A Resource Allocation System for Invasive Alien Plant Control on the St Francis Conservancy

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Declaration

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

B Reeves 30/01/2009

Signature Date

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ABSTRACT

Alien invasive species pose a great threat to ecosystems and human communities (Richardson & van Wilgen 2004; Hobbs & Humphries 1995). On private lands, there is a need to find institutional, administrative and scientific responses to the alien plant problem that operate beyond the scale imposed by property boundaries (Slocombe 1998; Grumbine 1994). One such response is the development and support of institutions, such as the St Francis Conservancy, that foster cross-boundary management of landscapes (Hurley *et al*. 2002).

This study seeks to promote collective decision-making and collaborative management by private landowners through the development of a resource allocation system for the control of alien invasive plants on the St Francis Conservancy.

The conservancy is located in the south-eastern lowlands of the Cape Floristic Region, between the villages of Cape St Francis and Oyster Bay, and is comprised of the properties of multiple landowners.

Mutliple Criteria Decision Analysis (MCDA) and Geographical Information Systems (GIS) techniques were integrated to provide a spatially explicit resource allocation system that considered environmental, social and economic concerns. The MCDA technique selected for use was the Analytical Hierarchy Process (AHP). This technique has a record of providing robust, defensible decisions and enabled the resource allocation decision-problem to be decomposed into a hierarchy of objectives, criteria and indicators.

Stakeholders participated in the development of the resource allocation system, especially through providing input into the determination of the relative importance of criteria and indicators through the assignment of weights. Various weighting scenarios were presented and these were interpreted into an

implementation plan. The costs and effort required to clear alien plants were estimated, and obstacles facing the implementation of the plan were identified.

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PREFACE

The focus of this dissertation is on aiding natural resource management through the integration of spatial data and analysis techniques with decision support tools. This study describes a resource allocation system for alien invasive plant control on the St Francis Conservancy. The conservancy was established as an attempt to create an institution for many landowners to cooperate in the management of a priority ecosystem within the Cape Floristic Region.

The dissertation is broken down into four chapters. The first chapter provides the rationale behind the study and describes the context of the study area. This includes a description of the study area, the threat posed by alien plants, the need for a coordinated approach to their control, the establishment of a conservancy as a means to achieve this, and the need for a decision-support system to allocate resources. The second chapter starts with a discussion on decision support in general, describes the generalised approach to Multiple Criteria Decision Analysis (MCDA), and then examines various MCDA techniques. The third chapter describes the application of a decision-support model to the alien invasive plant problem on the conservancy. The fourth chapter focuses on translating the results of the decision support model into implementation.

1. INTRODUCTION

1.1 Rationale for the study

In the past, approaches to nature conservation have focussed on the management of large areas of public land in statutory protected areas (Binning 1997). However, the statutory protected area network is strongly biased towards certain types of geographies and habitats, with reserves often being located in the least productive portions of the landscape. The vast majority of statutory protected area networks do not achieve the goal of comprehensively conserving biodiversity (e.g. Rouget *et al.* 2003, Gallo *et al.* 2009). In response to this, an emerging strategy is to strengthen the network of private conservation areas (see Sample 1994).

Conservation on private land brings with it a suite of issues. One of which is the division of land by property boundaries, which rarely correspond to the ecological boundaries of ecosystems and the dynamic processes that sustain them (Brunckhorst 1998). The move towards ecosystem management requires finding institutional, administrative and scientific ways of managing whole ecosystems instead of small arbitrary units, such as those imposed by property boundaries (Slocombe 1998; Grumbine 1994). On private lands, the move toward ecosystem management suggests the need for collective action by landowners across property boundaries (Hurley *et al*. 2002).

In 2003 funding became available from the Critical Ecosystem Partnership Fund (CEPF) to support private sector involvement in the conservation of the Cape Floristic Region (CFR), a recognised centre of diversity and endemism (Cowling & Hilton-Taylor 1994; Goldblatt & Manning 2002). In response to this opportunity, a project was developed through the Biodiversity Conservation Unit of the Wildlife and Environment Society of South Africa to support private landowner conservation in the Cape St Francis area, in the south-eastern lowlands of the CFR. The project aimed to support the establishment of a conservancy, as a mechanism for cross-boundary management cooperation between multiple landowners. While a number of small associations had previously existed between the landowners of this area, the extent of cross-boundary management that occurred was largely limited to issues such as the maintenance of common access roads and addressing security problems. The conservancy was conceived as a step towards achieving the vision of a consolidated, well-managed conservation area stretching from St Francis Bay and Oyster Bay.

The conservancy model was chosen for its potential to stimulate cross-property management without imposing onerous restrictions on landowners. Botha (2001) defines a conservancy as "a dedicated forum of farmers, private landowners and conservation bodies that interact and manage issues and common resources." Although conservancies do not have legal status, they are a successful option for landowners that are significantly affected by common resource issues. Conservancies were originally established to co-ordinate security on stock farms, but their utility is not limited to this. Conservancies have proved to be an effective model for co-operative management of alien vegetation, wildfires and block burns, game movement and catchment management. They also provide a focus with which groups of farmers can effectively lobby or negotiate with government agencies (Botha 2001).

In order to facilitate the establishment of the conservancy, the project had six main components:

1 Raise awareness– raise awareness of the project and the potential for a conservancy amongst direct stakeholders and the broader conservation community.

2 Form partnerships – form partnerships with agencies and organisations that could directly assist in the implementation of the project. This included the development of relationships with the conservation departments of both local and provincial government and with civil society organisations;

3 Establish the conservancy – develop the institutional structures of the conservancy and negotiate with individual landowners for their inclusion. Provide assistance in meeting the relevant legal, managerial and administrative requirements for the establishment of a conservancy;

4 Develop tools for the management of the conservancy – this largely the development an environmental management plan for the conservancy;

5 Increase the conservation status of the conservancy – increase the conservation status of the conservancy, or pockets of lands within the conservancy; and

6 Sustainability – identify potential funders and an investigate means of income generation to support the management of the conservancy.

Although conservancies require significant set-up time, the benefits they deliver usually far exceed the start-up and running costs (Botha 2001). The benefits for participating landowners were seen to be the following:

1 Pooling of resources – the costs of management activities (including alien vegetation eradication, fire management, wildlife introductions and road maintenance) could be shared among many property owners;

2 Easier access to assistance and funding – funding organisations and conservation agencies would likely be more willing to assist a body such as a conservancy than to interact with individual landowners;

3 Improved security – a well-managed conservancy would likely offer security benefits;

4 The emotive appeal of belonging to a conservancy;

5 Potential for increased property values – with increased environmental awareness and increasing levels of urbanisation, properties that are part of a well-run conservancy may experience appreciation in market value; and

6 Economic opportunities – conservancies offer wider opportunities for developing ecotourism, and other sustainable nature-based industries, over individual properties.

Most relevant to the current study is the potential that multiple landowners, such as those operating within a conservancy, have for the pooling of resources and collective management. This is especially relevant with regard to the management of alien invasive plants (Gunderson-Izurieta *et al.* 2008). Invasive plants are a problem that requires collective action (Klepeis *et al.* 2009), and control attempts are less likely to be successful if they occur by landowners working in isolation (see VanBebber 2003). the transboundary ecology of invasive plants, their control requires landowners to have the ability to cooperative effectively with neighbours and other key stakeholders (Klepeis *et al.* 2009).

A prerequisite for collective action is collective decision-making. However, collective decision-making is fraught with complexities, especially for multifaceted decisions that are coupled with personal interests and emotional factors (Liu & Wei 2000). This study seeks to promote collective decision-making and to reduce the potential for conflicts through the development of a transparent and unbiased resource allocation system and implementation strategy for the control of invasive plants. Elsewhere it has been shown that the development of the formal plans for ecosystem management is essential to obtain support for coordinated landscape-scale management by multiple landowners and managers (Brunson 2008).

This resource allocation system should be capable of identifying and prioritising areas for investment of resources through an evaluation of multiple criteria and in a manner that best achieves potentially competing objectives.

1.2 Description of the Study Area

1.2.1. Location

The study area is approximately 5 800 ha in extent and is situated about twenty kilometres south of the N2 highway at Humansdorp, in the south-western sector of the Humansdorp coastal plain between the villages of Cape St Francis, St Francis Bay and Oyster Bay (see Figure 1). The study area falls within the south-eastern lowlands of the Cape Floristic Region.

Figure 1: The location of the study area.

1.2.2. *Climate*

The study area has a warm temperate climate, which is mostly affected by a succession of east-moving cyclones (low-pressure systems) followed by high-pressure anti-cyclones that ridge in behind the lows. The frequency and intensity of these fronts is greatest in winter, when the area experiences cool weather and large amounts of rain. Summer rainfall is usually associated with the approach of cool post-frontal air from the south-west, moving over a relatively warm ocean, as a high-pressure system moves north-eastwards along the coast, or as a result of cut-off lows (Cowling 1984). Annual rainfall is between 700 and 800 mm, distributed throughout the year, but with a marked peak in the winter months. The driest months are normally from December to February (Cowling 1997).

The area experiences frequent strong to gale force winds, especially between September and December when south-westerly and south-easterly winds alternate. Winter winds are predominantly from the west and south-west, with an increase in east or south-east winds in summer (Lubke 1985). The calmest months are between March and May, although very strong berg winds may develop during this period (Cowling 1997).

1.2.3. *Topography and geology*

A feature of the landscape is the series of gravelled terraces at various contour intervals (roughly 30m, 60m, 100m and 200m) which are related to a descending sequence of high sea levels (Binneman 2001).

Besides the terraces at the coast, the topography of the study area is dominated by a series of fossil or fixed dunes aligned in a west-east direction (see Figure 2). These linear hairpin dunes are likely to have formed under a previous wind regime that was considerably stronger than present (Cowling 1984). The highest point within the study area is 110m above sea level.

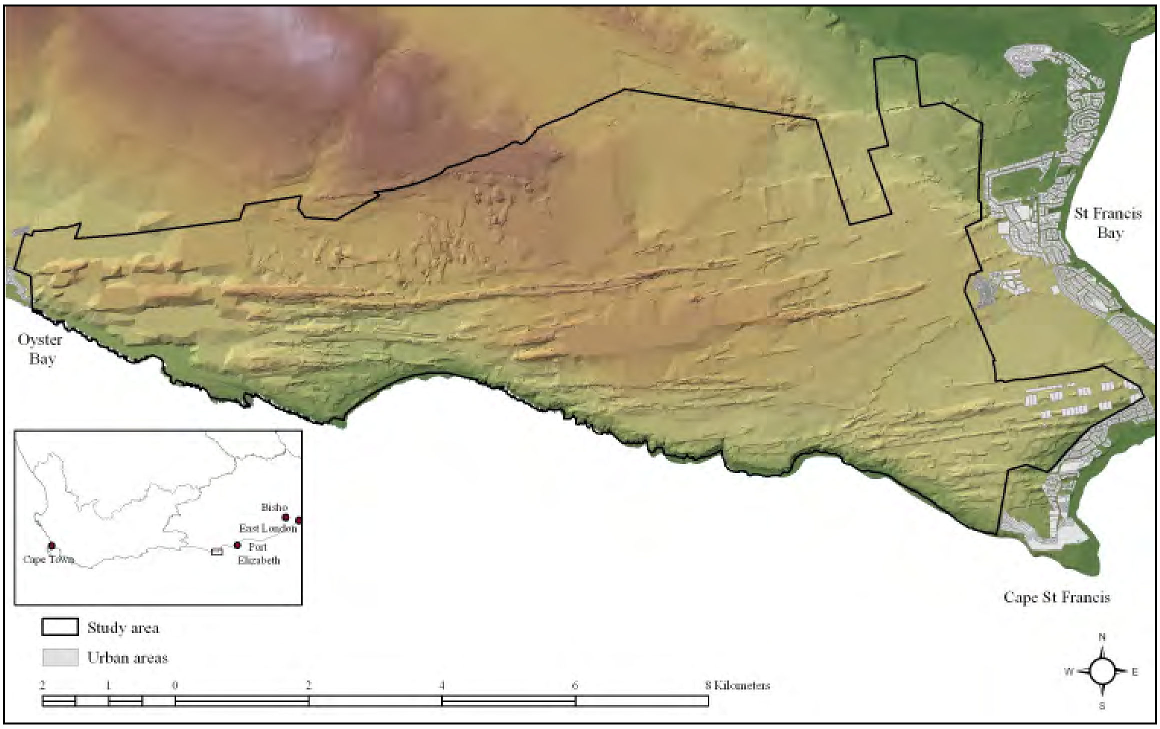


Figure 2: The topography of the study area is characterised by a series of linear hairpin aligned in a west-east direction.

The study area falls on one of the many isolated capes along South Africa's southern and south-eastern coast, which is a coast is characterized by a series of half-moon sandy bays that face eastward (Binneman 2001). These bays are formed as a result of the differential erosion rates of resistant quartzitic rocks of the Table Mountain Group and softer rocks to the north. The quartzitic rock outcrops form the headlands of these east-facing bays, while recent deposits of sands accumulate further eastward against the long curves of the bays (Cowling 1984).

Associated with the bays and headlands are headland bypass dunefield systems (see Figure 3 and Figure 4). The best remaining example of an active headland bypass dunefield on the south coast of South Africa is the Oyster dunefield (La Cock & Burkinshaw 1996). This dunefield forms a 17 km long corridor of shifting sand that moves in an easterly direction from its source at Oyster Bay, across the headland, to its sink, historically near the mouth of the Kromme estuary. South of the Oyster Bay dunefield, a smaller headland bypass dunefield has its source at Thysbaai and its sink at St Francis Bay. Transverse dunes, aligned at right angles to the prevailing south-west wind direction, predominate in the dunefields (Cowling 1984). In parts of dunefields, the basement deflation level has been reached, resulting in exposed rocky outcrops of Table Mountain sandstone.

Figure 3: Aerial view looking westwards onto the headland bypass dunefields of the study area.

Between 1917 and 1924 intensive efforts were made to stabilise the dunes. Further dune stabilisation efforts took place in 1964 when the township of St Francis Bay (then Seavista) was established (Daines *et al*. 1991). These attempts were only partially successful and the new road to St Francis Bay and Cape St Francis, which crosses over the eastern extreme of the Oyster Bay dunefield, is frequently inundated by sand.

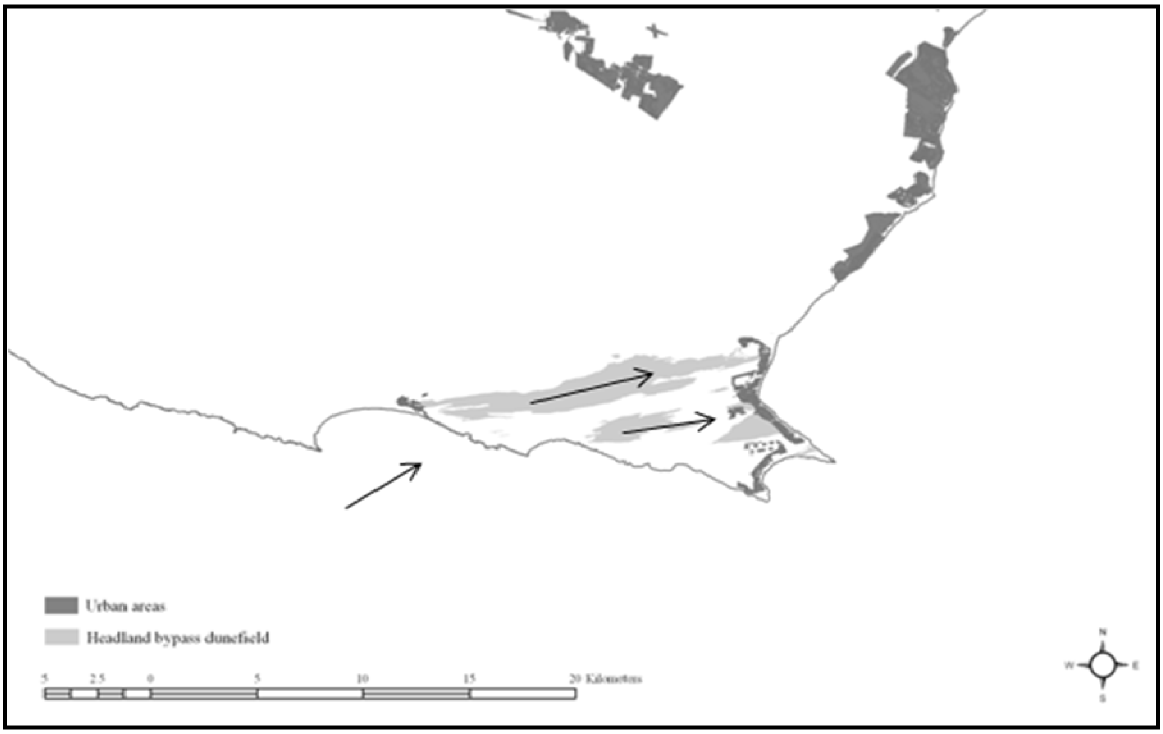


Figure 4: Historically the headland bypass dunefields would transport sand from Oyster Bay to St Francis Bay.

Since 1960 large-scale developments at the eastern end of this dunefield have resulted in major environmental problems associated with disruption of the transport function of the dunefield and the Sand River. Sand is calculated to be transported eastwards at an annual rate of 30 m^3 per metre width. This translates to an annual sand transport value of 9000 m^3 over a width of 300 m (La Cock & Burkinshaw 1996). The headland bypass dunefields are believed to be a source of sand to the shoreline of St Francis Bay (Daines *et al.* 1991), the beaches of which have been steadily eroding – a likely consequence of the stabilisation of the dunefields (Lubke & de Moor 1998).

The study area is located on recent sand and calcrete with exposed Table Mountain Sandstone unconsolidated along the shoreline (Cowling 1984). Soils are generally deep alkaline sands (Fernwood formation) which are rich in calcium. Although the soils of the study area have a high level of phosphates, this is unavailable for plants due to the high soil pH. Soil patterns of the dune sands are relatively simple, and are often strongly associated with dune topography. On the fossil vegetated dunes, the dune crests have deep soils but they are dry, excessively drained and contain little organic matter. Soils of the dune slopes are deep, calcareous and generally poorly developed. North facing slopes are exposed to greater solar radiation and the soils are only marginally wetter than the crests. In contrast, the cooler south facing slopes are wetter, with higher organic matter content (stained to a depth of about 0.5m). The dune troughs have the greatest soil moisture in the landscape as they receive runoff from the adjacent slopes and have impeded drainage due to the existence of a calcrete hardpan at variable depths below the soil surface. (Kruger & Cowling 1996) Here deep, seasonally waterlogged sands with an abnormal accumulation of organic matter occur. Soils of the mobile dune sands have little or no organic matter and are much drier than those of the vegetated dunes (Richard Cowling *pers comm.* 2009).

1.2.4. Natural vegetation

Cowling (1996) describes the distribution and occurrence of flora within the Humansdorp coastal plain as typical of the floristic complexity of the western part of the Eastern Cape, which is the meeting point of four of South Africa's five phytogeographical regions (namely Afromontane, Cape, Karoo-Namib and Tongaland-Pondoland). The area is characterised by a mosaic of vegetation types with affinities to different biogeographical areas, such as Cape fynbos with Tongaland-Ponoland forest and thicket.

Fire is an important determinant in vegetation composition on the Humansdorp coastal plain, governing the balance between fynbos and thicket. Historically, thicket was confined to areas in the landscape where some physical feature (e.g. rocky outcrop, or river valley or coastal margin) afforded fire protection. With the preclusion of fire, the long-lived thicket species begin to consolidate through vegetation growth and seedling establishment. Fynbos plants, which have relatively short life spans and poorly persistent seed banks, cannot tolerate the dense shade cast by the emerging thicket shrubs and eventually succumb due to senescence and competition. Thus, in the long-term absence of fire (in the order of 100 years), the species- and endemic-rich fynbos is replaced by relatively species-poor thicket, which comprises of plant species with wide distributions in southern Africa (Cowling 1997). Closed thickets are more widespread along the coastal margin where fire frequency is the lowest (Cowling 1984).

The dune crests have low vegetation cover, while the south-facing slopes have denser vegetation than the north-facing slopes. Dune slopes support a matrix of dune fynbos and dune thicket, while the often waterlogged dune troughs support predominantly herbaceous vegetation (see Figure 5).

Figure 5: A *Mariscus conjestus* dominated wetland within the fossil dune slacks (photo R.M. Cowling).

The vegetation of the study area was comprehensively studied by Cowling (1984). Subsequently, the spatial distribution of vegetation types for the area was mapped as part of the conservation assessments for the Fynbos (Cowling *et al.* 1999) and Subtropical Thicket Biomes (see Cowling *et al*. 2003). The conservation assessment for the Fynbos Biome used Broad Habitat Units (BHU's) as biodiversity surrogates. The BHU's were derived at a 1:250 000 scale from attributes that are good predictors of most vegetation patterns in the Cape Floristic Region (Cowling & Heijnis 2001). The study area was classified as primarily falling within the St Francis Fynbos Thicket Broad Habitat Unit (BHU). This BHU is one of six fynbos / thicket mosaic BHU's in the CFR, and prior to transformation by urbanisation, agriculture, forestry and alien plant invasions, covered some 25 924 ha (about 0.2% of the CFR).

The conservation assessment for the Subtropical Thicket Biome described the area as containing Algoa Dune Thicket, St Francis Dune Thicket (Algoa Dune Thicket in a mosaic with fynbos), Humansdorp Grassy Fynbos, Kromme Forest Thicket and South Coastal vegetation.

A more detailed vegetation mapping exercise for study area was conducted at a 1:10 000 scale by Kruger & Cowling (1996). Six primary vegetation groups were identified and subsequently divided into nine vegetation types (with two additional habitat types: sandy shores and driftsands) (see Figure 6 and Table 1).

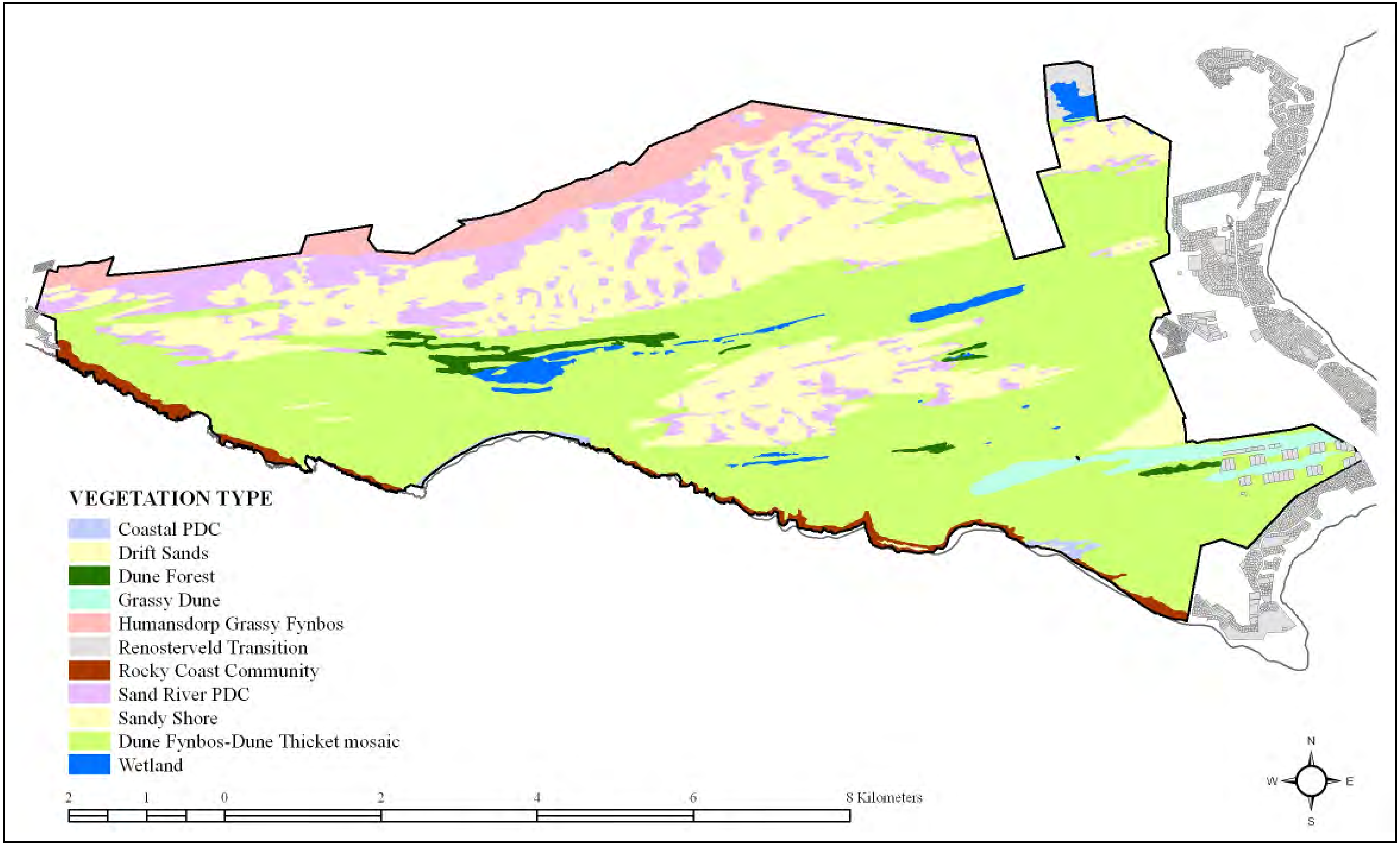


Figure 6: The vegetation types mapped for the study area by Kruger & Cowling (1996).

Table 1: Description of the vegetation types described for the area by Cowling & Kruger (1996).

Primary Veg. Type	Secondary Vegetation Type	Area
South Coast Dune Fynbos	**Dune Fynbos – Dune Thicket mosaic** A low closed ericoid shrubland with open restioid understory (Pierce & Cowling 1984). Characterised by a high proportion of shrubs such as *Passerina vulgaris*, *Rapanea gilliana*, *Chironia baccifera*, *Agasthosma apiculata* and *Myrica quercifolia*. This vegetation unit supports regionally endemic species such as *Silene primulae, Erica chloroloma, Agathosma stenopetala* and *Phylica litoralis*. Restioids such as *Ischyrolepis eleocharis* and *Restio leptoclados* are included in the understory. Interspersed within the fynbos matrix are dune thicket clumps, dominated by *Sideroxylon inerme, Pterocelastus tricuspidatis, Maytenus procumbens, Mystroxylon aethiopicum, Olea exasperata* and *Rhoicissus digitata*.	
	Grassy Dune Fynbos Generally found in the dune slacks, Grassy Dune Fynbos patches differ from the above vegetation type in that they support a higher proportion of grassland species, including *Stenotaphrum secundatum, Tristachya leucothrix, Themeda trianda, Imperata cylindrical*, *Aspalathus spinosa* and *Tetrarai cuspidata*, and geophytes, and that they lack the same diversity of shrub species.	
	Renosterveld – Dune Fynbos mosaic This unit is the transition zone between South Coast Dune Fynbos and South Coast Renosterveld, which is found primarily on Bokkeveld shales. The Renosterveld elements include *Elytropappus rinocerotis, Metalasia muricata, Themeda trianda, Sporobolus Africana, Cynodon dactylon* and *Cliffortia linearifolia* (Pierce & Cowling 1984).	
Forests	**South Coast Dune Forest** Dune forests, which are restricted to sheltered dune slacks, ranging in 3-10m in height. Characterised by *Sideroxylon inerme, Euclea racemosa, Pterocelastrus tricuspidatus, Hippobromus pauciflorus* and *Zanthoxylum capense*. These differ from the patches of dune thicket in that they are generally less spinescent, taller, form a well-established canopy and support a greater diversity of species.	

Pioneer Dune Communities	**Coastal Pioneer Dune communities** Found in the unstable foredunes abutting sandy shores. They are dominated by shrubs such as *Passerina rigida, Metalasia muricata, Helichrysum spp., Myrica cordifolia, Stenotaphrum secundatum* and *Chrysanthemoides monolifera*. The communities found in the less stabilised portions immediately abutting the sandy shores include species such as *Arctotheca populefolia, Carpobrotus deliciosus, Felicia echinata, Scaevola plumier* and *Gazania rigens*. **Sand River Pioneer Dune communities** These communities are similar to the coastal dune communities but they contain a high representation of geophytes and orchids (*Bonatea spp*. And *Satyrium spp.).* Wetlands are often associated with these patches. Woody vegetation comprises clumps of *Myrica cordata, Stoebe plumose* and *Rhus crenata.*
Rocky coast vegetation	**Rocky Coast communities** The species found in vegetation pockets on the rocky coast are often halophytes such as *Chenolea diffusa, Limonium scabrum* and *Disphyma crassifolia*. Where large amounts of sand collect, pockets of vegetation similar to the pioneer dune communities may establish. Included in these pockets would be species such as *Plantago carnosa* and *Gazania rigens* and grasses such as *Sporobolus virginicus* and *Stenotaphrum secundatum.*
Wetlands	**Wetland** These are generally vleis associated with calcrete hardpans, dominated by sedges such as *Fuirena hirsuta, Mariscus congestus*, *Juncus spp*. And *Scirpus spp*.
Drift sands	**Drift sands** Drift sand generally associated with the headland bypass dunefields and lacks vegetation.
Sandy shore	**Sandy shores** Sandy shore along the major beaches.

1.2.5. Human habitation

The study area has an exceptionally rich array of archaeological and paleontological sites (Binneman 2001), suggesting human habitation of the area for up to a million years. More recently, the area was inhabited by the San (from about 10 000 years ago) and by the Khoi peoples (from about 1 700 years ago). Bantu-speaking people, the forerunners of the Xhosa, were established in the region from about 1 400 years ago (Binneman 2001).

Early Stone Age stone tools (e.g. hand axes and cleavers) dating back between 1 million and 200 000 years ago can be found inland from Thysbaai. Vast Middle Stone Age deposits dating between 125 000 and 70 000 years ago (including artefacts, tools and fossilised bone remnants) have also been found in the mobile dune fields (Binneman 2001; Cowling 1997). Much of the archaeology of the area, however, relates to the Late Stone Age (from 30 000 years ago). The coastline also has a large concentration of Late Stone Age shell middens and fish traps occur at two sites (Cowling 1997).

Shell middens are composed of marine food waste and cultural remains. Middens close to rocky shores generally contain shellfish (e.g. brown mussel *Perna perna*, limpets *Patella spp*. and periwinkle *Oxystele sp.*) while middens located along open beaches generally consist of the white sand mussel *Donax serra*. Middens may also yield marine and terrestrial mammal remains, stone artefacts, bone tools and pottery. The majority of shell middens are located within 300 m of from the high water mark, although some middens may occur up to 5 km from the coast (Webley & Hall 1998).

These sites represent single moments in time and individually provide limited information. However, the study area is important from an archaeological perspective because the large number of sites within it can yield valuable information on settlement strategies and social and economic patterns of prehistoric peoples (Binneman 2001).

1.2.6. Land ownership

The migration of people from metropolitan to rural areas for leisure reasons has resulted in rural landscapes that have an increasingly heterogeneous mix of landholders and land use. The percentage of inhabitants who fall outside the traditional category of fulltime farmer is growing. The majority of newcomers seek a 'rural' lifestyle in a rural context, consuming its amenity value rather than living off the land (Klepeis *et al*. 2009). This is reflected in the study area, which is composed of properties with various land uses and owned by different types of landowners.

The study area is comprised of a total of 70 properties owned by 47 landowners. In order to understand ownership patterns within the study area, landowners were placed into categories (see Table 2). The term 'lifestyler' is used by Klepeis *et al.* (2009) to describe individuals who take up landownership in rural areas of parcels of land that are used primarily for their amenity value rather than their production value. These landowners acquire land principally for lifestyle or cultural reasons. Klepseis *et al.* (2009) distinguishes between 'part-time' (those whose primary residence is not on the rural properties in question) and 'fulltime' lifestylers (those who reside on the rural properties in question). Although the lifestylers are the most numerous of landowners (33 in total, with 31 part-time and 2 fulltime) within the study area, they also possess the smallest properties (28 ha on average) (see Table 3). The lifestyler landowners within the study area can generally be classified as falling into a number of loose associations of landowners, including the Rebelsrus Conservation Association, the Mosterthoeks landowners association and the group of landowners at Thyspunt (see Figure 7).

Table 2: Landowner types within the study area.

Landholder type	Characteristics
Full time lifestylers	Full time residents; may have a secondary residence elsewhere; main or only source of income is off-property; amenity use
Part-time lifestylers	'Weekenders' or occasional visitors; primary residence elsewhere; rely on off-property income; amenity use
Farmer	Full time residents. Main source of income is farming in the broader area. Have farms that extend into the study area, but do not necessarily farm within the study area.
Property developer / land investor	Riside off-property. Own land that is currently developed or is planned to be developed, mostly for housing estates.
Ecotourism operator	Riside off-property, but with managers who riside on the properties. Pursue commercial ecotourism operations on properities.
Parastatal	Own land set aside for public infrastructure development.

Landowners classified as farmers in this study are those that derive their income from farming. These landowners generally reside on their farms. Although their farms extend into the study area, their agricultural practises generally occur outside of it (except for occasional use for grazing of cattle). There are five farmers within the study area, and the portions of their farms within the study area are comparatively moderate in size (144 ha on average).

Property developers are those that have purchased or kept parcels of land with the intention of developing them, largely as housing estates. There are seven property landowners (or consortia of landowners) classified as property developers, and their properties are generally large (276 ha on average) and in various stages of development. The properties classified as being owned by property developers include Rocky Coast Farms (currently undeveloped), the St Francis Field (airfield and housing estate), the St Francis Links (golf course and housing estate) and the Sand River Sanctuary (housing estate).

One landowner (operating through the closed corporation Macohy cc) is pursuing commercial ecotourism operations on five properties within the study area (a total area of 567 ha). These operations include a game reserve and a guesthouse. The landowner resides abroad, but the operations are run by resident managers.

The single largest landowner within the study area is Eskom, the parastatal corporation responsible for power production and distribution in South Africa. Eskom owns 16 properties at Thyspunt, covering a total of 1 499 ha of the study area. The site is managed by a non-resident estates and conservation manager.

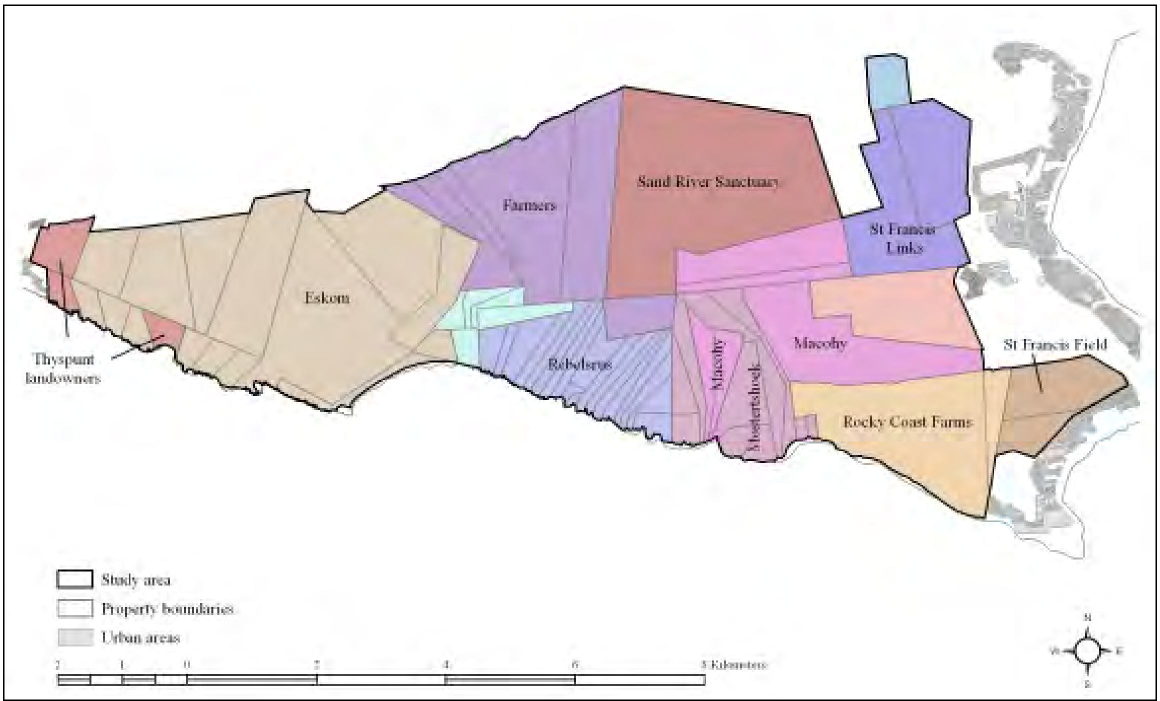


Figure 7: The ownership and landowner groupings of the study area.

Table 3: Ownership of the study area

Owner category	Total area (ha)	No. of properties	Owner / category	Properties / owner	Area / owner (ha)
Full-time lifestyler	55	2	2	1.0	28
Part-time lifestyler	868	33	31	1.1	28
Farmer	722	7	5	1.4	144
Property developer / land investor	1932	7	7	1.0	276
Ecotourism operator	567	5	1	5.0	567
Parastatal	1499	16	1	16.0	1499
Total	5644	70	47	1.5	120

1.2.7. Conservation significance of the study area

The two biome-level conservation assessments covering the study area have both identified the area as a conservation priority. The St Francis Fynbos Thicket Mosaic Broad Habitat Unit (identified in the conservation assessment for the Fynbos Biome) was classified as being irreplaceable thus essential for the achievement of conservation targets for the Cape Floristic Region.

10 369 ha of the remaining, intact habitat of the St Francis Fynbos Thicket Broad Habitat Unit is required to effectively conserve it into the future. Since this comprises about 129% of the remaining habitat, all remaining intact habitat of this BHU is required to achieve the conservation target. No further loss of this habitat type, which is mostly in private hands, can be afforded if conservation targets are to be met.

St Francis Dune Thicket (identified within the conservation assessment for the Subtropical Thicket Biome) was classified as being an endangered ecosystem that cannot withstand significant further loss of natural area.

The reservation status of the ecosystems within the study area is low, with only 2.2% conserved in a statutory reserves.

Vlok & Euston-Brown (2002) state that most of the St Francis Dune Thicket unit is threatened by alien woody plants and urban development and that most of the species endemic to this dune system in this unit are threatened with extinction.

The study area has an exceptionally rich array of plant species of special concern, including Red Data Book plants and plants with very limited distributions. Cowling (1997) recorded 250 species of plants in 72 families and 188 genera in his management plan for the Thyspunt area. He states that this diversity is exceptional and that a far fewer number of species would generally be recorded from an equivalent-sized dune area in the generally more species rich south-western part of the Cape Floristic Region. This high diversity is partly as a result of the overlap of subtropical and temperate (Cape) floras in the Eastern Cape (Cowling 1984).

1.3 The alien invasive plant problem

Alien invasive species pose a great threat to ecosystems and the human communities by reducing natural capital, destabilising ecosystems and threatening economic productivity (Richardson & van Wilgen 2004; Hobbs & Humphries 1995). Globally, invasion by alien species is recognized as the second largest threat to biodiversity. In South Africa, estimates have suggested that about 10 million hectares are invaded by alien plants, with the Fynbos biome being the most invaded biome. The most important invaders are trees and shrubs in the genera *Acacia*, *Hakea* and *Pinus* (Richardson & van Wilgen 2004).

Alien plant species cause damage to ecosystems through their excessive use of resources (especially water) or their addition of resources to ecosystems (especially nitrogen) (see Richardson & van Wilgen 2004; Levine *et al.* 2003). They may also promote intense wildfires, stabilize sand movement and promote erosion. A common consequence of the impacts of alien invasive plants on natural systems is a reduced ability of ecosystems to provide environmental goods and services (or ecosystem services). This is most often referenced in accounts of how alien plants impact on water production and supply, but alien plants also impact on a variety of other ecosystem services. For example, in coastal zones invasion by rooikrans *Acacia cyclops* has led to the stabilization of naturally mobile sand dunes through increased plant cover and root biomass. This has altered coastal sediment movements (an ecosystem service that replenishes beach sand) and has led to massive beach depletion in the Eastern and Western Cape provinces (Richardson & van Wilgen 2004).

Richardson & Kluge (2008) summed up the main impacts that the invasive acacias have in natural and semi-natural ecosystems in South Africa:

1. In fynbos they change plant community structure and increase flammability through increased fuel loads;
2. In fynbos they increase soil erosion due to water repellancy induced by very intense fires in dense stands;
3. Especially in fynbos, they change soil chemistry which transforms ecosystems leading to, among other things, increased invasion by alien grasses;
4. Especially in fynbos, dense stands lead to reduced native species diversity;
5. Especially in fynbos, dense stands reduce water production in catchments;
6. In riparian habitats in all biomes, dense stands transform native communities with marked alteration of ecosystem functioning;
7. In all biomes, dense stands lessen the aesthetic, recreational and scientific value of plant communities; and
8. In coastal ecosystems, dense stands stabilise naturally mobile sand dunes, altering coastal sediment movement and leading to extensive beach erosion.

The inherent characteristics of the Australian acacias, such as their rapid growth rates and copious seed production, together with the absence of natural enemies, give them a competitive advantage over native plant species (Richardson & Kluge 2008).

Alien invasive plants constitute the single largest threat to biodiversity on the conservancy, with almost two-thirds of the area exhibiting some degree of invasion (3 600 ha or 62%) and almost half (2 300 ha or 40%) of the area containing either dense or moderate stands. While Port Jackson *Acacia saligna* and other invaders occur in small pockets, rooikrans *Acacia cyclops* is by far the most important alien invader (Kruger & Cowling 1996) on the conservancy. Rooikrans was determined by Robertson *et al.* (2003) to be the thirteenth highest priority alien plant invader in South Africa. This conclusion was reached after an examination of seventeen criteria and the assessments of several experts on priority ratings for the most important alien invasive plants in South Africa. With annual seed production of between 1 400 and 5 100 seeds per square metre of canopy, rooikrans exhibits a higher fecundity in the fynbos of South Africa than in its native land (Holmes *et al* 1987). Its seeds are dispersed by birds, and rapid spread is inevitable, particularly in areas that have been disturbed (Kruger & Cowling 1996).

The management of alien invasive plant species thus represents a major challenge facing the landowners of the conservancy. Given their potential for rapid increase, in excess of 7% per year (Cowling 1997), alien invasive plants may encroach much of the remaining un-invaded area of the study area if no control measures are put in place. The negative impacts associated with this scenario are significant, and include:

1. Loss of biodiversity, including endangered ecosystems, rare and endangered species and critical ecological processes;
2. Destruction of the archaeological sites of the area;
3. Reduction in the scenic value of the area as the natural vegetation is replaced by a dense stands of rooikrans;
4. Reduced accessibility to areas as impenetrable thickets of rooikrans become established;

5. Creation of large standing fuel loads and exacerbation of the dangers of wildfires;
6. A legal problem – landowners are compelled to control alien invasive plant species on their properties in terms of the Conservation of Agricultural Resources Act; and
7. Potential loss of economic opportunities (e.g. through ecotourism).

Addressing the alien invasive plant problem is likely to be a long-term, and expensive, undertaking. With costs for initial clearing of acacia species ranging from approximately R1 500 - R5 000 per ha (Andrew Knipe *pers comm.* 2008), the effective control of alien plants on the conservancy will require the investment of tens of millions of rands.

1.4 Decision support and alien plant control

There is an increasing awareness in the scientific community of the necessity to use interdisciplinary approaches in the management of alien invasive species (Ceddia et al. 2008).

Given the magnitude of the alien plant invasion on the conservancy, the high cost of controlling alien plants and the professed limited resources of the landowners of the conservancy, it is important that alien plant control operations are carefully planned and that resources are invested in the most efficient and effective way. Unfocussed approaches to alien plant control can easily result in wasted resources. Marais *et al.* (2004) suggest that the choice of appropriate courses of action regarding the clearing of invasive alien plant infestations can be assisted by the development of decision-support models. Although landscape-scale models have been developed, these have not been scaled-down to the site level and have not yet been put into practice (Marais *et al.* 2004).

Although the magnitude of the alien plant problem on the conservancy is sufficient in itself to warrant the development of such a system, its necessity is further emphasized by the fact that the conservancy contains a large number of landowners, each of whom would be more inclined to spend their resources within the boundaries of their own properties rather than on conservancy-wide priority areas. The rationale for the establishment of the conservancy is the creation of an institution that enables cross-property management of an entire ecosystem, but the adoption and implementation of such an approach would only be possible if landowners believe that resources are allocated in an equitable manner. In this context, a resource allocation system requires a combination of characteristics:

1. It should allow participation by the landowners in its development;
2. It should be simple and easily understood;
3. It should take the a range of relevant decision-making criteria into account and allow for debate around the relative importance of these criteria;
4. It should enable explicit and defensible decisions; and
5. It should be capable of being easily updated, as conditions change and implementation proceeds.

The next chapter describes approaches to decision-support and resource allocation and, from these, describes appropriate technologies that meet the above criteria and that would be appropriate for use on the conservancy.

2. DECISION-SUPPORT APPROACHES

2.1 Introduction

Decision-making in environmental projects is typically a complex and confusing exercise, requiring trade-offs between socio-political, environmental, and economic factors (Linkov *et al.* 2004). Various approaches have been developed to incorporate consideration of these trade-offs in order to aid decision-making. One such approach is Multiple Criteria Decision Analysis (MCDA). MCDA is a branch of Operations Research that deals with decision-making in a structured and systematic manner.

2.2 Multiple Criteria Decision Analysis

The philosophical departure point from classical Operations Research approaches is the representation of several conflicting criteria (Geldermann & Rentz 2000). Roy (1996) states that MCDA is "a decision-aid and a mathematical tool allowing the comparison of different alternatives or scenarios according to many criteria, often contradictory, in order to guide the decision maker(s) towards a judicious choice".

Mendoza & Martins (2006) conducted a critical review of MCDA in natural resource management. They state that "it is clear that MCDA offers a suitable planning and decision-making framework for natural resources management. Because it is inherently robust, it can also provide a convenient platform that lends itself well in bridging the gap between the soft qualitative planning paradigm and the more structured and analytical quantitative paradigm".

MCDA provides a rigorous foundation to support the decision-making process and holds many advantages over informal judgement. Well-developed MCDA approaches usually have a number of characteristics in common (adapted from Anon. 2007):

1. They are transparent and explicit;
2. They are highly flexible and enable the capture of quantitative and qualitative data and issues;
3. The choice of objectives and criteria is open to analysis and to change;
4. Scores and weights are also explicit and can be cross-referenced to other sources of information;
5. They are relatively simple for clients and stakeholders to use;
6. They permit the development of many alternative scenarios;
7. They allow the exploration of trade-offs;
8. They enable the stakeholder to factor results into decision-making processes; and
9. They provide an important means of communication within the decision-making body and with the wider community.

MCDA problems normally involve six components (Sener 2004):

1. The decision-maker or a group of decision-makers and their preferences with respect to the evaluation criteria;
2. A goal or a set of goals the decision-maker wishes to achieve;
3. A set of evaluation criteria (objectives and/or physical attributes);
4. The set of decision alternatives;
5. The set of uncontrollable (independent) variables or states of nature (decision environment); and
6. The set of outcomes or consequences associated with each alternative attribute pair.

MCDA is primarily concerned with how to combine the information from several criteria to form a single index of evaluation (Chang *et al.* 2008). This index can be used to identify the most preferred alternative, to rank alternatives, to short-list alternatives for subsequent detailed evaluation, or to distinguish acceptable from unacceptable alternatives.

A set of systematic MCDA procedures for have been developed for analyzing complex decision problems in a number of contexts. These procedures include dividing the decision problems into smaller more understandable parts, analyzing each part, and integrating the parts in a logical manner to produce a meaningful solution (Malczewski, 1997).

Most of the MCDA techniques were developed during the latter part of the 20th century. The various techniques are similar in that they make the alternatives and their contribution to the different criteria explicit and that they all require the exercise of judgement. They differ principally in the manner that they combine and utilize data (Spackman *et al.* 2000). MCDA approaches can be classified into three broad groups on the basis of the major components of the decision analysis (Sener 2004): 1) Multiple Objective Decision Making (MODM) versus Multiple Attribute Decision Making (MADM); 2) individual versus group decision maker problems; and 3) decisions under certainty versus decisions under uncertainty.

Table 4: Characteristics of MADM and MODM models (Chakhar & Martel 2003)

Multiple Attribute Decision-Making	**Multiple Objective Decision-Making**
Restricted set of alternatives	High or infinite number of feasible solutions
Explicitly defined set of alternatives	Implicitly defined set of feasible solutions
Aggregation based on outranking relation or utility function	Uses a local and an interactive aggregation algorithms
Requires a priori information on the decision maker's preferences	Requires much less a priori information on the decision maker's preferences

MADM deals with a discrete and usually limited (finite) number of pre-specified alternatives whereas MODM deals with variable decision values to be determined within a continuous (infinite) number of choices (see Table 4) (Chakhar & Martel 2003). MODM analyses a subset of continuous vector space by locating all efficient solutions before determining the optimum, based on the user's preferences (Geldermann & Rentz 2000). MADM problems require that choices be made among alternatives that are described by their attributes and the set of attributes is given explicitly. Unlike MADM, MODM requires that means-ends relationships be specified, since they deal explicitly with the relationship of attributes to higher level objectives. MODM involves designing the alternatives, with each alternative being defined implicitly in terms of the decision variables and evaluated by means of objective functions (Malczewski 1997) MODM methods are suitable for operational planning. However, for the comparison of several alternatives or for strategic planning, the MADM approaches should be used (Geldermann & Rentz 2000).

MADM and MODM problems can be further classified as individual or group decision-making, according to their goal-preference structure. If there is a single goal preference, the problem is considered as individual decision-making regardless of the number of decision-makers involved. However, if the individual or interest groups are characterized by different goal preferences, the problem is considered as group decision-making (Malczewski 1997).

A next level of classification of MCDA approaches depends on the certainty of the decision. If the decision-maker has perfect knowledge of the decision environment, the decision is considered as decision under certainty. Although certainty is assumed for many techniques, most of the real world decisions involve some aspects that are unknown and difficult to predict. These types of decisions are classified as decisions under uncertainty. The decisions under uncertainty can be further subdivided into fuzzy and probabilistic decision-making. The probabilistic decisions are handled by probability theory and statistics. If the situation is ambiguous, fuzzy set theory is employed and the problem is structured as the degree of how much an event belongs to a class (Sener 2004).

2.3 General model of Multiple Attribute Decision-Making

The main advantage of MADM is that it can give decision-makers many dimensions to consider related elements, and evaluate all possible options under variable degrees (Chu *et al*. 2007).

MADM principally comprises of two mathematical steps: 1) aggregation of judgements with regard to each criterion and each alternative, and 2) the ranking of alternatives according to aggregation rules (Geldermann & Rentz 2000). The primary requirement of most MADM techniques is a performance table or performance matrix that contains the evaluations or scores of a set of alternatives on the basis of a set of criteria. The next requirement is the aggregation of the different criteria scores using a specific aggregation procedure. Decision-maker preferences are taken into account by weights that are assigned to different criteria. The aggregation of criteria scores allows the decision-maker to make comparison between the different alternatives on the basis of these scores (Chakhar & Martel 2003).

The following is a generalised procedure is common to MADM techniques:

1. Identification of objectives;
2. Identification of alternatives for achieving the objectives;
3. Identification of the criteria to be used to compare the alternatives;
4. Scoring the alternatives;
5. Weighting; and

6. Selection of alternatives.

2.3.1. *Identification of objectives*

Sound decision-making should begin with the setting of clear objectives. An objective is a statement about the desired state of the system under consideration (Chankong & Haimes 1983). Objectives can be classified into to ultimate and immediate objectives. Ultimate objectives are usually framed in terms of strategic or higher-level deliverables whereas immediate objectives are those that can be directly linked with the outputs of the project. Consideration of the proposed alternatives requires a focus on those criteria which contribute to the immediate objectives, which in turn contributes to the achievement of the ultimate objectives.

2.3.2. *Identification of alternatives for achieving objectives*

After the objectives have been defined, alternatives for the achievement of the objectives are identified. Chakhar & Mousseau (2007) define decision alternatives as alternative courses of action among which the decision-maker must choose. A spatial decision alternative consists of at least two elements: an action and a location.

The set of spatial decision alternatives may be discrete (MADM) or continuous (MODM). In the case of MADM, the problem involves a discrete set of pre-defined decision alternatives. Spatial alternatives are often modelled through a combination of the basic spatial primitives, namely points, lines, or polygons. In the case of MODM there are a high or infinite number of decision alternatives, often defined in terms of constraints. For practical reasons, the set of potential alternatives is often represented in a “discretised" form where each raster represents an alternative. Alternatives may also be constructed as a collection of rasters (square grid cells) (Chakhar & Mousseau 2007).

2.3.3. *Identification of criteria*

After the identification of the alternatives, the MCDA practitioner must decide on how to compare the contribution of the different alternatives to the achievement of the objectives. This requires the selection of criteria to reflect performance in meeting the objectives. Each criterion must be measurable, i.e. it must be possible to measure how well a particular alternative is expected to perform in relation to the criteria (Spackman *et al.* 2000). Constraints or admissibility criteria are natural or artificial restrictions on the potential alternatives. Constraints may be used in the pre-analysis steps to divide alternatives into two categories: “acceptable" or “unacceptable". An alternative is acceptable if its performance on one or several criteria exceeds a minimum or does not exceed a maximum (Chakhar & Mousseau 2007).

In the context of spatial decision-making, evaluation criteria are associated with geographical entities and relationships between entities, and are represented in the form of criteria maps. These criteria maps are not merely simple map layers, but are models of the preferences of the decision-maker concerning a particular concept. This is in contrast to simple map layers, which are representations of some spatial real data. A criterion map represents subjective preferential information and different individuals may assign different values to the same mapping unit in a criterion map (Chakhar & Mousseau 2007).

Keeney & Raiffa (1993) suggested the following checklist to support the selection of evaluation criteria:

1. Completeness – all relevant criteria are reflected;
2. Measurability – the performance of the alternatives against all the criteria can be measured;
3. Decomposability – the criteria cannot be refined into further sub-criteria; and

4. Absence of duplicates – a criterion does not reflect the performance of an attribute already considered under another criterion.

2.3.4. Scoring the alternatives

An alternative's attributes are the measurable quantities reflecting the degree to which objectives (defined by the criteria) are achieved by the alternative (Sener 2004). In order to score the attributes of an alternative for a given objective, two properties, comprehensiveness and measurability, should first be satisfied. An attribute is comprehensive if its value sufficiently indicates the degree to which the objective is met. And it is measurable if it is reasonably practical to assign a value in a relevant measurement scale. The ratio, interval, ordinal and binary scales are suitable for measurement of attributes, whereas nominal scale is not since it does not allow an ordering of the alternatives (Janssen, 1992).

The evaluation of scores for the alternatives may be either quantitative or qualitative. Qualitative data can be converted into quantitative data by the assignment of numeric values to such data. This requires the definition of a measurement scale. The most frequently used measurement scale is the Likert-type scale, which is composed of the same number of favourable and unfavourable levels with a neutral level between these. An example of a Likert-type scale with five levels is: very unfavourable, unfavourable, neutral, favourable, very favourable. The quantification procedure consists of constructing a measurement scale like the one mentioned above and then associating numerical values with each level of the scale. For example, the numbers 0, 1, 2, 3 or 4 may be associated with the five-point scale from very unfavourable, unfavourable, neutral, favourable to very unfavourable (Chakhar & Mousseau 2007).

Although the evaluation of alternatives may be expressed according to different scales, a large number of MCDA methods require that all of their criteria are expressed in a similar scale. This allows for comparison between and within criteria. There are many standardization procedures, but the most frequently used is the linear transformation procedure. This procedure obtains the percentage of the maximum value for each alternative within each criterion (Chakhar & Mousseau 2007).

A standard feature of MADM-type MCDA is a performance matrix, or consequence table, in which each row describes an alternative and each column describes the performance of the alternatives against each criterion (see Table 5).

Table 5: Example of a performance matrix (using arbitrary information).

	Criterion 1		**Criterion 2**		
Alternatives	**Indicator 1a**	**Indicator 1b**	**Indicator 2a**	**Indicator 2b**	**Indicator 2c**
Alternative 1	5	3	4	2	5
Alternative 2	3	1	4	3	1
Alternative 3	2	3	1	5	4

2.3.5. *Weighting*

In MCDA, it is usual for the decision-maker to consider one criterion to be more important than another. This is accounted for by assigning weights to the different criteria. Assigning weights accounts for the changes in the range of variation for each evaluation criterion and the different degrees of importance being attached to these ranges of variation (Kirkwood 1997). Criteria weight determination is the key point in obtaining the total scores of alternatives and most importantly the conclusion of MCDA problems (Aldian & Taylor 2005).

The criteria weight determination methods can be classified into two main groups, namely objective approaches and subjective approaches. In the objective approaches, criteria weights are derived from information contained in each criterion through mathematical models (Aldian & Taylor 2005). In subjective approaches, criteria weights are derived from the decision-maker's judgement. These judgments are often obtained through a series of designed questions that are posed to the decision-maker (Aldian & Taylor 2005). Although techniques have been developed to assist in the assigning of weights, Geldermann & Rentz (2000) state that in most MCDA situations, the setting of weights for the simultaneous consideration of the numerous investigated criteria will remain subjective to a certain extent. The four principle techniques for assigning weights are: ranking, rating, pairwise comparison and trade-off analysis methods (Sener 2004).

The simplest method for evaluating the importance of criteria and determining appropriate weights is *ranking*. In ranking, every criterion under consideration is ranked in the order of decision-maker's preferences. Weights are then assigned in proportion to the preference ranking of the criteria. This method is attractive to many MCDA practitioners due to its simplicity. However, it becomes less appropriate as the number of criteria used increases. Another disadvantage of this method is its lack of theoretical foundation (Sener 2004).

Rating requires that a decision-maker estimate the relative importance of the criteria, and thus the weights, on the basis of a predetermined scale. One of the simplest rating methods is the point allocation method (or fixed-point method). Under this approach, criteria are allocated points on a scale of 0 to 100, where 0 indicates that the criterion can be ignored, and 100 represents a situation where only one criterion requires consideration. The point allocation method forces the decision-maker to consider trade-offs among criteria because it is not possible to assign a higher weight to one criterion without reducing the weight assigned to one or more of the other criteria (Prato & Herath 2006). Another rating method is the ratio estimation procedure, a derivative of the point allocation method. A score of 100 is assigned to the most important criterion and proportionally smaller weights are given to criteria lower in the order. Ratios are calculated by dividing the score assigned to each criterion by the score of the least important criterion. Disadvantages of this method are that assigned weights may be difficult to justify, and the lack of theoretical foundation for this approach (Sener 2004).

The *pairwise comparison method* requires pairwise comparisons between the criteria. A ratio matrix is created from these comparisons and this ultimately results in the production of relative weights as an output. The pairwise comparison method involves the following steps (see Sener 2004; Boroushaki & Malczewski 2008):

1. The development of a pairwise comparison matrix (see Table 6): The method uses a scale to define the pairwise relative importance of criteria with values range from 1 to 9 (see Table 7).

Table 6: An arbitrary example of a pairwise comparison matrix.

	Criterion A	Criterion B	Criterion C	Criterion D
Criterion A	1	2	3	4
Criterion B	1/2	1	2	3
Criterion C	1/3	1/2	1	4
Criterion D	1/4	1/3	1/4	1

Table 7: Scale for determining pairwise relative importance of criteria (from Sener 2004)

Intensity of importance	Definition
1	Equal importance
2	Equal to moderate importance
3	Moderate importance
4	Moderate to strong importance
5	Strong importance
6	Strong to very strong importance
7	Very strong importance
8	Very to extremely strong importance
9	Extreme importance

2. Computation of the weights: computation of the weights involves three steps. First the values in each column of the matrix are summed. Then, each element in the matrix is divided by its column total and the resultant normalized pairwise comparison matrix is obtained. Next the elements in each row are averaged by dividing the sum of normalized scores for each row by the number of criteria. These averages provide an estimate of the relative weights of the criteria being compared.

3. Estimation of the consistency ratio: the consistency ratio assists in determining if the comparisons between criteria are consistent or not and involves following operations:

a. Determine the weighted sum vector: multiply the weight for the first criterion by the first column of the original pairwise comparison matrix, then multiply the second weight times the second column, and so on. These values are then summed across the rows of the matrix;
b. Determine the consistency vector: divide the weighted sum vector by the criterion weights determined previously;
c. Compute lambda (λ) and the Consistency Index (CI): lambda is the average value of the consistency vector. CI provides a measure of departure from consistency and has the formula below:

   $$CI = (\lambda - n) / (n-1)$$

d. Calculation of the consistency ratio (CR): using the formula below:

   CR = CI / RI

   Where RI is the random index and depends on the number of elements being compared (see Table 8).

Table 8: Randomness Index for *n* number of criteria (Seener 2004).

n	RI	n	RI	n	RI
1	0.00	6	1.24	11	1.51
2	0.00	7	1.32	12	1.48
3	0.58	8	1.41	13	1.56
4	0.90	9	1.45	14	1.57
5	1.12	10	1.49	15	1.59

If CR < 0.10, the ratio indicates a reasonable level of consistency in the pairwise comparison. If CR ≥ 0.10, the values of the ratio indicates inconsistent judgments.

The advantages of the pairwise comparison method to determine weights is its simplicity – only two criteria are considered at a time – and the fact that it can be implemented in a spreadsheet environment and incorporated into GIS-based decision making procedures. Disadvantages are that, the relative importance of evaluation criteria is determined without considering the scales on which the criteria are measured and that decision systems with many criteria will require a very large amount of pairwise comparisons (Sener 2004).

Trade-off analysis requires that the decision-maker compare two alternatives with respect to two criteria at a time in order to assess which alternative is preferred. Trade-offs are then used define unique set of weights that assigns the same overall value to all of the equally preferred alternatives. This method assumes that the trade-offs that the decision maker is willing to make between any two criteria do not depend on the levels of other criteria. The disadvantage of this method is that the decision-maker is presumed to obey the axioms and the fact that small differences in the consistency of judgements can significantly alter the outcome (Ref).

2.3.6. Selection of alternatives

The final stage of the decision making process is the selection of an alternative. This should be seen as a separate stage because MCDA cannot incorporate into the formal analysis every judgement which should be taken into account in the final decision. At this stage it may be decided that a further alternative or alternatives should be considered and the analysis revisited.

2.4 An overview of MCDA techniques

Within MADM, aggregation procedures generally fall into two categories, with two corresponding schools of practice: 1) value/utility function-based, and 2) outranking relation-based (Chakhar & Martel 2003). The American school focuses on the value/utility function-based approaches while the European (primarily French) school is based on outranking approaches (Geldermann & Rentz 2000).

The American school assumes that the decision-maker has an accurate understanding of the value/utility of the alternatives against the criteria, and also of the relative importance of the criteria. The most prominent approaches are Simple Additive Weighting (SAW), the Analytical Hierarchy Process (AHP) and the Multi Attribute Value / Utility Theory (MAVT/MAUT). The European school assumes that preferences are not apparent to the decision-maker and that decision-support is necessary for structuring the problem and for providing insight into the consequences of different weightings. The emphasis under this school is on the recognition of limits of objectivity. The prominent methods within this philosophy are ELECTRE, PROMETHEE, ORESTE and TOPSIS (Geldermann & Rentz 2000).

Geldermann & Rentz (2000) compared the results of models from the American (using SAW and MAUT) and the European (using PROMETHEE) schools and found striking similarities. They found that, despite the philosophical differences in the approaches, their mathematical outcomes are similar. The formation and existence of the two schools is often subject of heated debate, primarily because there is no common view about a psychologically "correct" method of modelling human value judgements. Without such a generally accepted paradigm, it has been stated that there are only competing schools and sub-schools and that much of the work has been directed at building new methods within the existing foundations, rather than on debating across the philosophies (Geldermann & Rentz 2000).

2.5 Value/utility function-based approaches

A value-function (or in certain cases, a utility-function) describes the conversion from the "natural scale" of the alternative for a given criterion to the value scale, which can be continuous or discrete. Depending on the approach, either direct numerical judgements or indifference methods may be used in declaring and measuring value. Direct numerical judgements measure value through methods such as direct rating, ratio estimation, category estimation and curve estimation. In these cases, the MCDA practitioner or respondents are asked to make direct estimates of strengths of preferences on a numerical scale. Indifference methods require that pairs of evaluation objects are varied until a match is established in their respective strengths of preference. Techniques include difference standard sequences, bisection, variable probability and variable certainty equivalent methods (Seppälä 2003).

In order to describe the mathematical foundation for the approaches, the following basic MADM notation is used: Consider the set A of T alternatives to be ranked, and set F of K criteria to be optimised:

A : $= \{a_1,\dots,a_T\}$: Set of discrete alternatives or techniques a_t ($t = 1\dots T$)

F : $= \{f_1,\dots,f_K\}$: Set of criteria relevant for the decision f_k ($k = 1\dots K$)

A decision matrix or performance matrix D: = $(x_{tk})_{t=1,\dots,T\,;\,k=1,\dots,K}$, which is a ($T \times K$) matrix, can be constructed and whose elements $x_{tk} = f_k(a_t)$ indicate evaluation of alternative a_t, with respect to criterion f_k (Geldermann & Rentz 2000):

$$D = \begin{bmatrix} x_{11} & \cdots & x_{1K} \\ \vdots & x_{tk} & \vdots \\ x_{T1} & \cdots & x_{TK} \end{bmatrix} := \begin{bmatrix} f_1(a_1) & \cdots & f_K(a_1) \\ \vdots & f_k(a_t) & \vdots \\ f_1(a_T) & \cdots & f_K(a_T) \end{bmatrix}$$

2.5.1. *Simple Additive Ranking (SAR)*

This approach considers only the measure ranks of the alternatives. The alternatives are first assigned a rank for each criterion and then the weighted ranks are summed (Pelkonen 2003). Both descending (most preferred alternative has a rank of 1) and ascending rank (most preferred alternative has a rank equal to the total number of alternatives) orderings can be defined and result in the same rank order. It has to be defined which ordinal score is to be given, if two or more alternatives score even on a certain criterion (Geldermann & Rentz 2000).

The overall value $v(a_t)$ of the alternatives is thus the summed weighted ordinal ranks with respect to all the criteria, represented by the following equation (Geldermann & Rentz 2000):

$$v(a_t) = \frac{1}{T}\sum_{k=1}^{K} w_k . R_k\left(f_k(a_t)\right) \quad \text{with } w_k \geq 0 \text{ and } \sum_{k=1}^{K} w_k = 1$$

Where:

a_t: Alternative t

f_k: Criterion k

w_k: Weighting factor of criterion k

T: The total number of alternatives

R_k: Ordinal rank of an alternative with regard to criterion k

If $f_k(a_t) \rightarrow$ max: $R_k(\max\{f_k(a_t)\}) = T$ and $R_k(\min\{f_k(a_t)\}) = 1$

If $f_k(a_t) \rightarrow$ min: $R_k(\max\{f_k(a_t)\}) = 1$ and $R_k(\min\{f_k(a_t)\}) = T$

Normalisation is not necessary for obtaining a rank order of the regarded alternatives, but allows a more intuitive graphical representation (Geldermann & Rentz 2000). In spite of the crude differentiation between the alternatives, SAR is a widely used approach. A drawback of SAR is that it is prone to rank reversals (the so called Borda Count effect) (Geldermann & Rentz 2000).

2.5.2. Simple Additive Weighting (SAW)

Simple Additive Weighting (SAW) is an additive model based on the evaluation of the value of alternatives with regard to each criterion, the subsequent aggregation of the weighted values and the optional normalisation of the results that are obtained (Geldermann & Rentz 2000). Unlike SAR, which considers only the aggregated ranks, SAW aggregates the relative performance of the alternatives (Pelkonen 2003).

In SAW, the values $v(a_t)$ of the alternatives against the criteria are declared through value functions, which may be linear or non-linear. Direct numerical judgements are used to measure value, most often through direct rating, ratio estimation, category estimation and curve estimation (Geldermann & Rentz 2000).

Values are standardised to a common dimensionless scale to allow for comparison between criteria with different scale ranges (Sener 2004). The simplest method to do this is to normalise the data by dividing the raw value (v') of the alternative for a given criterion by the maximum raw value returned from the set of alternatives for that criterion:

$$v_k(f_k(a_t)) = \frac{v_k'\,(f_k(a_t))}{v_k'\,(f_k(A))^{max}}$$

Where:

$v_k(f_k(a_t))$: Standardised value of alternative t with respect to criterion k

$v_k'(f_k(a_t))$: Raw data value for alternative t with respect to criterion k, determined through a value function

$v_k'(f_k(a_T))^{max}$: Maximum raw data value from the score from the set of *A* alternatives for criterion *k*.

Once the values of the alternatives with respect to the various criteria are standardised to a common scale, the values are multiplied by the criteria weights and summed across the criteria for each alternative to obtain an overall ranking, as represented by the following equation:

$$v(a_t) = \sum_{k=1}^{K} w_k . v_k\ (f_k(a_t)) \quad with\ w_k \geq 0\ and \sum_{k=1}^{K} w_k = 1$$

Where:

$v(a_t)$: Overall value of the alternative *t* (the summed weighted values of the alternative against all of the criteria)

w_k: Weighting factor of criterion *k*

K: Total number of *k* criteria

The result is a preference order of the alternatives, with the higher ranking alternatives indicating higher preference or value over the lower ranking alternatives (Geldermann & Rentz 2000). Most MCDA approaches use the SAW model and it has a well-established record of providing robust and effective support to decision-makers working on a range of problems and in various circumstances. The model can only be applied if it can be reasonably assumed that criteria are preferentially independent of each other and if uncertainty is not formally built into the MCDA model (Spackman *et al.* 2000; Critto *et al.* 2006).

2.5.3. *Analytical Hierarchy Process (AHP)*

The Analytical Hierarchy Process (AHP) was developed by Saaty in 1980 who states that it is a useful tool to analyse decisions in complex social and political problems. It is suitable for cases where many interests are involved and a number of people participate in the judgement process (Saaty 1980; Ananda & Herath 2003). It is a variant of SAW that is based on three principles: decomposition, comparative judgment and synthesis of priorities (Sener 2004).

Decomposition involves breaking a complex decision problem into simpler decision problems to form a decision hierarchy. The decision hierarchies within AHP typically include four levels. Sener (2004) uses the four levels of goal, objectives, attributes and alternatives. Berliner (2005) used the levels of principles, criteria, indicators and verifiers. The top level of the decision hierarchy is the ultimate goal of the decision. The hierarchy then reduces from general to specific until the level of the attributes is reached. Operations take place independently at different hierarchy levels and then each level is linked to the next higher level. This allows for smaller numbers of factors to be taken into account at a time (Spackman *et al.* 2000).

Once the decision problem is decomposed into a hierarchy, the objectives and alternatives are scored. Procedures for deriving weights at the various levels of the hierarchy are usually based on pairwise comparisons, supported by a comparison matrix. Thus, in assessing weights, the decision-maker is asked a series of questions on how important one particular element of the hierarchy is relative to another for the decision being addressed (Spackman *et al.* 2000). Pairwise comparison reduces the complexity of decision-making since two components are considered at a time. The consistency of the weight estimation procedure can be checked through determining the consistency ratio for the pairwise comparisons at each level of the hierarchy (Sener 2004).

There has been substantial debate amongst MCDA practitioners on the strengths and weaknesses of the AHP. An advantage of this approach is that users generally find pairwise comparison of hierarchical elements to be straightforward and convenient. However, concerns have been raised over the theoretical foundations of the AHP (Spackman *et al.* 2000).

2.5.4. Multi-attribute value theory / utility theory (MAVT / MAUT)

The multi-attribute value theory (MAVT) and multi-attribute utility theory (MAUT) model were derived in the 1940s and 1950s. These approaches are based on the hypothesis that a value or utility function can be defined for the considered alternatives which the decision-maker wishes to maximise. The difference between MAVT and MAUT is that MAVT is based on value functions whereas MAUT is based on utility functions (Seppälä 2003). A value function combines the multiple evaluation measures (attributes) into a single measure of the overall value of each alternative. Utility functions involve the computation of an expected utility of an alternative. Prominent authors with respect to MAVT / MAUT are Keeney and Raiffa (see Keeney & Raiffa 1993). They used the theoretical foundations provided by the earlier researchers to develop a set of procedures which allow decision-makers to evaluate multi-criteria alternatives in practice (Spackman *et al.* 2000).

The three building blocks for MAVT / MAUT procedures are 1) the performance matrix, 2) procedures to determine whether criteria are independent of each other, and 3) estimation of the parameters in a mathematical function which results in a single number index, U, to represent the decision-maker's overall valuation of an alternative in terms of the value of its performance on each of the separate criteria. In MAVT / MAUT value or utility functions are generated by indifference methods, where pairs of evaluation objects are varied until a match is established in their respective strengths of preference. Difference standard sequences and bisection are the indifference methods used in MAVT, whereas variable probability and variable certainty equivalent methods are used in MAUT (Seppälä 2003).

Although the approach is well regarded and effective, it is relatively complex due to the fact that it formally builds uncertainty into the model and secondly that it does not allow attributes to interact with each other in a simple, additive fashion. Although it may be important to build one or both of these factors into the analysis, it is often better to ignore them in practice to allow for simpler and more transparent decision support (Spackman *et al.* 2000).

2.5.5. *Outranking relation-based approaches*

The value/utility function-based approaches all suffer from the same drawback; that of complete 'compensation'. Complete compensation between attributes occurs when a sufficiently large gain in a lesser attribute, or a number of lesser attributes, is able to compensate for a small loss in a far more important attribute (Stewart 1992).

In order to overcome the problem of complete compensation and of the existence of a 'true' ranking of the alternatives, the outranking or concordance methods were developed within the European school of MADM. Outranking rather takes into account the facts that preferences are 1) not constant in time, 2) are not unambiguous, and 3) are not independent of the process of analysis (Geldermann & Rentz 2000).

Outranking methods seek to eliminate alternatives that are 'dominated'. Dominance, in terms of outranking, uses weights to give more influence to some criteria than others. One alternative outranks another if it outperforms the other on enough criteria of sufficient importance (as reflected by the sum of the criteria weights) and is not outperformed by the other alternative by recording a significantly inferior performance on any one criterion. All alternatives are then assessed in terms of the extent to which they exhibit sufficient outranking with respect to the full set of alternatives being considered as measured against a pair of threshold parameters (Spackman *et al*. 2000).

Outranking can thus be defined as follows: alternative a_t outranks $a_{t'}$, if there is a "sufficiently strong argument in favour of the assertion that a_t is preferable to $a_{t'}$ from the decision maker's point of view". Accordingly, the outranking relation is the result of pairwise comparisons between the alternatives with regard to each criterion (Geldermann & Rentz 2000).

'Classical' decision making, like SAW, is based on strict preference ($a_t\ P\ a_{t'}$), i.e. alternative a_t is strictly preferred to $a_{t'}$, and indifference ($a_t\ \ I\ a_{t'}$), i.e. a_t is as good as $a_{t'}$. But in reality situations may exist in which neither of a pair of alternatives outranks the other. If the decision-maker consequently cannot declare a_t better than $a_{t'}$ or vice versa, the outranking methods allow explicitly for incomparability ($a_t\ R\ a_{t'}$). Moreover, the concept of weak preference (a_t Q $a_{t'}$) is used, if for example the decision maker declares alternative a_t to be just slightly better than $a_{t'}$ (Geldermann & Rentz 2000). The outranking methods require similar input data to that required for the value/utility function procedures. They also require the specification of alternatives, assessment of the performance of the alternatives against a series of criteria and determination of weights to express the relative importance of the criteria (Spackman *et al.* 2000).

The main disadvantage of outranking methods is that they are dependent on arbitrary definitions of what constitutes outranking and how the threshold parameters are set and manipulated by the decision-maker. The advantages of outranking methods are that they encourage greater interaction between the decision-maker and the model and that they capture some of the political realities of decision-making. In particular outranking methods downgrade alternatives that perform badly on any one criterion. Thus they recognize that alternatives may prove to be unacceptable to stakeholders if the alternatives score poorly on one criterion, even if the alternatives perform well against the other criteria. However, due to their complexity, the potential for widespread use of the outranking methods appears to be somewhat limited (Spackman *et al.* 2000). The most widely applied outranking method is the Elimination and Choice Translating Reality (ELECTRE I), and several of its derivatives (ELECTRE II, III, IV, PROMETHEE I and II) (Sener 2004).

2.5.6. *Ideal point methods*

The ideal point method involves ranking alternatives according to their degree of separation from an ideal point, which is defined as the most desirable, weighted, hypothetical alternative. The degree of separation is measured in terms of metric distance and the alternative closest to the ideal point is the declared to be the preferred alternative. (Sener 2004).

The most popular ideal point method is the Technique for Order Preference by Similarity to the Ideal Solution (TOPSIS) developed by Hwang and Yoon in 1981 (Sener 2004). Although the ideal point methods can be implemented both in raster and vector GIS, the technique is more suited to the raster GIS environment. Ideal point methods regard an alternative as an inseparable collection of attributes (defined by the criteria), and this method is thus attractive when the dependency among attributes is difficult to test or verify (Sener 2004).

2.6 Sensitivity Analysis

The uncertainty associated with any decision situation requires that a sensitivity analysis be conducted. This allows the decision-maker to test the consistency of a given decision or its variation in response to any modification in the input data and/or in the decision maker preferences (Chakhar & Martel 2003). Sensitivity analysis is conducted by varying the decision weights to investigate their impacts on the rank ordering of the alternatives (see Geneletti 2008).

2.7 Integrating MCDA and GIS

Spatial MCDA differs from conventional MCDA techniques in that it includes an explicit geographic component. In contrast to conventional MCDA, spatial MCDA requires information on criterion values and the geographical locations of alternatives in addition to the decision-makers' preferences with respect to a set of evaluation criteria. The results of the analysis thus depend not only on the geographical distribution of attributes, but also on the value judgments involved in the decision-making process (Al-Shalabi *et al.* 2006).

Geographic Information Systems (GIS) integrate several components and different subsystems to collect, store, retrieve and analyze spatially-referenced data. Although the numerous practical applications of GIS have shown that it is a powerful tool for acquiring, managing and analyzing spatial data, many management science specialists share the opinion that GIS are limited with respect to spatial decision-aid. This is largely because of its lack of more powerful analytical tools that would enable it to deal with spatial problems involving several conflicting criteria (Chakhar & Martel 2003).

Chakhar & Martel (2003) summarise the criticisms that have been addressed to GIS technology by numerous authors as follows:

1. The decision-maker's preferences (e.g. criteria weights) are not taken into account by current GIS.
2. In most GIS packages spatial analytical functionalities lie mainly in the ability to perform deterministic overlay and buffer operations, which are of limited use when multiple and conflicting criteria are concerned;
3. Current GIS do not permit the assessment and comparison of different scenarios. They identify only solutions satisfying all criteria simultaneously;
4. Analytical functionalities found in most GIS are oriented towards the management of data not towards an effective analysis of them;
5. Overlaying techniques in standard GIS become more difficult to comprehend as the number of layers increases; and
6. Overlaying methods consider that all features are of equal importance.

Chakhar & Martel (2003) believe that the solution to some of the shortcomings of GIS with regard to decision-support can be overcome by integrating GIS with Operations Research or Management Science tools. They contend that the most suitable family of tools for integration with GIS is MCDA, which is highly complementary to GIS: while GIS is a powerful tool for managing spatially referenced data, MCDA is an efficient tool for modelling spatial problems (Chakhar & Martel 2003). Malczewski (2006) defines GIS-based MCDA as "a process that integrates and transforms geographic data (map criteria) and value judgments (decision maker's preferences and uncertainties) to obtain overall assessment of the decision alternatives." The integration of analytical techniques designed to cope with multicriteria problems in a Geographic Information System (GIS) can provide the user with a valuable addition to the functionality of GIS (Carver 1991).

2.8 Case Studies that integrate MCDA and GIS

The use of MCDA in resource allocation decisions for invasive species is rare (Cook & Proctor 2007). However, numerous studies have incorporated GIS and MCDA technologies in order to support spatial decision-making.

Mendoza *et al.* (2002) used MCDA to evaluate military training areas for restoration in the Fort Hood military training area in Texas. Three measures were used to determine land condition, namely: erosion status, percent vegetative cover and range condition. MCDA methods were integrated with GIS to make their land condition assessment spatially explicit. From this they developed a GIS-based land repair allocation model to identify and prioritize critical areas for restoration. By making use of MCDA techniques, they were able to move away from assessing land condition based on a single factor, erosion status, as had been done previously and were able to produce a meaningful assessment of suitable areas for land restoration.

Setiawan *et al.* (2004) integrated MCDA and GIS techniques to identify and map areas most likely to be affected by peat swamp forest fires in Pahang, Malaysia. They examined factors such as land use, road network, slope, aspect and elevation in order to develop a spatially weighted index model, which was interpreted into a fire hazard assessment model. They validated their model against maps of actual fire occurrence and showed that most of the actual fire spots were located in the very high and high fire risk zones identified by their model.

Berliner & Macdonald (2005) used a combination of questionnaires and MCDA to identify nodes for the South African Environmental Observatory Network (SAEON). In the questionnaire component, experts were asked to identify potential sites and then to rank these together with the thirteen sites that were originally proposed for the network. The MCDA component involved analysis of the suitability of potential sites based on eight selection criteria. By using expert knowledge combined with MCDA and GIS, the investigators were able to make recommendations on suitable individual sites and suitable combinations of sites for the SAEON Fynbos node.

Genelettti (2008) used GIS and MCDA to determine suitable areas in northern Italy for the development on new ski sites. The following indicators were used: ecosystem loss and fragmentation, soil erosion, geomorphologic hazards, interference with flora and fauna and visibility. MCDA was used to generate composite indices and to rank ski areas according to their overall suitability.

Geneletti & van Duren (2008) applied MCDA techniques in a spatial context to support the zoning of the Paneveggio-Pale di S. Martino Natural Park in Italy. The park was zoned into three protection levels, ranging from strict conservation to tourism and recreation. The park was partitioned into homogenous land units and an MCDA-GIS based land suitability analysis was carried out for each unit.

Ultimately the study should assist the park's management and other stakeholders in determining appropriate zoning by providing a zoning approach that is scientifically sound and practical.

2.9 Shortcomings with MCDA

The following shortcomings have been noted with regard to MCDA:

- In additive models, the aggregation function may be multiplicative instead of additive if the criteria are non-compensatory (i.e. when good performance on one criterion does not compensate for poor performance on another) (Hajkowicz & Higgins 2008).
- Criteria transformations may be non-linear, often concave or convex forms more accurately capture decision maker preferences (Hajkowicz & Higgins 2008).
- Sometimes additive models produce only very minor differences in the final value for the alternatives, which may be insufficient to differentiate performance (Hajkowicz & Higgins 2008).

2.10 Selection of MCDA technique

The criteria suggested by Spackman *et al.* (2000) for selection of MCDA methods for practical application in real world decision-making problems were used as a guide in selecting techniques for application on the conservancy. These criteria are (Spackman *et al.* 2000):

1. Internal consistency and logical soundness of the method;
2. Transparency of the method;
3. The method's ease of use;
4. Data requirements for the method are not inconsistent with the magnitude of the issue being considered;

5. Realistic time and manpower resource requirements for the analysis process;
6. The ability of the method to provide an audit trail; and
7. The availability of software, where needed, for the application of the method.

The primary consideration in the selection of an appropriate MCDA technique for application on the conservancy was that the decision-making process must easy to understand and interrogate by stakeholders. Stakeholders not versed in the intricacies of sophisticated methods and technologies often meet their application with scepticism (van der Merwe & Lohrentz 2001). This notion is supported by Cook & Proctor (2007) who state that if stakeholders cannot understand the MDCA technique, and how it generated a result, the results are unlikely to be used. Hajkowicz & Higgins (2008) also support the contention that ease of understanding should be a primary concern in the selection of a MCDA technique. They compared different MCDA techniques for water resource management and found strong agreement between the techniques. They reported that there were only a few cases were different techniques generated markedly different results. Thus they state that the selection of MCDA technique is typically of lesser importance than the initial structuring of the decision problem, which includes selection of criteria, selection of decision options, weighting of criteria and obtaining performance measures against the criteria. The following was observed in a case where MCDA was scrutinised in a court of law in the Netherlands: "The main methodological challenge is not in the development of more sophisticated MC[D]A methods. Simple methods, such as weighted summation, perform well in most cases. More important is the support of problem definition and design" (Hajkowicz & Higgins 2008).

Therefore, the more advanced techniques, including the outranking methods, were avoided (see Spackman *et al.* 2000) in favour of simple additive methods, which are relatively easy to understand and implement and which can be modelled in a simple spreadsheet. The Analytical Hierarchy Process (AHP) was selected because it is an intuitive method that can be easily understood by stakeholders and has a record of providing robust and effective support to decision-makers working on a wide range of problems in a various circumstances (Spackman *et al.* 2000)

3. ANALYSIS OF CLEARING PRIORITY AREAS

3.1 Introduction

This chapter deals with the application of MCDA techniques to identify priority areas for invasive alien plant control on the conservancy.

3.1.1. Planning area

The planning area for the analysis corresponded with the boundary of the conservancy at the time of the study (see Figure 8). The extent of the planning area roughly corresponds with the area bounded by Oyster Bay in the west, Cape St Francis in the east, the northern extent of the headland bypass dune system in the north and the Indian Ocean in the south.

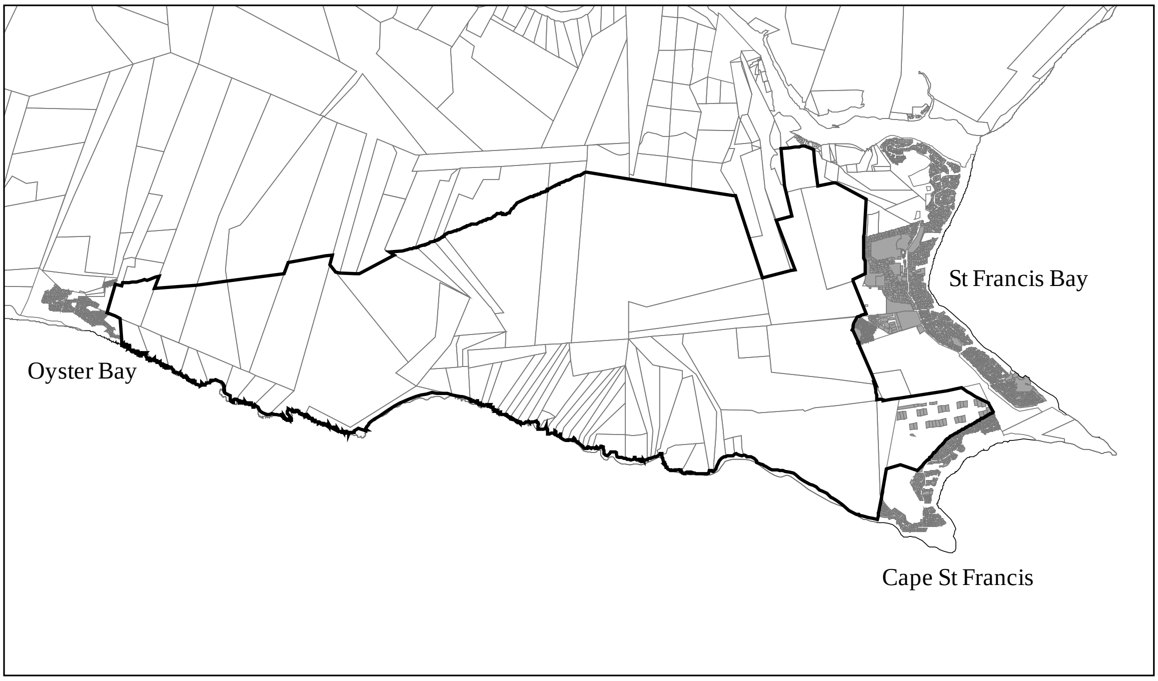


Figure 8: The planning area for the MCDA analysis. This corresponds to the boundary of the St Francis Conservancy.

3.1.2. *Planning units*

The planning area was divided in planning units, with each planning unit representing a decision alternative (a notional candidate area for alien control operations). The planning units were obtained by dividing the planning area into 1 ha grid cells and then intersecting the grid cells with property boundaries (see Figure 9), using the Geoprocessing Intersect function within the ESRI ArcView 3.2 GIS software package. The planning units thus had a maximum area of 1 ha while many were smaller than this due to the cleaving of grid cells by the property boundaries. The intersection enabled recommendations stemming from the analysis to be used on a conservancy-wide scale but also as a guide to landowners operating within the boundaries of their own properties. A total of 6808 planning units were defined.

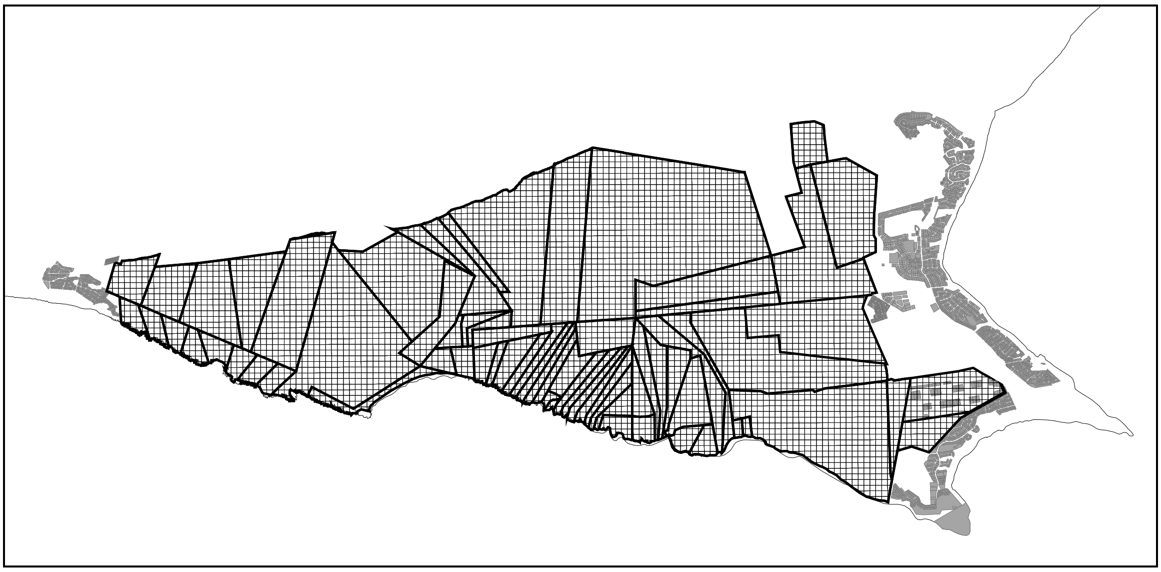


Figure 9: The planning units for the MCDA analysis.

3.2 Development of the Analytical Hierarchy Process model

An important preliminary step in the process was to define the desired goals, objectives, or purpose of the project (Strager 2004). The objectives for the analysis were extracted from the outcomes of a dedicated conservancy subcommittee meeting. The ultimate objective was phrased as follows: "to protect and restore the sensitive natural environment of the conservancy for the benefit of landowners and the community". Contributing to the ultimate objective, immediate objectives included: 1) to integrate biological, social and heritage concerns in the selection of priority areas for alien plant control, 2) to make the most efficient use of resources allocated towards alien plant control, and 3) to select priority control areas in an objective and unbiased manner.

The above was used as a guide in the selection of criteria. The AHP model was then developed by decomposing the criteria into relevant indicators (see Figure 10).

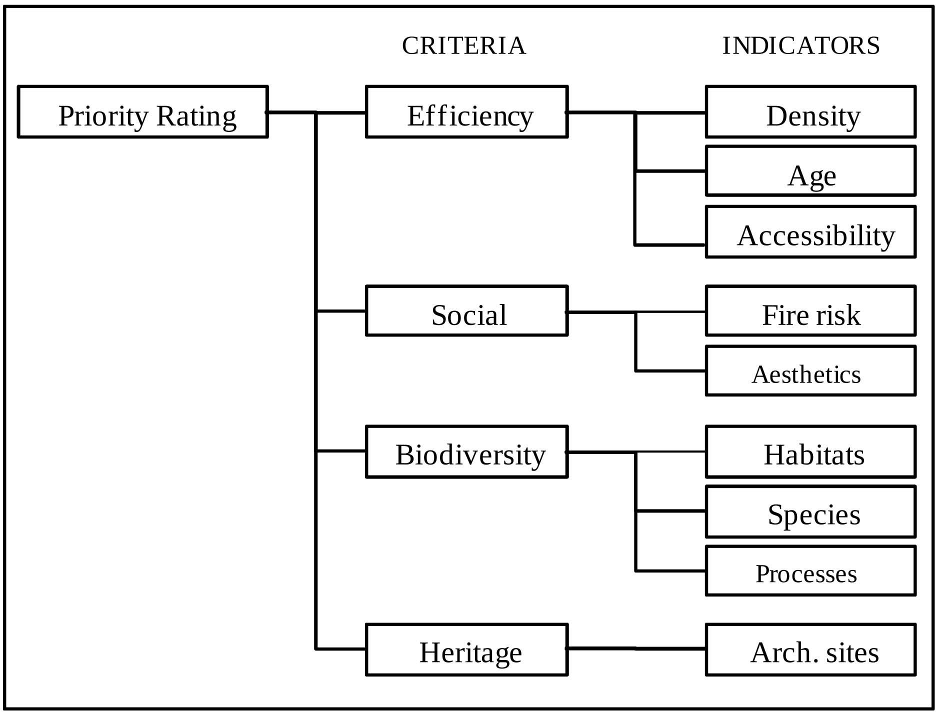


Figure 10: The AHP decision hierarchy developed for the analysis.

3.3 Criteria

Four criteria were identified for the evaluation of the decision alternatives (planning units): efficiency, social significance, biodiversity significance and heritage significance.

The efficiency criterion was included in order to prioritise areas where maximum cost efficiency could be achieved for control operations. Alien plant control is labour intensive and expensive and there are limited resources available to direct towards clearing operations. Resources should thus be used in the most efficient way. The social significance criterion was included in order to prioritise areas that are important to conservancy members. This includes areas where alien plants have the potential to impact on the safety of members or to damage their property (through increased risk of fire), or where alien plants impact upon the amenity value of the area. The biodiversity significance criterion was included to prioritise areas that are important from a biodiversity conservation perspective. This criterion is especially important, considering the conservation significance of the area and considering that strategies that prioritize sites of high biodiversity value mitigate the threat posed by alien plants to native plant diversity best (Higgins *et al.* 2000; Hobbs & Humphries 1995). The heritage significance criterion was included to consider sites with heritage value in the prioritisation of areas for control operations.

3.4 Indicators

The criteria themselves are not directly measurable. In order to determine the performance of the planning units against the criteria, the criteria were decomposed into indicators, which are the measurable elements of the AHP hierarchy (see Table 9):

Table 9: A description of the criteria and indicators used in the analysis.

Criterion	Indicator	Description
Efficiency	Density	The density of alien plants.
	Age	The age class of alien plants.
Social	Accessibility	The accessibility of areas.
	Fire risk	Areas that should be cleared to protect infrastructure from fire.
	Aesthetics	Areas that are highly visible and where alien plants can

		impact on scenic value.
Biodiversity	Ecosystem status	Areas containing habitats that are conservation priorities.
	Species of special concern	Areas containing species of special concern.
	Ecological processes	Areas that are important for the continued functioning of important ecological process.
Heritage	Archaeological sites	Areas containing important archaeological sites.

The rationale for the inclusion of each of the indicators is discussed below.

3.4.1. Alien plant density

Higgins *et al.* (2000) explored different strategies for clearing alien plants and found that strategies that prioritise clearing of high-density stands are expensive and time consuming compared to strategies that prioritise low-density stands. The rationale behind prioritising sparse stands over dense stands includes:

1. Dense stands are often impenetrable and difficult to work in;
2. Sparse stands have the potential to become dense stands and hence more costly to clear (Marais 2000);
3. The ratio of area cleared to effort required is greater in sparse stands. Thus sparse stands are less expensive to clear than dense stands and larger areas can be restored to a natural state for the same unit effort when working in sparse stands compared to dense stands (see Higgens *et al.* 2000; van Wilgen *et al.* 2000); and
4. The disturbance caused by clearing high density stands often creates opportunities for the subsequent re-establishment of alien plants (Higgens *et al.* 2000).

The most rapid and cost-effective strategy is therefore to begin operations by clearing low-density stands, leaving high-density stands of adult plants until last (Higgens *et al.* 2000).

Therefore, in constructing the model, planning units with predominantly sparse stands should therefore be preferred over those with predominantly dense stands.

3.4.2. Age of alien plants

It is less expensive and easier to clear juvenile plants before clearing adult plants (see Table 10). Stem diameter of rooikrans is a function of age (Higgens *et al.* 2000). Seedlings and young juvenile plants (up to approximately 6 months old) can be hand-pulled from the ground while older juvenile plants (up to approximately 3 years old) have small stem diameters which are relatively easy to cut through with non-mechanised cutting equipment (see Figure 11). In contrast, old plants have large stem diameters that require substantial effort to fell.

Table 10: A comparison of the cost (in South African Rands and using figures from 1997) of clearing different density classes of alien plant species in fynbos for mature and juvenile stands (adapted from Versfeld et al.1997).

Age class	**Sparse (< 25%)**	**Moderate (25 - 75%)**	**Dense (> 75%)**
Mature	686	1295	4487
Juvenile	525	945	1421

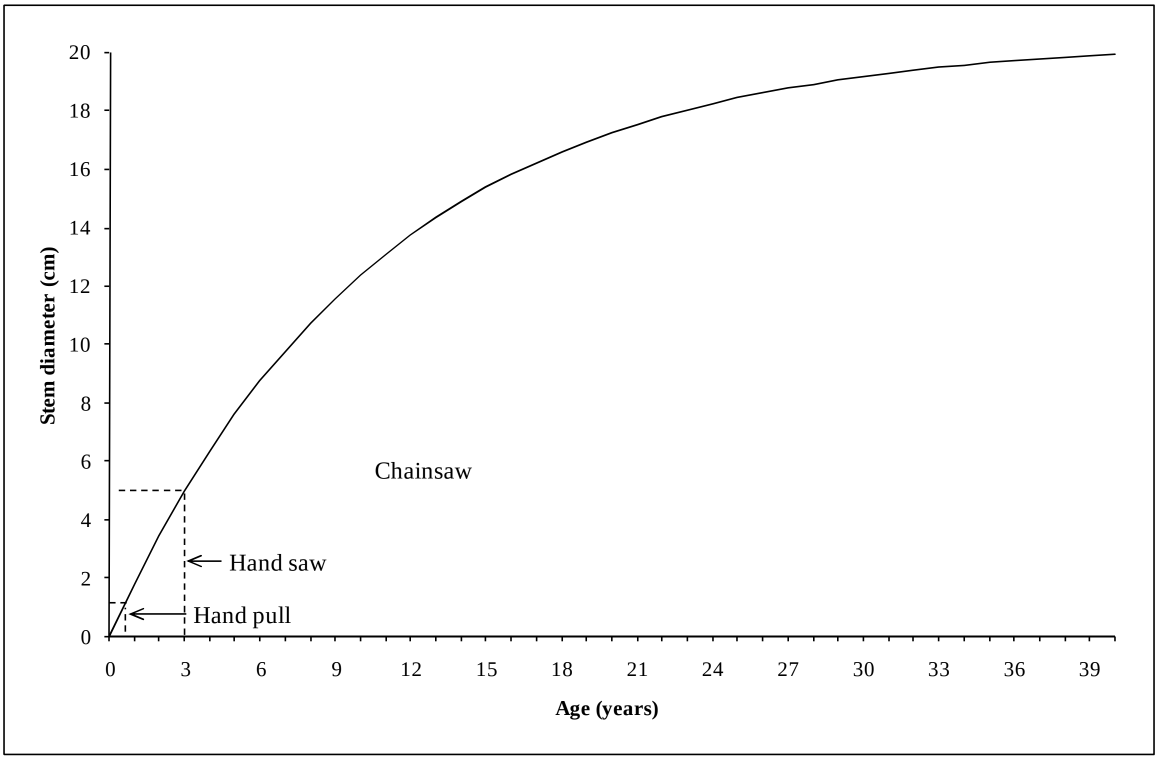


Figure 11: Growth of A. cyclops, measured as a function of stem diameter, and mechanical clearing method for the different age classes (Higgens *et al.* 2000).

Therefore, in constructing the model, planning units with young plants should be preferred for clearing over planning units with older plants. A problem here is that age classes were mapped in 2005, and provided that no new fires have occurred within the study area, all stands would have aged by the time of completion of this study. Nonetheless, assuming no new fires, the distribution of age classes between areas should remain unchanged.

3.4.3. Access to clearing sites

Easily accessed sites are more cost-effective to clear than remote sites. In easily accessed sites, clearing teams can spend less time travelling and transporting equipment, and thus can devote more time each day to clearing compared to less accessible sites.

The model should therefore be constructed to prioritise planning units that are easily accessible over planning units that are more remote.

3.4.4. *Fire risk*

Rooikrans originates from a fire prone environment and is adapted to fire and highly flammable. Dense stands of rooikrans create elevated fuel loads and promote more intense fires than those that occur in un-invaded fynbos (see Table 11) (Chapman & Forsyth 2000).

Table 11: Fuel loads and fire intensity for fires in fynbos compared to stands of exotic species (Chapman & Forsyth 2000).

	Fuel loads (gm^{-2})	**Fire intensity (kWm^{-1})**
Fynbos	1 000 - 3 000 (max 7 000)	20 000 - 30 000
Acacia cyclops	9 000	20 000 - 60 000
Pinus sp.	18 000 - 40 000	no data
Eucalyptus sp.	42 000	60 000 (max 100 000)

The presence of rooikrans thus exacerbates the danger of wild fires and constitutes a threat to houses and other infrastructure within the conservancy. This threat is aggravated by the fact that most of the landowners are not resident. Wilson & Ferguson (1986) have shown that unattended houses are more likely to be destroyed by fire than those that were actively defended. Small fires may easily be extinguished by residents, but in unattended houses, those small fires may eventually destroy the house.

In order to minimize the threat of wildfires on buildings, the model should be constructed to prioritise planning units around buildings in order to reduce fuel loads in these areas (see Chapman & Forsyth 2000, Anon. 2006).

3.4.5. *Aesthetics*

For the majority of the landowners, the study area is important for its amenity and recreation values. Natural character, including landscape image and identity and natural variety are the important in this regard (Krause 2001). The amenity value of a landscape is often expressed by the general presence of nature. This quality often has more to do with feeling and knowledge than visual perception alone. Many of the elements that result in lack of ecological integrity are also often perceived as a visual disturbance (Fry *et al.* 2008).

Fry *et al.* (2008) report on numerous aspects that contribute to the visual character of an area. Relevant aspects include:

1. Complexity – diversity, richness of landscape elements and features, interspersion of pattern;
2. Naturalness – closeness to a preconceived natural state;
3. Stewardship – sense of order and care, perceived accordance to an ''ideal'' situation reflecting human care through active and careful management; and
4. Lack of disturbance – contextual fit and coherence, lack of constructions and interventions.

The presence of alien invasive plants impacts negatively on all of the above aspects. Rooikrans forms dense monospecific stands which are monotonous and detract from the area's scenic quality. In addition, rooikrans grows taller than the native vegetation and dense stands can result in views being obscured. Thus, in order to maintain and improve the aesthetic character of the study area, the model should be constructed to prefer highly visible planning units for alien plant clearing operations than those that are less visible.

3.4.6. Archaeological sites

Alien invasive plants pose a severe threat to the conservation of archaeological sites (Grapow & Blasi 2004). Binneman (2001) states that alien invasive plants are one of the two main factors threatening the archaeological sites of the study area. Encroaching alien plants obscure sites and prevent access to them, hindering their future study. They also have the potential to cause direct (root systems may disturb the integrity of the sites) and indirect damage (through the intense fires associated alien invasive plants) to the sites. The model should therefore be constructed to prioritise planning units that contain archaeological sites over those that do not.

3.4.7. Habitat value

The conservation significance of the habitats within the planning area should be taken into account in the development of the model. One means of doing so is to consider the Ecosystem Status categories of vegetation types.

The Ecosystem Status categories, as defined in the conservation assessment for the Subtropical Thicket Biome, were based on the conservation status of vegetation types as determined by the area of each vegetation type required to achieve its biodiversity-based target and the remaining area of its extant habitat. Vegetation types that can no longer meet their targets because of habitat transformation were classified as *Critically Endangered*. Vegetation types that have much more extant habitat than what is required for target achievement were classified as *Currently Not Vulnerable* (now *Least Threatened*). The *Endangered* and *Vulnerable* categories fell between these two extremes (Cowling *et al.* 2003; see also Pierce & Mader 2006).

Four Ecosystem Status categories were defined in the conservation assessment for the Subtropical Thicket Biome:

- *Critically Endangered* – the original extents of these ecosystems have been reduced to the extent that they are under threat of collapse or disappearance. They can withstand no further loss of natural area;
- *Endangered* – the original extents of these ecosystems have been severely reduced to the extent that their functioning and existence of are under serious threat.
- *Vulnerable:* Much of the original extents of these ecosystems remain. Further disturbance or destruction could harm their health or functioning, and they can withstand only limited loss of natural area.
- *Least Threatened:* Most of the original extents of these ecosystems remain. They are mostly undamaged, healthy and functioning and can withstand some loss of natural area.

Because there are fewer opportunities to effectively conserve vegetation types within the higher Ecosystem Status categories, the model should be constructed to prioritise planning units containing higher Ecosystem Status vegetation types over planning units with vegetation types of lower Ecosystem Status.

Areas where the natural habitat of the conservancy area has been severely degraded or transformed, should be considered to be of lower conservation value. Certain types of habitat transformation are restorable (e.g. light to moderate grazing) while others may not have good prospects for restoration (especially transformation types that disturb soil structure e.g. ploughing and heavy grazing). The model should thus be constructed to reduce the priority of planning units that have been severely degraded or transformed.

3.4.8. Species of special concern

In addition to the conservation value of habitats within the study area, the conservation value of certain species, especially those that are threatened or narrowly distributed, warrant consideration. Although it is difficult to obtain presence / absence data for species, the model should be constructed to prioritise planning units that are known to contain species of special concern. This prioritisation should reflect the status of the species in question.

3.4.9. Ecological processes

Ecological processes are the processes between biota, and those between biota and their physical environments, that generate and maintain biodiversity.

Although a multitude of important ecological processes are likely to occur within the study area, only those whose spatial components are capable of being mapped can be included in the model.

Wetland areas play an important role in terms of the hydrology of the study area (in terms of flood mitigation, water quality) but also provide important habitat for keystone pollinators and avifauna.

The headland bypass dune systems are important in terms of sand movement and for the unique yet ephemeral habitats they provide to biota. Associated with the sand movement corridors are a series of wetlands which form when the underlying calcrete hardpan is exposed during periods of high rainfall (see La Cock & Burkinshaw 1996). At least one Red Data Book species, *Satyrium hallackii*, is dependent on this process.

The sand movement corridors are relatively free of invasive alien plants, possibly because the dynamic nature of the systems does not allow for their recruitment and establishment. However, alien plants are able to gain a foothold in the more stable environments on the periphery of these corridors and, in time as the fringe areas become stabilised, they have the potential to expand incrementally deeper into the corridors.

In order to ensure the persistence of critical ecological process, the model should be constructed to prioritise planning units that occur within the spatial components of these ecological processes.

3.5 Data collection and scoring of indicators

Spatial data were collected for each of the indicators and mapped as layers on a GIS. Indicator maps were then generated by using the data represented in these layers to derive preference information for each planning unit for each indicator. A standardised Likert-type scale was devised to rate the planning units against the indicators. The scale ranged from zero to four, where zero represented the least preferred alternative and four represented the most preferred alternative.

Indicators were scored using rule-based models of preference for the planning units. Rule-based models are function-free and they are usually expressed in symbolic forms, such as “if-then” decision rules. The main advantage of rule-based models of preference is their natural and easy interpretation. (Greco *et al.* 2001).

3.5.1. Alien plant density

Data on alien plant density and distribution for the planning area was obtained from a previous study (Kruger & Cowling 1996). These data were refined through a ground-truthing exercise and updated to reflect changes in alien plant density and distribution that had occurred over time (see Figure 12). Alien plants were assigned to three density categories, as follows:

Dense = 75 – 100 % cover; plants spaced < 0.1 canopy diameters apart;

Moderate = 25 – 75 % cover; plants spaced at 0.1 to 2 canopy diameters apart; and

Sparse = 0 – 25 % cover; plants spaced > 2 canopy diameters apart.

3606 ha (almost two thirds) of the planning area (and almost the entire vegetated extent) is invaded to some degree. There is a marked occurrence of dense stands on the fringes of the mobile dunefields, with high density stands occurring in swathes than run parallel to the fossil dune ridges. The cores of the dunefields are relatively un-invaded. Rooikrans is the predominant invader. Port Jackson is rarely found on the coastal strip, but occurs on sites further inland of the coast (Kruger 1996) on older, more acid dunes (Cowling 1997).

Table 12: Extent of alien plant invasion within the planning area

Category	Area (ha)	Percent
Dense	1462	25.3
Moderate	835	14.5
Sparse	1309	22.7
Not invaded	2163	37.5
Total	5769	

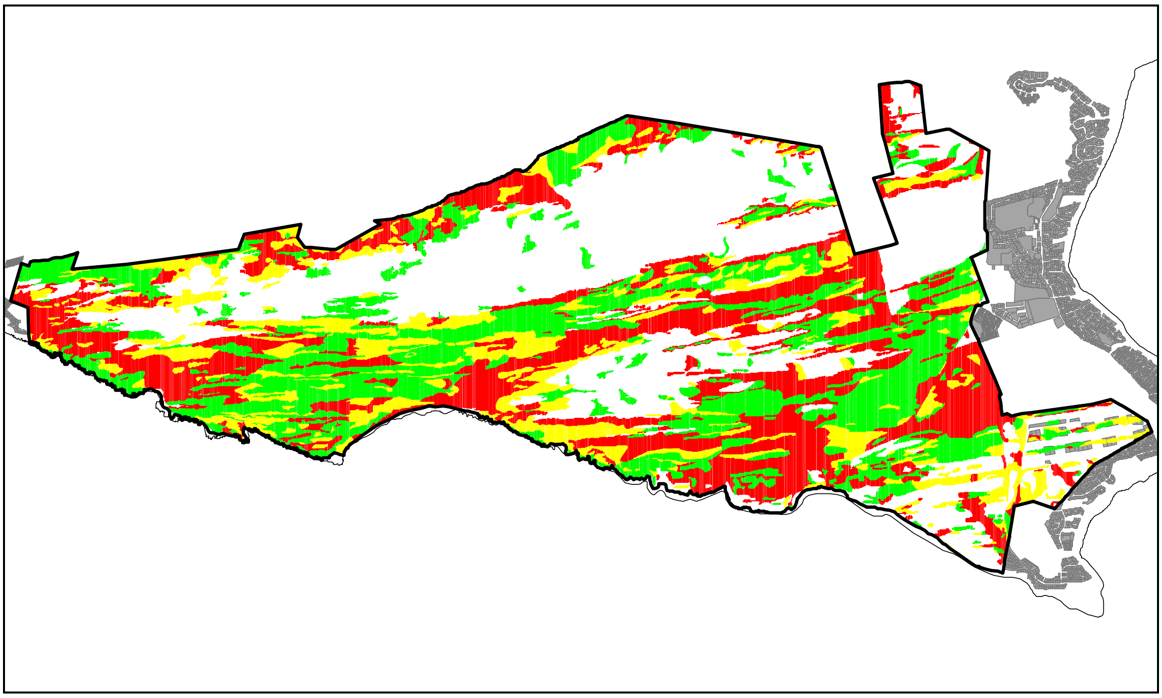


Figure 12: The density and distribution of alien plants within the planning area (red – dense, yellow – moderate and green – sparse).

In order to obtain the alien plant density and distribution for each planning unit, the Geoprocessing: Intersect function within the ESRI ArcView 3.2 GIS software package was used to intersect the planning units with the data on alien plant density and distribution. The XTools extension for ArcView 3.2 was then used to calculate the invaded area of the each of the planning units for the respective density categories. The values returned for the density categories for each planning unit were converted to a single value, the condensed area (see Marais *et al.* 2004), to allow for comparison between planning units. The condensed area is the equivalent area that the alien plants would occupy if they were condensed to provide a completely closed canopy cover.

Since the density classes are broad categories for a continuous range of density values (e.g. dense = 75 – 100% cover) the mean value of the range of each class was used to calculate condensed area (87.5% for dense, 50% for moderate and 12.5% for sparse). Condensed area was calculated according to the following formula:

$$C(a_t) = \sum_{x=D,M,S} i_x d_x$$

Where:

$C(a_t)$:	Condensed area of alternative a_t
i:	invaded extent for the density class
d:	mean value of the range of the density class
D:	Dense density class
M:	Moderate density class
S:	Sparse density class

For example, if a planning unit contained 0.3 ha of dense stands, 0.5 ha of moderate stands and 0.1 ha of sparse stands the condensed area would be:

Condensed area (ha) = (0.3 ha x 87.5%) + (0.5 ha x 50%) + (0.1 ha x 12.5%)

= 0.26 ha + 0.25 ha + 0.01 ha

= 0.52 ha

Since planning units are not all the same size invaded area of each planning was expressed as a percentage of the total area of the planning unit. Due to the use of mean values for range of the density classes in the above equation, the maximum possible result is 87.5% of the area of the planning unit.

By obtaining condensed area, a single density value could be determined for each planning unit and this allowed for planning units to be compared with each other in terms of their alien plant density. The following rule-based preference model was used to assign the value, $v_{density}(a_t)$, of the alternatives with respect to alien plant density:

If: $70.0\% \leq C(a_t) \leq 87.5\%$; then $v_{density}(a_t) = 0$

If: $52.5\% \leq C(a_t) < 70.0\%$; then $v_{density}(a_t) = 1$

If: $35.0\% \leq C(a_t) < 52.5\%$; then $v_{density}(a_t) = 2$

If: $17.5\% \leq C(a_t) < 35.0\%$; then $v_{density}(a_t) = 3$

If: $0.0\% \leq C(a_t) < 17.5\%$; then $v_{density}(a_t) = 4$

Planning units returning high values for the above preference model have low condensed areas, implying either an absence of dense stands or relatively low proportions of dense stands. Conversely, planning units returning low values for the value function have high condensed areas, implying significant proportions of dense stands.

3.5.2. Age of alien plants

Broad age groups were mapped for the planning area based on a combination field visits by the author and data collected on fire history by the Fourcade Botanical Group, a natural interest group from nearby St Francis Bay. Because most recruitment occurs after fire in fynbos, most stands of plants, including alien plants, tend to be even aged in fynbos. For this reason it is reasonable to ignore within-stand differences in age (Higgins *et al* 2000).

Three broad age classes were considered; juvenile, young adults and old adults. The collected data were mapped in ESRI ArcView 3.2. Thereafter, the resultant map was intersected with the planning units using the Geoprocessing: Intersect function to obtain a map of the age class distribution of each planning units. The area of the polygons within each planning unit was calculated using the XTools extension for ArcView 3.2 and planning units were then assigned to values, $v_{age}(a_t)$, based on the predominant age class that occurred within them according to the following rule-based preference model:

If $area_{old\ adult} > area_{young\ adult} + area_{juvenile}$; then $v_{age}(a_t) = 0$

If $area_{young\ adult} > area_{old\ adult} + area_{juvenile}$; then $v_{age}(a_t) = 2$

If $area_{juvenile} > area_{old\ adult} + area_{young\ adult}$; then $v_{age}(a_t) = 4$

In this way, planning units that contained mostly juvenile plants were prioritised over planning units that contained mostly young adult plants, which in turn were prioritised over planning units that contained mostly old adult plants.

3.5.3. Access to clearing sites

In order to determine accessibility, the planning area's road network was mapped off orthorectified aerial photographs. Five concentric buffers of 100 m intervals were drawn around the roads using the Create Buffers feature in ArcView 3.2 (see Figure 13).

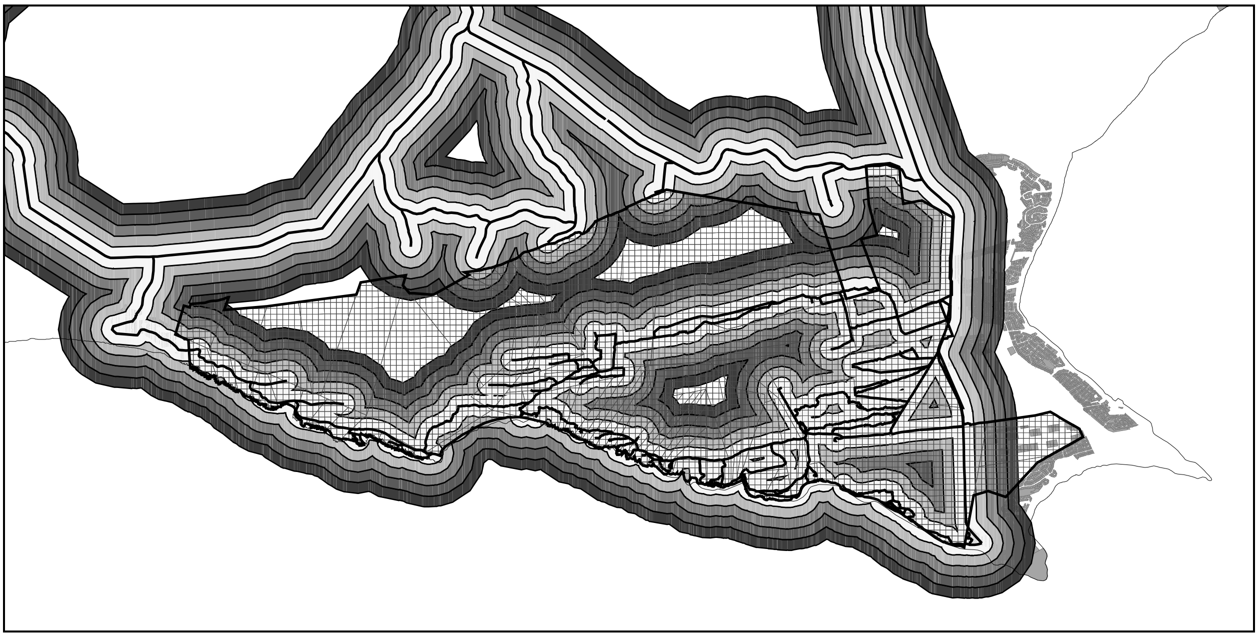


Figure 13: The planning area's road network was mapped and buffers with 100 m were drawn around the road network.

Planning units were assigned values, $v_{access}(a_t)$, according to the buffer in which they fell based on the following rule-based preference model:

If a_t is > 500 m from a road; then $v_{access}(a_t) = 0$

If a_t is > 400 and ≤ 500 m a road; then $v_{access}(a_t) = 1$

If a_t is > 300 and ≤ 400 m a road; then $v_{access}(a_t) = 2$

If a_t is > 100 and ≤ 200 m a road; then $v_{access}(a_t) = 3$

If a_t is ≤ 100 m from a road; then $v_{access}(a_t) = 4$

Planning units that were completely enclosed by dense stands of invasive plants were assigned a value of zero for this indicator because of the impenetrable nature of such stands.

Planning units that returned higher values for this preference model were considered to be a higher priority for clearing in terms of this indicator than planning units that returned lower values because they are more accessible.

3.5.4. *Fire risk*

Houses and other infrastructure (such as bird hides) were mapped off orthorectified aerial photography or from GPS coordinates captured in the field. Three concentric buffers with an interval of 100 m were generated around each structure using the Create Buffers function in ArcView 3.2 (see Figure 14).

Figure 14: Buffers were created at 100m intervals around buildings and other infrastructure.

Planning units were assigned to values, $v_{fire}(a_t)$, in terms of this indicator according to the following rule-based value function:

If a_t is > 200 m from infrastructure; then $v_{fire}(a_t) = 0$

If a_t is > 100 and ≤ 200 m from infrastructure; then $v_{fire}(a_t) = 2$

If a_t is ≤ 100 m from infrastructure; then $v_{fire}(a_t) = 4$

Planning units that returned higher values for the above preference model are closer to infrastructure and therefore considered a greater priority for clearing than planning units with lower values.

3.5.5. *Aesthetics*

A Triangulated Irregular Network (TIN) model was created in the 3D Analyst extension for ESRI ArcMap 9.2 GIS. A TIN model is an efficient way for representing continuous three-dimensional surfaces as a series of linked triangles. TINs can be generated from point, polygon and line datasets that contain x (latitude), y (longitude) and z (elevation) values.

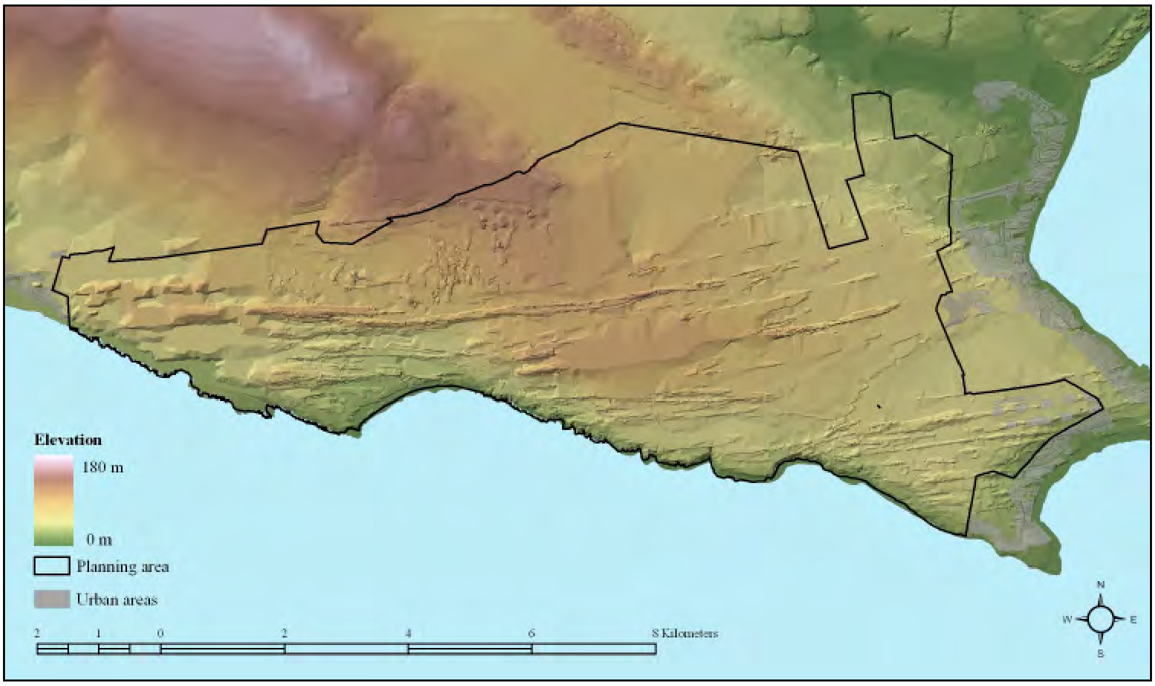


Figure 15: The TIN, generated from 5m and 20m contours, for the planning area.

For the eastern and central sections of the study area, the TIN was generated from 5m contour data. However, five meter contours were not available for the western sector of the study area, and 20m meter contours were used (see Figure 15). A viewshed analysis was then conducted on the TIN model (Figure 16). Viewshed analyses are used to identify areas that can be seen from one or more observation points or lines. Each cell in the output raster receives a value that indicates the number of observation points it from which it can be seen. Cells that cannot be seen from the observer points are given a value of zero. The landowners' houses were used as observation points in order to provide an indication of the visibility of areas within the study area from the places where the residents spend most of their time. Areas returning high scores in the viewshed analysis are highly visible from the landowners' houses while areas returning low scores area less visible.

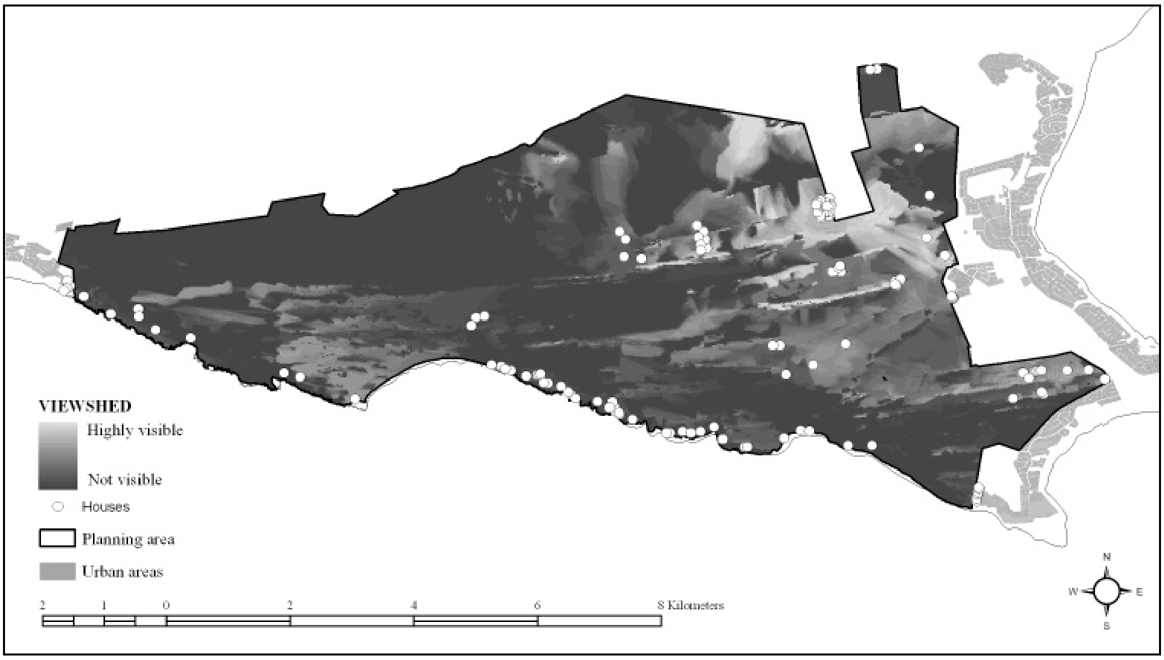


Figure 16: Viewshed analysis of the study area using 5m TIN and the landowners' houses as observation points.

Values, $v_{aesthetics}(a_t)$, were then assigned to the planning units based on the visibility scores returned by the viewshed analysis according to the following rule-based preference model (categorised by quantile intervals):

If: $15 <$ *viewshed score* $(a_t) \leq 55$; then $v_{aesthetics}(a_t) = 0$

If: $6 <$ *viewshed score* $(a_t) \leq 15$; then $v_{aesthetics}(a_t) = 1$

If: $2 <$ *viewshed score* $(a_t) \leq 6$; then $v_{aesthetics}(a_t) = 2$

If: $0 <$ *viewshed score* $(a_t) \leq 2$; then $v_{aesthetics}(a_t) = 3$

If: *viewshed score*$(a_t) = 0$; then $v_{aesthetics}(a_t) = 4$

Planning units that returned high values for the above preference model are in areas that are highly visible, and were a priority for clearing operations in terms of this indicator.

3.5.6. Archaeological sites

Broad areas housing known archaeological sites were mapped from Binneman (2001). One hundred meter concentric buffers were drawn around these areas using the Create Buffers function in ESRI ArcView GIS 3.2 (see Figure 17). The buffers were then intersected with the planning units using the ArcView's Geoprocessing Intersect function.

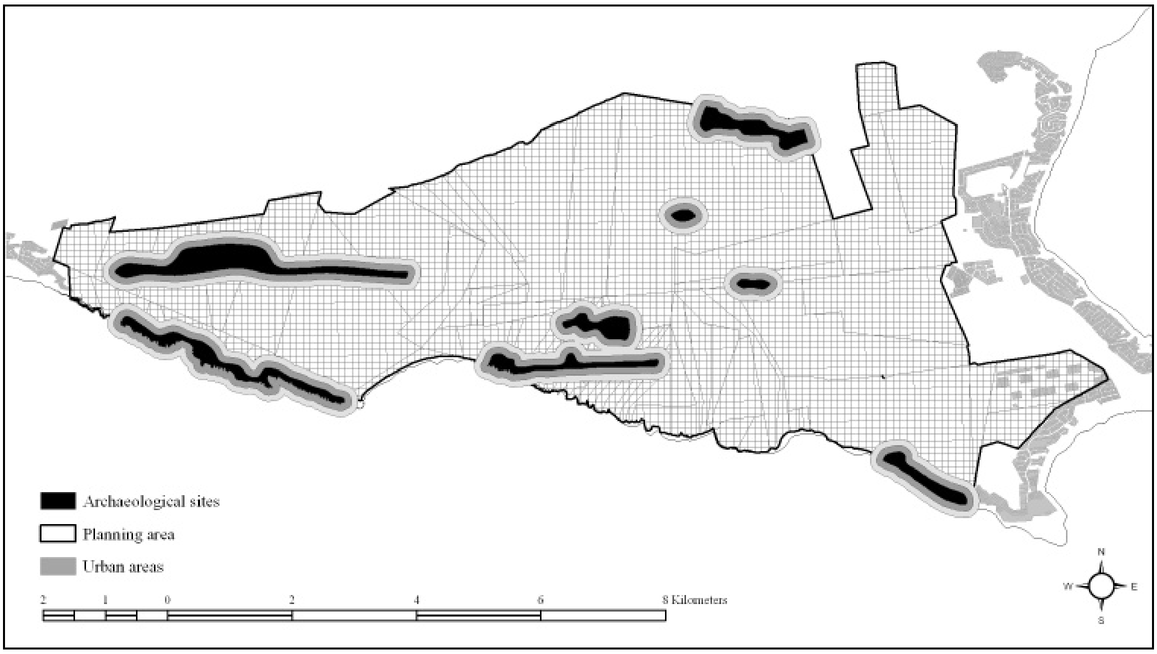


Figure 17: 100m buffers were drawn around the known archaeological sites.

Planning units were then assigned values, $v_{arch}(a_t)$, according to the following rule-based preference model:

If a_t is > 200 m from an archaeological site; then $v_{arch}(a_t) = 0$

If a_t is > 100 and ≤ 200 m from an archaeological site; then $v_{arch}(a_t) = 2$

If a_t is ≤ 100 m from an archaeological site then $v_{arch}(a_t) = 4$

Planning units that returned high scores for the above preference model are near to archaeological sites and should be prioritised in terms of this indicator.

3.5.7. Habitat value

The conservation assessment for the Subtropical Thicket Biome assigned Ecosystem Status categories for the vegetation types (mapped at a 1:100 000 scale) within its planning domain (see Figure 18). The vegetation of the study area, however, was mapped at a finer scale (1:10 000) by Kruger & Cowling (1996).

Ecosystem Status categories for the finer-scale vegetation types were therefore inferred by using the Ecosystem Status of vegetation types from conservation assessment for the Subtropical Thicket Biome as a guide (see Table 13 and Figure 19).

Table 13: Analogies between the conservation assessment for the Subtropical Thicket Biome and the vegetation mapped by Kruger & Cowling (1996).

Subtropical Thicket Biome conservation assessment	**Kruger & Cowling *et al.* (1996)**
St Francis Dune Thicket (*Endangered*)	South Coast Dune Fynbos Rocky Coast Community
South Coastal Vegetation (*Least Threatened*)	Drift Sands Coastal Pioneer Dune Community Sand River Pioneer Dune Community
Humansdorp Grassy Fynbos (*Least Threatened*)	Misc. Grassy Fynbos
Kromme Fynbos / Renosterveld Mosaic (*Vulnerable*)	Renosterveld Transistion

Due its relatively coarse nature, the dune forests and wetlands mapped by Kruger & Cowling (1996) do not appear on the map of the vegetation types for the Subtropical Thicket Biome. These vegetation types were assigned to the *Critically Endangered* category due to their protection status within national legislation. The vegetation type classified as Algoa Dune Thicket (Vulnerable) was not distinguished in the vegetation map of Kruger & Cowling (1996), but was included in the South Coast Dune Fynbos fynbos / thicket mosaic vegetation type.

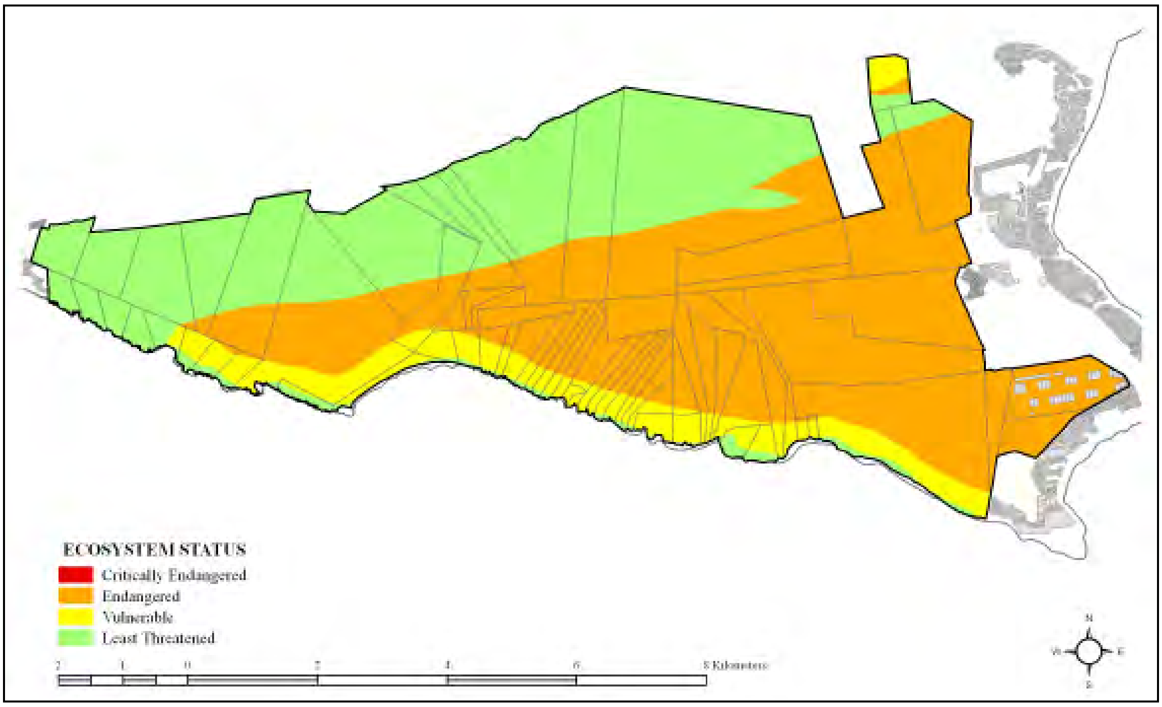


Figure 18: The STEP Ecosystem Status categories for vegetation within the planning area.

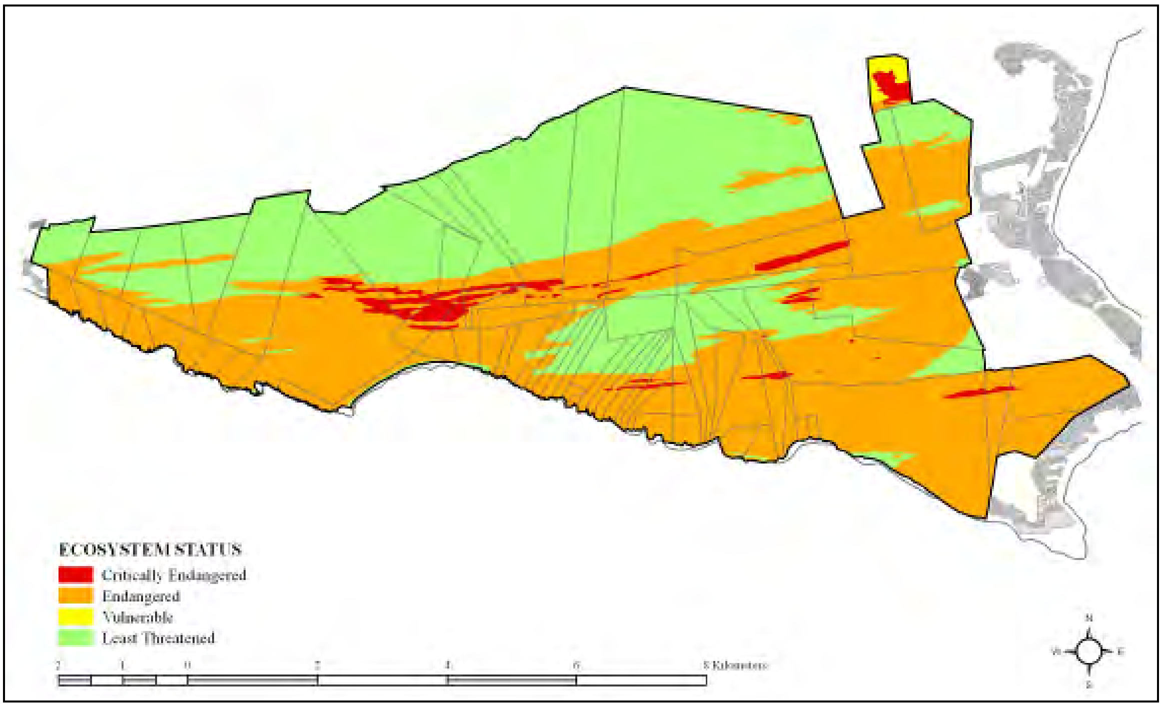


Figure 19: Inferred Ecosystem Status of the vegetation within the planning area.

Planning units were assigned values, $v_{EcoStatus}(a_t)$, based on the following rule-based preference model:

If: a_t is predominantly Critically Endangered; then $v_{EcoStatus}(a_t) = 4$

If: a_t is predominantly Endangered; then $v_{EcoStatus}(a_t) = 3$

If: a_t is predominantly Vulnerable; then $v_{EcoStatus}(a_t) = 1$

If: a_t is predominantly Least Threatened; then $v_{EcoStatus}(a_t) = 0$

The results of the above preference model were adjusted according to the transformation status of the vegetation types. Transformed and degraded areas were mapped from aerial photography and from field inspection. The following rule-based preference model was used to adjust the results and to obtain an overall indication of habitat value, $v_{Habitat}(a_t)$:

If: a_t is predominantly untransformed; then $v_{Habitat}(a_t) = v_{EcoStatus}(a_t)$

If: a_t is predominantly degraded; then $v_{Habitat}(a_t) = v_{EcoStatus}(a_t) -1$

If: a_t is predominantly transformed; then $v_{Habitat}(a_t) = 0$

Planning units that returned high values for the above preference model contain untransformed vegetation types of high Ecosystem Status.

3.5.8. Species of special concern

Data on rare and endemic plant species were collected between 2003 and 2005 by the Fourcade Botanical Group as part of the Custodians of Rare and Endangered Wildlife (CREW) Programme of the South African National Biodiversity Institute (SANBI). Twenty nine sites within the planning area were visited by the Fourcade Botanical Group. The list of special species included in the study was determined by the group, experts from SANBI and local botanists (including Prof. Richard Cowling and Mrs Caryl Logie).

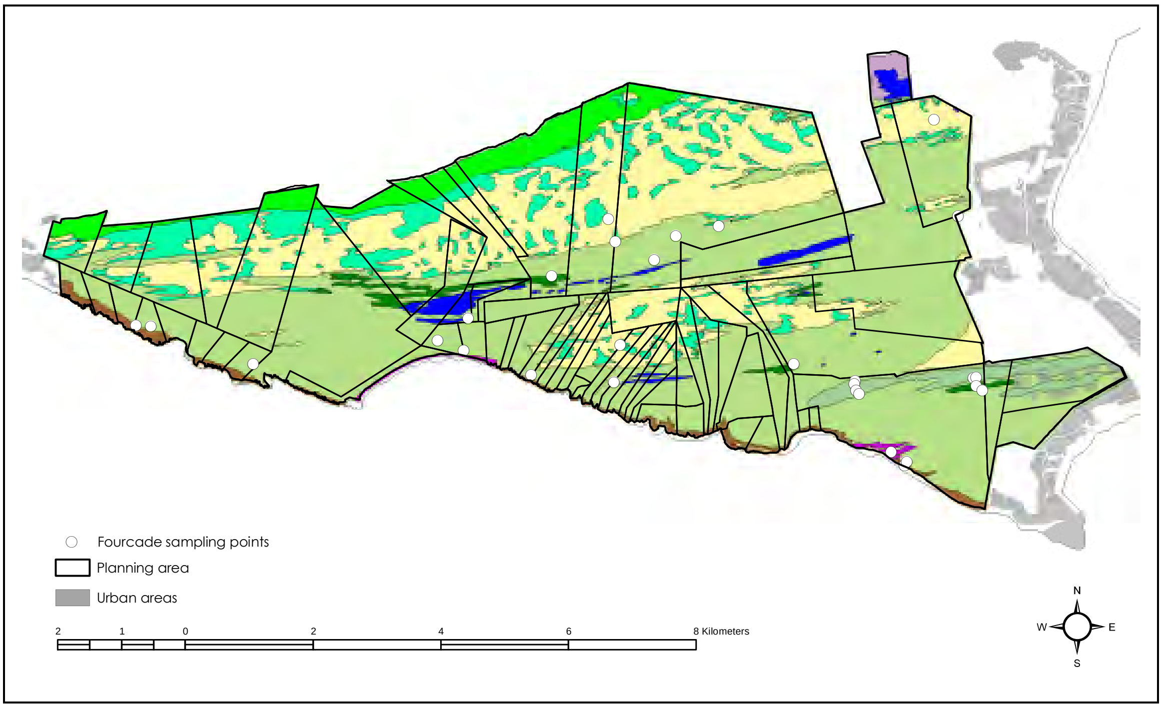


Figure 20: Survey sites for species of special concern.

The sampling sites were mapped and one hundred meter buffers were drawn around the point location for each site. These buffers were intersected with the planning unit and the number of specials occurring in each planning unit. Planning units were then assigned values, $v_{SSC}(a_t)$, based on the highest status species recorded for each site, according to the following rule-based preference model:

If: a_t contains Critically Endangered or Endangered species;	then $v_{SSC}(a_t) = 4$
If: a_t contains Vulnerable species;	then $v_{SSC}(a_t) = 3$
If: a_t contains other special of special concern;	then $v_{SSC}(a_t) = 2$
If: a_t contains no species of special concern;	then $v_{SSC}(a_t) = 0$

Planning units that returned high values for the above preference model contain species of special concern, or are in close proximity to these species, and were prioritised within the model.

3.5.9. Ecological processes

The periphery and source of the mobile dunefields (at the western extent of the dunefields where marine sand is deposited) were mapped and these polygons were intersected with the planning units using the Geoprocessing Intersect function in ESRI ArcView GIS 3.2. Planning units were then assigned values, $v_{Process}(a_t)$, according to the following rule-based preference model:

If: a_t contains source areas; then $v_{Process}(a_t) = 4$

If: a_t contains peripheral areas; then $v_{Process}(at) = 2$

If: a_t does not contain either of the above; then $v_{Process}(a_t) = 0$

Planning units returning values of greater than zero for the above preference model were considered to be important for the continued function of the mobile dunefields and were prioritised in terms of the model.

3.5.10. Indicator preference maps

The values returned for each of the preference models for the indicators were mapped to provide a spatial indication of priority areas with respect to the indicators (see Figure 21 to Figure 29).

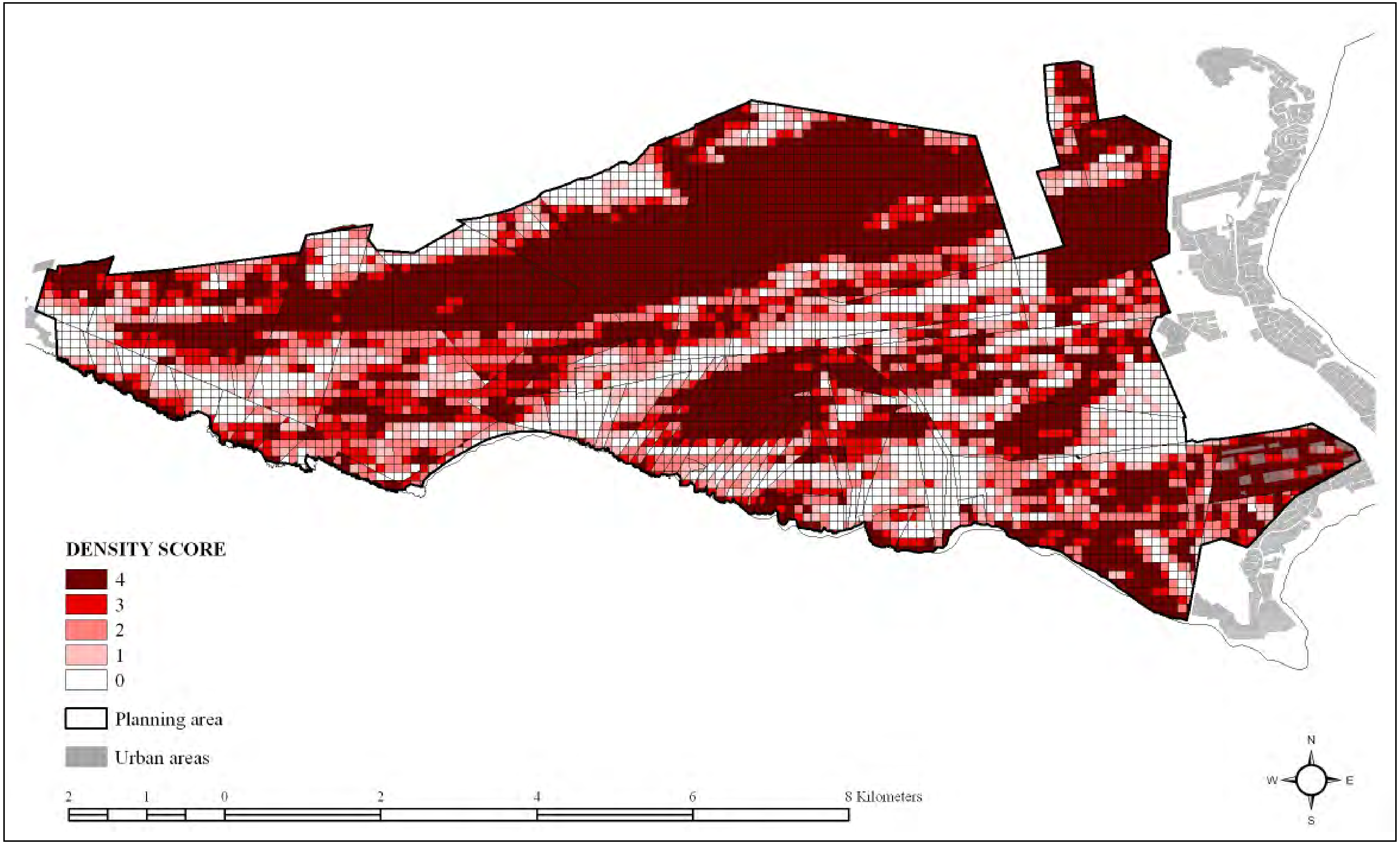


Figure 21: Density indicator

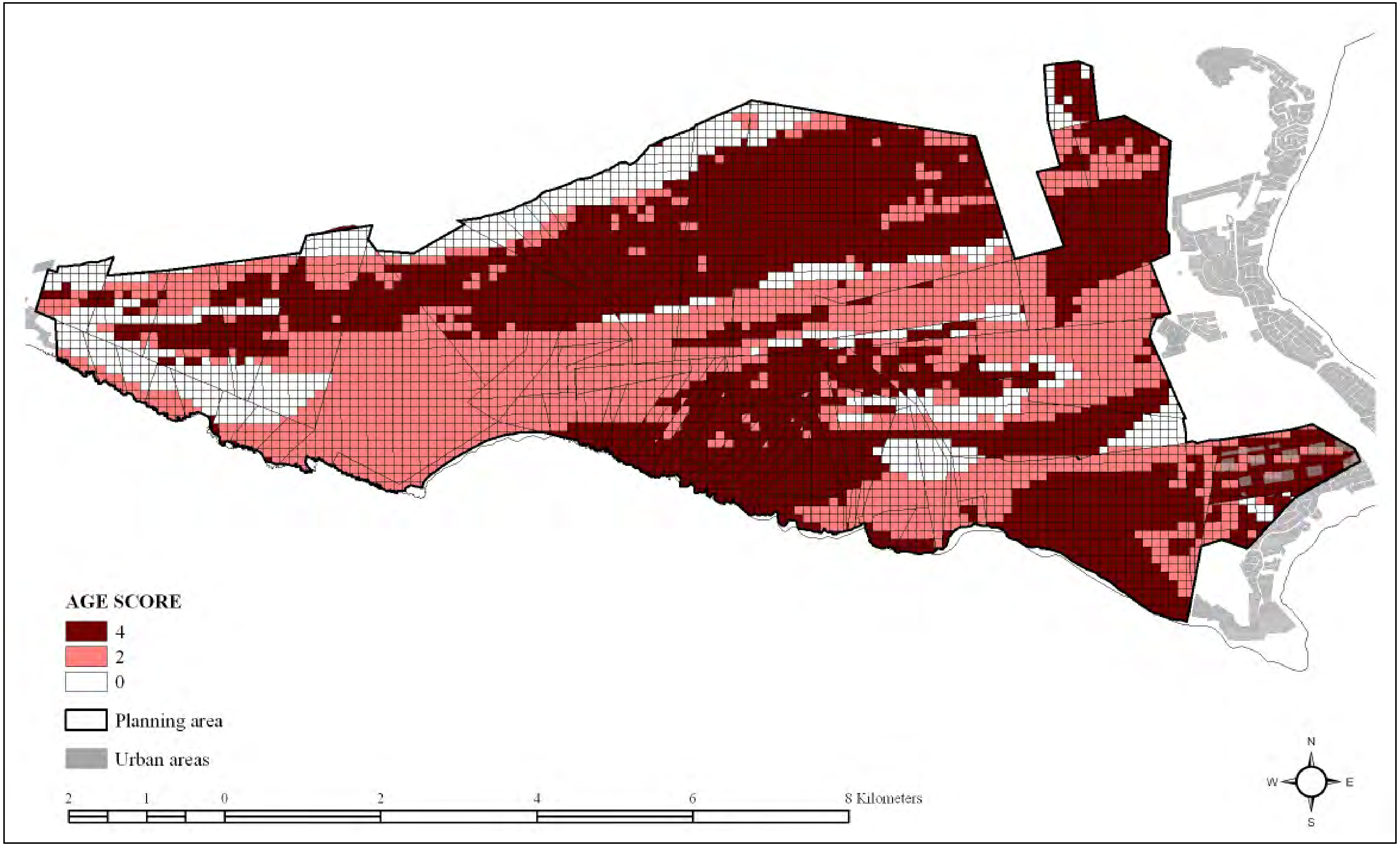


Figure 22: Age indicator

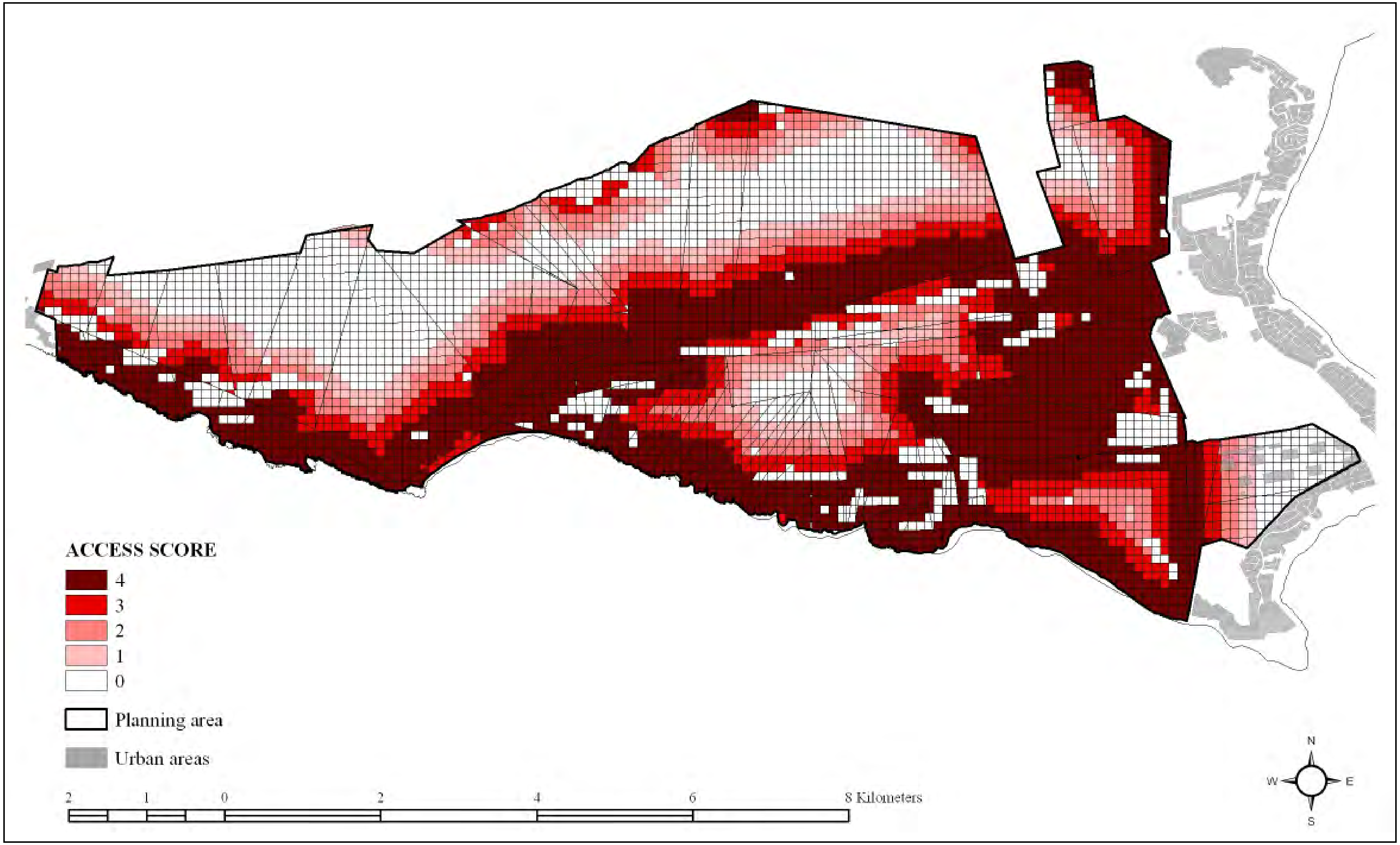


Figure 23: Access to clearing sites indicator score

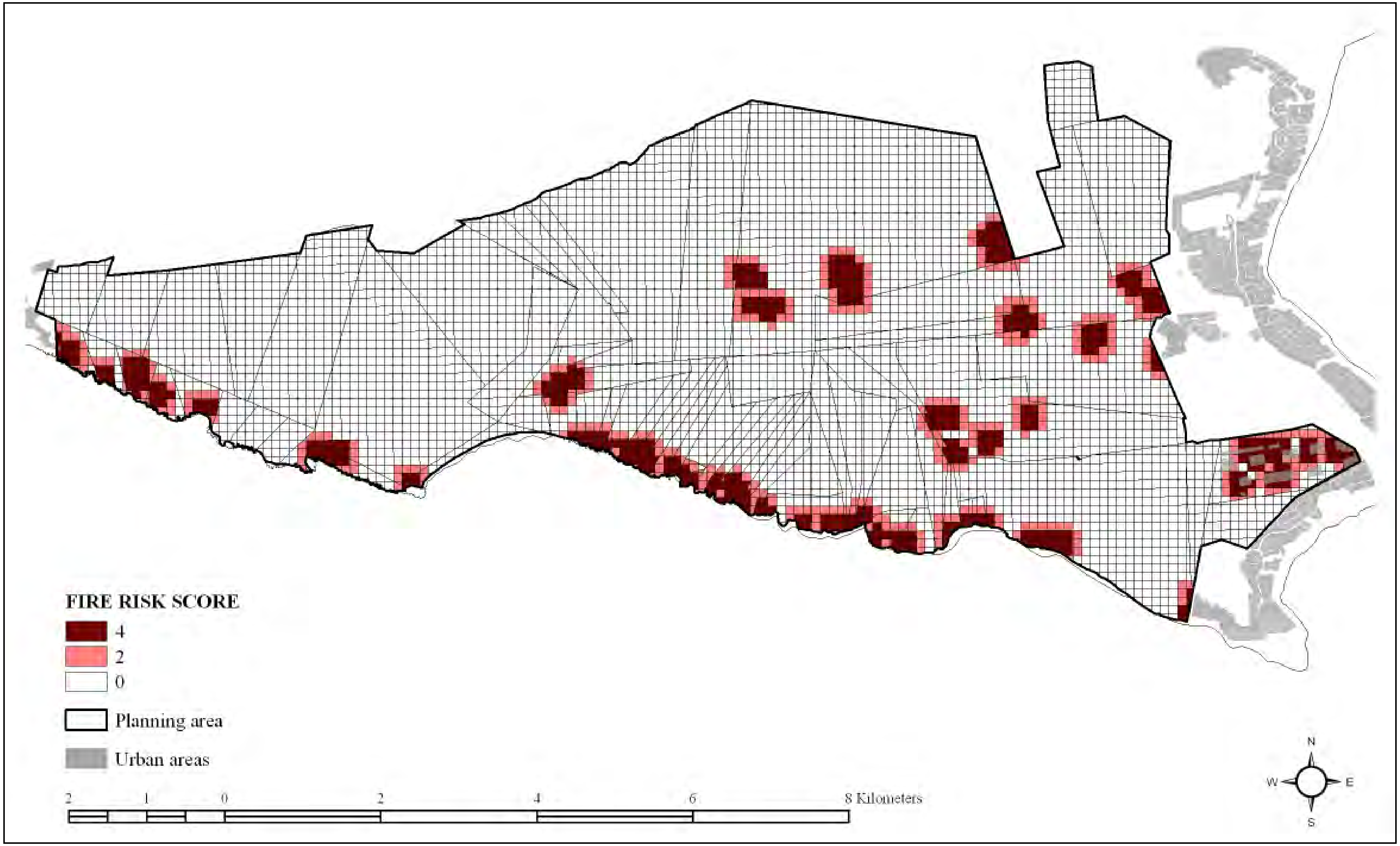


Figure 24: Fire Risk indicator score

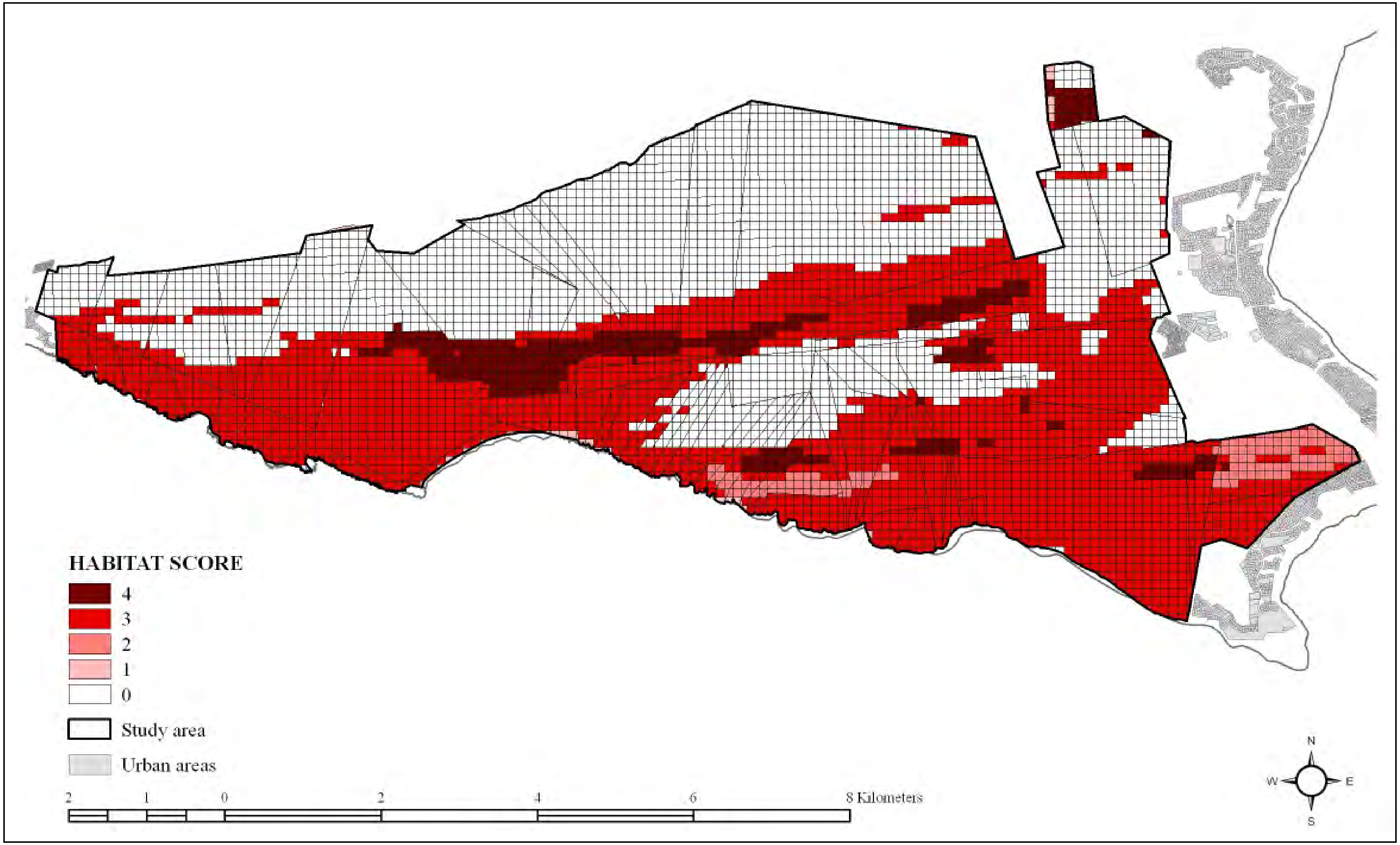

HABITAT SCORE
4
3
2
1
0
Study area
Urban areas
2
1
0
2
4
6
8 Kilometers
N
W
E
S

Figure 25: Habitat indicator score

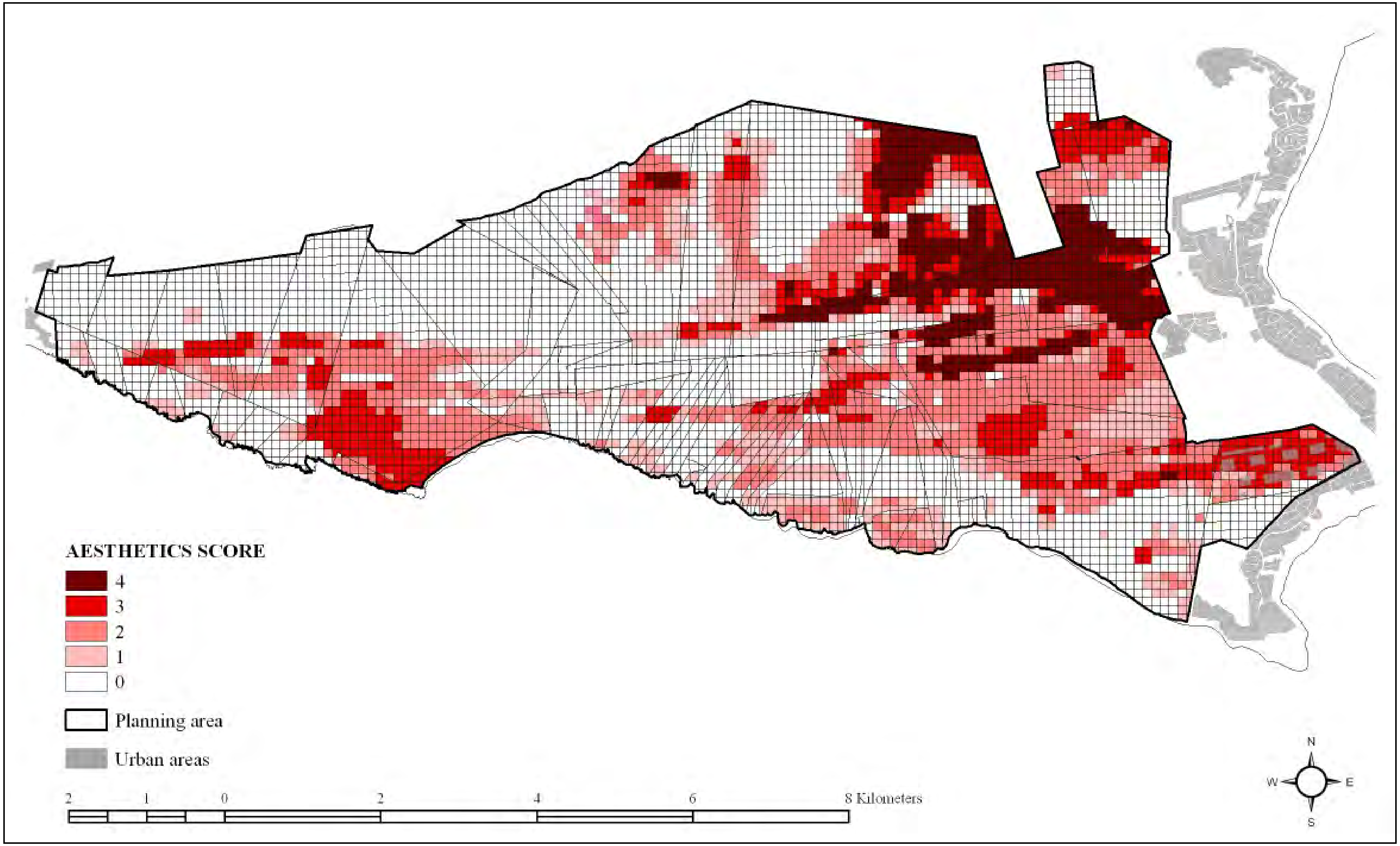


Figure 26: Aesthetics indicator score

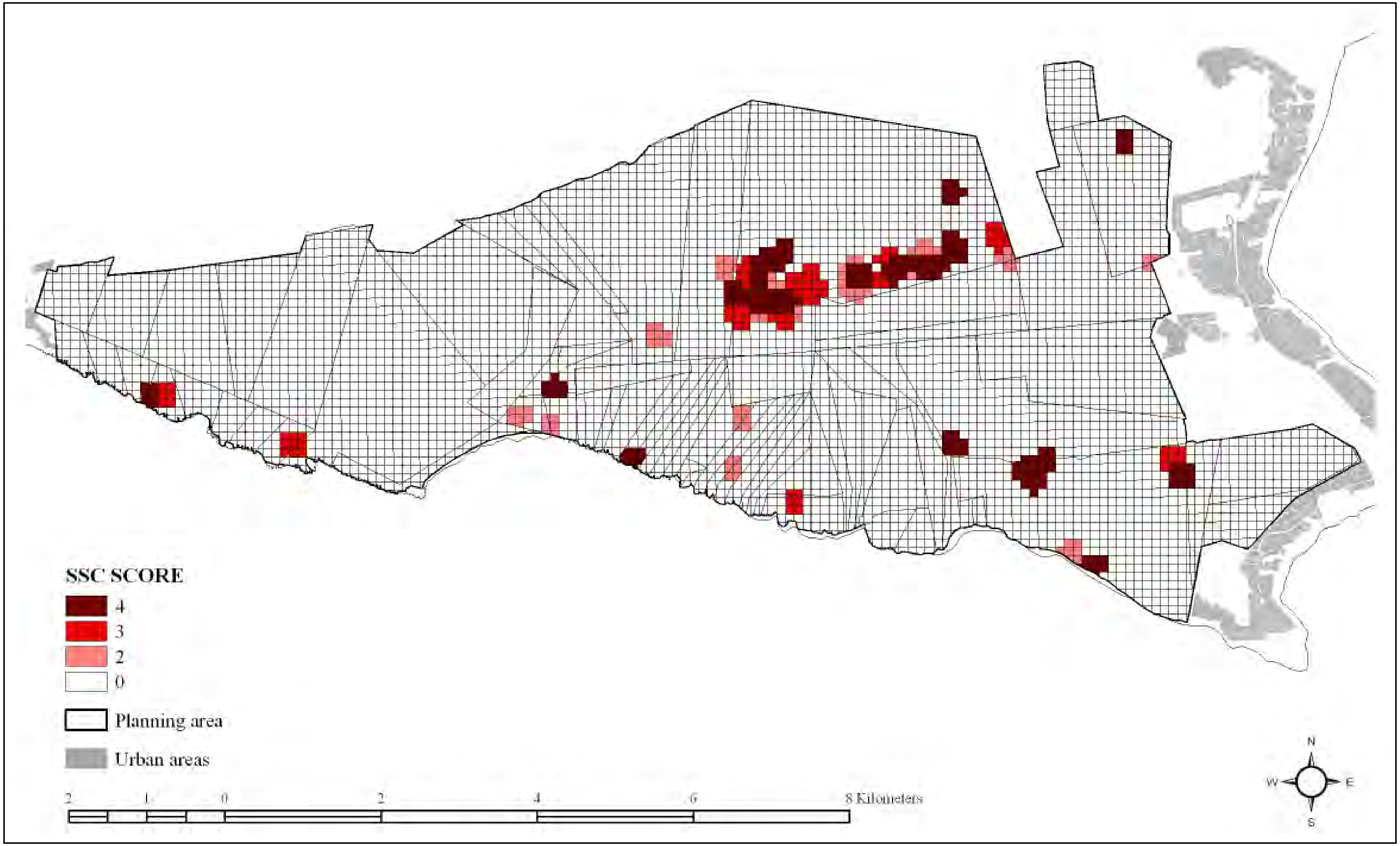


Figure 27: Species of Special Concern indicator score

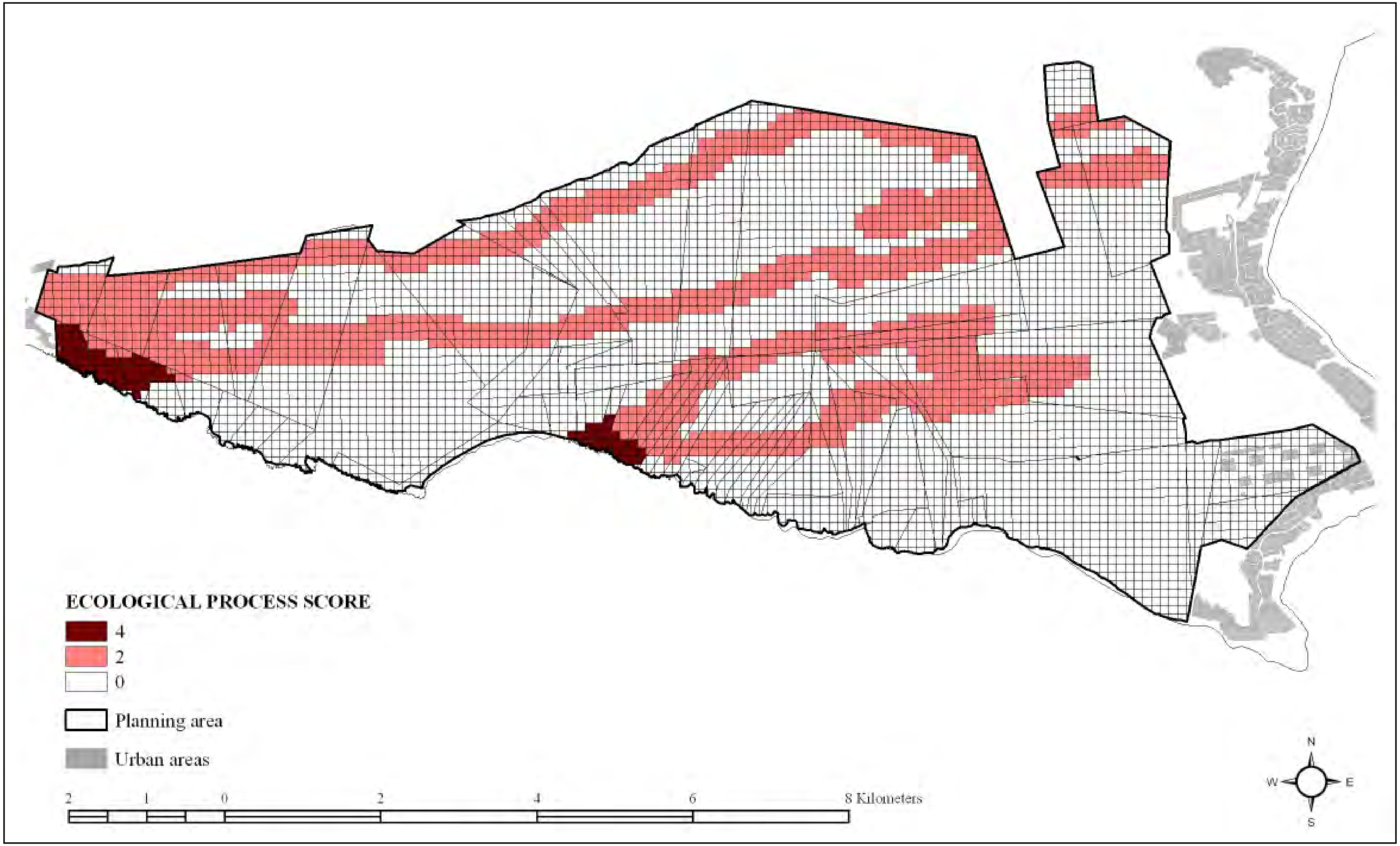


Figure 28: Ecological process indicator score

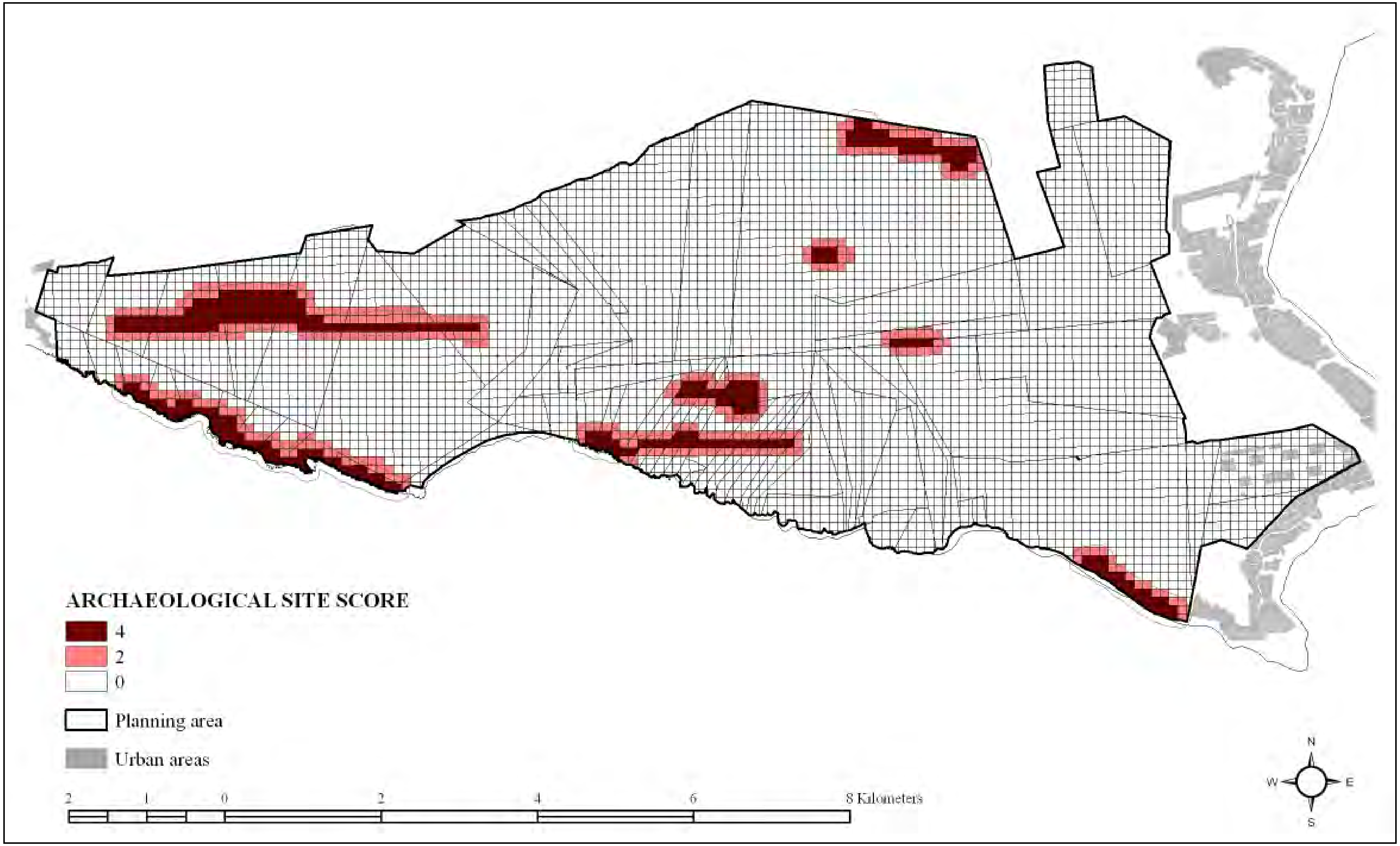


Figure 29: Archaeological site indicator score

3.5.11. Performance matrix

The various indicators were overlaid, using the Geoprocessing Union function in ESRI Arcview 3.2, to generate a single map layer with the values for planning units that were returned from the preferences models. The attribute table for this new map layer contained 59 472 entries (a 6608 row by 9 column table) and represents the performance matrix for the analysis.

3.5.12. Determining weights through stakeholder input

In order to determine weights for the criteria and indicators, a questionnaire was distributed to landowners and individuals with experience in alien plant control (for example conservation managers from other areas). Questionnaires hold a number of advantages over other methods of obtaining information (Walonick 2004):

1. They are cost effective when compared to face-to-face interviews. They become even more cost effective as the number of research questions increases;
2. They are easy to analyze. Data entry and tabulation for nearly all surveys can be easily done with many computer software packages;
3. They are familiar to most people;
4. They reduce bias. There is uniform question presentation and the researcher's own opinions are less likely to influence the respondent's answers (for example, there are no verbal or visual clues to influence the respondent); and
5. They are less intrusive than telephone or face-to-face surveys. When a respondent receives a questionnaire in the mail, it can be completed in his own time. Unlike other research methods, the respondent is not interrupted by the research instrument.

Questionnaires also have drawbacks (Walonick 2004):

1. A major disadvantage of questionnaires is the possibility of low response rates. Response rates vary widely (10%-90%), however well-designed studies can produce high response rates;
2. Another disadvantage of questionnaires is the inability to probe responses. Questionnaires are structured instruments and they allow little flexibility to the respondent with respect to response format; and
3. Gestures and other visual cues are not available with questionnaires. The lack of personal contact will have different effects depending on the type of information being requested. A questionnaire requesting factual information will probably not be affected by the lack of personal contact. A questionnaire probing sensitive issues or attitudes may be severely affected.

As a general rule, longer questionnaires get fewer responses than shorter questionnaires. Because response rate is the single most important indicator of the confidence that can be placed in the results (Walonick 2004), it was important to keep the questionnaire as brief as possible.

The questionnaire was distributed to landowners, academics, NGO and civil-society representatives and government officials. Forty three questionnaires were returned (see Table 14).

Table 14: Types of questionnaire respondents

Respondent	**Count**
Non-farming Landowner	15
Farming Landowner	3
Conservation Manager	13
NGO Representative	9
Academic	7
Government Official	7

Most respondents were non-farming landowners or conservation managers (see Table 14). Only three respondents were farmers, but none of these respondents farm within the planning area. Respondents who owned land in the area were asked to state their level of experience (no experience, some experience or considerable experience) with regard to alien plant control. Only four respondents stated that they had no experience in alien plant control. Similar numbers of respondents stated that they had either some experience (19) or considerable experience (20) in controlling alien plants.

For the remaining section of the questionnaire, respondents were asked if each of the criteria and indicators were important according to the following scale: *Strongly Disagree* (1), *Disagree* (2), *Neutral* (3), *Agree* (4) and *Strongly Agree* (5). The sum of the responses and the number of times respondents chose the maximum category (5) were recorded. Biodiversity conservation criteria scored highest, followed by economic criteria, social criteria and heritage criteria.

Table 15 Respondents' rating of the criteria and indicators

	Econ.	*Soc.*	*Biod.*	*Herit.*	*Dens.*	*Age*	*Acc.*	*Fire*	*Aes.*	*Hab.*	*Spec.*	*Proc.*
Sum	158	128	211	126	168	150	134	151	131	199	197	177
No. of 5's												
Mean	3.67	2.98	4.91	2.93	3.91	3.49	3.12	3.51	3.05	4.63	4.58	4.12
Standard Error	0.20	0.22	0.06	0.22	0.16	0.18	0.15	0.19	0.21	0.09	0.10	0.16
Median	4	3	5	3	4	3	3	4	3	5	5	4
Mode	4	3	5	3	5	3	3	4	3	5	5	5
Std. Dev.	1.29	1.42	0.37	1.44	1.06	1.16	1.00	1.26	1.34	0.62	0.66	1.05
Sample Variance	1.65	2.02	0.13	2.07	1.13	1.35	1.01	1.59	1.81	0.38	0.44	1.11
Kurtosis	-0.69	-0.88	19.08	-1.13	-0.42	-0.74	-0.40	-0.87	-1.08	1.14	0.62	-0.47
Skewness	-0.69	0.04	-4.27	0.03	-0.55	-0.21	0.35	-0.48	-0.03	-1.47	-1.34	-0.89
Range	4	5	2	4	4	4	4	4	4	2	2	3
Minimum	1	0	3	1	1	1	1	1	1	3	3	2
Maximum	5	5	5	5	5	5	5	5	5	5	5	5
Confidence (95.0%)	0.40	0.44	0.11	0.44	0.33	0.36	0.31	0.39	0.41	0.19	0.20	0.32

Table 16 was devised to facilitate pairwise comparison. Using Table 16, the mean values of the responses returned from the questionnaire for the criteria and indicators were compared against each other in a pairwise fashion to determine relative importance according to the scale developed by Saaty (1980) (see Table 7).

Table 16: Matrix for converting mean scores for criteria and indicator importance for pairwise comparison.

			Criterion / Indicator B								
	Mean Score		**1.0 Very Low**	**1.5**	**2.0 Low**	**2.5**	**3.0 Mod**	**3.5**	**4.0 High**	**4.5**	**5.0 Very High**
Criterion / Indicator A	**1.0**	**Very Low**	**1**	2	3	4	5	6	7	8	9
	1.5		2	**1**	2	3	4	5	6	7	8
	2.0	**Low**	3	2	**1**	2	3	4	5	6	7
	2.5		4	3	2	**1**	2	3	4	5	6
	3.0	**Mod**	5	4	3	2	**1**	2	3	4	5
	3.5		6	5	4	3	2	**1**	2	3	4
	4.0	**High**	7	6	5	4	3	2	**1**	2	3
	4.5		8	7	6	5	4	3	2	**1**	2
	5.0	**Very High**	9	8	7	6	5	4	3	2	**1**

Pairwise comparison matrices were then constructed for each element of the AHP hierarchy (see Table 17 to Table 20).

Table 17: Pairwise comparison matrix for the criteria

	Biodiversity	Economic	Social	Heritage
Biodiversity	1	4	5	5
Economic	1/4	1	2	2
Social	1/5	1/2	1	1
Heritage	1/5	1/2	1	1

Table 18: Pairwise comparison matrix for the Economic indicators

	Density	Age	Access
Density	1	2	3
Age	1/2	1	2
Access	1/3	1/2	1

Table 19: Pairwise comparison matrix for the Social indicators

	Fire risk	Aesthetic
Fire risk	1	2
Aesthetic	1/2	1

Table 20: Pairwise comparison matrix for the Biodiversity indicators

	Habitat	SSC	Processes
Habitat	1	1	2
SSC	1	1	2
Processes	1/2	1/2	1

The values in the comparison matrices were normalized to produce normalized comparison matrices. From the normalized comparison matrix the average of the rows were calculated to determine an estimate of the relative weights for the criteria and indicators (see Table 21). The consistency ratios for each of the pairwise comparison matrices were determined, and the comparisons were found to be acceptably consistent.

Table 21 Weights determined through stakeholder input

Criterion	Weight
Economic	0.1907
Social	0.1067
Biodiversity	0.5960
Heritage	0.1067
Indicator	Weight
Density	0.5390
Age	0.2973
Access	0.1638
Fire risk	0.6667
Aesthetics	0.3333
Habitats	0.4000
Species	0.4000

Processes	0.2000

3.5.13. *Alternative weight scenarios*

To factor in that stakeholders might have different views on the relative importance of the criteria, four aggregation scenarios were developed using different weight sets for the criteria (see Geneletti & van Duren 2008) (see Table 22).

Table 22: Weight sets for emphasising individual criteria

Criterion	**Economic-heavy**	**Biodiversity-heavy**	**Social-heavy**	**Heritage-heavy**
Economic	0.40	0.20	0.20	0.20
Biodiversity	0.20	0.40	0.20	0.20
Social	0.20	0.20	0.40	0.20
Heritage	0.20	0.20	0.20	0.40

3.5.14. *Aggregation*

For each criterion, the values achieved by the planning units for its subordinate indicators were multiplied by their respective weights. This provided an overall value for each criterion (see Figure 33, Figure 34, Figure 35 and Figure 36). The values for the criteria were then multiplied by the criteria weights to attain an overall value of the planning units for the analysis, which were subsequently mapped in ArcView 3.2.

Five maps were produced, representing the following criteria weight scenarios: 1) weights determined through stakeholder input (see Figure 37), 2) efficiency-heavy weight scenario (see Figure 38), 3) biodiversity-heavy weight scenario, 3) social-heavy weight scenario and 4) heritage-heavy weight scenario.

The overlap between the different scenarios is presented in Figure 39. Overlap was calculated by determining which planning units performed relatively well (overall score > 1.4), moderately well (overall score 0.8-1.4), and those that scored poorly (overall score < 0.8) in all of the scenarios.

A problem shared with all additive models is the potential for critical features within one of the indicators to be 'lost' on aggregation if those features do not score well on the other indicators. In order to account for this, and to prevent the dilution of critical features, a separate map was generated to depict planning units containing features that achieved the maximum score in any of the indicators that were determined to be critical (see Figure 40). The efficiency indicators and the aesthetic indicator were excluded because these were not considered to contain features of critical importance. The features that are depicted on this map thus are critically endangered habitat types, endangered species, archaeological sites and the areas immediately adjacent to houses (in order to reduce the risk posed by fire on the houses).

3.6 Results

3.6.1. Outputs of the analysis

Similar patterns emerged in the various weight scenarios, with certain areas performing either consistently well or consistently poorly regardless of the change in criteria weights (see Figure 38 and Figure 39). The largest difference between the scenarios is noted in the areas that achieved moderate scores.

The priorities on the analysis emerged as follows:

- The coastal strip was as a priority in all of the weight scenarios due to: the proximity to houses, the location of archaeological features along the coast, the easy accessibility of this area and the generally low density of alien plants;
- The area between the two mobile dunefields also emerged as a priority in all of the scenarios. This is because of the presence of *Critically Endangered* habitats in the western extent of this area, (wetlands and forest), the large number of species of special concern, the relative low density of alien plants, and the future presence of the residences associated with the Sand River Sanctuary development. The eastern extent of this area is also relatively accessible; and
- The more inland sections of the Rocky Coast Farm area, the northern section of the St Francis Field and the central section of the Macohy Thula-Moya property emerged as moderate to high priorities, especially within the scenario that emphasized efficiency. This is largely due to the low density of alien plants in these areas, but also because they are easily accessed.

The following areas did not emerge as priorities:

- The inland section of the Mostertshoek area, the western inland sections of the Thyspunt area, the western inland sections of the Rebelsrus area and the eastern section of the Macohy Thula-moya propery: this is largely due to the high density of rooikrans in these areas and also due to the presence of older alien plants in some of these areas;
- The cores of the mobile dunefields: although these areas have low densities of alien plants they are relatively inaccessible, generally contain few species (except for the wetland areas which contain species of special significance) and are not required to be cleared to protect infrastructure from fire. The mobile dunefields were also assigned a low Ecosystem Status by the conservation assessment for the Subtropical Thicket Biome. Areas that emerged as priorities within the dunefields were those that contained archaeological sites, special species (wetlands) and the edges of the dunefields, which are being encroached by alien plants;
- A large area within the St Francis Links and a portion of the inland section of Rebelsrus achieved low overall scores due to the habitat degradation and transformation status of these areas;
- The area to the north of the Oyster Bay dunefield: this area did not perform well in the analysis because of the relatively low ecosystem status of Humansdorp Grassy Fynbos and because of the dense invasions in this area.

3.6.2. *Sensitivity analysis*

A sensitivity analysis was conducted assess the stability of the results and to test the effects caused by changes in the weights assigned to the criteria and indicators (see Geneletti 2008; Ananda & Herath 2003). In order to do this, the weight of one criterion or indicator is varied while keeping the weights for the other criteria and indicators constant.

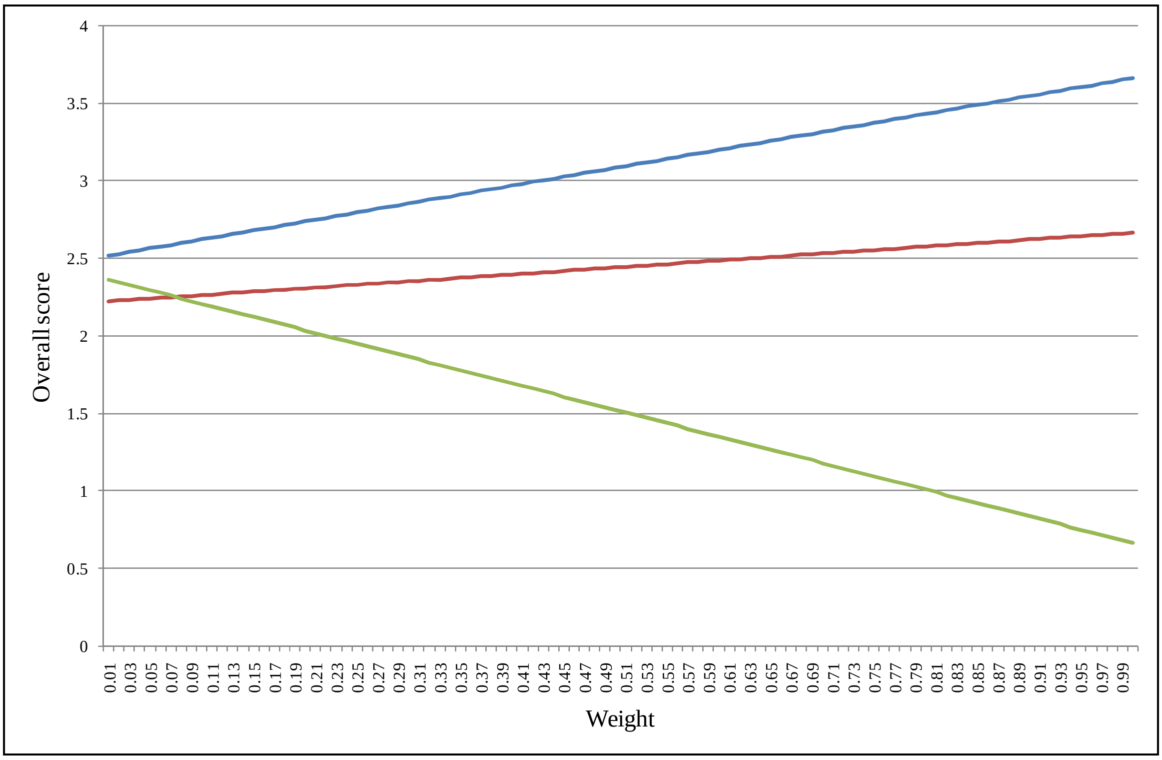


Figure 30: An arbitrary example of how the performance of planning units is affected by altering the weights of one of the criteria or indicators.

Figure 30 shows the results of an arbitrary sensitivity analysis, presented for explanatory purposes. On the *X*-axis, the range of possible weight values is represented, and on the *Y*-axis, the overall score or performance of the alternatives is represented. When two lines cross, a rank reversal between two alternatives occurs (the alternatives switch rankings in terms of their value). For example, a rank reversal occurs between the alternative represented by the green line and the alternative represented by the red line at a weight value of 0.07 for criterion examined in Figure 30. Below the value of 0.07 for the weight, the alternative represented by the green line exhibits better performance, while beyond this value the alternative represented by the red line exhibits better performance. The blue line does not intersect with either of the other lines, meaning that this alternative performs better than the other two alternatives regardless of changes to the value of the weight for the criterion considered.

In order to conduct the sensitivity analysis, 30 planning units were randomly selected. This was done by deriving random numbers in Microsoft Excel and then using these random numbers to select planning units according to their unique identity numbers. The performance of these planning units was computed for all possible values for the weights at every level in the AHP hierarchy. The ratios between the remaining weights within each level of the of AHP hierarchy were kept constant.

For each of the criteria and indicators, graphs were plotted for the variations in the overall scores of each of the selected planning units as the weights were manipulated (see Appendix 1). The intersection between any two lines within any of the graphs was calculated by following the procedure set out below:

The functions for line 1 (representing the performance of planning unit 1) and line 2 (representing the performance of planning unit 2) are:

$$y_1 = m_1x_1 + c_1$$

$$y_2 = m_2x_2 + c_2$$

For each of these lines the gradients, m, can be calculated by dividing the change in y value by the change in x value, as follows:

$$m = \Delta y \ / \ \Delta x$$

The point where these lines cross the Y-axis, c, can then be calculated by substituting values for x, y and the gradient for the lines calculated above into the functions for each of the lines.

At the point of intersection, $y_1 = y_2$ and $x_1 = x_2$, therefore:

$m_1x + c_1 = m_2x + c_2$

The x value for the intersection (i.e. the weight value at which a rank reversal occurs), can be calculated by rearranging the above equation and substituting the m and c values for each of the lines into the following equation:

$x = (c_2 - c_1)/(m_1 - m_2)$

The number of intersections and the range between the average lowest intersection and the average highest intersection were calculated for each of the criteria and indicators (see Figure 31). Criteria and indicators that exhibit a high number of rank reversals and that have rank reversals occurring within a small range are more sensitive to affecting the overall outcomes of the model than those that have fewer rank reserves and where rank reversals happen over a larger range.

As would be expected, manipulation of the criteria weights had a greater influence on the ranking of planning units than manipulation of the indicator weights (see Figure 31). Of the criteria, the efficiency criterion had the greatest average range between rank reversals with alteration of weights, this means that a large change in the weight is generally required for rank reversals to occur based on the weight of this criterion. In contrast, the heritage criterion generally only required a small change in weight for rank reversals to occur.

Manipulation of the biodiversity indicator weights generally had the greatest impact on the overall ranking of planning units compared to the other indicators, but changes to ranking generally only occurred with large changes to the weight values. Manipulation of the efficiency indicator weights had a smaller affect on the overall rankings, but changes to rankings occurred over a smaller range in the change of the weights for these indicators. Changes to the social indicator weights resulted in the smallest change in the overall ranking of planning units and required the greatest average manipulation of the weights for overall ranking to be affected.

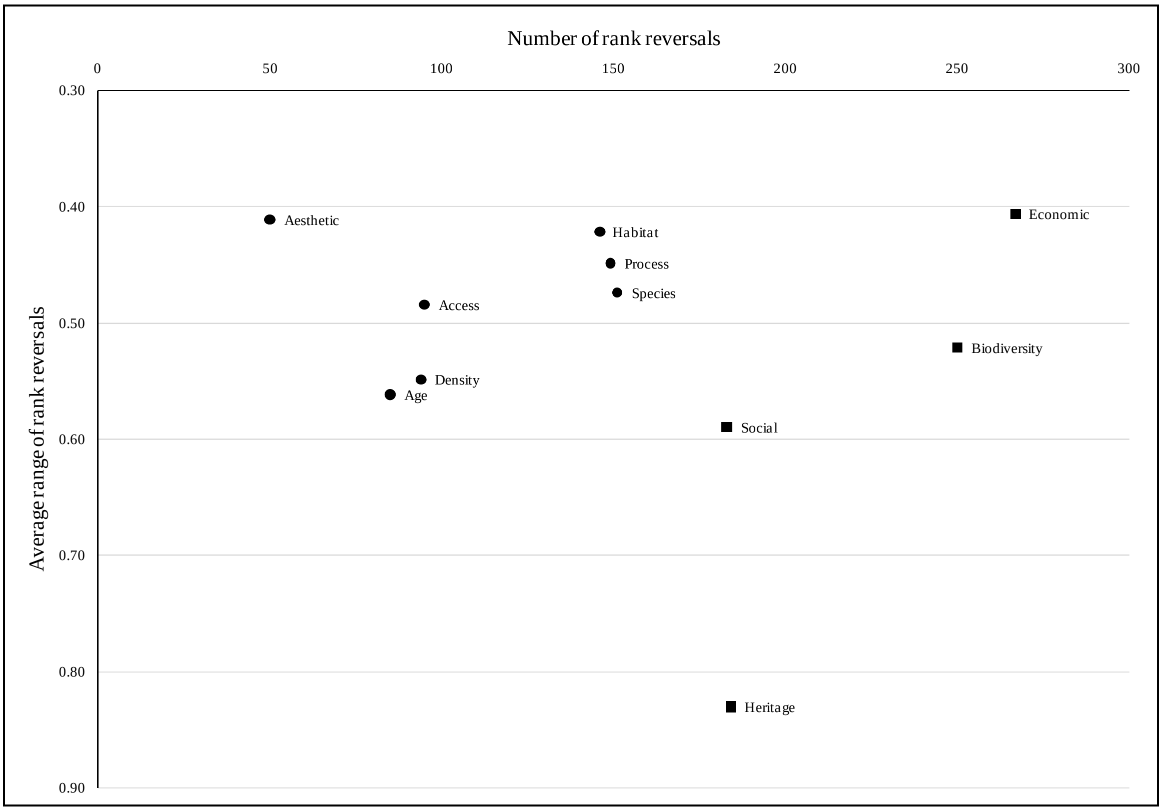


Figure 31: The number of rank reversals and the average range of the rank reversals for the criteria and indicators.

The average change in the overall score, on manipulation of the weights, of the planning units and the proportion of planning units affected were also calculated (Figure 32). Manipulation of the weights of certain criteria and indicators affected the overall score of planning units to a greater degree than manipulation of the weights of others. The proportion of planning units affected by manipulation of weights also varied.

As would be intuitive, manipulation of the criteria weights generally had a greater affect on overall score and ranking of planning units than manipulation of the indicator weights. All of the planning units were affected by the manipulation of the criteria weights and the average change in overall score of the planning units was relatively high. Manipulation of the biodiversity indicator weights had a much greater affect on the overall score of planning units than manipulation of the efficiency indicator and social indicator weights, but more planning units were affected by the manipulation of the efficiency indicator weights. Manipulation of the social indicator weights had the lowest impact on the overall score of the planning units and the least number of planning units were affected.

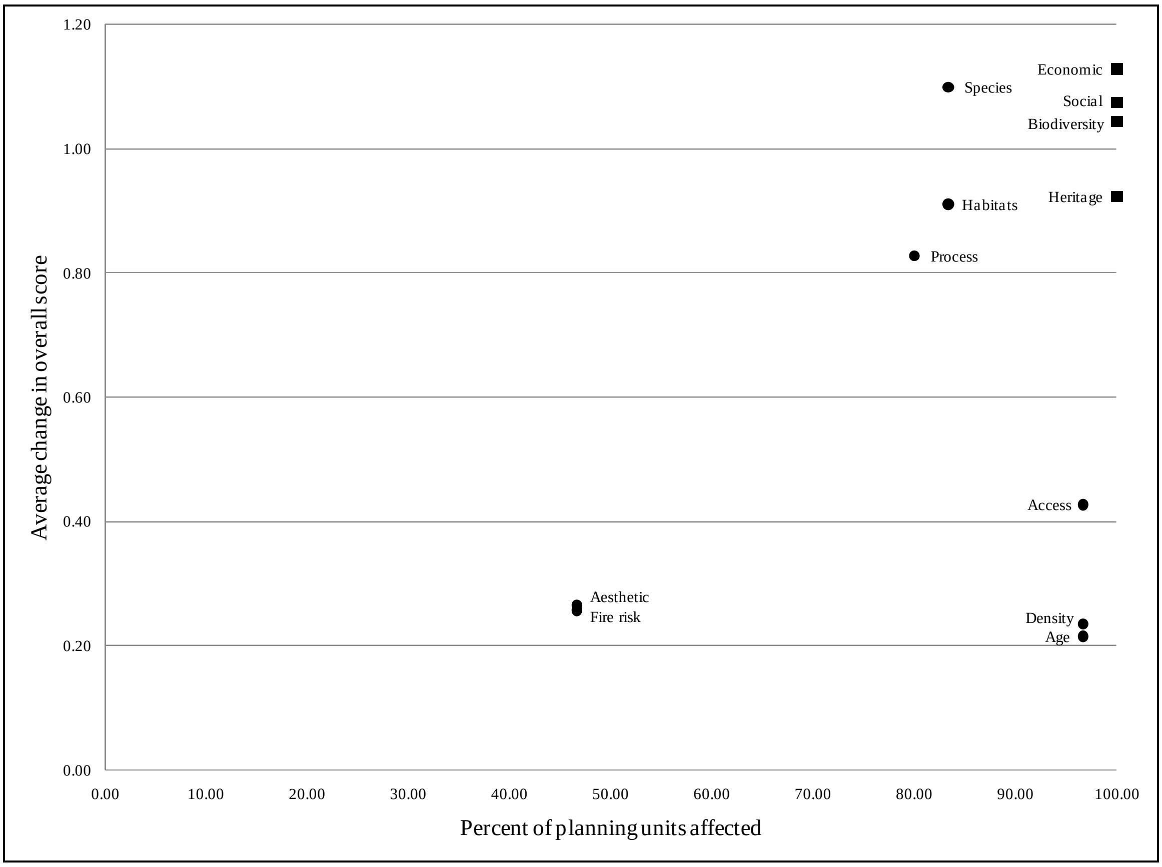


Figure 32: Average change in overall score and the percent of planning units affected by weight manipulation.

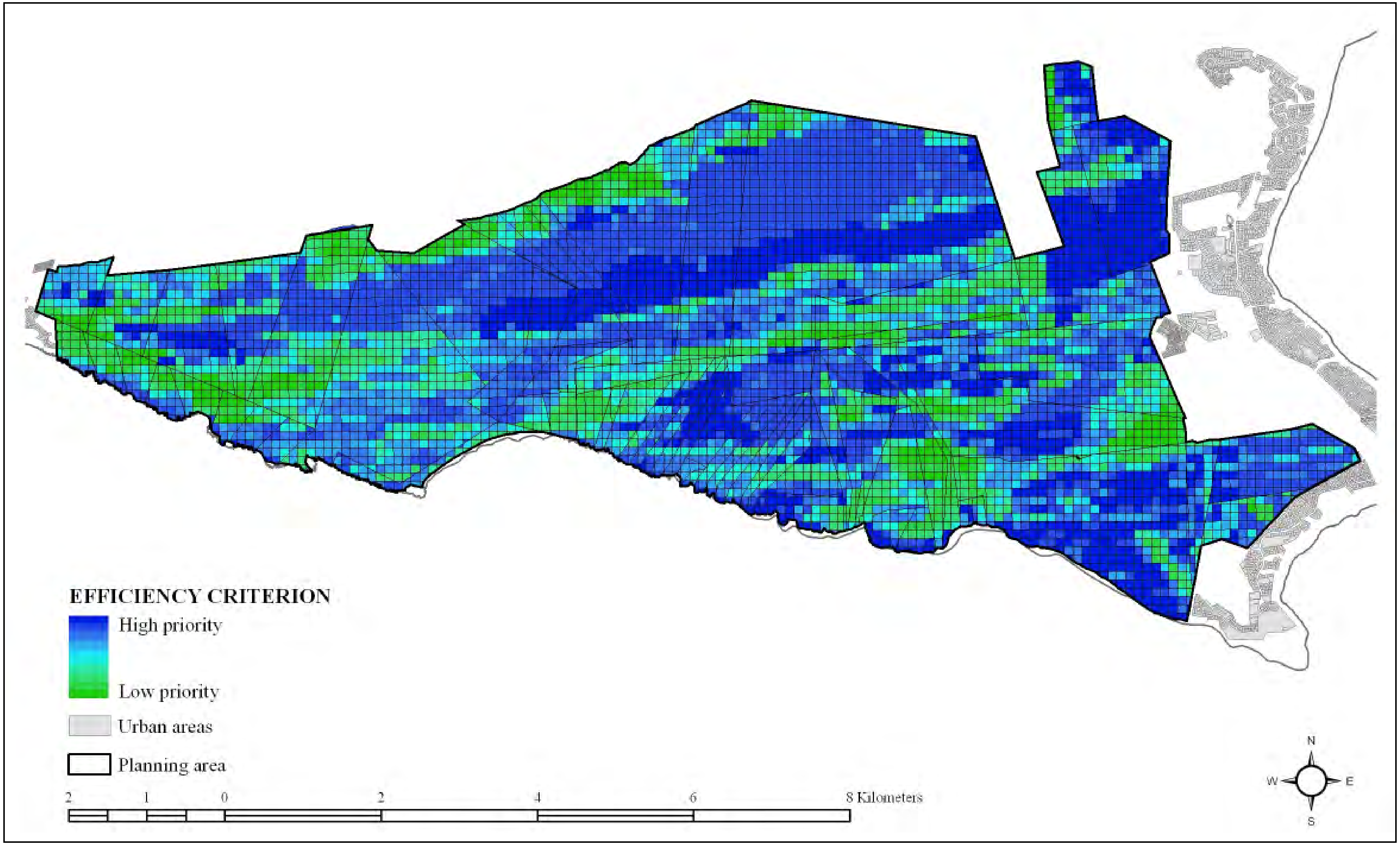


Figure 33: Aggregated efficiency criterion score.

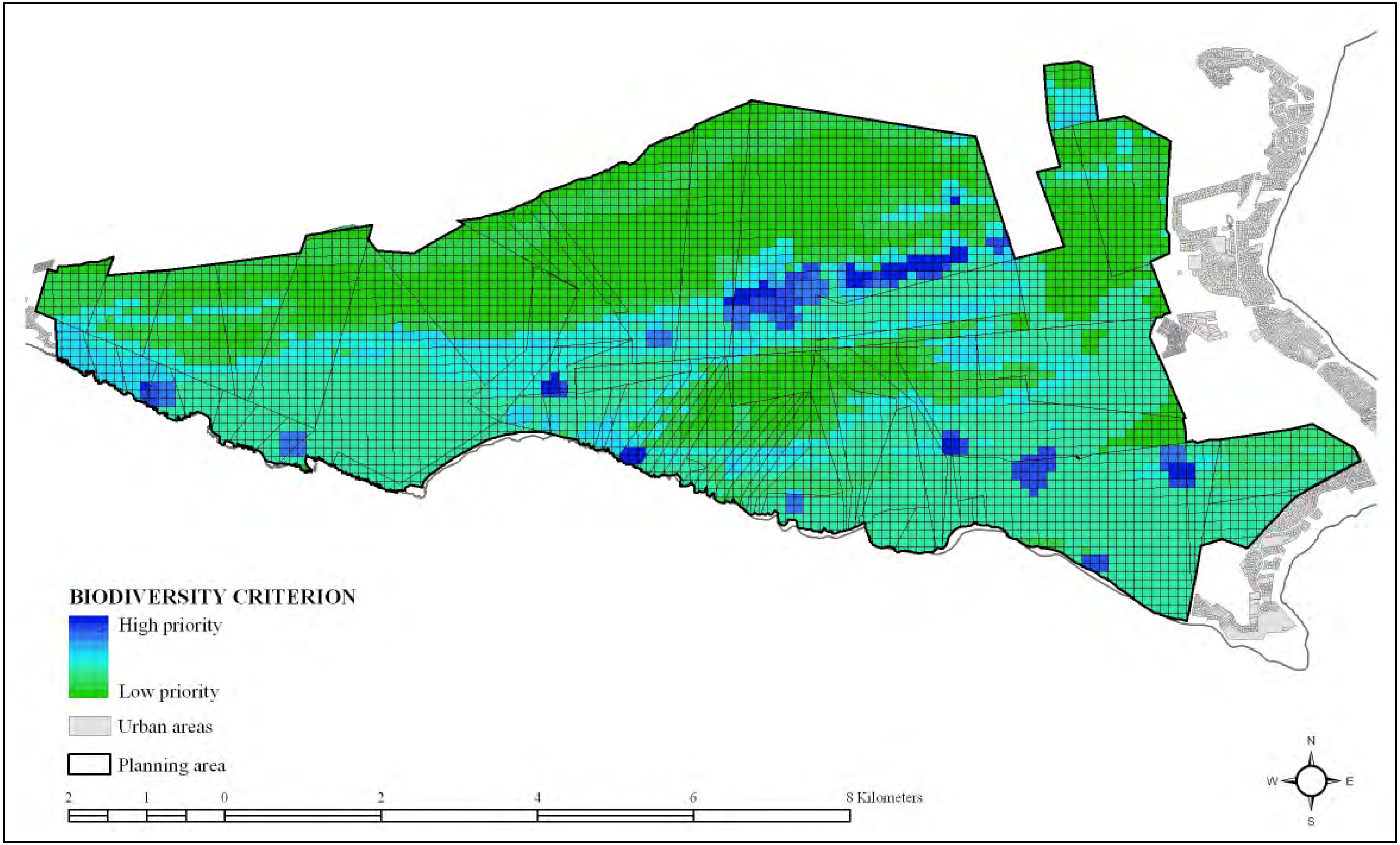


Figure 34: Aggregated biodiversity criterion score.

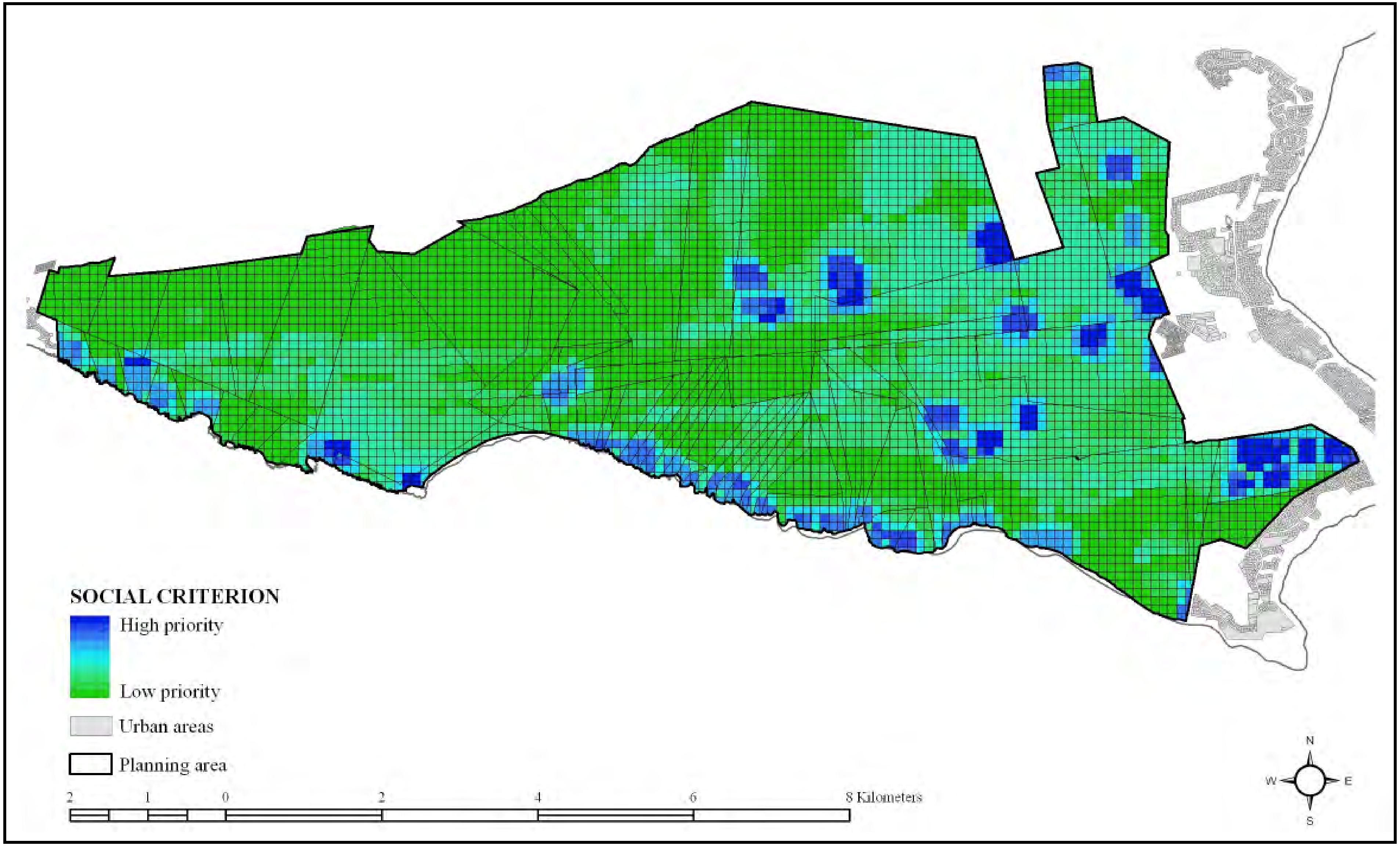


Figure 35: Aggregated social criterion score.

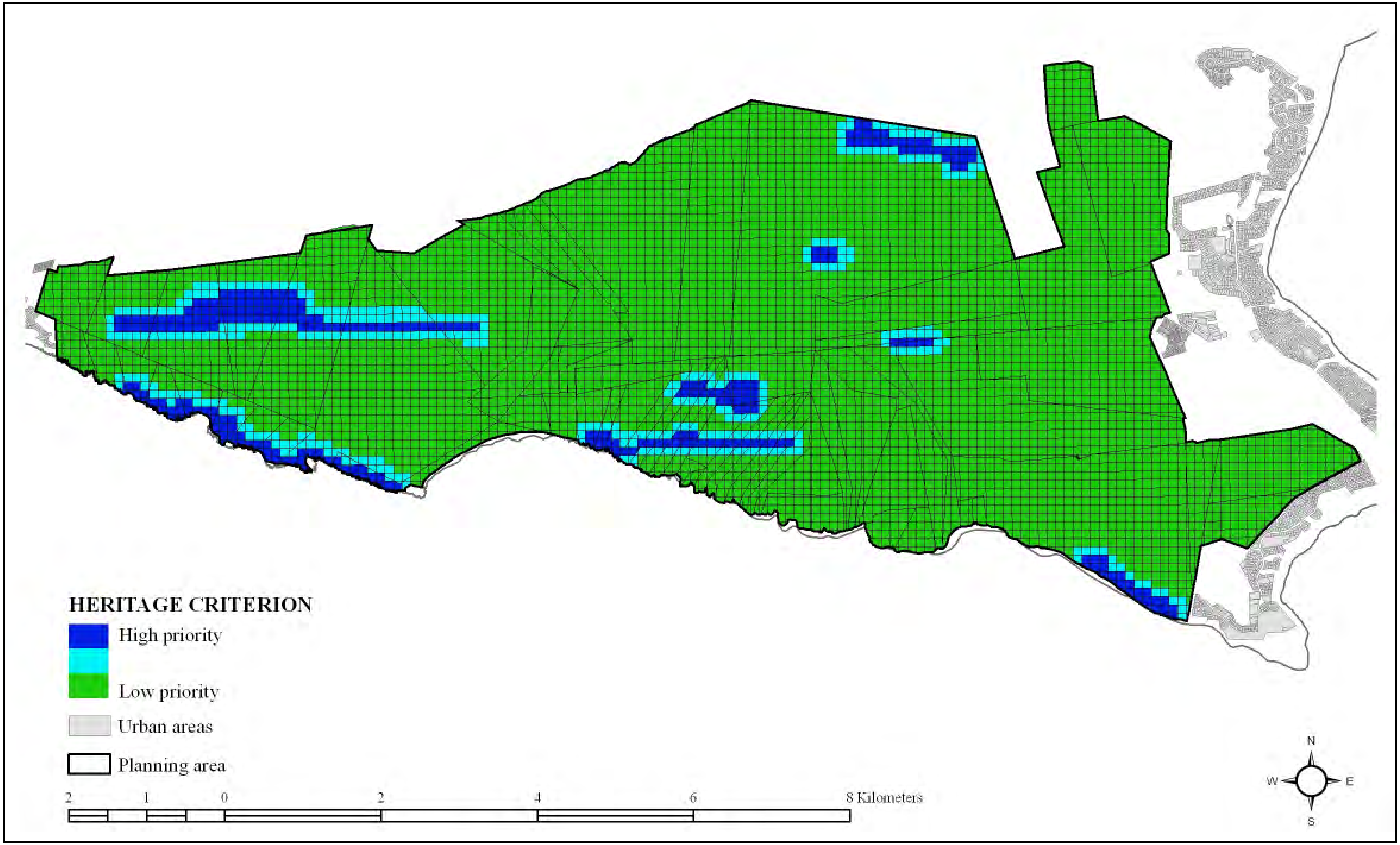


Figure 36: Heritage criterion score.

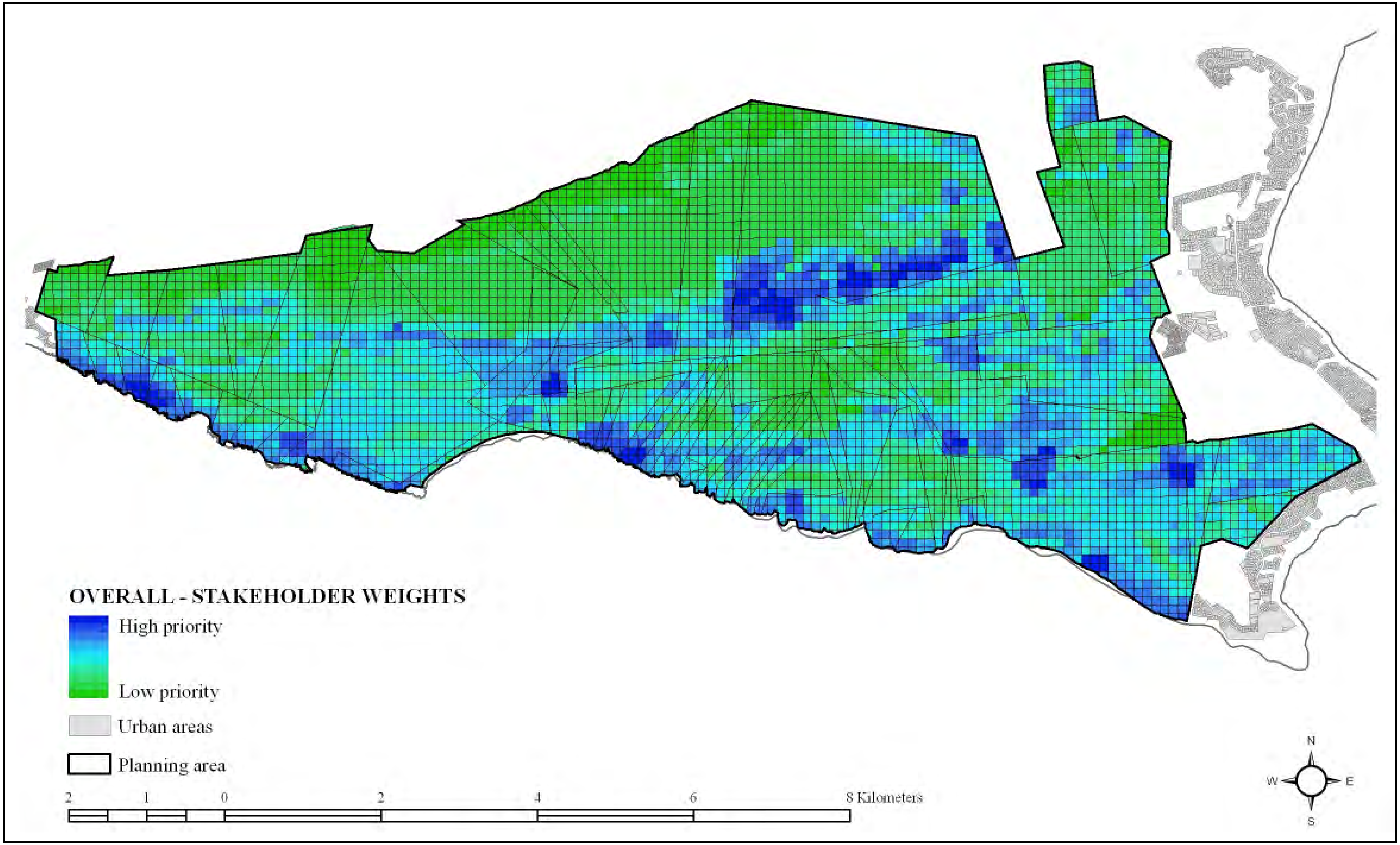
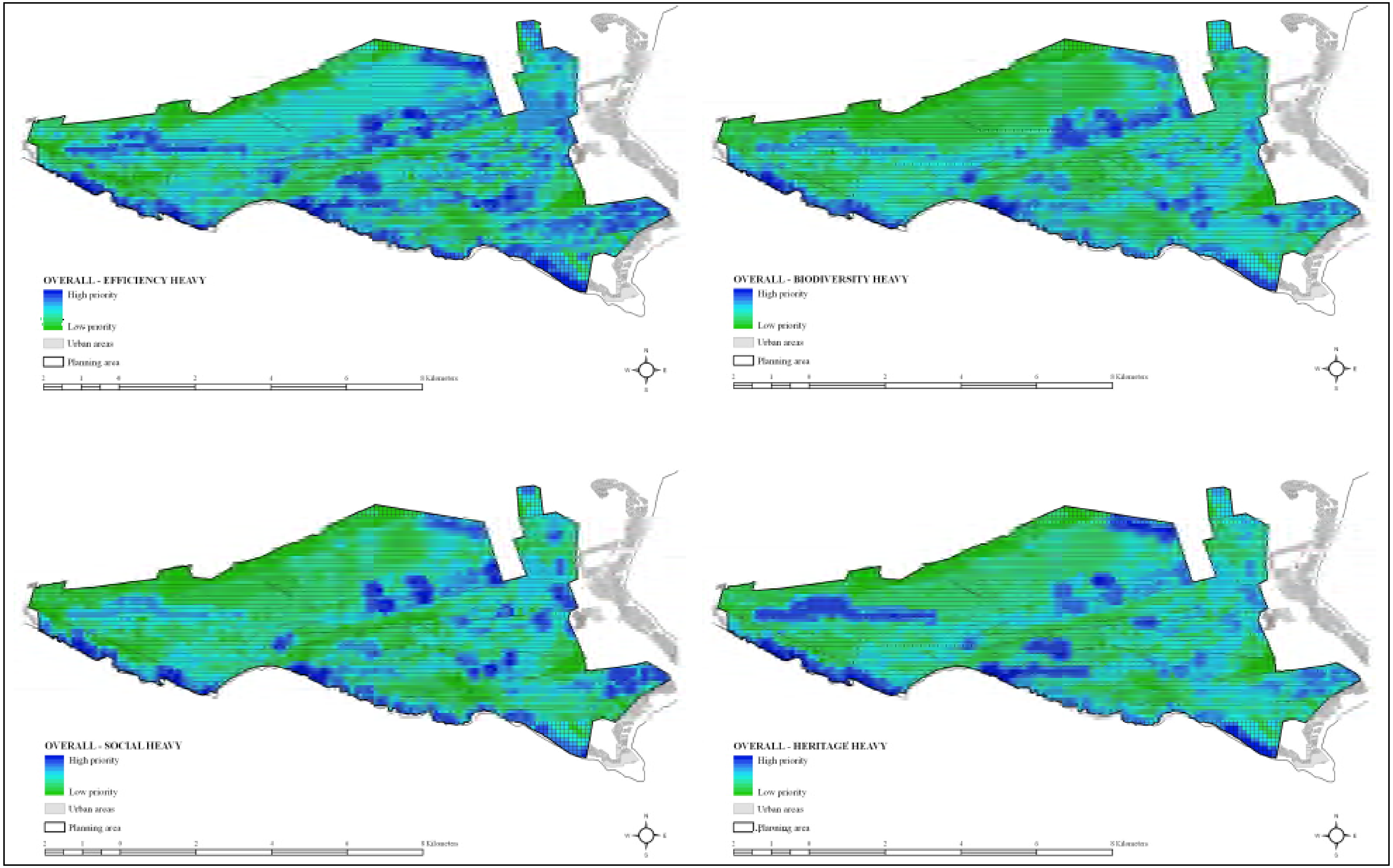


Figure 37: Overall results of the MCDA analysis using weights determined through stakeholder input.



OVERALL - EFFICIENCY HEAVY
High priority
Low priority
Urban areas
Planning area
8 Kilometers
OVERALL - BIODIVERSITY HEAVY
High priority
Low priority
Urban areas
Planning area
8 Kilometers
OVERALL - SOCIAL HEAVY
High priority
Low priority
Urban areas
Planning area
8 Kilometers
OVERALL - HERITAGE HEAVY
High priority
Low priority
Urban areas
Planning area
8 Kilometers

Figure 38: Weighting scenarios emphasising each of the criteria.

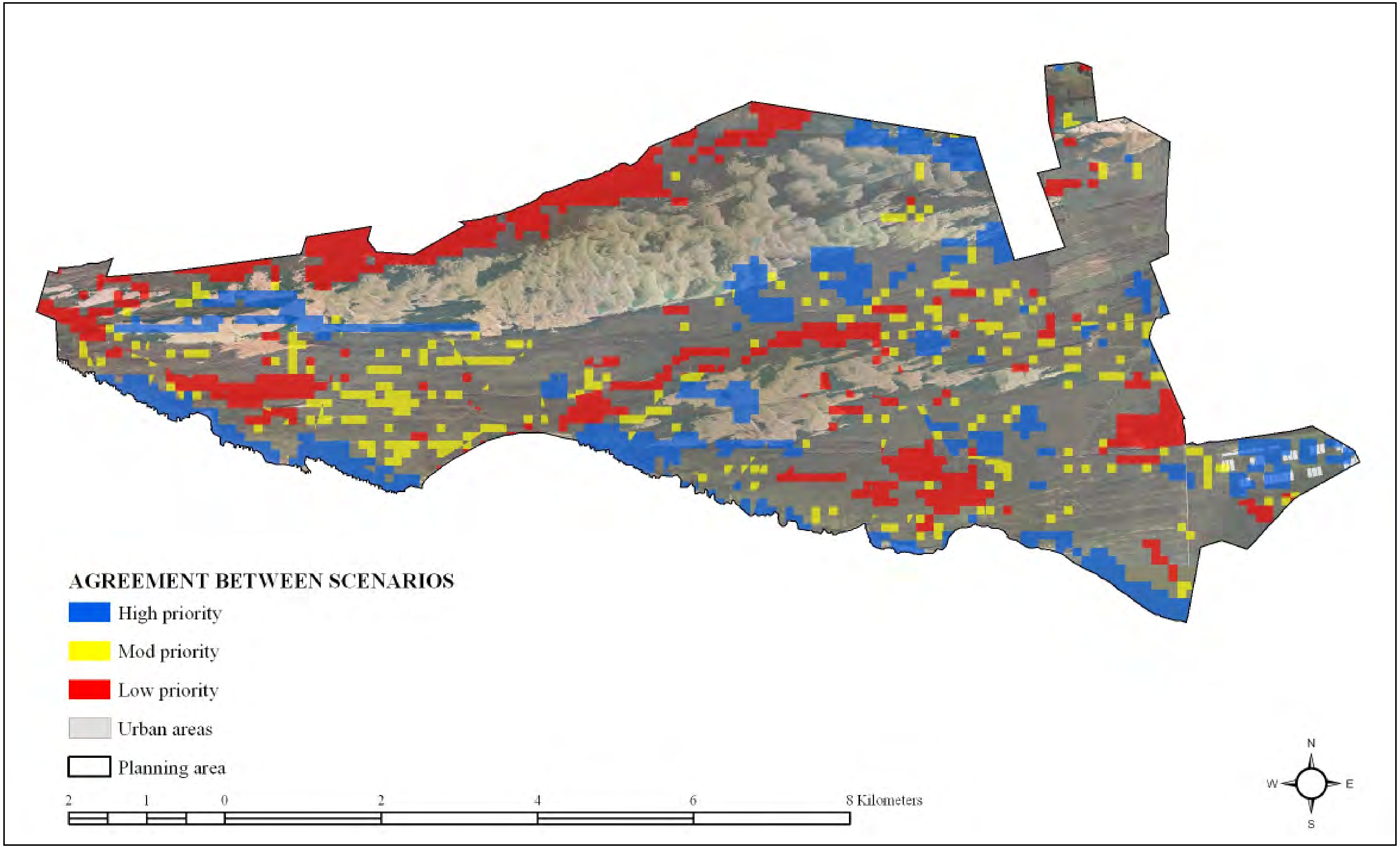


Figure 39: Overlap between the scenario maps.

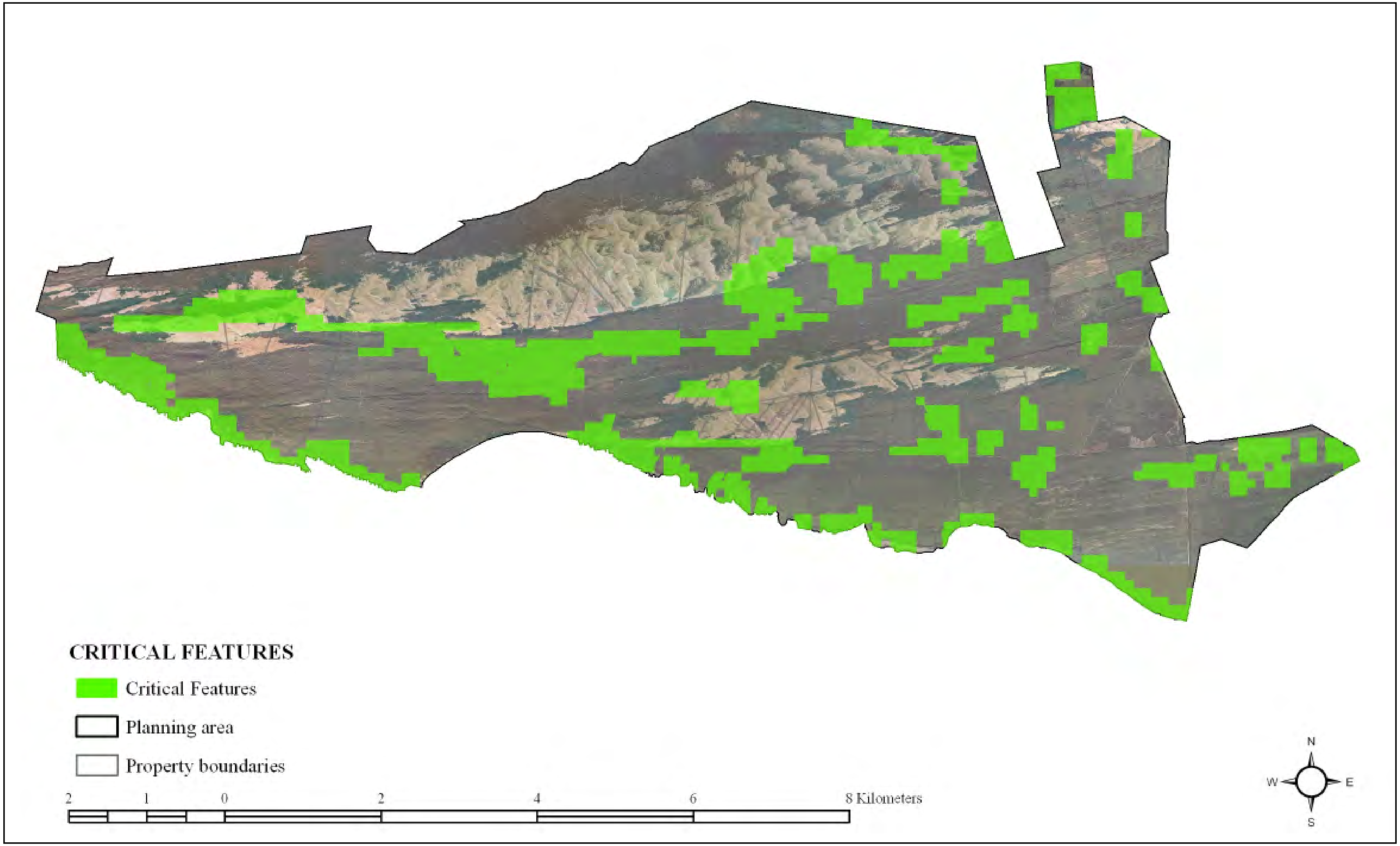


Figure 40: Critical features within the planning area.

3.7 Discussion

The nature of the alien plant problem on the conservancy inherently requires a consideration of the distribution of relevant spatial features, and of numerous, potentially conflicting criteria. The integration of GIS and MCDA techniques provided a means to account for these requirements, while also serving as unbiased and defensible method for resource allocation. As experienced by other authors (e.g. Strager 2004), the AHP framework proved to be useful in the structuring of the decision-problem as a hierarchy of goals, objectives and criteria.

While other alien plant control strategies are often based on experience of individual land managers, who intuitively take a number of considerations into account, the model allows for a quantified and explicit representation of factors influencing decision-preferences. A comparison of early results from the model with strategies developed for individual properties within the study area showed striking similarities in terms of the identified priority areas for clearing. For example, the priority areas determined by the model for the Eskom land at Thyspunt matched with the areas identified as priorities by the conservation manager for the properties (Gert Greeff *pers. comm.* 2005). Similarly, the priority areas for the Macohy Thula-Moya property determined by Coetzee (2003) showed strong agreement to those determined by the model. These similarities suggest that the model accurately quantifies features and preferences that would be intuitively taken into account by land managers.

There are few documented cases of MCDA-GIS based resource allocation systems for alien plant control. However, it is believed that the simplicity and flexibility of the approach, allows for the methodology to be easily reproduced for other areas. The data requirements for the development of the model are not excessive, and are believed to be appropriate to the scale of the current application. However, there is scope to improve the approach, especially with regard to the identification of relevant criteria for the analysis. In addition, several of the criteria included in the model could be refined in order to add further defensibility to the results. For example, Ecosystem Status is only one component of habitat value and other factors, such as relative rarity of habitat types, could be considered for inclusion. As an illustration of this, the mobile dune systems still encompass much of their original extents and thus have low Ecosystem Status, but are nonetheless significant due to the fact that they are regionally rare (Cowling 1997). Similarly, it is likely that a host of other ecological processes could be incorporated into the Ecological Process indicator.

The ease of understanding of the model allowed for stakeholder participation in the determination of criteria and their relevant importance of the criteria. Because stakeholders had input into the development of the model, the likelihood of the results being accepted is increased.

A strength of the approach is that stakeholder preferences can be incorporated by simply altering weights at any level of the decision-hierarchy. The method for determining weights from stakeholder preferences was a divergence from standard approaches, which normally require a direct pairwise comparison of the hierarchical decision-elements by the stakeholders. However, this is justified considering the exceedingly large amount of comparisons and would be required of the stakeholders under more conventional approaches.

The utility of a questionnaire to elicit stakeholder preferences is debatable. Since questionnaire responses were averaged, widely divergent views by different stakeholders may be diluted. However, differing views were accounted for by the consideration of different weighting scenarios, and of areas that are priorities across scenarios. This assisted in determining areas that would be priorities regardless to moderate changes to the weighting of criteria. The different weight scenarios resulted in similar patterns in terms of the overall performance of the planning units, indicating that model is not highly sensitive to small changes in the weighting system. Areas that showed no overlap between weighting scenarios (either as high priorities or low priorities) are those areas that will exhibit the greatest change in priority status on alteration of weights. It is these areas where stakeholders are mostly likely to disagree on, depending on their own preferences, when it comes to the allocation of resources.

A shortcoming of additive models, complete compensation, is avoided by presenting critical features as a separate layer. These are features that should be considered, because of their significance within a single criterion, as priorities for clearing, regardless of the changes to the weighting system.

A fortunate outcome of the results from the model is the distribution of priority areas across the conservancy. The priority areas were distributed across the spectrum of property ownership, and did not show a distinct favouring of the property of any one landowner. This should assist in implementation, because the majority of landowners have at least a portion of their properties identified as a priority for clearing on implementation.

The paucity of accessible implementation-focussed literature on resource allocation systems for alien plant control at the cadastral scale, and the pressing need for effective control strategies, suggests that future work should be focussed in this direction. Another aspect that warrants further attention, and which was superficially covered in the current study, is the human component of alien plant control strategy. This is especially true for landscapes comprised of multiple landowners and land uses, and where the need for ecosystem management requires cooperation between landowners. The development of institutions, such as conservancies, for stimulating collaborative management has occurred elsewhere successfully (Brunckhorst 1998), however their utility is not fully appreciated locally.

4. IMPLEMENTATION

In this chapter, the outputs of the analysis are interpreted into an implementation plan to guide planning for alien plant control operations.

4.1 Implementation plan

4.1.1. Clearing priority areas

The implementation plan was developed by classifying areas as either immediate, short-, medium- or long-term priority areas for clearing. These priority areas represent clearing blocks and were selected through an examination of the overall results of the MCDA analysis (see Figure 37), the overlap between the different scenarios (see Figure 39) and critical features (see Figure 40). The extent of alien plant invasion within the clearing blocks are presented in Table 23. Clearing blocks should ideally be delineated by topographical features in order to assist clearing teams in orientating themselves in the field (Gert Greef pers comm. 2005). The dune ridges within the study area were therefore included in Figure 41. The density of alien plants within the clearing blocks was also depicted in this map to assist with prioritisation within the blocks (darker areas representing less dense stands).

Table 23: Extent of invasion within the priority clearing categories.

Priority	Dense (ha)	Moderate (ha)	Sparse (ha)	Not invaded (ha)	Total (ha)
Immediate	393	352	526	560	1832
Short-term	204	205	407	185	1001
Medium-term	270	115	226	1273	1885
Long-term	594	163	149	145	1051
Total	1462	835	1309	2163	5769

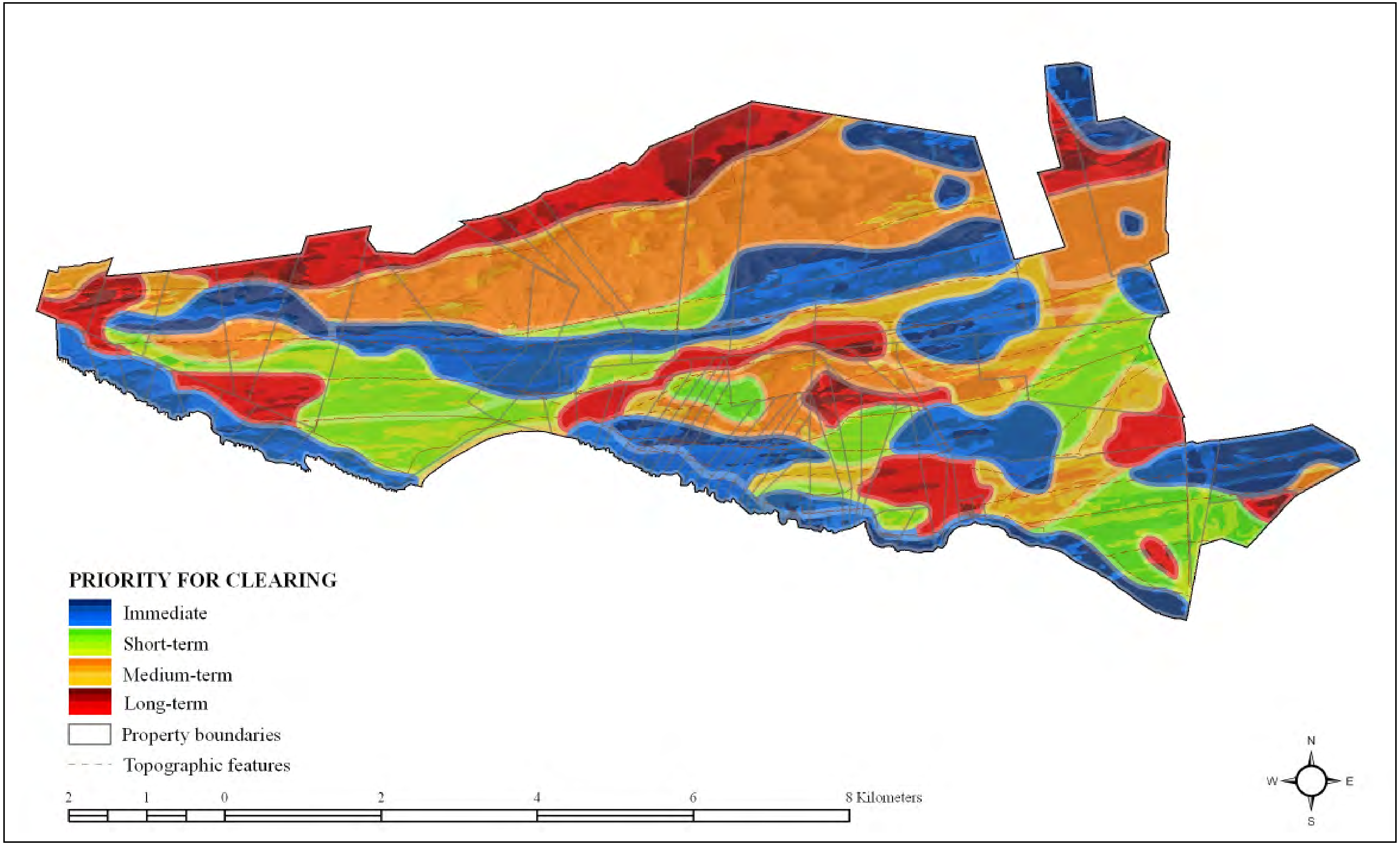


Figure 41: Immediate, short-term, medium terms and long-term clearing priority areas.

4.1.2. *Estimated clearing costs*

The figures in Table 24 reflect the per hectare costs to clear rooikrans. Clearing costs were obtained from data from the CSIR (Anon. 2004) and, for comparison, from a representative of the Working for Water (WfW) Programme (Andrew Knipe *pers comm* 2008). Figures from the CSIR were adjusted to current date by allowing for 10% inflation per year.

Table 24: Costs per ha for clearing *Acacia cylops*.

Source	**Treatment**	**Dense (>75% cover)**	**Mod (25-75% cover)**	**Sparse (<25% cover)**
CSIR Projected 2009 cost / ha (Anon. 2004)	Initial clearing	R 3,655	R 2,666	R 861
	1st follow-up	R 2,486	R 1,322	R 549
	2nd follow-up	R 1,229	R 758	R 299
	3rd follow-up	R 615	R 379	R 150
WfW (Andrew Knipe *pers comm.* 2008)	Initial clearing	R 5,000	R 3,500	R 1,500

The figures from the WfW Programme were considerably higher than those obtained from the CSIR data. A conservative estimate of the costs to clear the clearing blocks was calculated using the CSIR figures (see

Table 25). From this, the estimate cost to complete initial clearing and three follow-up treatments within the entire study area is approximately R18 million. The majority of this total cost is from the costs associated with clearing dense stands (approx. R10.5M). In comparison the cost to clear and follow-up the sparse stands is much lower (approx. R2.2M). The control area classified as immediate priorities will cost approximately R6.0M to clear.

Table 25: Estimated cost to treat priority clearing sites.

Priority	Dense	Moderate	Sparse	Total
Immediate				
Initial clearing	R 1,437,643	R 939,496	R 453,267	R 2,830,405
1st follow-up	R 977,709	R 465,690	R 289,121	R 1,732,520
2nd follow-up	R 483,628	R 267,179	R 157,617	R 908,424
3rd follow-up	R 241,814	R 133,589	R 78,809	R 454,212
Total	R 1,966,825	R 1,233,305	R 789,683	R 5,925,561
Short-term				
Initial clearing	R 745,195	R 546,345	R 350,222	R 1,641,763
1st follow-up	R 506,791	R 270,813	R 223,393	R 1,000,996
2nd follow-up	R 250,686	R 155,373	R 121,785	R 527,844
3rd follow-up	R 125,343	R 77,686	R 60,893	R 263,922
Total	R 1,628,015	R 1,050,217	R 756,292	R 3,434,525
Medium-term				
Initial clearing	R 988,562	R 306,008	R 194,926	R 1,489,495
1st follow-up	R 672,299	R 151,682	R 124,336	R 948,316
2nd follow-up	R 332,555	R 87,024	R 67,783	R 487,362
3rd follow-up	R 166,278	R 43,512	R 33,891	R 243,681
Total	R 2,159,694	R 588,226	R 420,936	R 3,168,856
Long-term				
Initial clearing	R 2,171,368	R 435,296	R 128,447	R 2,735,111
1st follow-up	R 1,476,699	R 215,768	R 81,931	R 1,774,398
2nd follow-up	R 730,455	R 123,792	R 44,666	R 898,913
3rd follow-up	R 365,228	R 61,896	R 22,333	R 449,457
Total	R 4,743,750	R 836,752	R 277,377	R 5,857,879
Grand Total	R 10,498,284	R 3,708,500	R 2,244,288	R 18,386,820

4.1.3. Estimated clearing effort

The estimated effort to clear the study area of alien plants was calculated using figures from Marais (2000) (see Table 26). Substantially greater effort is required to clear dense stands of rooikrans in comparison to that required to clear sparse stands.

Table 26: Effort required to clear invasive *Acacia spp.* (adapted from Marais 2000).

Density	Effort (person days / ha)	Density category	Ave. Effort (person days /ha)
0-1%	0.7	Sparse (<25%)	4.3
1-5%	1.8		
5-25%	10.5		
25-50%	11.3	Moderate (25-75%)	16.5
50-75%	21.8		
75-100%	29.9	Dense (>75%)	29.9

The following assumptions were made in calculating clearing effort: A person day is one production worker working eight hours per day. Supervisor and / or management personnel were not counted as person days for the purposes of estimating effective amount of production labour. Supervision and management costs are however included in the cost per person day. Clearing teams consist of eight production workers each. This accounts for eight person days per team per day. A production year was assumed to be 195 effective productive days (78% of the workdays available per year) (Marais 2000).

In order to calculate required effort, the effort required for each density class was multiplied by the area occupied by the density classes (see Table 27). The dense stands within the planning area will require approximately eight times more effort to clear than the sparse stands (43 681 person days compared to 5637 person days). The control blocks classified as immediate priorities will require approximately 20 000 person days to clear.

Table 27: Estimated clearing effort (in person days) required to clear the priority clearing sites.

Priority	Dense	Moderate	Sparse	Total
Immediate	11754	5830	2267	19851
Short-term	6092	3390	1752	11235
Medium-term	8082	1899	975	10956
Long-term	17752	2701	642	21096
Total	43681	13820	5637	63138

The magnitude of the alien plant problem is only fully appreciated when considering the time and effort required to address the problem (see Table 28). For example, a single clearing team will require in excess of forty years to complete initial clearing of the study area. This level of response is clearly not appropriate, especially when considering that thickening up of sparse stands and the expansion of alien plants into new areas have not been taken into consideration.

Table 28: Estimated time required to clear the study area, depending on the number of teams employed.

Number of teams & control clocks		Days				Years			
		Dense	Mod	Sparse	Total	Dense	Mod	Sparse	Total
1 team									
	Immediate	1469	729	283	2481	7.5	3.7	1.5	12.7
	Short-term	762	424	219	1404	3.9	2.2	1.1	7.2

Medium-term	1010	237	122	1370	5.2	1.2	0.6	7.0
Long-term	2219	338	80	2637	11.4	1.7	0.4	13.5
Total	5460	1728	705	7892	28.0	8.9	3.6	40.5
2 teams								
Immediate	735	364	142	1241	3.8	1.9	0.7	6.4
Short-term	381	212	109	702	2.0	1.1	0.6	3.6
Medium-term	505	119	61	685	2.6	0.6	0.3	3.5
Long-term	1110	169	40	1319	5.7	0.9	0.2	6.8
Total	2730	864	352	3946	14.0	4.4	1.8	20.2
3 teams								
Immediate	490	243	94	827	2.5	1.2	0.5	4.2
Short-term	254	141	73	468	1.3	0.7	0.4	2.4
Medium-term	337	79	41	457	1.7	0.4	0.2	2.3
Long-term	740	113	27	879	3.8	0.6	0.1	4.5
Total	1820	576	235	2631	9.3	3.0	1.2	13.5
4 teams								
Immediate	367	182	71	620	1.9	0.9	0.4	3.2
Short-term	190	106	55	351	1.0	0.5	0.3	1.8
Medium-term	253	59	30	342	1.3	0.3	0.2	1.8
Long-term	555	84	20	659	2.8	0.4	0.1	3.4
Total	1365	432	176	1973	7.0	2.2	0.9	10.1

4.1.4. Control methods

Sustainable management of invasive plants demands the integration of biological, chemical and mechanical control options, along with various forms of cultural control including, for example, prescribed burning and restoration programmes (Richardson & Kluge 2008).

Mechanical control options include the physical felling or uprooting of plants, their removal from the site, often in combination with burning. In the control of alien acacias and other aliens that accumulate large stores of hard-coated seeds in the soil, burning is a useful method for reducing their seed bank via triggering mass germination and mortality (Holmes *et al.* 2005). When fire occurs within the study area, follow-up operations should take place to control emerging seedlings (within two years of the fire). Fire stimulates the germination of indigenous seeds and may destroy alien seeds. Where unplanned fires occur without prior clearing, it is frequently still cost effective to conduct follow-up operations within two years. This serves the same purpose as normal follow-up operations as it prevents regenerating seedlings from setting seed. In addition, the reduction of the vegetation density that results from the fire reduces the difficulty of walking and makes the invasive alien plants more visible. All areas that have been burnt by wildfires should therefore be inspected within 6 months to determine whether follow-up operations are appropriate. Mechanical control is labour-intensive and thus expensive to use in extensive and dense infestations, or in remote or rugged areas (van Wilgen *et al.* 2000).

Herbicides are not required for rooikrans, which rarely coppices on felling. Herbicides should be applied to Port Jackson to prevent sprouting of cut stumps. However, there are legitimate concerns over the use of herbicides in terms of potential environmental impacts (see Hobbs & Humphries 1995). Although newer herbicides tend to be less toxic, have shorter residence times, and are more specific, concerns over detrimental environmental impacts still remain. The use of chemical control is often governed by legislation, and the effective and safe use of herbicides requires a relatively high level of training; both of these factors can restrict the use of chemical control on a large scale (van Wilgen *et al.* 2000).

Biological control (or "biocontrol") involves using species-specific insects or other invertebrates, and diseases, from the alien plant's region of origin (see Hobbs & Humphries 1995). Most invasive alien plants show no "weedy" behaviour in their natural ranges - their ability to grow vigorously and produce huge amounts of seeds is kept in check by a host of co-evolved organisms. Some species, when transported to a new region without the attendant enemies, grow more vigorously and produce many more seeds than in their native ranges, and become aggressive invaders. Biocontrol aims to reduce the effects of this phenomenon, and to achieve a situation where the formerly invasive alien plant becomes a non-invasive naturalised alien. Biological control has many potential benefits, including its potential cost-effectiveness, and the fact that it is (usually) environmentally benign (van Wilgen *et al.* 2000).

4.1.5. Guidelines for clearing

The highest priority for clearing should always be given to follow-up operations. Before commencing with any new clearing operations, follow-up clearing should be conducted on any area that has already been cleared of alien plants. If allowance is not made for follow-up operations, the money initially invested will be wasted and the invasive alien plants will rapidly return to previous levels of infestation. Only once provision has been made for follow-up clearing, should additional areas be selected for initial clearing.

The timing of follow-up operations should never exceed the juvenile period of the plants (the time between germination / sprouting and the production of seed) (Cowling 1997). Thus, all areas that have been cleared must be followed up before any regenerating plants can set seed. The juvenile period for rooikrans is between 2-3 years, although it is recommended that follow-up operations occur before this (within a year), as young plants can be more easily eradicated (i.e. can be hand-pulled) than older plants.

The second highest priority for clearing, after follow-up operations, should then be given to sparse category infestations (individual plants or open stands) within the immediate priority clearing blocks. Only once the required follow-up operations have taken place and all the areas categorised as sparse have been cleared in these blocks should efforts be directed to the moderate category within these control blocks. Similarly, no effort should be expended on the dense category until follow-up operations are complete and the lower categories have been cleared.

The principle of working in relatively lightly infested areas first obligates that workers are spaced further apart than they would be in more heavily infested areas. This increases the logistical complexity of operations and requires a relatively high management to worker ratio.

4.2 Obstacles to implementation

The following factors lead to poor alien plant control in landscapes that are used primarily for the amenity value: financial constraints, a poor awareness of the weeds problem, absenteeism and time constraints and landowner values that lead them to be unconcerned with weeds (see Klepeis *et al* 2009).

4.2.1. Financial constraints

The major constraint preventing the institution of control measures is likely to be a financial one. Landowners may not have the funds to invest in alien plant control operations, or they may have the funds but be disinclined spend them on control operations.

Conservancy stakeholders were questioned on who should fund alien plant control measures. Most felt strongly that government and landowners should share the responsibility. Respondents placed lower responsibility on donor agencies for funding alien plant control (see Table 29).

Table 29: Respondents' perception of who should fund alien plant control

Funder	Score	No 5's
Landowners	163	22
Government	189	28
Donors	120	7

Support has already been obtained for indirect control methods, such as biological control, on the conservancy. In 2005, the 500 individuals of rooikrans seed weevil *Melanterius servulus* were donated by the Plant Protection Research Institute for release on the conservancy. However, the possibility of obtaining support from government-based alien control programmes for direct control methods, such as mechanical clearing, appears to be small. Attempts to obtain support from the Working for Water and the Working for Wetlands Programme were unsuccessful (Elizabeth Rautenbach *pers comm.* 2008). This may be in part because of previous negative experience that these programmes have had in dealing with private landowners. For example, landowners have elsewhere not met their obligation to conduct maintenance clearing after initial clearing and follow-up operations have been conducted by the Working for Water Programme on their properties (Patrick Marsh *pers comm.* 2005). In addition, several attempts to obtain financial support from donor agencies that support conservation projects were unsuccessful. Even though these applications pitched as being linked to job creation opportunities for neighbouring disadvantaged communities, it is possible that donors were reluctant to fund projects on land perceived to be owned by affluent landowners.

It thus seems likely that landowners will be required to finance control operations themselves. A possible mechanism would be for the conservancy to levy a monthly fee on its membership for control operations. These funds could then be pooled to employ clearing teams for the conservancy. The efficiencies that could be gained from the pooling of resources in this manner include those gained from sharing equipment costs, training requirements, management costs and transport costs. However, the landowners' willingness to contribute towards a central clearing team that would not be focussed exclusively on their own properties is likely to be low, especially without external pressure to address the problem. In this regard, Hershdorfer *et al.* (2007) found that managing alien plants requires locally enforceable alien plant control regulations.

The legal mechanism for exerting this pressure already exists in form of the Conservation of Agricultural Resources Act. The Act imposes various obligations on land users on whose land species listed as invasive occur. Land users are compelled to control such plants using measures that are appropriate for the species and ecosystem concerned. Refusal or failure to comply with these obligations constitutes an offence. In addition, if a direction has been issued to a particular land user to comply with certain control measures, and the land user fails or refuses to do so, he is guilty of an offence. However, directives have not been issued under this legislation to any landowner in the area, and despite the commencement of CARA and its regulations over twenty years ago, there has not been one successful conviction under this legislation (Paterson 2006).

Le Maitre *et al.* (2004) state that innovative approaches are needed to help landowners realize that the expenditure on alien plant control is in their own best interest.

4.2.2. *Awareness of the problem of alien plants*

Paterson (2006) states that perhaps the greatest problem in addressing alien plant invasion is lack of public awareness regarding the nature and extent of the problem. Landowners are not likely to implement control measures if they are not aware of the negative impacts of alien plants, and of the consequences of not addressing the problem. When stakeholders were questioned about the how they perceive the threat of alien plants, the majority (42) indicated that alien plants are a problem while only one indicated that alien plants are not a problem. However, the question posed was not structured in a manner that enabled the determination of the perceived magnitude of the problem and landowners' attitudes towards addressing the problem.

4.2.3. *Absenteeism*

The fact that most of the landowners are non-resident is likely to contribute to lack of action in addressing the alien plant problem on the conservancy. Klepeis *et al.* (2009) notes that absentee landholders tend to be less well informed about local environmental problems. In addition, these authors state that absentee landholders are less likely to implement weed control measures and participate in neighbourhood and community-scale responses.

In addition, absenteeism erodes the potential for the development of the social capital required for the implementation of a cooperative approach to alien plant management conservancy. Klepeis *et al.* (2009) states that the social capital that functions when rural lands are used by full time primary producers is ill-suited to a landscape increasingly dominated by residents who value the land for its amenity, and whose ideas about land and rural life are diverse.

4.2.4. Distribution of land use on the properties

Because most of the landowners of the conservancy only utilise small sections of their properties (the sections immediately adjacent to the coastline) intermittently for recreation, there may be disinterest in managing the remaining sections of the properties (Klepeis *et al.* 2009).

5. CONCLUSION

In order to successfully address the threat posed by alien invasive plants to the St Francis Conservancy, a new approach is required. This approach will need to be characterised by commitment from landowners, and their willingness to cooperate and share resources.

Successful cooperation between landowners is difficult to achieve, requiring substantial time and effort to initiate and maintain. However, the potential benefits of landowner cooperation promise to outweigh their costs and, besides expanding the possibilities for alien invasive plant control, will also serve to strengthen communities (VanBebber 2003).

Landowner cooperation in alien invasive plant control should be guided by a strategy that makes the best use of limited resources and satisfies the objectives of the landowners, while also preserving the values of the area. It is hoped that the current study will serve this purpose and also provide the foundation for fair collective decision-making.

Whatever form they take, it is imperative that alien invasive control measures are implemented as a matter of urgency. Higgins *et al.* (2000) showed that delaying the initiation of clearing operations has a strong effect on both the total cost of clearing and on the impacts that alien plants have on ecosystems. Already, adequate responses to the alien invasive plant problem on the conservancy will require substantial inputs of time and funds. The situation is likely to become considerably more difficult to contain without timely and effective action.

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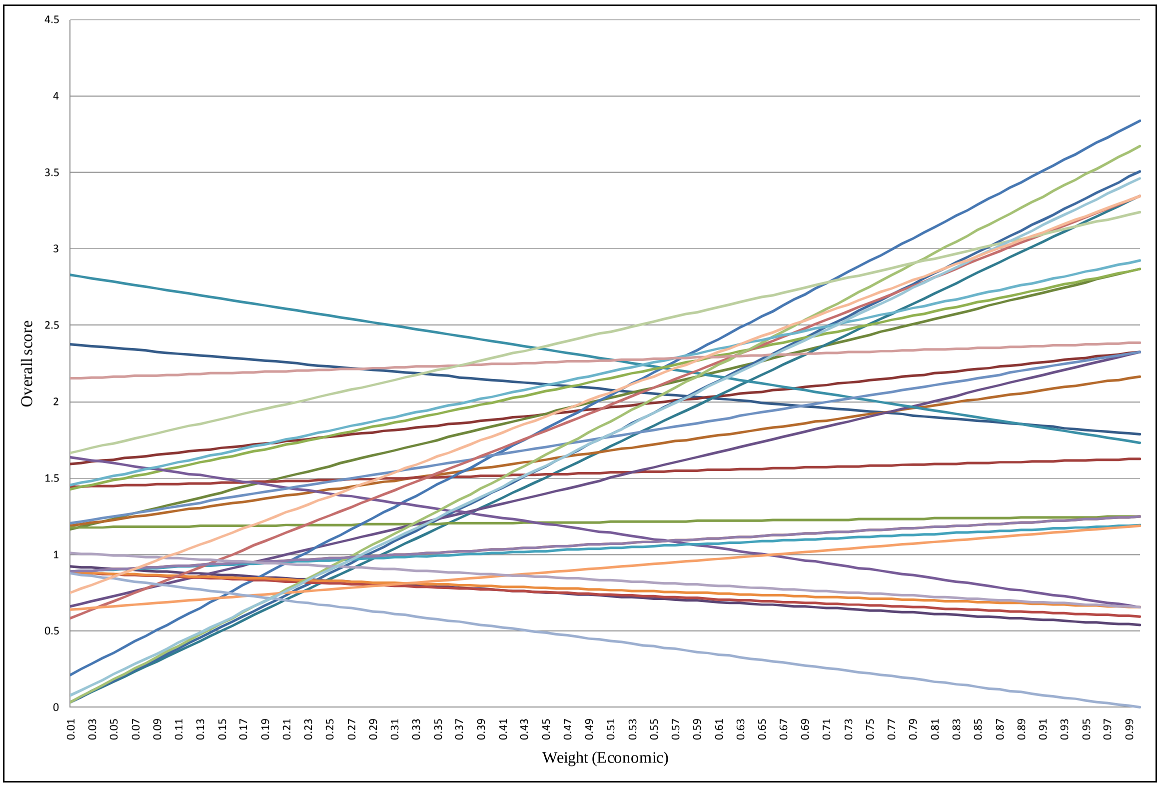
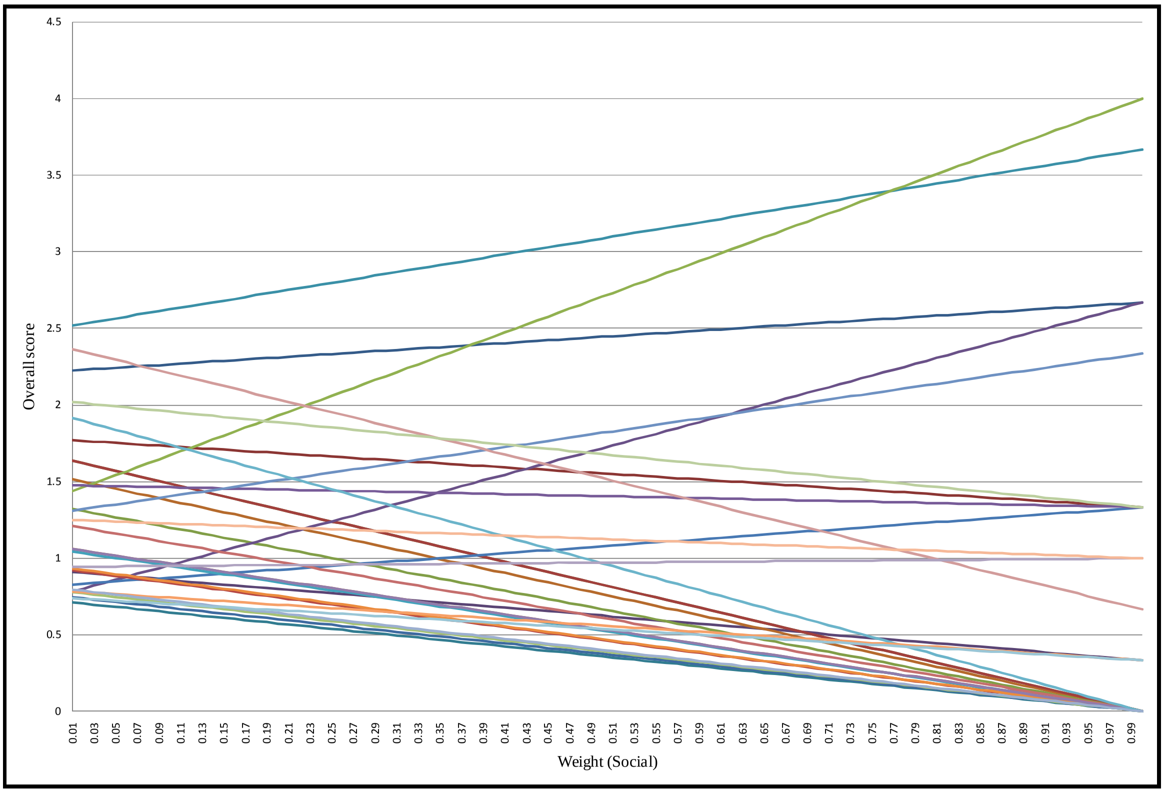
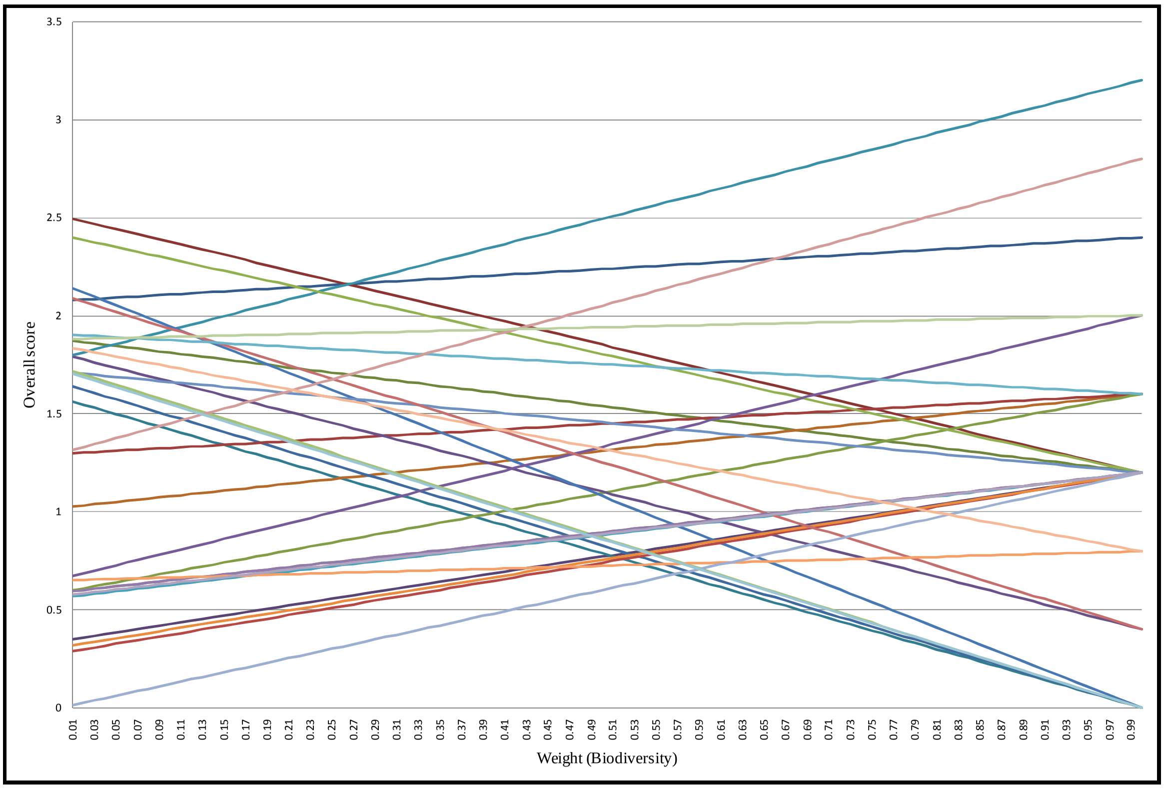
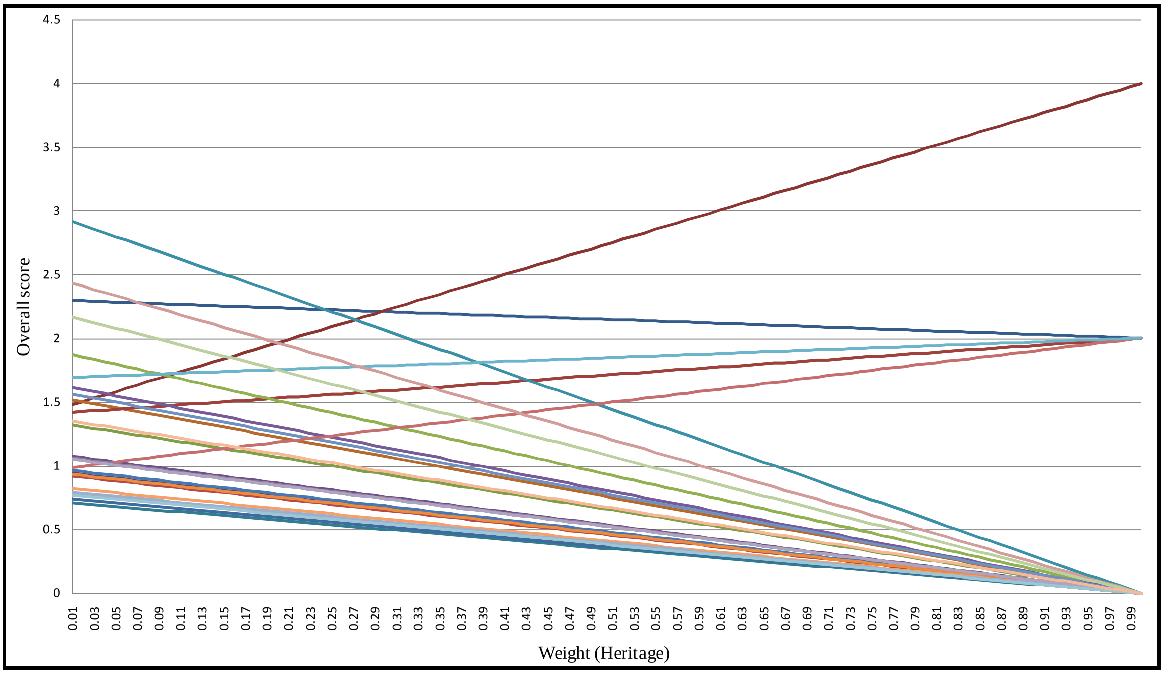
7. APPENDICES

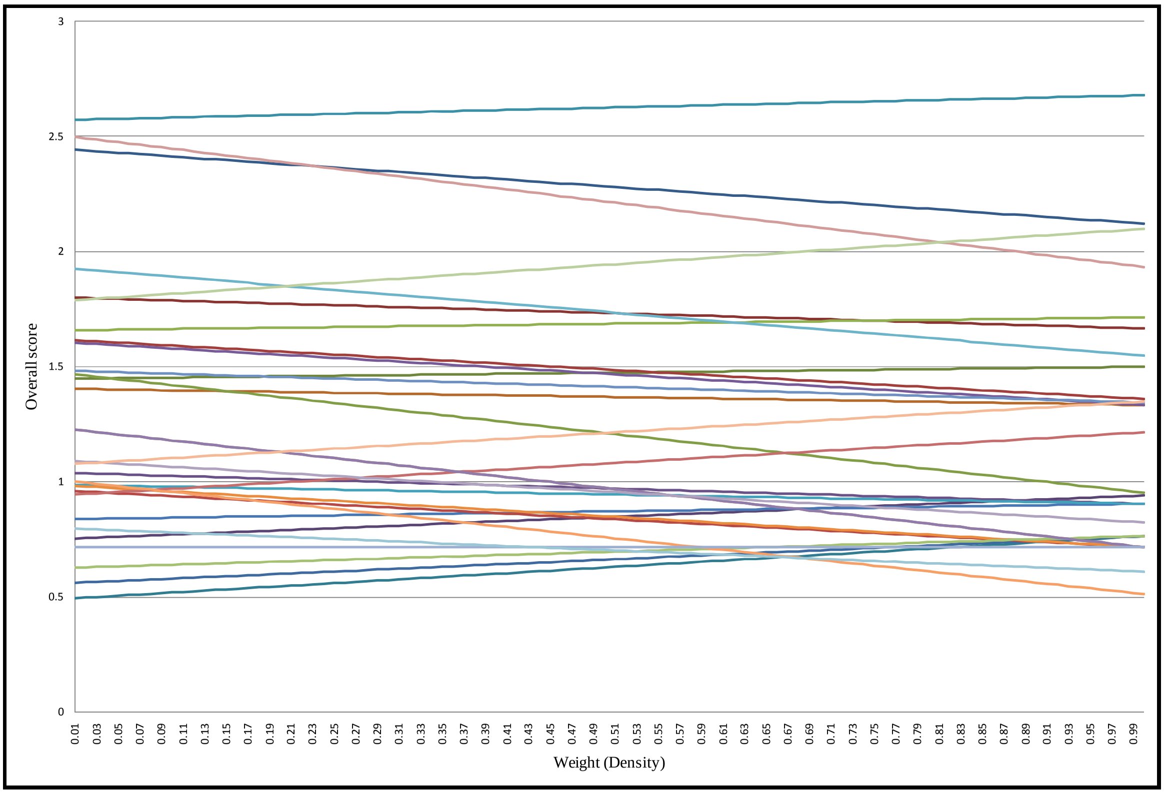
Appendix 1: Sensitivity analysis

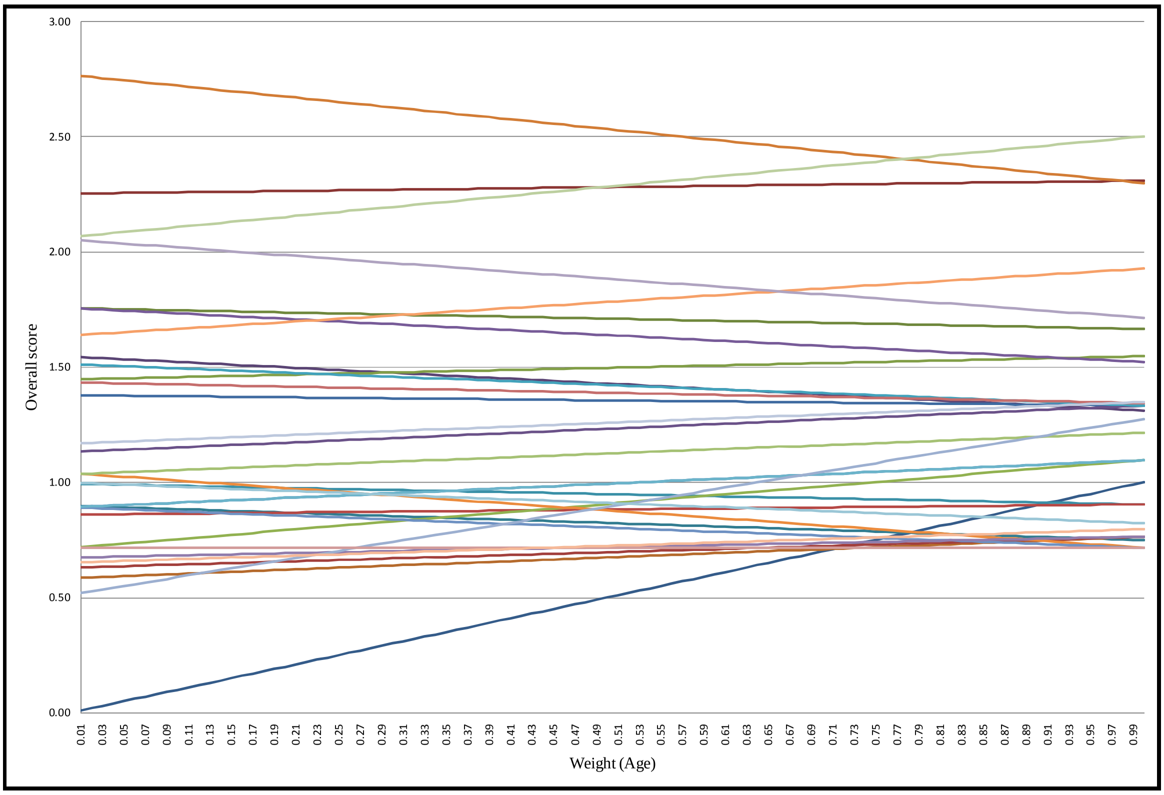


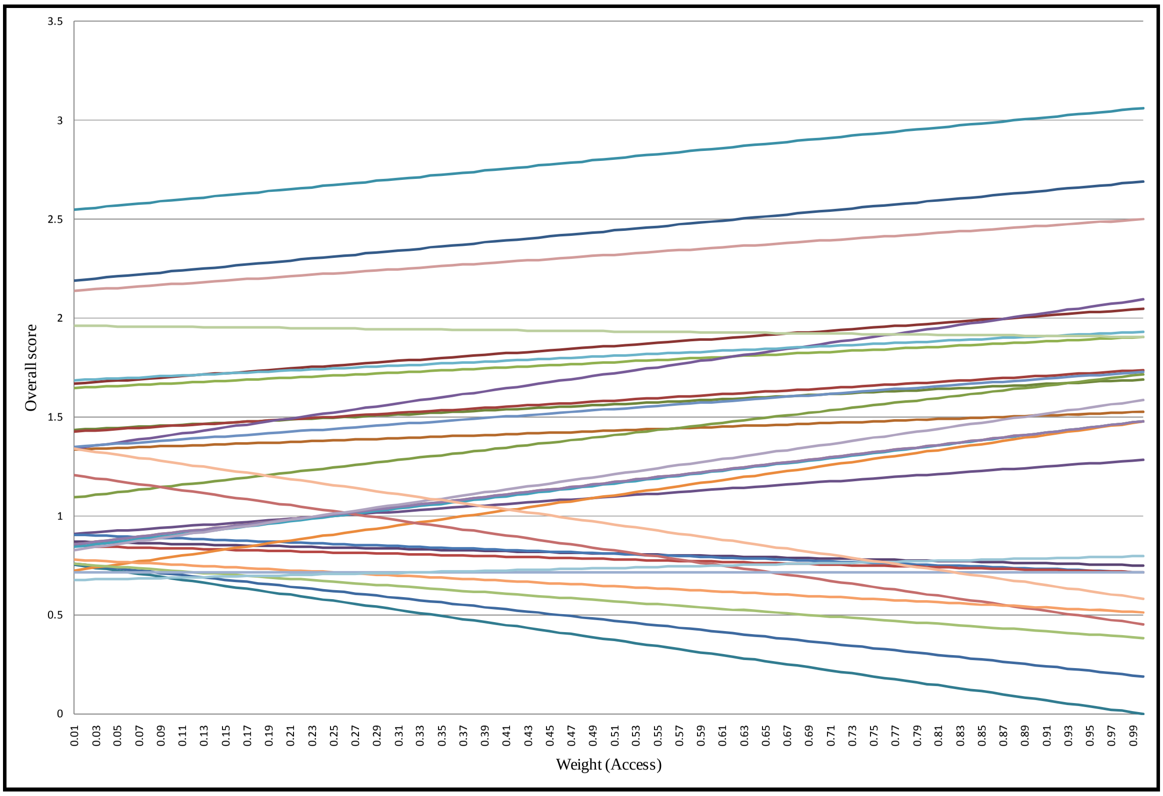


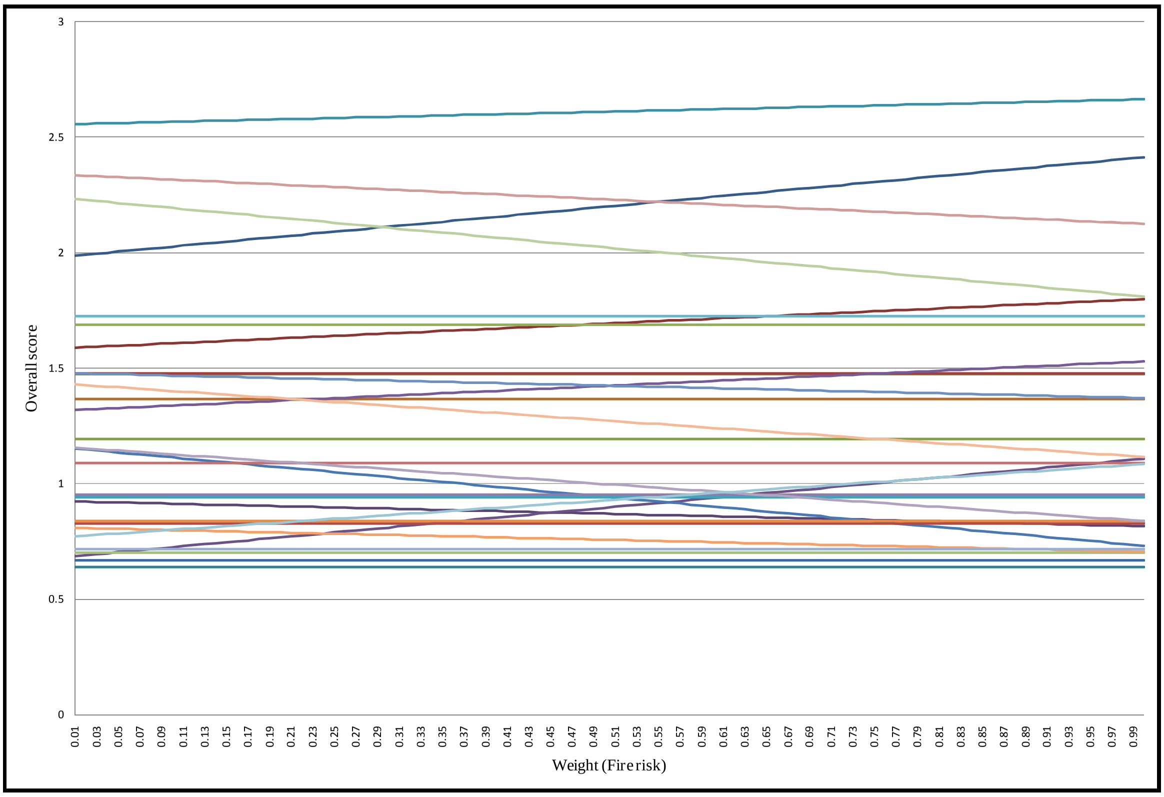

Overall score
Weight (Biodiversity)

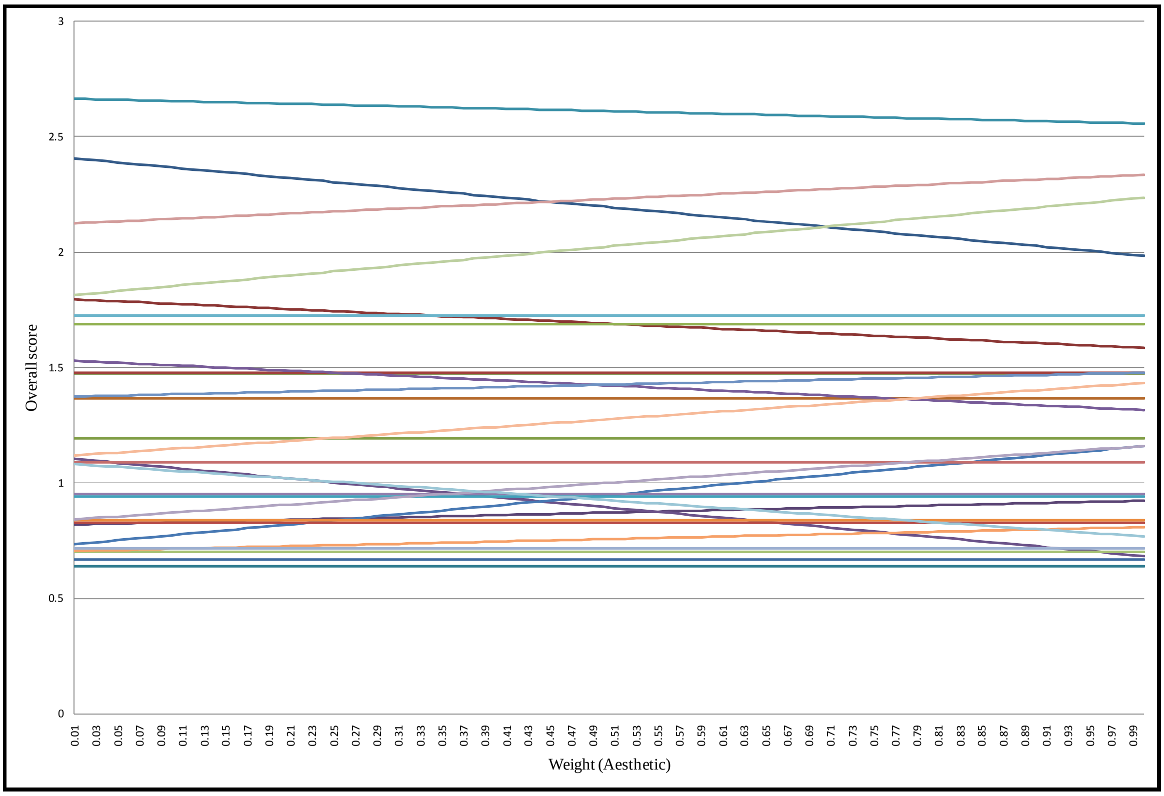

Overall score
Weight (Heritage)


Overall score
3
2.5
2
1.5
1
0.5
0
Weight (Density)

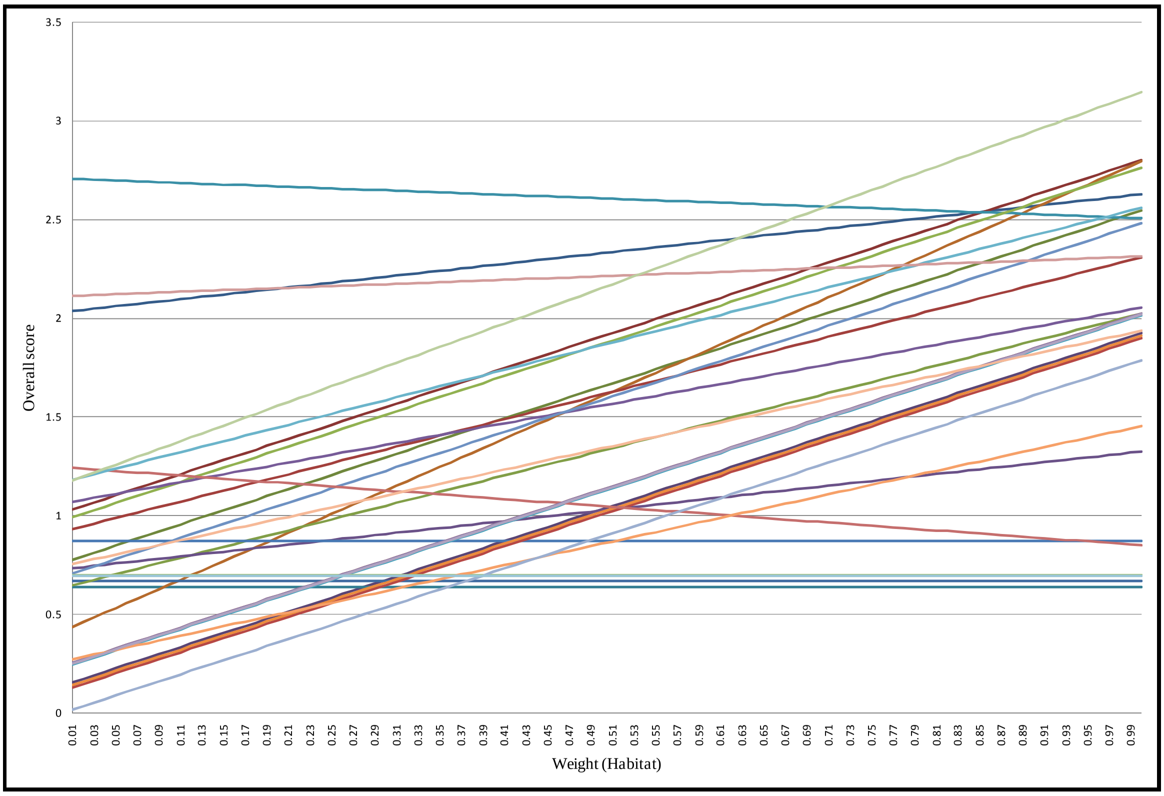

Overall score
3.00
2.50
2.00
1.50
1.00
0.50
0.00
Weight (Age)


Overall score
Weight (Access)

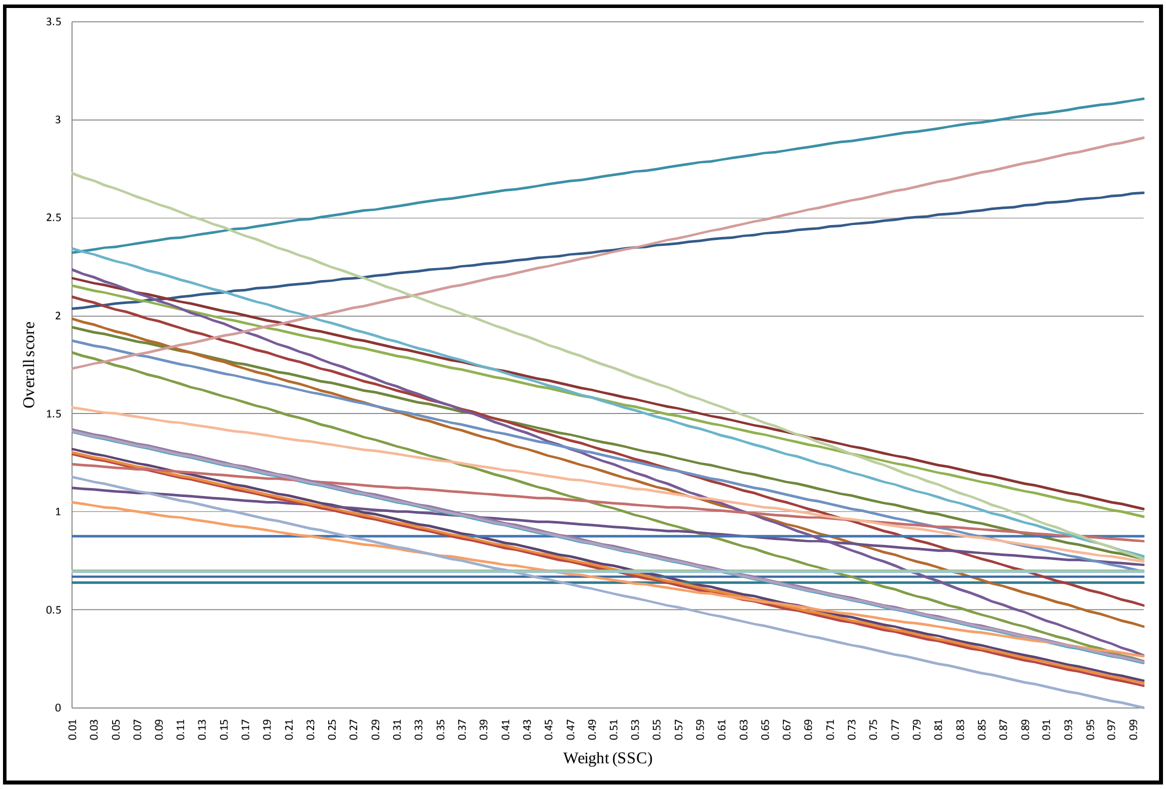

Overall score
Weight (Fire risk)

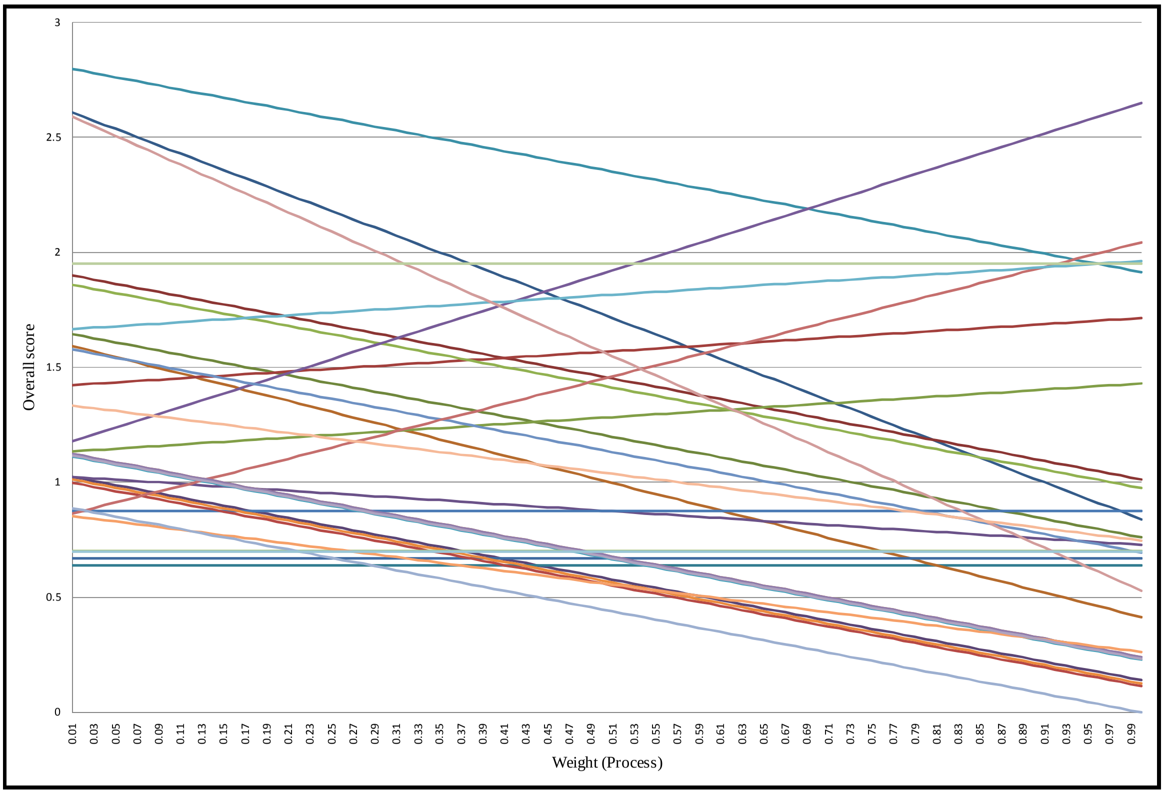


Overall score
Weight (Aesthetic)



Overall score
Weight (Habitat)


Overall score
Weight (SSC)


Overall score
Weight (Process)

Appendix 2: Cost and effort to clear priority areas

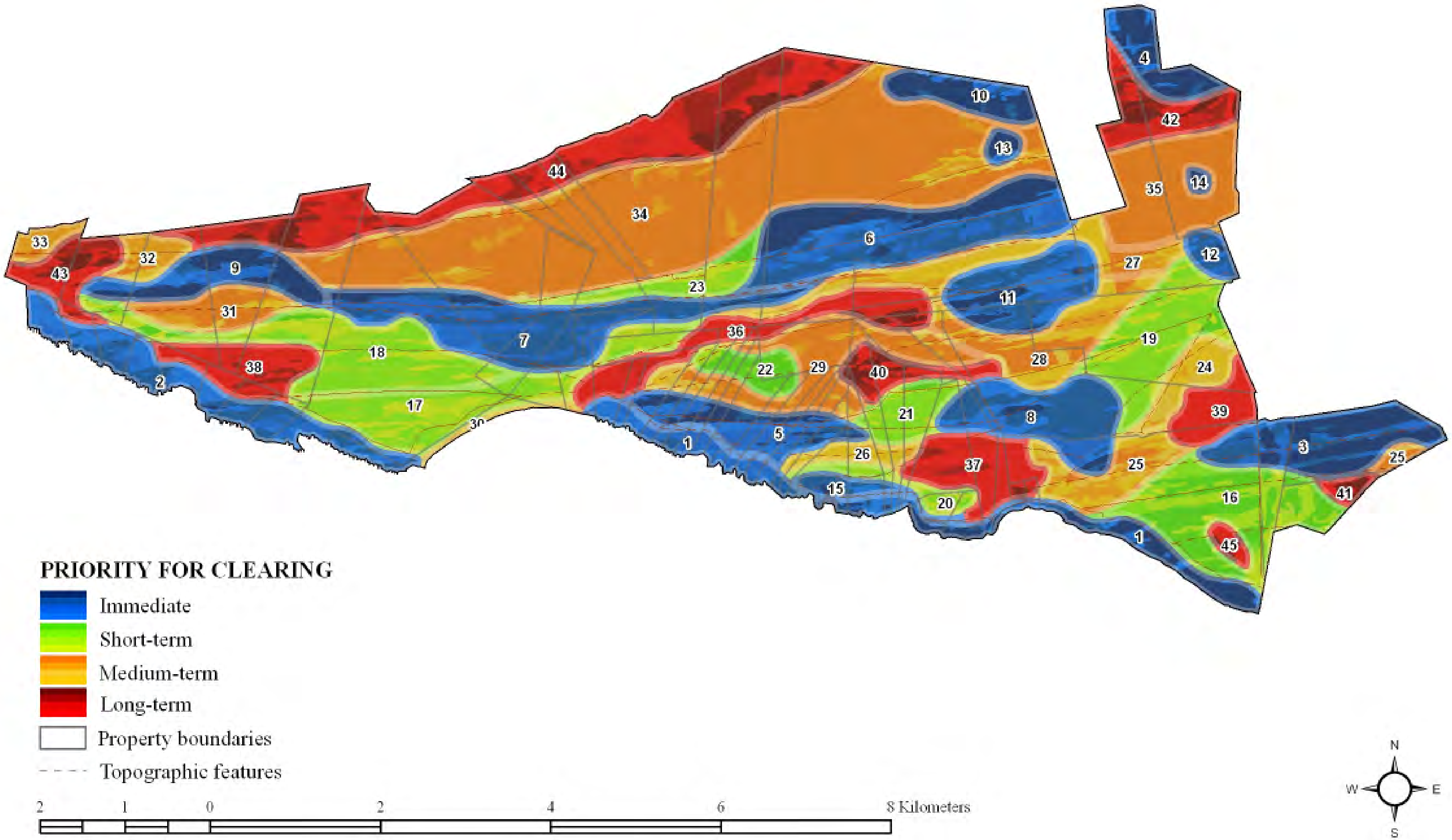


Figure 42: Clearing sites.

Table 30: Initial clearing costs and effort per site

	Extent (ha)					Cost (ZAR)				Effort (person days)			
Site	**Area**	**Clear**	**Dense**	**Mod**	**Sparse**	**Dense**	**Mod**	**Sparse**	**Total**	**Dense**	**Mod**	**Sparse**	**Total**
1	196	78	47	46	25	172807	121987	21851	316644	1413	757	109	2279
2	180	5	71	34	71	259376	89840	61026	410242	2121	557	305	2983
3	162	88	16	25	33	57946	66418	28161	152525	474	412	141	1027
4	72	55	13	1	3	45985	3589	2811	52385	376	22	14	412
5	119	35	39	31	13	144108	82500	11571	238179	1178	512	58	1748
6	274	100	47	53	73	173294	142047	62731	378072	1417	881	314	2612
7	265	26	24	86	129	87675	230067	110848	428591	717	1428	554	2699
8	158	7	46	19	85	168086	51476	73000	292561	1374	319	365	2059
9	113	67	12	27	7	43064	72034	5869	120967	352	447	29	828
10	72	55	7	5	5	24048	13903	4310	42262	197	86	22	304
11	136	18	53	15	50	193746	39878	43230	276853	1584	247	216	2048
12	27	3	4	6	15	14146	14986	12616	41749	116	93	63	272
13	16	12	1	1	1	5027	3558	1218	9804	41	22	6	69
14	8	8				0	0	0	0	0	0	0	0
15	35	3	13	3	16	48333	7213	14024	69571	395	45	70	510
16	216	108	21	54	33	76659	144208	28141	249008	627	895	141	1662
17	189		49	61	79	179121	161578	68445	409144	1464	1003	342	2809
18	217	11	57	45	105	206842	119383	90078	416304	1691	741	451	2882
19	161	9	37	12	102	135465	32743	88243	256451	1108	203	441	1752
20	39		14	8	17	50322	20261	14946	85529	411	126	75	612
21	62	5	8	6	43	29334	16934	36640	82908	240	105	183	528
22	50	40	2	4	4	8878	11798	3067	23743	73	73	15	161
23	68	13	16	15	24	58574	39441	20660	118674	479	245	103	827
24	56	0	36	13	6	133194	35889	5244	174327	1089	223	26	1338
25	105	22	33	18	32	120572	49266	27393	197231	986	306	137	1428
26	35		28	3	4	102986	8275	3614	114876	842	51	18	911
27	131	20	73	15	23	265333	39827	20220	325381	2169	247	101	2518
28	134	50	55	9	20	201133	23772	17452	242358	1644	148	87	1879
29	178	134	7	22	15	25904	58975	13256	98135	212	366	66	644
30	16	1	11	3	1	38845	7157	1181	47183	318	44	6	368
31	71	48	5	9	10	17305	23146	8774	49225	141	144	44	329
32	37	5	7	4	21	25839	10028	18345	54212	211	62	92	365
33	24	4	1	0	19	3249	419	16750	20418	27	3	84	113
34	929	825	14	18	71	52779	49063	61039	162882	432	304	305	1041
35	167	165	0	0	2	1423	189	1656	3268	12	1	8	21
36	137	12	106	18	2	387409	47170	1498	436078	3167	293	7	3468
37	108	2	87	9	10	317054	24583	8796	350433	2592	153	44	2789
38	89		65	12	13	236427	31040	11080	278546	1933	193	55	2181
39	64		62	0	2	226638	228	1314	228180	1853	1	7	1861
40	59	20	31	1	7	112352	2685	6292	121328	919	17	31	967
41	18	8	6	3	0	22725	7274	385	30385	186	45	2	233
42	93	26	35	16	15	129364	42628	13244	185236	1058	265	66	1388
43	74	13	36	11	14	130885	30066	11826	172777	1070	187	59	1316
44	394	60	155	93	85	565145	249035	73529	887710	4620	1545	368	6534
45	16	3	12	0	1	43369	588	482	44439	355	4	2	361
Tot	5769	2163	1462	835	1309	5342768	2227145	1126862	8696775	43681	13820	5637	63138

Table 31: Cost for follow up treatment per site.

	1st Follow-up (ZAR)				2nd Follow-up (ZAR)				3rd Follow-up				Grand
Site	Dense	Mod	Sparse	Total	Dense	Mod	Sparse	Total	Dense	Mod	Sparse	Total	Total
1	117522	60466	13938	191926	58133	34691	7598	100422	29066	17346	3799	50211	342560
2	176396	44532	38926	259854	87255	25549	21221	134025	43627	12775	10611	67013	460891
3	39408	32922	17963	90293	19493	18888	9793	48174	9747	9444	4896	24087	162554
4	31273	1779	1793	34845	15469	1021	977	17468	7735	510	489	8734	61046
5	98004	40894	7381	146279	48478	23462	4024	75964	24239	11731	2012	37982	260225
6	117854	70410	40014	228277	58297	40396	21814	120507	29148	20198	10907	60253	409037
7	59626	114040	70706	244372	29494	65428	38546	133468	14747	32714	19273	66734	444574
8	114311	25516	46564	186391	56545	14639	25385	96568	28272	7319	12692	48284	331243
9	29287	35706	3744	68736	14487	20486	2041	37013	7243	10243	1020	18507	124256
10	16355	6892	2749	25996	8090	3954	1499	13543	4045	1977	749	6771	46310
11	131762	19767	27575	179103	65177	11341	15033	91550	32588	5670	7516	45775	316428
12	9621	7428	8047	25096	4759	4262	4387	13408	2379	2131	2193	6704	45208
13	3419	1763	777	5960	1691	1012	424	3127	846	506	212	1563	10650
14	0	0	0	0	0	0	0	0	0	0	0	0	0
15	32870	3575	8946	45392	16259	2051	4877	23188	8130	1026	2438	11594	80173
16	52134	71481	17950	141565	25788	41011	9786	76585	12894	20505	4893	38292	256442
17	121816	80091	43659	245566	60257	45951	23801	130008	30128	22975	11900	65004	440578
18	140669	59176	57457	257302	69582	33951	31324	134857	34791	16975	15662	67428	459588
19	92127	16230	56287	164644	45571	9312	30685	85568	22785	4656	15343	42784	292996
20	34223	10043	9534	53800	16929	5762	5197	27888	8464	2881	2599	13944	95631
21	19949	8394	23371	51715	9868	4816	12741	27425	4934	2408	6371	13713	92852
22	6038	5848	1957	13842	2987	3355	1067	7408	1493	1678	533	3704	24955
23	39835	19550	13178	72563	19704	11216	7184	38105	9852	5608	3592	19052	129720
24	90582	17790	3345	111717	44807	10206	1824	56837	22403	5103	912	28418	196972
25	81998	24420	17473	123891	40561	14010	9525	64097	20280	7005	4763	32048	220036
26	70039	4102	2306	76446	34645	2353	1257	38255	17322	1177	628	19128	133829
27	180447	19741	12898	213086	89259	11326	7031	107617	44629	5663	3516	53808	374511
28	136786	11783	11132	159702	67662	6761	6069	80491	33831	3380	3034	40246	280439
29	17617	29233	8456	55305	8714	16772	4610	30096	4357	8386	2305	15048	100449
30	26417	3548	753	30718	13067	2035	411	15514	6534	1018	205	7757	53989
31	11769	11473	5597	28839	5821	6582	3051	15455	2911	3291	1526	7727	52021
32	17572	4971	11702	34245	8692	2852	6379	17923	4346	1426	3190	8962	61130
33	2209	208	10684	13101	1093	119	5824	7037	546	60	2912	3518	23656
34	35894	24320	38935	99148	17755	13953	21226	52934	8878	6976	10613	26467	178548
35	968	93	1056	2118	479	54	576	1108	239	27	288	554	3780
36	263468	23381	956	287805	130326	13415	521	144261	65163	6707	260	72131	504197
37	215621	12185	5611	233417	106658	6991	3059	116708	53329	3496	1529	58354	408479
38	160789	15386	7067	183242	79535	8827	3853	92215	39767	4414	1926	46107	321564
39	154132	113	838	155083	76242	65	457	76764	38121	32	229	38382	270228
40	76408	1331	4013	81752	37795	764	2188	40747	18898	382	1094	20373	142872
41	15455	3606	246	19306	7645	2069	134	9847	3822	1034	67	4924	34077
42	87977	21130	8448	117555	43518	12123	4606	60247	21759	6061	2303	30123	207925
43	89012	14903	7543	111458	44030	8550	4112	56693	22015	4275	2056	28346	196498

44	384343	123442	46902	554686	190117	70822	25569	286508	95058	35411	12784	143254	984447
45	29494	291	307	30093	14590	167	168	14924	7295	84	84	7462	52480
Tot	3633497	1103954	718780	5456230	1797325	633367	391851	2822543	898662	316684	195926	1411272	9690045

Appendix 3

Table 32: An example of an annual budget for supporting an alien plant eradication team (Derek Cook *pers comm.* 2005).

Subject	Description	Rate		Based on	Total	1 Year
Capital	Chainsaw and accessories	R 8,460.00	Per Year	2 chainsaws	R 8,460.00	R 8,460.00
	Hand Tools/Safety Equipment & parts	R 15,842.00	Per Year		R 15,842.00	R 15,842.00
					R 24,302.00	R 24,302.00
Manager	Salary	R 300.00	Per day	R150 \ hour	R 300.00	R 75,600.00
	Vehicle	R 125.00	Per day	25km \ day	R 125.00	R 31,500.00
	Fuel	R 21.00	Per day	25km \ day	R 21.00	R 5,292.00
	Service	R 5.00	Per day	25km \ day	R 5.00	R 1,260.00
					R 451.00	R 113,652.00
Staff	Salary – Foreman	R 100.00	Per day	1 individual	R 100.00	R 25,200.00
	Salary – Chainsaw Operator	R 80.00	Per day	2 individuals	R 160.00	R 40,320.00
	Salary - Labour	R 55.00	Per day	6 individuals	R 330.00	R 83,160.00
					R 590.00	R 148,680.00
Consumables	Fuel/Lubrication/Parts – X2 Chainsaws					
Petrol	5L per day @ R6.00 per litre	R 30.00	Per day	2 chainsaws	R 60.00	R 15,120.00
Oil -2stroke	0.100L per day @ R17.00 per litre	R 1.70	Per day	2 chainsaws	R 3.40	R 856.80
Cutter Bar Oil	500 ml per day @ R7.00 per litre	R 3.50	Per day	2 chainsaws	R 7.00	R 1,764.00
Chains	6 a year per chainsaw @ R250.00 ea	R 1,500.00	Per Year	2 chainsaws	R 3,000.00	R 3,000.00
Bars	3 a year per chainsaw @ R250.00 ea	R 750.00	Per Year	2 chainsaws	R 1,500.00	R 1,500.00
					R 4,500.00	R 22,240.80
Grand Total						R 308,874.80