education, incentives and opportunities. "Anything else simply buys time", and time is what is needed.

CONCLUSIONS: WINDOWS OF OPPORTUNITY

"The world is deluged with panaceas, formulas, proposed laws, machineries, ways out, and myriads of solutions. It is significant that every one of these… deals with… the structure of society, but none concerns the substance itself – the people. This despite the eternal truth of the democratic faith that the solution always lies with the people". (Saul D. Alinsky, In: Benarde, 1989)

1. SUMMARY

A total of 1 192 vascular species have been recorded across 59 remnants of natural and semi-natural habitats that are representative of the Cape Flats flora. These sites total an area of 64.8 km^2, roughly 10% of the Cape Flats. The presence of Red Data List species (RDLs) (Hilton-Taylor, 1996a, b; 1997) is high, at *ca* 4% (47 RDLs).

Although two previously protected areas are no longer feasible for conservation purposes (Blouvlei/Century City and Driftsand Nature Reserve), the remaining protected areas still do a remarkably good job of protecting biodiversity. Species and habitats on reserved sites contribute approximately 72% towards achieving reservation targets aimed at conserving each biodiversity feature (i.e. one subpopulation of a species and one unit of heterogeneity) at least once in a minimum set. These findings strongly suggest that previous accounts of the adequacy of the Cape Flats reserve network may have been unduly conservative.

However, the challenges for preserving the imperiled biodiversity of the Cape Flats are overwhelming, not only in terms of alleviating biologically threatening processes, but also garnering civil society support and institutional interest for protecting reserves and acquiring additional land for reserves. The likelihood of assigning conservation status to currently unconserved sites are unlikely. An additional minimum of 3 692.6 ha (20 sites), a relatively modest acquisition by global standards, is required to protect the unreserved species and habitats at least once, an effort necessary to prolong the persistence of all the species and the maintenance of some ecological process prevalent on the Cape Flats. Achieving this goal is hampered by the lack of alternative land for housing on the Cape Flats. Fortunately, the continual shrinkage of these remnants to smaller sizes does not automatically depress species richness. Rather, it has been established that maintaining habitat heterogeneity and keeping threats at bay, are the determining factors controlling species richness.

2. CONSERVATION PROSPECTS ON THE CAPE FLATS

This thesis mirrors what many people already know, that as we head into the 21st century, increases in species loss on the Cape Flats will continue unabated. It cannot be disputed that human-related activities in urban areas have a negative impact on the natural environment. Social norms and needs often drive these negative impacts (Matowanyika, 1994; Folke *et al.*, 1996).

2.1 CIVIL SOCIETY AND INSTITUTIONAL ASPECTS

The complex social dimensions of the Cape Flats (Wood *et al.*, 1994; Khan, 1999) do not readily justify conservation unless nature reserves play multi-purpose roles (Ngeleza, 1990; Nguta, 1992; Wood *et al.*, 1994; Wood & Gibbs, 1997). The persistence of indigenous remnants has conservation implications, and simultaneously represents an extraordinary potential to provide educational and recreational amenities. Limited successes, which have been attributed to limited (institutional) inputs, have been achieved with school groups, homes for the aged and tertiary institutions visiting these nature reserves (e.g. Ngeleza, 1990; Nguta, 1992). One of the success highlights has been the transformation of the Edith Stevens Nature Reserve to an area that now enjoys the support of local residents (Maze & Rebelo, 1999).

In an attempt to create an ethos of awareness and appreciation of natural resources on the Cape Flats, various community-based organisations (CBOs) have embarked on gardening projects (*pers. obs.*). These projects mainly highlight the horticultural and economic value of indigenous plants. Sadly, many CBOs, particularly those operating in low-income areas, are short-lived, primarily because they are totally dependent on public sector funds (Khan, 1999). Non-governmental organisations (NGOs), particularly the Botanical Society of South Africa, that focus on conservation implementation are making excellent contributions to conservation mainly through proactive, interventionist approaches towards 'conservation-friendly' urban development on the Cape Flats (Mckenzie & Rebelo, 1997; Maze & Rebelo, 1999). This is mediated by current policy and legislative frameworks, and to a lesser extent, through land acquisition negotiations (K. Maze & J. Wood, *pers. comm.*, 1997-2001). These NGOs are financially well-established and have a wide national support network generally amongst strata of high-income earners.

Greater financial investments should be afforded to CBOs on the Cape Flats because their particular function is complementary to that of NGOs (F. Khan, *pers. comm.*, 1999). Failure to do so will result in the fragmentation and alienation of stakeholders, especially the urban poor, and the chances of achieving even modest reservation targets will be unlikely. This represents a hindrance to the implementation of conservation plans. 'One-sided' approaches to conservation in urban areas (where the interests and concerns of local communities are not seriously addressed in implementation strategies) do not auger well for conservation and are not socially sustainable.

There is an emerging trend in South Africa, and indeed globally, for the corporate sector to be engaged in community development projects, particularly through environmental and conservation initiatives. A large proportion of undeveloped land on the Cape Flats (e.g. Somchem and AECI, > 2 000 ha) is owned by well-established, big commercial businesses (McKenzie & Rebelo, 1997). This sector should be regarded as an important stakeholder that is in a good position to make a contribution to preserving the natural environment.

2.2 INFORMATION AND RESEARCH

Globally, single-species plant translocations† have not enjoyed high successes (e.g. Griffith *et al.*, 1989), and the restoration of entire plant communities have not faired much better (Bullock, 1998). In some cases where plant populations have been threatened by a development on the Cape Flats, species translocations to safer sites such as nature reserves, and seed collections for storage in seedbanks, have been undertaken. However, there are only a few such instances (C. Hilton-Taylor, *unpublished data*), and concerns have been raised by managers and academics of the genetic implications, as well as protocols for effective translocations (Milton *et al.*, 1999). Translocation should be based on empirical ecological information. *Erica verticillata, Euphorbia marlothiana, Isoetes stellenbossiensis, Serruria aemula* var. *congesta* and *S. aemula* var. *foeniculacea* should be key species considered for *ex situ* cultivation, followed by re-introductions to sites from which they were collected. Special efforts should be made to rediscover (additional subpopulations) of *Serruria furcellata*, a

† 'Translocation' is defined as the "augmentation of populations within their natural range, re-introductions of species to their former range and introductions of indigenous species to sites where they have not been recorded but which are edaphically and biologically similar to their original habitat" (Milton *et al.*, 1999).

species suspected of being extinct, in addition to resolving poorly understood taxonomic relationships.

There is conflicting clarity on which species are indeed threatened and which species are endemic to the Cape Flats. These issues could be resolved by consulting the relevant taxonomic authorities, databases, literature sources and monitoring field observations. There is a responsibility to census, monitor and document the populations of currently recognised threatened plant species. By doing this, a precedent can be set for urban areas experiencing a similar biodiversity loss. A mechanism for consolidating field observations would also be useful for monitoring important plant populations. This should not be extremely time-consuming.

Documenting field-based monitoring data can facilitate *in situ* and *ex situ* conservation (e.g. propogation, seedbank storage, translocations, etc.) priorities. Moreover, this sort of monitoring work will shed more light on the prediction of the extinction rates of plant species and the erosion of ecological processes. This will assist in refining ever-changing conservation goals.

The importance of corridors that connect fragments on the Cape Flats has been recognised (McDowell *et al.,* 1991; Wood *et al.*, 1994). The identification of corridors might have contributed to achieving an overall conservation goal that addressed persistence (*Chapter 4*). Increasing opportunities for dispersal for many species between fragments may be essential for long-term persistence. However, corridors may be conduits for invasive plant species, and it has not been explicitly proven that corridors with a largely artificial character are beneficial to conservation (Hobbs, 1992). Vacant remnants on the Cape Flats which represent potential corridors, may not necessarily serve the best interests of maintaining plant biodiversity because they themselves are subjected to threatening processes (e.g. alien invasive encroachment, irregular fires and dumping of waste). Therefore, consideration should be given to corridors only on the *proviso* that they are alleviated from threatening processes.

A contribution of this study was to apply conservation planning principles to identify conservation-worthy priority sites within a hostile urban environment. Some interesting results showed that the consideration of threat and support indicators makes very little difference to the outcome of reserve selection procedures. This strongly suggests that ideal- and real-world solutions are highly congruent in heavily transformed urban areas where the biota is highly diverse, a finding similar to studies undertaken in pristine landscapes.

The thesis found that small-sized remnants can still be important for conservation, and this needs to be verified further for the viability of their populations. A limitation of this thesis is that qualitative species data relating to life-histories/reproductive strategies and abundance data (e.g. number of subpopulations or numbers of individuals) were not used (this data is only available for a few species). This would have added an interesting aspect towards understanding the risk of species extinction in relation to how well species might be able to respond to threats (e.g. species with narrow niches). The incorporation of this sort of data into reserve selection procedures might have addressed persistence particularly in relation to high-risk species, but the acquisition of this data is costly and time-consuming. Using this data to establish persistence goals may be an overtly strenuous exercise ('high investment, low returns') since conservation-worthy sites are unlikely to exist on the Cape Flats, say over the next 100/200/300…1 000 years (i.e. over some evolutionary time scale).

3. CONCLUSIONS

Should the situation arise within the next 10 to 15 years that the only natural remnants remaining on the Cape Flats are the 12 existing reserves, at least all the strict Cape Flats endemics will be protected. More widely occurring endemics and RDLs recorded on the Cape Flats also occur elsewhere outside the study area where opportunities for conservation of remnants are probably more likely to succeed than on the Cape Flats.

The conservation of existing remnants is highly unlikely to enable the persistence of ecological processes over an evolutionary time-scale, even if persistence goals are addressed in conservation plans. Achieving persistence in a fragmented landscape where the matrix is hostile to biodiversity, such as the Cape Flats, may not be feasible. The spatial components of most evolutionary processes have been irreversibly transformed. Persistence, therefore, can only be achieved through management strategies, and this will be very costly for the relevant conservation agencies.

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Appendix 1 List of the 59 study sites. Formal names are used wherever possible. * denotes a protected area, the definition of which is given in *Chapter 2*.

SITE CODE	SITE NAME
75MB*	75 Brigade Military Base* (formerly called Sixth Base Ordinance Depot)
AECI	AECI industrial site
BELHAR	Belhar road verge site
BLKFAUR	Bleak House-Faure road verge
BLOUFYN	Blouberg Fynbos site
BLOUREN	Blouberg Renosterveld koppie site
BLOUSTR	Blouberg Strandveld site
BLOUVL*	Blouvlei/Century City*
BDOWNS	Blue Downs urban fringe
BRACK	Brackenfell Granite outcrop
CFNR*	Cape Flats Nature Reserve*
CAPRI*	Capricorn Science Park*
CROSS	Crossroads open space
DRIFTSR*	Driftsands Nature Reserve*
DUINEF	Duinefontein Road-Lansdowne Road open space
DURBRC	Durbanville Racecourse
DURBNR*	Durbanvillle Nature Reserve*
EDITH	Site adjacent to Edith Stephens Nature Reserve
EDITHNR*	Edith Stephens Nature Reserve*
EERQR	Eerste River Quarry industrial site
HANOV	Hanover Park open space
KENRC	Kenilworth Racecourse
KHAYOS	Khayelitsha open space
KRAAIF	Kraaifontein Forest Reserve
LANJAN	Lansdowne Road-Jan Smuts Drive open space
LANOS	Lansdowne open space
LANRD	Lansdowne Road open space
MACASS	Macassar Dunes
MEW	Khayelitsha Mew Way open space
MILNLM	Milnerton lagoon mouth
MILNRC*	Milnerton Racecourse*
MKFLAT	Mitchells Plain-Khayelitsha Flats
N7N1	N7-N1 road interchange
NEWF	Newfields open space
NORTHP	Northpine open space
NWPEN	Northwest of Penhill Road
NYANGR	Nyanga Railway open space
PELZEEK	Pelican Park-Zeekoeivlei Flats
PENEST	Penhill Estate site
PENEER	Penhill/Eerste River open space
PHILEAST	Phillipi East site
PHILWET	Phillipi Wetland
PLATTE*	Plattekloof Natural Heritage Site*

Continues overleaf

RIET*	Rietvlei Protected Area*
RONDCO	Rondebosch Common
RONDEA	Rondebosch East open space
RONDVL*	Rondevlei Nature Reserve*
SAND	Sand Industria site
SOMCH*	Somchem industrial site*
STRANDF	Strandfontein Road open space
SWART	Swartklip open space
TOKFR	Tokai Forest Reserve
TYGNR*	Tygerberg Nature Reserve*
WELM	Farm Klein Welmoed
WETT	Wetton open space
WING	Wingfield Military Base
WOLFNR*	Wolfgat Nature Reserve*
WOOD	Woodbridge Island
YOUNG	Youngsfield Military Base

Appendix 2 Habitat types present on the study sites (developed from Figure 2.2). Each habitat is a unique combination of the variables. * denotes a protected area. Empty cells represent an absence. Site names are given in Appendix 1.

SITE CODE	SOIL TYPE	SOIL PROPERTIES	RELIEF	GEOLOGY	WATERBODY	HABITAT
75MB*	Acidic	deep	>120 m (flat & subdued)			9
75MB*	Clay/loam	shale derived	>120 m (flat & subdued)			29
75MB*	Acidic	deep	>120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	55
AECI	Alkaline	embryo	<120 m (flat & subdued)			10
AECI	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	1
AECI	Clay/loam	shale derived	<120 m (flat & subdued)		Standing (marshy/boggy, seasonal)	8
AECI	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	18
BDOWNS	Acidic	shallow	<120 m (flat & subdued)			23
BDOWNS	Alkaline	non-regic	<120 m (flat & subdued)			3
BELHAR	Alkaline	regic	<120 m (flat & subdued)			20
BLKFAUR	Alkaline	regic	<120 m (flat & subdued)			20
BLKFAUR	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
BLKFAUR	Acidic	deep	<120 m (flat & subdued)		Flowing (perennial, fresh)	34
BLOUFYN	Acidic	deep	>120 m (flat & subdued)	ferricrete		35
BLOUFYN	Clay/loam	granite derived	>120 m (flat & subdued)			36
BLOUFYN	Alkaline	regic	>120 m (flat & subdued)			37
BLOUFYN	Clay/loam	shale derived	>120 m (flat & subdued)			29
BLOUFYN	Acidic	deep	>120 m (flat & subdued)			9
BLOUREN	Acidic	shallow	>120 m (flat & subdued)			27
BLOUREN	Clay/loam	granite derived	>120 m (flat & subdued)			36
BLOUREN	Clay/loam	shale derived	<120 m (undulated/steep & dissected)			38
BLOUREN	Clay/loam	shale derived	>120 m (undulated/steep & dissected)	shale		39
BLOUSTR	Alkaline	non-regic	<120 m (flat & subdued)			3
BLOUSTR	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	1
BLOUSTR	Alkaline	regic	<120 m (flat & subdued)	limestone		2
BLOUVL*	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	1
BLOUVL*	Clay/loam	shale derived	<120 m (flat & subdued)	shale	Standing (marshy/boggy, perennial, fresh)	40
BLOUVL*	Acidic	shallow	<120 m (flat & subdued)		Standing (marshy/boggy, seasonal)	41
BLOUVL*	Acidic	shallow	<120 m (flat & subdued)			23

Continues overleaf

SITE CODE	SOIL TYPE	SOIL PROPERTIES	RELIEF	GEOLOGY	WATERBODY	HABITAT
BRACK	Clay/loam	shale derived	>120 m (flat & subdued)			25
BRACK	Clay/loam	shale derived	>120 m (flat & subdued)	granite		50
BRACK	Acidic	shallow	>120 m (flat & subdued)	sandstone		51
CAPRI*	Alkaline	regic	<120 m (flat & subdued)			20
CAPRI*	Alkaline	regic	<120 m (flat & subdued)	calcrete	Standing (marshy/boggy, perennial, fresh)	43
CFNR*	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	1
CFNR*	Alkaline	regic	<120 m (flat & subdued)	limestone		2
CFNR*	Alkaline	non-regic	<120 m (flat & subdued)			3
CFNR*	Alkaline	regic	<120 m (flat & subdued)	limestone	Standing (marshy/boggy, seasonal)	4
CROSS	Alkaline	regic	<120 m (flat & subdued)			20
DRIFTSR*	Alkaline	non-regic	<120 m (flat & subdued)			3
DRIFTSR*	Alkaline	non-regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	26
DUINEF	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
DURBNR*	Clay/loam	shale derived	<120 m (flat & subdued)			24
DURBNR*	Acidic	shallow	<120 m (flat & subdued)			23
DURBRC	Acidic	shallow	<120 m (flat & subdued)			23
DURBRC	Clay/loam	shale derived	<120 m (flat & subdued)			24
DURBRC	Acidic	shallow	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	7
DURBRC	Acidic	shallow	<120 m (flat & subdued)	ferricrete		25
EDITH	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	54
EDITHNR*	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
EDITHNR*	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	1
EDITHNR*	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	54
EDITHNR*	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	18
EERQR	Acidic	shallow	<120 m (flat & subdued)			23
EERQR	Clay/loam	granite derived	<120 m (flat & subdued)			33
HANOV	Acidic	shallow	<120 m (flat & subdued)	granite		32
KENRC	Acidic	deep	<120 m (flat & subdued)	granite		5
KENRC	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
KENRC	Acidic	shallow	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	7
KENRC	Clay/loam	shale derived	<120 m (flat & subdued)		Standing (marshy/boggy, seasonal)	8
KHAYOS	Alkaline	regic	<120 m (flat & subdued)			20

Continues overleaf

SITE CODE	SOIL TYPE	SOIL PROPERTIES	RELIEF	GEOLOGY	WATERBODY	HABITAT
KHAYOS	Alkaline	regic	<120 m (flat & subdued)	limestone	Standing (marshy/boggy, perennial, fresh)	42
KRAAIF	Acidic	deep	>120 m (flat & subdued)			9
LANJAN	Acidic	deep	<120 m (flat & subdued)			19
LANOS	Acidic	deep	<120 m (flat & subdued)			19
LANRD	Acidic	deep	<120 m (flat & subdued)			19
MACASS	Alkaline	embryo	<120 m (flat & subdued)			10
MACASS	Alkaline	non-regic	>120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	11
MACASS	Alkaline	embryo	<120 m (flat & subdued)		Pan	12
MACASS	Alkaline	non-regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	13
MACASS	Alkaline	non-regic	<120 m (flat & subdued)	limestone		14
MACASS	Alkaline	non-regic	<120 m (flat & subdued)	limestone	Standing (marshy/boggy, seasonal)	15
MEW	Alkaline	regic	<120 m (flat & subdued)			20
MEW	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	18
MILNLM	Alkaline	regic	<120 m (flat & subdued)			20
MILNLM	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	1
MILNRC*	Acidic	deep	<120 m (flat & subdued)	ferricrete		56
MILNRC*	Acidic	deep	<120 m (flat & subdued)	ferricrete	Standing (marshy/boggy, seasonal)	57
MILNRC*	Alkaline	regic	<120 m (flat & subdued)			20
MILNRC*	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
MKFLAT	Alkaline	non-regic	<120 m (flat & subdued)			3
N7N1	Acidic	deep	<120 m (flat & subdued)			19
N7N2	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, seasonal)	30
N7N3	Acidic	deep	<120 m (flat & subdued)	ferricrete	Flowing (perennial, estuarine)	31
NEWF	Acidic	deep	<120 m (flat & subdued)			19
NORTHP	Acidic	shallow	<120 m (flat & subdued)			23
NWPEN	Acidic	deep	<120 m (flat & subdued)			19
NYANGR	Acidic	deep	<120 m (flat & subdued)			19
PELZEEK	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
PELZEEK	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	18
PELZEEK	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	1
PENEER	Acidic	deep	<120 m (flat & subdued)			19
PENEST	Acidic	deep	<120 m (flat & subdued)			19

Continues overleaf

SITE CODE	SOIL TYPE	SOIL PROPERTIES	RELIEF	GEOLOGY	WATERBODY	HABITAT
PHILEAST	Alkaline	regic	<120 m (flat & subdued)	limestone	Standing (marshy/boggy, perennial, fresh)	42
PHILEAST	Alkaline	non-regic	<120 m (flat & subdued)			3
PHILEAST	Alkaline	regic	<120 m (flat & subdued)	calcrete	Standing (marshy/boggy, perennial, fresh)	43
PHILWET	Acidic	deep	<120 m (flat & subdued)			19
PHILWET	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
PLATTE*	Acidic	deep	>120 m (flat & subdued)			9
PLATTE*	Alkaline	non-regic	<120 m (flat & subdued)			3
PLATTE*	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
PLATTE*	Clay/loam	shale derived	>120 m (flat & subdued)	shale	Standing (marshy/boggy, seasonal)	16
RIET*	Acidic	shallow	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	60
RIET*	Acidic	shallow	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	7
RIET*	Alkaline	embryo	<120 m (flat & subdued)		Pan	12
RIET*	Clay/loam	shale derived	<120 m (flat & subdued)	shale	Standing (marshy/boggy, seasonal)	61
RIET*	Acidic	shallow	<120 m (flat & subdued)		Flowing (perennial, fresh)	62
RIET*	Alkaline	regic	<120 m (flat & subdued)			20
RIET*	Acidic	shallow	<120 m (flat & subdued)			23
RIET*	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	1
RONDCO	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
RONDEA	Acidic	deep	<120 m (flat & subdued)			19
RONDEA	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
RONDVL*	Alkaline	non-regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	17
RONDVL*	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	18
RONDVL*	Acidic	deep	<120 m (flat & subdued)			19
RONDVL*	Alkaline	regic	<120 m (flat & subdued)			20
SAND	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
SOMCH*	Alkaline	embryo	<120 m (flat & subdued)			10
SOMCH*	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	1
SOMCH*	Alkaline	regic	<120 m (flat & subdued)	calcrete		52
SOMCH*	Alkaline	regic	<120 m (flat & subdued)	limestone		2
SOMCH*	Alkaline	regic	<120 m (flat & subdued)		Pan	53
STRANDF	Alkaline	non-regic	<120 m (flat & subdued)			3
STRANDF	Alkaline	non-regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	17

Continues overleaf

SITE CODE	SOIL TYPE	SOIL PROPERTIES	RELIEF	GEOLOGY	WATERBODY	HABITAT
STRANDF	Alkaline	regic	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, estuarine)	18
SWART	Alkaline	regic	<120 m (flat & subdued)		Flowing (perennial, fresh)	58
SWART	Alkaline	non-regic	>120 m (flat & subdued)			59
SWART	Alkaline	non-regic	>120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	11
TOKFR	Acidic	shallow	>120 m (flat & subdued)			27
TOKFR	Clay/loam	granite derived	>120 m (flat & subdued)	granite		28
TOKFR	Clay/loam	shale derived	>120 m (flat & subdued)			29
TYGNR*	Clay/loam	shale derived	>120 m (flat & subdued)		Flowing (seasonal)	44
TYGNR*	Clay/loam	shale derived	<120 m (undulated/steep & dissected)		Flowing (seasonal)	45
TYGNR*	Clay/loam	shale derived	>120 m (undulated/steep & dissected)		Flowing (seasonal)	46
TYGNR*	Acidic	shallow	>120 m (flat & subdued)		Flowing (seasonal)	47
TYGNR*	Clay/loam	shale derived	>120 m (undulated/steep & dissected)	shale	Standing (marshy/boggy, seasonal)	48
TYGNR*	Clay/loam	shale derived	<120 m (undulated/steep & dissected)	shale		49
TYGNR*	Clay/loam	shale derived	>120 m (undulated/steep & dissected)	shale		39
TYGNR*	Acidic	shallow	>120 m (flat & subdued)			27
WELM	Acidic	shallow	<120 m (flat & subdued)	granite		32
WELM	Clay/loam	granite derived	>120 m (flat & subdued)	granite		28
WELM	Clay/loam	shale derived	>120 m (flat & subdued)			29
WETT	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
WETT	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
WING	Acidic	deep	<120 m (flat & subdued)			19
WING	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6
WING	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, seasonal)	30
WOLFNR*	Alkaline	embryo	<120 m (flat & subdued)			10
WOLFNR*	Alkaline	non-regic	>120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	11
WOLFNR*	Alkaline	regic	<120 m (flat & subdued)			20
WOLFNR*	Alkaline	regic	>120 m (flat & subdued)	calcrete		21
WOLFNR*	Alkaline	regic	>120 m (flat & subdued)	limestone		22
WOOD	Alkaline	regic	<120 m (flat & subdued)			20
YOUNG	Acidic	deep	<120 m (flat & subdued)			19
YOUNG	Acidic	deep	<120 m (flat & subdued)		Standing (marshy/boggy, perennial, fresh)	6