The distribution and abundance of herpetofauna on a Quaternary aeolian dune deposit: Implications for Strip Mining

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ABSTRACT

Exxaro KZN Sands is planning the development of a heavy minerals strip mine south of Mtunzini, KwaZulu-Natal, South Africa. The degree to which mining activities will affect local herpetofauna is poorly understood and baseline herpetofaunal diversity data are sparse. This study uses several methods to better understand the distribution and abundance of herpetofauna in the area. I reviewed the literature for the grid squares 2831DC and 2831 DD and surveyed for herpetofauna at the study site using several methods. I estimate that 41 amphibian and 51 reptile species occur in these grid squares. Of these species, 19 amphibian and 39 reptile species were confirmed for the study area. In all, 29 new unique, grid square records were collected.

The paucity of ecological data for cryptic fauna such as herpetofauna is particularly evident for taxa that are difficult to sample. Because fossorial herpetofauna spend most of their time below the ground surface, their ecology and biology are poorly understood and warrant further investigation. I sampled fossorial herpetofauna using two excavation techniques. Sites were selected randomly from the study area which was expected to host high fossorial herpetofaunal diversity and abundance. A total of 218.6 m^3 of soil from 311 m^2 (approximately 360 metric tons) was excavated and screened for herpetofauna. Only seven specimens from three species were collected. All were within approximately 100 mm of the surface even though some samples removed soil 1 m below the surface. There was no detectable difference in fossorial herpetofaunal density (individuals.m^{-2}) between methods or from areas under different land uses. Neither soil compaction nor land use nor soil texture predicted fossorial herpetofaunal density or abundance. The data suggest that fossorial herpetofauna occur at extremely low densities in the area. This finding has implications for population estimates and conservation measures for these species.

In order to better understand the effects of land use on herpetofaunal diversity, I used sample-based rarefaction curves to compare the diversity of the herpetofaunal species assemblages occurring in each of the four main land uses on the study site. Forest areas hosted significantly higher diversity than grasslands and the two agricultural mono-cultures, *Eucalyptus* and sugarcane plantations. Additionally I demonstrated empirically that riparian woodlands host higher species richness and herpetofaunal abundance than non-riparian areas. Potential reasons for the apparently suppressed diversity of these areas include the use of pesticides and/or herbicides, harvesting regimes, and the reduction in habitat heterogeneity. The potential value of riparian woodlands as refugia and corridors that could facilitate recolonisation of revegetated areas post-mining is discussed.

Negative influences of mining activities on local herpetofauna are of particular interest given the potential and verified presence of several threatened taxa in the area including *Bitis gabonica*, *Python natalensis*, *Afrixalus spinifrons*, *Hemisus guttatus* and *Hyperolius pickersgilli*. These, as well as the “conservation needy” species proposed in a specialist report on the impacts of the mine on local herpetofauna are discussed in the light of my fieldwork. Mitigatory measures are required to reduce the negative impacts likely to be experienced by certain threatened taxa. I discuss a proposal for the development of a wetland reserve targeting, among other amphibian species, *H. pickersgilli*.

DECLARATION

I declare that this dissertation is my own, unaided work unless specifically acknowledged in the text. It has not been submitted previously for any degree or examination at any other university, nor has it been prepared under the aegis or with the assistance of any other body or organization or person outside of the University of the Witwatersrand, Johannesburg, South Africa.

Bryan Maritz July, 2007

DEDICATION

This work is dedicated to the loving memory of my late parents Phyllis and Hendry Maritz. I will be forever grateful for the love and opportunities you provided for me.

Acknowledgments

Special thanks go to the Jansen, Prinsloo, Brooks and the extended Best family for their hospitality. You all contributed to me enjoying my time in Mtunzini, a critical factor in the success of this project. Sean Jansen helped sieve soil, install and check traps and collect specimens. Your company and companionship are greatly appreciated.

Exxaro KZN Sands provided funding, logistic support and access to land. Special thanks go to Rob Hattingh for providing the opportunity to conduct this research as well as his insight and thoughts. Support from Ivan Beukes and Jacollette Adam was invaluable. The Exxaro KZN Sands CPC Laboratories performed the Particle Size Distribution analysis. Jonathan Cromhout, Lize Nel, Dan Nel and Derick and Douglas Saint provided logistic help and access to land during fieldwork.

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All protocols used during this research were passed by the Animal Ethics Screening Committee of the University of the Witwatersrand under permit 2005/58/1. Ezemvelo KZN Wildlife permitted the research under permit 2541/2005.

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Chapter 1: Introduction

1.1 Mining and conservation in South Africa

The South African National Environment Management Biodiversity Act (NEMBA) was promulgated in 2004 with the objective of providing for the management and conservation of biological diversity within South Africa. With ever increasing degrees of habitat transformation throughout the country (Driver et al., 2005), this legislation is becoming increasingly important, particularly with respect to industry and development. The need for greater sensitivity to environmental considerations has also been recognised for mining, and minimum standards for environmental responsibility are now incorporated into the Mineral And Petroleum Resources Development Act, 2002 (Act No. 28 of 2002), as part of its developmental regulations.

Mining plays a critically important role in the social and economic development of many countries (United Nations, 2002). Additionally, natural resources play an integral part of modern lives and, as human populations increase globally, there is an increasing demand for minerals, with global pressure for "environmentally friendly" extraction techniques (Tilton, 2002). As a result, the integration of mining policies and protocols with biodiversity conservation has become increasingly necessary and has resulted in the initiation of multi-stakeholder dialogues on the topic (e.g., the workshop on mining and biodiversity, initiated by the IUCN and International Council on Mining and Metals – ICMM; Pretoria, 2005) and the production of documents highlighting case studies from around the world (e.g., IUCN and ICMM, 2004).

While global biodiversity conservation efforts continue to intensify (Novacek and Cleland, 2001), efforts to assess and understand the constituent units of the biodiversity are often overlooked. Particular attention needs to be given to taxa that have previously been overlooked, such as the herpetofauna, as well as taxa and ecosystems that are poorly understood. This study provides an assessment of the potential impacts of a proposed mine on local herpetofaunal populations.

Exxaro KZN Sands (formerly Ticor-South Africa) has begun the development of a heavy minerals strip mine near Fairbreeze, northern KwaZulu-Natal, South Africa. The mine will be extracting Titanium-rich heavy minerals including Ilmenite ($FeTiO_2$), Rutile (TiO_2), Zircon ($ZrSiO_4$) and Leucoxene from the aeolian sand deposits (Norman and Whitfield, 2006). The Fairbreeze mine will be the second of two such heavy mineral mines in the area. The other, Hillendale, is approximately 20 km north of Fairbreeze and has been operational since 2001.

1.2 Study site

All investigations were conducted at locations on and around the Exxaro KZN Sands Fairbreeze C Extension mine, immediately south of the town of Mtunzini, KwaZulu-Natal, South Africa (28.961 S; 31.749 E). The area forms part of the Maputaland-Pondoland-Albany Biodiversity Hotspot (Mittermeier et al., 2005). Biodiversity hotspots are a construct of the conservation organization, Conservation International (www.conservation.org), and serve to focus conservation efforts on areas where they are most needed and will produce the best results for the allocated funds (Myers et al., 2000). Historically coastal forest and grassland, the area was transformed by agriculture, initially to sugarcane, and subsequently to *Eucalyptus* plantations in the 1930s (van der Elst et al., 1999) (Fig.1.1). These crops dominate today but are interspersed with "semi-natural" forested areas. The area is underlain by Quaternary sands deposited approximately 350 000 to 400 000 years ago (Maud and Botha, 2000). Climate in the area is sub-tropical receiving more than 1200 mm rain per annum (Shulze, 1997).

Currently, habitats in the area show varying levels of disturbance. While large tracts of land have been converted to sugarcane and timber production, areas of "semi-natural" woodland habitat remain. Less obviously, agricultural activities in recent years have probably reduced mean annual runoff in the two major catchments in the area, the Amanzinyama and Siyayi River Catchments (Shepherd et al., 2004; van der Elst et al., 1999), potentially disturbing habitats.

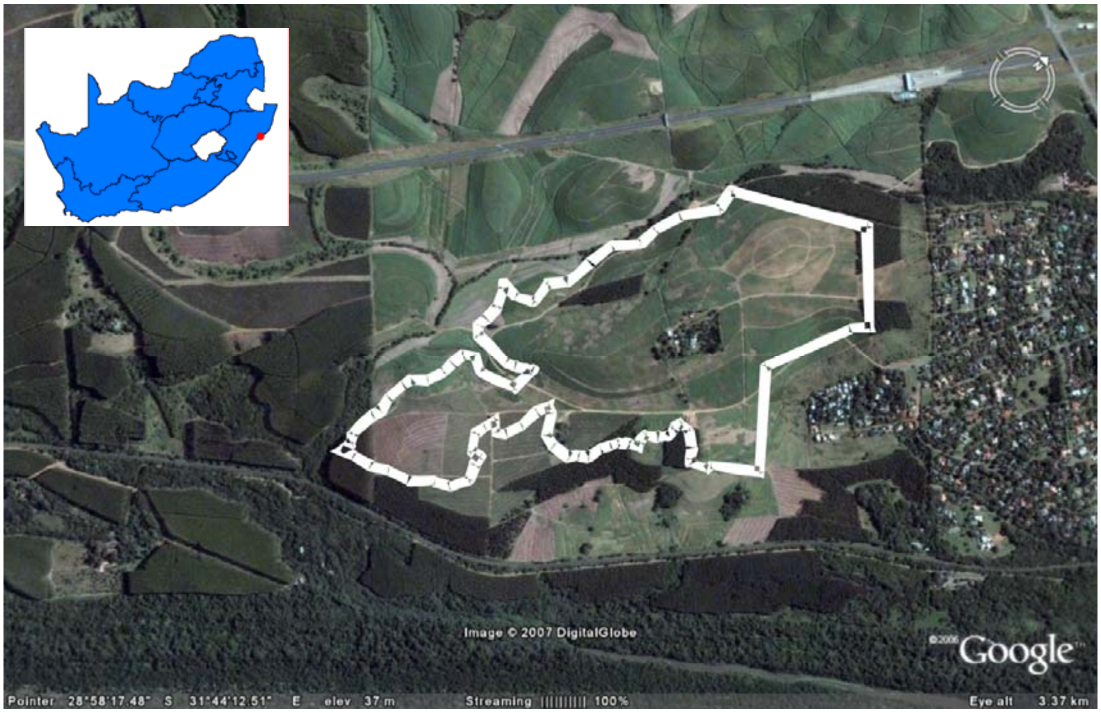

Figure 1.1: Aerial view of the study area showing area of interest showing approximate position of the Fairbreeze C Ext ore body (white). The town of Mtunzini forms the northern limit of the mining area. Mining will be limited to trasnformed habitat. Image courtesy of Google Earth.

1.3 Mining

The mining process begins with the removal of a layer of topsoil from the surface of the area to be mined. This topsoil is stockpiled for use in the dune reconstruction process. The mineral-rich sand is mined using high-pressure water cannons. The resultant slurry is pumped from the mining area to a processing plant where it undergoes several screening and separation processes (Ticor South Africa, *undated*). Slimes generated during the process are treated with flocculent and pumped to a residue dam so as to recover water to be used during further mining processes.

Run-off associated with mining activities is not likely to be of reduced quality (Shepherd et al., 2004). While total dissolved solids (TDS), minerals and flocculants are likely to increase, Shepherd et al. (2004) indicate that this impact will be localised and ameliorated through the natural filtration and buffering resulting form the stability of the ecosystem. Haigh and Davies-Coleman (1997) argue that the effect a perturbation has on a biotic community cannot be understood until the effects on individual taxa are understood. Given that the effects of increased TDS, minerals and flocculants (and their potential synergistic interactions) on herpetofaunal and specifically amphibian communities remains poorly understood (Haywood, 2004), the conclusion that water quality will not be compromised by mining activities may be premature. However, for the purposes of my study, I will cautiously accept the conclusion of Shepherd et al. (2004) pending further research and assume that changes in water quality resulting from mining activities will not significantly influence herpetofaunal populations.

Mining activities are likely to affect the area in two major ways: severe habitat transformation on the actual mining site resulting from sand extraction techniques is probable. It should be noted though, that only transformed habitats (generally those under agriculture) will be mined. Secondly, changes in the hydrology resulting from additional water input into the ecosystem from mining activities (Fig. 1.2; Shepherd et al., 2004) affecting fauna and flora.

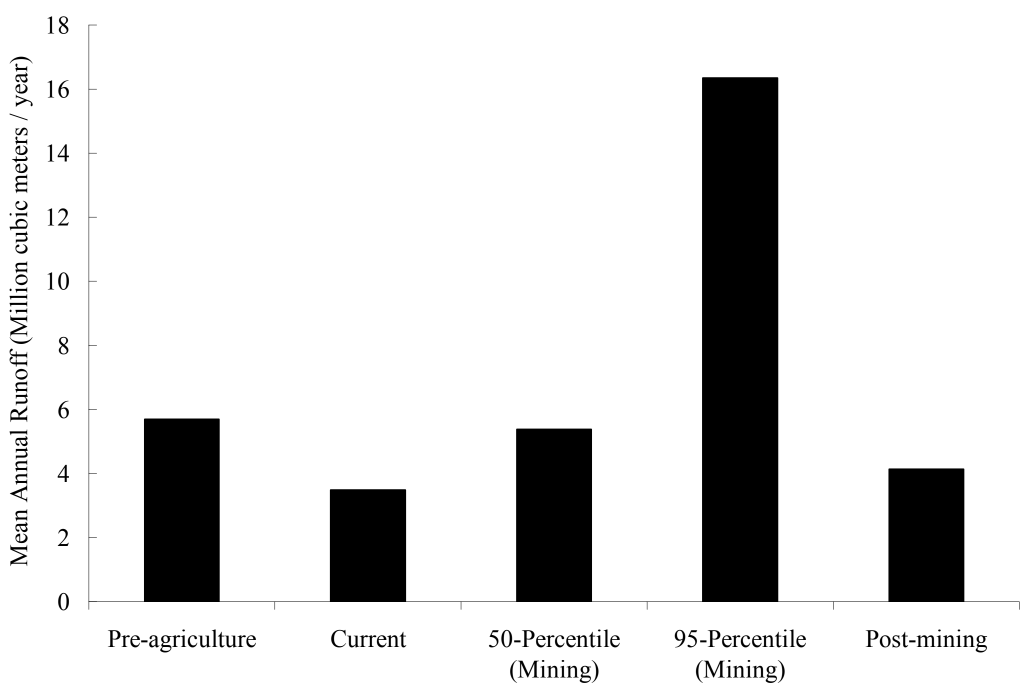

Figure 1.2: Modelled Mean Annual Run-off for the Siyayi Catchment (adapted from Shepherd et al., 2004).

1.4 Approach

The restoration and rehabilitation of mined habitats (either to a semi-natural state or to pre-mining state of transformation) requires that certain “baseline” pre-mining data exist in order to provide a measure of the appropriate target state. Additionally, such data are essential in monitoring the progress and evaluating the success of post-mining restoration activities. Unfortunately such data do not exist for many habitat types or for many taxa, especially cryptic taxa such as herpetofauna.

Many species of reptiles and amphibians have remarkably cryptic life histories and numerous species have gone undetected in a particular area for many years (e.g., Bauer et al., 2003; Bishop, 2004) despite intensive search efforts, making assessment of how these species might respond to perturbation very difficult. Additionally, many reptiles and amphibians lack the ability to disperse effectively, as evidenced by the large number of southern African species seemingly derived through the process of allopatric speciation (e.g., Branch et al., 2006). As a result, these species may not be capable of responding to threats such as mining by fleeing (as would be expected for most birds or large mammals). Alternatively, reptiles and amphibians are likely to take refuge, clearly an ineffective response given that refuge sites are likely to be destroyed. The implications are that mining is likely to have very different impacts on certain herpetofaunal species, making extrapolations of impacts, based on other taxa, inappropriate.

By gathering baseline information of herpetofaunal species assemblages, local distribution, relative abundance and habitat associations, the effect that mining activities are likely to have on local herpetofaunal populations can be better gauged, providing for better conservation planning tools. With such tools, conservation concerns can be addressed proactively rather than the more usual reactive approach. While it is the purpose (either implicitly or explicitly) of the Environmental Impact Assessment (EIA) process and its constituent specialist reports to highlight such conservation concerns, the brevity and scope of these processes does not often, if ever, provide a thorough assessment. This report, through its constituent chapters, aims to provide exactly that, by clarifying and directing conservation efforts targeting herpetofaunal populations to be negatively influenced by the proposed heavy mineral strip mine.

Chapter 2 investigates broad-scale biogeographic distribution patterns of southern African “east-coast” herpetofauna. Understanding the biogeographic patterns of species distributions relative to the study site can help improve predictions as to the herpetofaunal species assemblage that characterises the study area, infer characteristics of certain herpetofaunal populations in the study area, and inform decisions as to the occurrence of certain cryptic species (some of conservation concern) on the study site. Additionally, Chapter 2 reviews the available herpetofaunal distribution data, at quarter degree square (QDS) resolution, to better gauge levels of species richness relative to the rest of South Africa, and more accurately define the constituents of the species assemblage. I include the results of my field surveys and finally produce a systematic account of the herpetofauna of the grid squares 2831DC and 2831DD onto which the study site falls.

Chapter 3 provides insight into the ecology and biology of fossorial herpetofauna, particularly in the study area, but also across South Africa using GIS mapping techniques. Mean fossorial herpetofaunal density (individuals.m^{-2}) is estimated for the entire study site as well as for areas under different land use. Apart from the application of the results to the development of the proposed mine, the research proposes and empirically compares a new quantitative method for surveying fossorial herpetofauna to a previously published method (Measey, 2003). I also discuss the relative advantages and disadvantages of each method, as well as the difficulties involved in surveying fossorial herpetofauna and how these may be overcome as to progress the science of fossorial herpetofaunal ecology.

Chapter 4 indicates how land use appears to be affecting the herpetofaunal community on the study site. Areas under different land uses are compared relative to their mean and predicted species richness, community composition and herpetofaunal abundance. Information contained in this chapter is important in understanding how the levels of transformation influence the current diversity and abundance of herpetofauna in the study area.

Chapter 5 discusses conservation concerns that may arise from the development of a heavy minerals strip mine in KwaZulu-Natal, South Africa. This chapter includes discussion around the threatened herpetofaunal species known from the area, and, where applicable, mitigatory measures that aim to reduce the potential negative impacts that the proposed mine could have. It also discusses previous herpetofaunal specialist studies conducted in the area by Everard and Van Wyk (1996) and later, Alexander (2004a).

Finally, Chapter 6 summarises the findings of this study as well as the recommendations resulting from my research.

The investigations that I conducted and present in this report will, through the multi-scale approach used, provide the information that meets the needs of a thorough environmental management strategy for the proposed mine. Results from this work should additionally inform current protocols and future planning by highlighting the taxa and areas of concern and providing recommendations for mitigation of mining activities on these subjects.

Chapter 2: Herpetofauna of the greater Mtunzini area, KwaZulu-Natal, South Africa: Diversity and Biogeography

2.1 Introduction

Reptiles and amphibians are among the most poorly studied vertebrate taxa globally (Fazey et al., 2005), especially in old world regions. Southern Africa is no exception with little information available on local herpetofaunal species. Distribution data are lacking for many species, although the South African Frog Atlas Project (SAFAP) and South Africa Reptile Conservation Assessment (SARCA) have aimed to remedy this. The paucity of distribution data makes it very difficult to assess which species occur in a particular area, making herpetofaunal management of those areas difficult.

I employ techniques during this project that, to a degree, allowed me to overcome some of these data shortage problems. I assess herpetofaunal diversity at multiple scales. By incorporating a broad-scale interpretation of herpetofaunal diversity using biogeographic principals with fine-scale assessment, proven surveying techniques and knowledge on potentially resident species, I develop an improved understanding of the herpetofaunal community on the study site. The resultant information allowed me to predict which species occur on the study site and how populations of these species are likely to react to the disturbance posed by the proposed mining operation.

2.2 Biogeography of the greater Zululand region of KwaZulu-Natal, South Africa

Biogeography is a discipline that documents and assesses the spatial patterns in biodiversity (Lomolino et al., 2006). It is fundamentally pattern-based, yet has many applications, not least of which is conservation, particularly in the face of anthropogenic impacts at global scales (Lomolino et al., 2006). An understanding of the biogeography of the Mtunzini area is thus essential to predict the occurrence of herpetofaunal species on the study site and give insight into the ecology and conservation status of the species that occur in the area.

Several authors have reviewed the biogeography of southern African herpetofauna (**Amphibia**: Alexander et al., 2004; Bruton and Haacke, 1980; Drinkrow and Cherry, 1995; Poynton, 1964; **Reptilia**: Alexander, 1984; Hewitt, 1923; Poynton and Broadley, 1978; Stuckenberg, 1969). It is not within the scope of this report to critically review these treatments, suffice to say that at a coarse scale, the geographic distributions of the herpetofauna of coastal KwaZulu-Natal can be split into three groups. These include those species with “tropical” distributions, those with “temperate” distributions and those with “transitional” distributions (Broadley, 1980).

I mapped the extent of the distributions of herpetofaunal species along the KwaZulu-Natal coast, from the Mozambique border to Port Edward. I extracted locality data from Minter et al. (2004) for amphibians, and Bourquin (2004) for reptiles. These two publications present the most complete locality data sets for the province. I inferred the distribution of species, in most cases assuming that distributions were continuous. I excluded the chameleon genus *Bradypodion* from the analysis as its taxonomic status is unresolved, making it difficult to allocate isolated populations to defined groups. However, given that species of this genus rarely occur in sympatry (Tolley et al., 2006), *Bradypodion* spp. are not likely to significantly effect biogeographic patterns along the coast.

To establish changes in species richness along the coast, I summed the number of species occurring in each latitudinal class. These classes were 0.25° in extent (approximately 30 km), corresponding with the Quarter Degree Square (QDS) resolution of the amphibian distributional data (the coarser of the data sets). I performed a correlation analysis to determine the degree of correlation between latitude and species richness, separately for reptiles, amphibians and all herpetofauna (Fig. 2.1). Additionally, I classified each species as “Temperate”, “Transitional” or “Tropical” based on their known distributions. These classifications were used to assess the degree to which each biogeographic group contributes to the herpetofaunal assemblage along the coast.

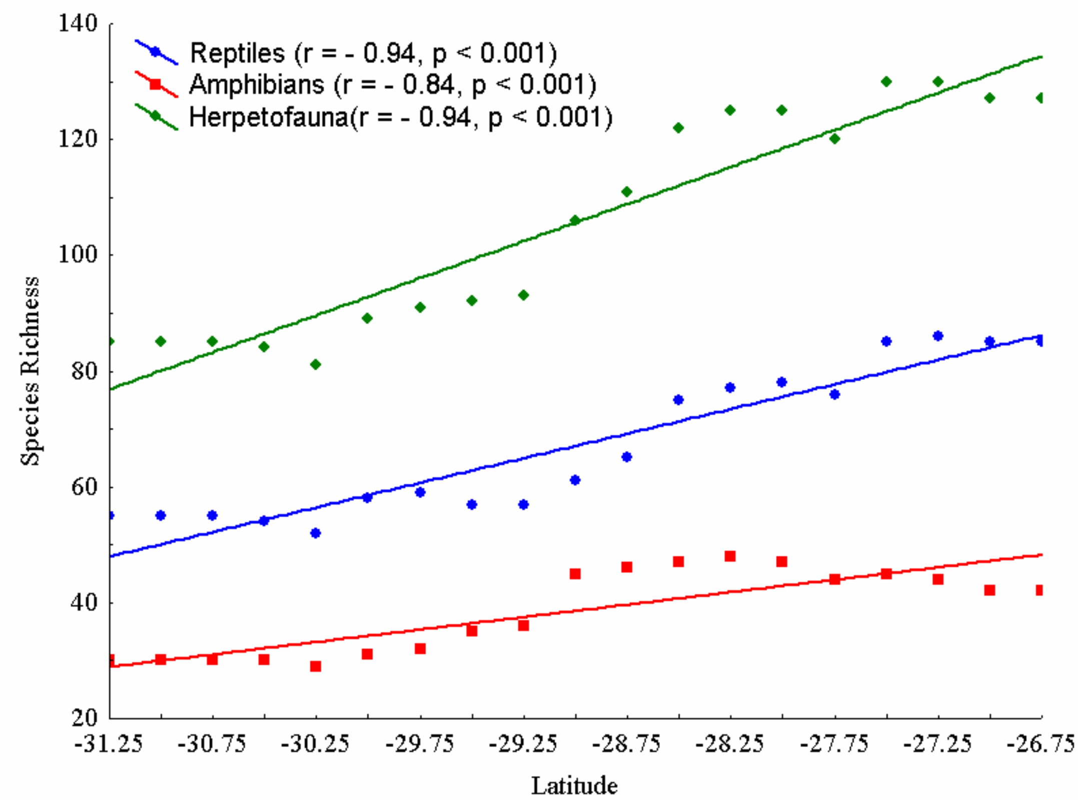

Figure 2.1: Correlation analyses showing changes in reptile (blue), amphibian (red), and all herpetofauna (green) species richness with latitude.

Herpetofaunal species richness decreases with increasing latitude ($r = -0.94$), a trend that holds true for both reptiles ($r = -0.94$) and amphibians ($r = -0.84$) (Fig. 2.1). From the perspective of this study, the position of Mtunzini this area of species subtraction is important (Fig. 2.2).

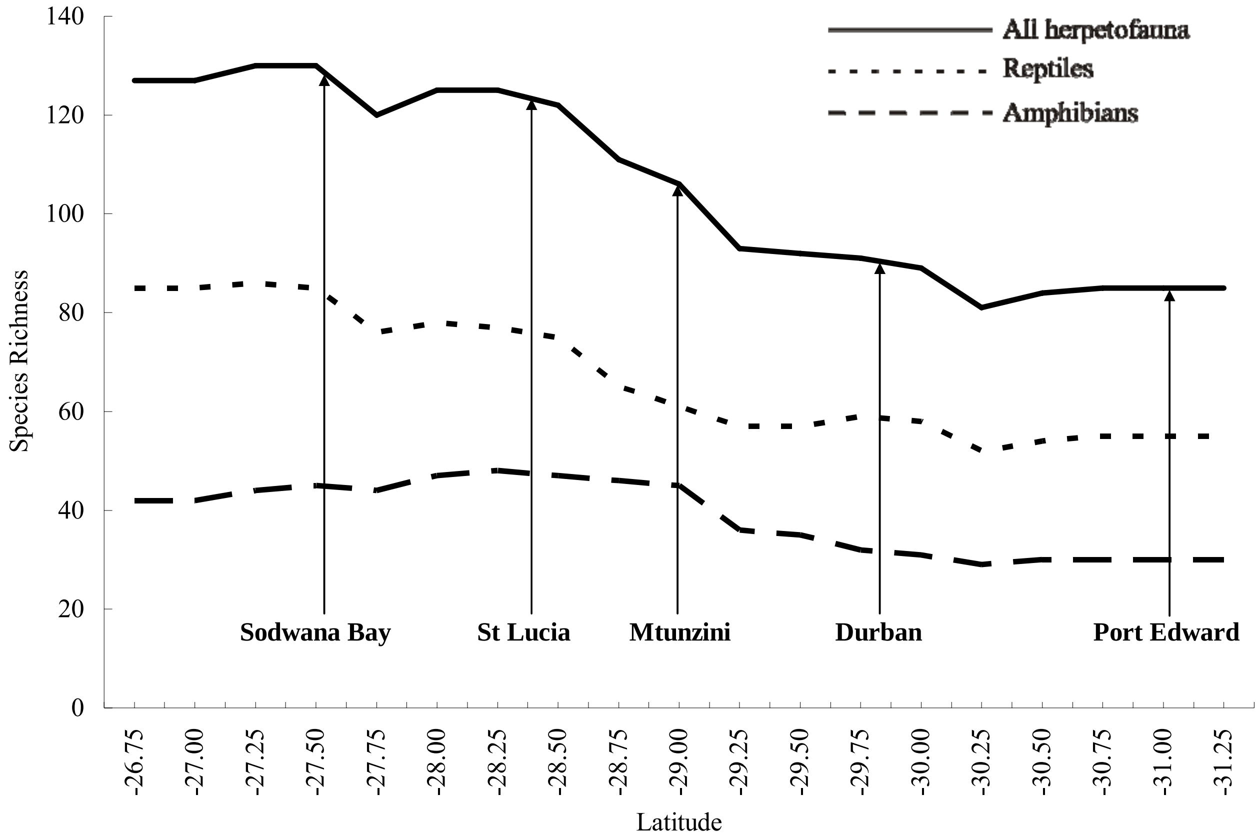

Figure 2.2: Changes in herpetofaunal species richness with latitude along the KwaZulu-Natal coast. The approximate latitudes of Sodwana Bay (-27.555), St Lucia (-28.373), Mtunzini (-28.952), Durban (-9.855) and Port Edward (-31.061) are indicated as reference points.

Amphibian diversity begins to decline at the approximate latitude of Mtunzini (29° S). However, the slope and position of this decline may be partially a sampling artefact (Poynton, 1980). Mtunzini has hosted many amphibian studies and the amphibian diversity of this area is well known. Sites further south, such as Amatikhulu (approximately 20 km south of Mtunzini), may host higher diversity than portrayed (Fig. 2) as these areas have not been as extensively surveyed. Reptile species richness begins to decline well north of Mtunzini with an approximate reduction in reptile species richness of 11 % between Sodwana Bay and St Lucia, and a further 20 % reduction between St Lucia and Mtunzini. The strong decline in herpetofaunal species richness south of St. Lucia indicates that several species reach their southern limit in this area. Importantly, St. Lucia also marks the southern limit of the Maputaland region coinciding with the narrowing and disappearance of the Mozambique coastal plain as well as the southern limit of the Lebombo Mountains (Watkeys et al., 1993), a factor that may well be contributing to the subtraction of reptile species along the coast.

Herpetofaunal assemblages are dominated by “tropical” species, with increasing contribution from “Temperate” species with increased latitude (Fig. 2.3). Approximately 67 % of the herpetofaunal species in the Mtunzini area have “Tropical” affinities, approximately 14 % have “Temperate” affinities and approximately 19 % have “transitional” affinities (Fig. 2.3).

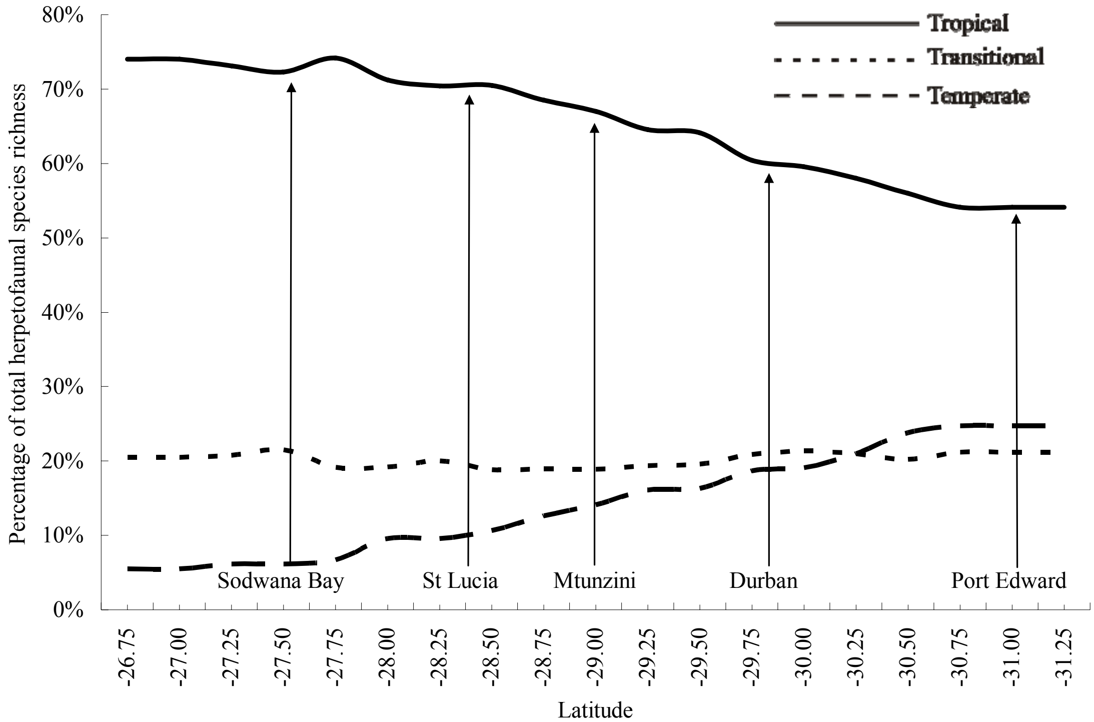

Figure 2.3: The percentage contribution of each biogeographic category to the total herpetofaunal species community along the KwaZulu-Natal coast.

The trends for reptiles and amphibians indicate that many herpetofaunal species reach their southern limits near Mtunzini which has implications for the characteristics of many of the herpetofaunal populations in the area. Simply, marginal populations (populations at or near to a species geographic limit) are often more susceptible to extirpation. Such susceptibility is not unexpected given that species distributions are not continuous, but rather tend to consist of metapopulations (Gaston, 2003). Furthermore, it is the periodic extirpation of populations that occur near to the limit of the distribution of a species that result in areas of absence, and thus define the limits of the range. Conversely, it should be noted that colonisation also takes place at such edges resulting in range expansion. Gaston (2003) discusses the factors that may result in such a situation.

Conversely, the fragmented nature of marginal populations may provide, through the isolation of populations, the precursors for allopatric speciation. Allopatric speciation is the process where isolated populations diverge significantly from "sister" populations resulting in new species (Lomolino et al., 2006), and has been implicated as a major driver in southern African amphibian diversity (Poynton, 1964). Allopatric speciation is also likely to have been critical in the formation of many southern African reptile species (Poynton and Broadley, 1978).

Given the high levels of habitat transformation in the area, many species that may have occurred in the area historically may represent now extirpated populations. Thus a simple assessment of the species richness in the area based on historical records may overestimate actual current species richness. Additionally, some remaining populations may be susceptible to environmental perturbations and thus be at risk of extirpation. The implications are obvious: local isolated populations that are either directly or indirectly detrimentally affected by the proposed mine are more susceptible to extirpation.

In summary, the study area hosts high herpetofaunal species richness, dominated by species showing "Tropical" distributions. However, species richness is negatively correlated with latitude indicating that many herpetofaunal species reach their southern limit along the KwaZulu-Natal coast. The resultant area, that hosts numerous range limits, provides an insight into the likelihood of localised populations being extirpated, either through natural or anthropogenic processes.

2.3 Herpetofaunal survey of the greater Mtunzini area (2831DC and 2831DD)

2.3.1 Methods

I searched two pertinent literature sources (Bourquin, 2004; Minter et al., 2004) for records of the herpetofaunal species recorded from the grid squares 2831DC and 2831DD over which the study site falls. While the Fairbreeze mining area falls mainly in 2831DC, this grid square was poorly sampled. Conversely, the town of Mtunzini is mainly in 2831DD, which is relatively well sampled. Since the species assemblages of the two grid squares are not likely to be significantly different, and since I collected several opportunistic records from the town, I included both grid squares in my area of interest. Any species indicated in the literature as having been recorded in 2831DC or 2831DD were included in the systematic review of the herpetofauna of the area.

I used quarter degree square (QDS) resolution as in Minter et al. (2004) and re-sampled the data from Bourquin (2004), which is at a finer scale, to allow me to collate data for all species of herpetofauna. While this coarse scale is not ideal, the poor quality of herpetofaunal distribution records necessitated large sample units. The result however is the inclusion of several species from areas inside the area of interest that are not likely to occur on the actual study site. For example, the two selected grids cover part of the Ngoye Mountains. Species such as the amphibians *Natalobatrachus bonebergi* and *Arthroleptella hewitti*, and the lizards *Pseudocordylus m. melanotus* and *Bradypodion* sp. nov. Dhlinza (In Bourquin (2004) as *Bradypodion* sp. J: see Reisinger et al. (2006)) are not likely to occur in the coastal parts and were not detected during my survey efforts.

Using several techniques I surveyed herpetofauna in the study area and immediate surroundings. Methods included trapping using pitfall traps, funnel traps and drift fences (Campbell and Christman, 1982; Gibbons and Semlitch, 1981; Maritz et al., In Press), excavating soil pits in search of fossorial species (Measey et al., 2003), active searches (Branch, 1998), road cruising (Simmons, 2002) and provision of a free "problem animal" removal service. Here I provide a brief description of each method. Detailed accounts of certain methods are included in the relevant chapters.

Trap arrays were installed throughout the study site to collect herpetofauna. Each array consisted of five 20-litre pitfall traps and eight funnel traps installed in conjunction with approximately 28 m of plastic drift fencing, adapted from Campbell and Christman (1982) and Gibbons and Semlitch (1981) (Fig. 2.4).

Arrays were installed at 21 sites, in various habitats, throughout and adjacent to the study site. Sampled habitat types included sugarcane, *Eucalyptus* plantation, secondary grassland, secondary forest and riparian forest. Arrays were checked and maintained for periods of time ranging from approximately 2 weeks to 12 weeks (Fig. 2.5) depending factors beyond my control such as crop harvesting. In total, traps were active for 1146 array-days.

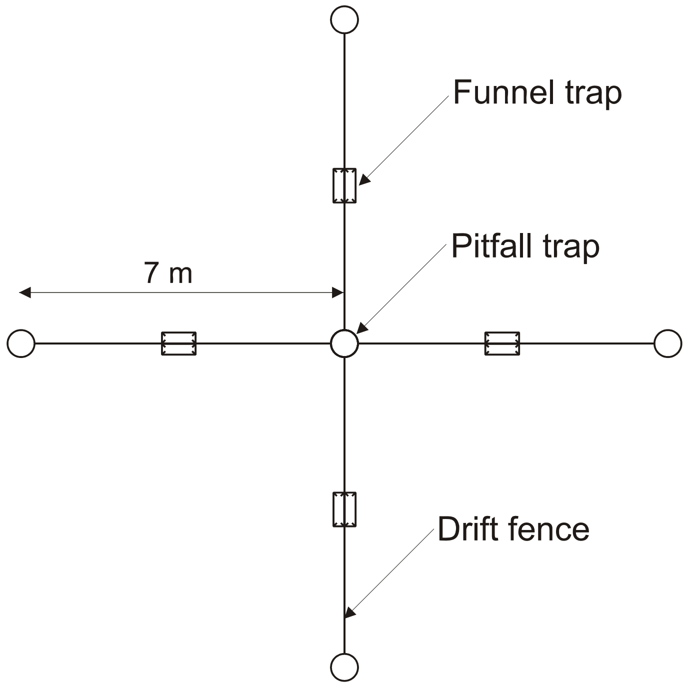

Figure 2.4: Plan view of terrestrial trap array showing drift fences, pitfall traps and funnel traps.

Traps were checked daily. All captured reptiles and amphibians were removed from traps and identified to species level. Most specimens were released at of point of capture. Some specimens were, however, preserved as museum voucher specimens.

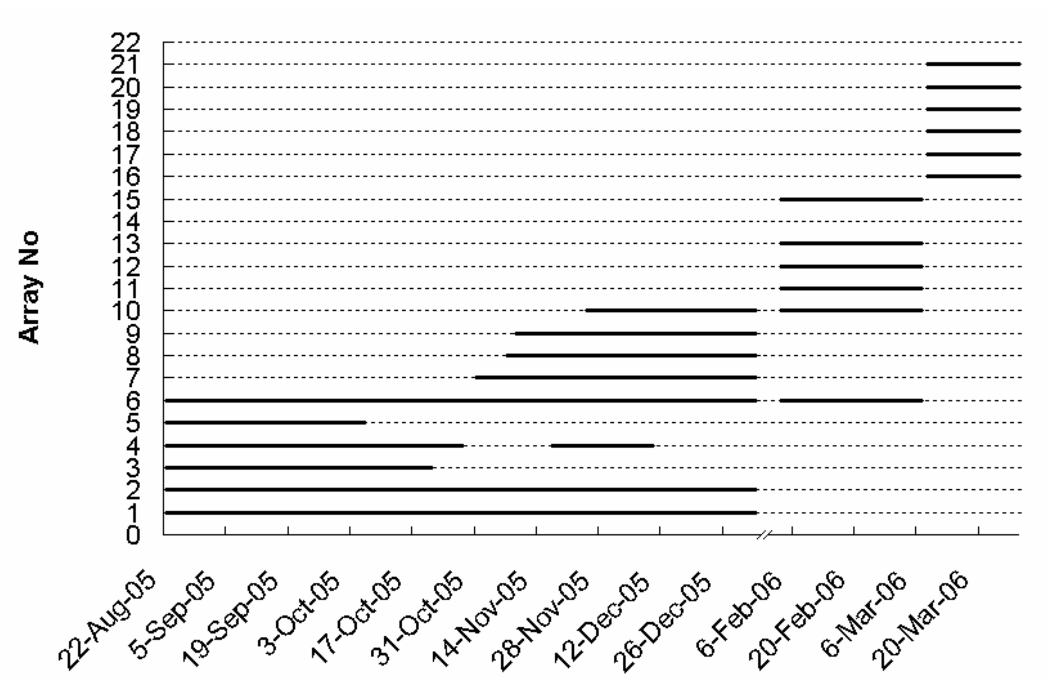

Figure 2.5: Dates during which trap arrays were active.

Details of the soil excavation technique are presented in Chapter 3. In summary, pits of varying sizes were excavated either with shovels (small pits: 1 m x 1 m x 0.3 m) or earthmoving equipment (large pits: 3 m x 3 m x 1 m). Removed soil was thoroughly searched by hand (in the case of small pits) or passed through a custom built sieve (in the case of large pits) to expose any small, fossorial herpetofauna dwelling in the soil. Pits were excavated in several habitats including sugarcane fields, *Eucalyptus* plantations, secondary grasslands, and restored forest.

Road cruising involves driving at low speeds, generally after sunset, with the objective of encountering reptiles and amphibians on the road surface. The technique is particularly useful for collecting snakes as these animals may move onto tarred roads during the early evening to absorb residual heat (Branch, 1998). Road cruising also allows one to visually survey a large, clear area (road surface) rapidly, during a period when nocturnal herpetofauna may be moving around. Animals that have been killed by motor vehicles are also encountered and often offer valuable distribution and ecological data (e.g., Maritz, In Press).

I searched suitable locations (e.g., underneath rocks, logs and other surface debris; in large leaf fronds) in various habitat types for reptiles and amphibians. Additionally, wetlands were searched at night, mainly with the intent of finding amphibians. Such amphibian surveys included audio surveys (frog advertisement calls are species specific and can be used to confirm the presence of certain species). Using spotlights I searched at night for chameleons.

Many people have an innate fear of snakes and do not like having these animals in their gardens, households or places of work (Shine and Koenig, 2001). I advertised a free “Problem Animal Removal Service” in the local newspaper (Maritz, 2005). By doing so, I hoped to collect presence data for many species of reptiles, particularly snakes.

2.3.2 Results

Forty-one amphibian species and 51 reptile species were listed in the literature for 2831DC and 2831DD. Of these, 38 amphibian species and 28 reptile species were recorded from only a single grid (2831DD). This finding is not surprising given the higher human population density in 2831DD, mostly a result of the town of Mtunzini.

In all, 41 species of amphibians and 51 species of reptiles were recorded from or around the study site. Importantly, the degree of match between the list of species that I collected and the list generated from the literature was surprisingly low. New amphibian QDS records were limited to 2831DC (2 species) while new reptiles species were recorded for 2831DC (19 species) and 2831DD (3 species). Additionally, 5 species were recorded for either QDS for the first time. These included the relatively abundant species *Panaspis walbergi*, *Acanthocercus atricollis* and *Philothamnus semivariegatus* (Table 2.1).

2.3.3 Discussion

Despite being in an area of high population density and an area that has hosted numerous herpetologists and naturalists in recent decades, the Mtunzini area (2831DC and 2831DD) are poorly represented by distribution records in the literature. The notable absence of abundant species in the literature (*Panaspis walbergi*, *Acanthocercus atricollis* and *Philothamnus semivariegatus*) indicates that such species are often overlooked by investigators who incorrectly assume that they have been previously collected because they are common.

2.4 Discussion

The biogeographic assessment presented above gives valuable insight into the species likely to occur on the site. Literature records for the relevant QDSs include several species that are excluded from the study site based on biogeographic factors. The rupicolous species listed such as *Pseudocordylus melanotus* provide just such an example. Intensive sampling in the study area was also valuable as it confirmed the presence of numerous species (many of which had not been recorded before) in the area and indicate those species that are not likely to occur in the area, or are very rare at best.

Table 2.1: New QDS distributional records detected through field surveys

Species	New Records
Breviceps mossambicus	2831DC
Hemisus guttatus	2831DC
Stigmochelys pardalis	2831DC
Python natalensis	2831DC
Aparallactus capensis	2831DC and 2831DD
Amblyodipsas concolor	2831DC
Amblyodipsas polylepis	2831DC
Lycophidion capense	2831DC
Mehelya capensis	2831DC
Mehelya nyassae	2831DC and 2831DD
Duberria lutrix	2831DC
Psammophis brevirostris	2831DC
Psammophis mossambicus	2831DD
Philothamnus semivariegatus	2831DC and 2831DD
Philothamnus hoplogaster	2831DC
Philothamnus natalensis	2831DD
Dispholidus typus	2831DC
Naja annulifera	2831DC
Naja melanoleuca	2831DC
Trachylepis striata	2831DC
Trachylepis depressa	2831DD
Trachylepis varia	2831DC
Panaspis walbergi	2831DC and 2831DD
Scelotes mossambicus	2831DC
Gerrhosaurus flavigularis	2831DC
Acanthocercus atricollis	2831DC and 2831DD
Chamaeleo dilepis	2831DC
Lygodactylus capensis	2831DC
Hemidactylus mabouia	2831DC

2.4 Systematic account of the herpetofauna of the greater Mtunzini area, KwaZulu-Natal, South Africa (2831DC and 2831DD)

The systematic account presented here represents all reptile and amphibian species previously recorded from the grid squares 2831DC and 2831DD as well as all the additional herpetofaunal species detected during my field surveys. Species listed in bold text represent species that I detected on the study site. Familial categorisations follow Branch (1998) and Frost et al. (2006). "Likelihood of occurrence" indicates the likelihood that a particular species occurs on the Exxaro Fairbreeze C Ext mining site.

CLASS: AMPHIBIA	Likelihood of occurrence
ORDER: ANURA	
FAMILY: ARTHROLEPTIDAE	
Genus: *Arthroleptis*	
Arthroleptis stenodactylus Pfeffer, 1893	Unlikely
***Arthroleptis wahlbergi* Smith, 1849**	Confirmed
Genus: *Leptopelis*	
Leptopelis mossambicus Poynton, 1985	Possible
***Leptopelis natalensis* (Smith, 1849)**	Confirmed
FAMILY: BREVICIPTIDAE	
Genus: *Breviceps*	
Breviceps adspersus Peters, 1882	Possible
***Breviceps mossambicus* Peters, 1854**	Confirmed
Breviceps sopranos Minter, 2003	Possible
Breviceps verrucosus Rapp, 1842	Possible

FAMILY: BUFONIDAE

Genus: *Amietophrynus*

***Amietophrynus gutturalis* Power, 1927**	Confirmed
Amietophrynus rangeri Hewitt, 1935	Possible

Genus: *Schismaderma*

***Schismaderma carens* (Smith, 1848)**	Confirmed

FAMILY: HEMISOTIDAE

Genus: *Hemisus*

***Hemisus guttatus* Rapp, 1842**	Confirmed

FAMILY: HYPEROLIIDAE

Genus: *Afrixalus*

***Afrixalus delicatus* Pickersgill, 1984**	Confirmed
***Afrixalus fornasinii* (Bianconi, 1849)**	Confirmed
Afrixalus spinifrons (Cope, 1862)	Possible

Genus: *Hyperolius*

Hyperolius acuticeps Ahl, 1931	Possible
***Hyperolius argus* Peters, 1854**	Confirmed
***Hyperolius marmoratus* Rapp, 1842**	Confirmed
Hyperolius pickersgilli Raw, 1982	Possible
***Hyperolius pusillus* (Cope, 1862)**	Confirmed
***Hyperolius tuberilinguis* Smith, 1849**	Confirmed

Genus: *Kassina*

Kassina maculata (Duméril, 1853)	Possible
Kassina senegalensis (Duméril and Bibron, 1841)	Possible

FAMILY: MICROHYLIDAE	
Genus: *Phrynomantis*	
Phrynomantis bifasciatus (Smith, 1847)	Possible
FAMILY: PYXICEPHALIDAE	
Genus: *Amietia*	
***Amietia angolensis* (Bocage, 1866)**	Confirmed
Genus: *Arthroleptella*	
Arthroleptella hewitti FitzSimons, 1947	Highly unlikely
Genus: *Natalobatrachus*	
Natalobatrachus bonebergi Hewitt and Methuen, 1913	Highly unlikely
Genus: *Pyxicephalus*	
Pyxicephalus edulis Peters, 1854	Unlikely
Genus: *Strongylopus*	
***Strongylopus fasciatus* (Smith, 1849)**	Confirmed
Strongylopus grayii (Smith, 1849)	Possible
Genus: *Tomopterna*	
Tomopterna cryptotis (Boulenger, 1907)	Possible
***Tomopterna natalensis* (Smith, 1849)**	Confirmed
FAMILY: PHRYNOBATRACHIDAE	
Genus: *Phrynobatrachus*	
***Phrynobatrachus mababiensis* FitzSimons, 1932**	Confirmed
***Phrynobatrachus natalensis* (Smith, 1849)**	Confirmed
FAMILY: PIPIDAE	
Genus: *Xenopus*	
***Xenopus laevis* (Daudin, 1802)**	Confirmed

FAMILY: PTYCHADENIDAE

Genus: *Ptychadena*

Ptychadena anchietae **(Bocage, 1867)**	Confirmed
Ptychadena mascareniensis (Dumeril and Bibron, 1841)	Possible
Ptychadena mossambica (Peters, 1854)	Likely
Ptychadena oxyrhynchus (Smith, 1849)	Likely
Ptychadena porosissima (Steindachner, 1867)	Likely

FAMILY: RHACOPHORIDAE

Genus: *Chiromantis*

Chiromantis xerampelina (Peters, 1854)	Unlikely

CLASS: REPTILIA

ORDER: TESTUDINES

FAMILY: PELOMEDUSIDAE

Genus: *Pelomedusa*

Pelomedusa subrufa (Lacépède, 1788)	Likely

Genus: *Pelusios*

Pelusios rhodesianus Hewitt, 1927	Unlikely
Pelusios sinuatus (Smith, 1838)	Unlikely

FAMILY: TESTUDINAE

Genus: *Stigmochelys*

Stigmochelys pardalis **(Bell, 1828)**	Confirmed

Genus: *Kinixys*

Kinixys belliana belliana Gray, 1831	Possible

ORDER: SQUAMATA

FAMILY: LEPTOTYPHLOPIDAE

Genus: *Leptotyphlops*

Leptotyphlops sylvicolus Broadley and Wallach, 1997	Likely

FAMILY: PYTHONIDAE

Genus: *Python*

***Python natalensis* Smith, 1840**	Confirmed

FAMILY: ATRACTASPIDIDAE

Genus: *Atractaspis*

***Atractaspis bibronii* (Smith, 1849)**	Confirmed

Genus: *Aparallactus*

***Aparallactus capensis* (Smith, 1849)**	Confirmed

Genus: *Amblyodipsas*

***Amblyodipsas concolor* (Smith, 1849)**	Confirmed
***Amblyodipsas polylepis polylepis* (Bocage, 1873)**	Confirmed

FAMILY: COLUBRIDAE

Genus: *Lycodonomorphus*

***Lycodonomorphus rufulus* Lichtenstein 1823**	Confirmed

Genus: *Lamprophis*

***Lamprophis capensis* (Dumeril and Bibron 1854)**	Confirmed
***Lamprophis inornatus* Dumeril and Bibron 1854**	Confirmed

Genus: *Lycophidion*

***Lycophidion capense capense* (Smith, 1831)**	Confirmed

Genus: *Mehelya*

***Mehelya capensis capensis* (Smith, 1847)**	Confirmed
***Mehelya nyassae* (Gunther, 1860)**	Confirmed

Genus: *Duberria*

***Duberria lutrix lutrix* (Linnaeus, 1758)**	Confirmed

Genus: *Psammophis*

***Psammophis brevirostris* Peters, 1881**	Confirmed
***Psammophis mossambicus* Peters 1882**	Confirmed

Genus: *Philothamnus*

***Philothamnus semivariegatus* (Smith 1840)**	Confirmed
***Philothamnus hoplogaster* (Gunther 1863)**	Confirmed
***Philothamnus natalensis natalensis* (Smith 1848)**	Confirmed

Genus: *Dasypeltis*

Dasypeltis inornata (Smith 1849)	Possible
***Dasypeltis scabra* (Linnaeus, 1758)**	Confirmed

Genus: *Crotaphopeltis*

***Crotaphopeltis hotamboeia* (Laurenti 1768)**	Confirmed

Genus: *Telescopus*

Telescopus semiannulatus semiannulatus (Smith, 1849)	Possible

Genus: *Dispholidus*

***Dispholidus typus* (Smith 1829)**	Confirmed

Genus: *Thelotornis*

***Thelotornis capensis capensis* (Smith 1849)**	Confirmed

FAMILY: ELAPIDAE

Genus: *Naja*

***Naja annulifera* (Peters 1854)**	Confirmed
***Naja melanoleuca* Hallowell 1857**	Confirmed
Naja mossambica Peters 1854	Possible

Genus: *Dendroaspis*

Dendroaspis polylepis (Gunther, 1864)	Likely
***Dendroaspis angusticeps* (Smith 1849)**	Confirmed

FAMILY: VIPERIDAE

Genus: *Causus*

***Causus rhombeatus* (Lichtenstein 1823)**	Confirmed

Genus: *Bitis*

Bitis arietans arietans (Merrem, 1820)	Possible
***Bitis gabonica* (Dumeril and Bibron 1854)**	Confirmed

FAMILY: SCINCIDAE

Genus: *Acontias*

***Acontias plumbeus* Bianconi 1849**	Confirmed

Genus: *Trachylepis*

***Trachylepis striata* (Peters, 1854)**	Confirmed
***Trachylepis depressa* (Peters, 1854)**	Confirmed
***Trachylepis varia* (Peters, 1867)**	Confirmed

Genus: *Panaspis*

***Panaspis walbergi* (A. Smith, 1849)**	Confirmed

Genus: *Scelotes*

***Scelotes mossambicus* Peters 1882**	Confirmed

FAMILY: CORDYLIDAE

Genus: *Pseudocordylus*

Pseudocordylus melanotus melanotus (A. Smith, 1838) Highly Unlikely

FAMILY: GERRHOSAURIDAE

Genus: *Gerrhosaurus*

***Gerrhosaurus flavigularis* Wiegman, 1829** Confirmed

FAMILY: AGAMIDAE

Genus: *Acanthocercus*

***Acanthocercus atricollis* (A. Smith, 1849)** Confirmed

FAMILY: CHAMAELEONIDAE

Genus: *Bradypodion*

Bradypodion sp. nov. Dhlinza Unlikely

Genus: *Chamaeleo*

***Chamaeleo dilepis* Leach, 1819** Confirmed

FAMILY: GEKKONIDAE

Genus: *Lygodactylus*

***Lygodactylus capensis* (A. Smith, 1849)** Confirmed

Genus: *Hemidactylus*

***Hemidactylus mabouia* (Moreau de Jonnes, 1818)** Confirmed

ORDER: CROCODYLIA

FAMILY: CROCODYLIDAE

Genus: *Crocodylus*

Crocodylus niloticus Laurenti, 1768 Likely

Chapter 3: Diversity, abundance and distribution of fossorial herpetofauna

3.1 Introduction to fossorial herpetofaunal ecology

Several terrestrial ecologists would rank soil as one of the least-studied micro-habitats on earth (Copley, 2000). Some of the most basic questions about the diversity and abundance of organisms in this micro-habitat remain almost entirely unknown, even for soil mega-fauna such as fossorial herpetofauna. These organisms may have important functions in the environment (Lavelle et al., 1997), constitute a high biomass, and contribute significantly to biodiversity (Measey, 2006), yet they remain poorly studied.

The fossorial herpetofauna are comprised of a suite of phylogenetically unrelated, and morphologically diverse, reptiles and amphibians. Measey (2006) defines fossorial herpetofauna as reptiles and amphibians that either utilise the soil and soil debris for refuge, or those that spend the majority of their lives living, feeding and breeding in the soil. This definition is open to debate. Although I agree in principal with Measey's (2006) definition, I do think it requires revision. Measey (2006) refers only to organisms inhabiting soil. I think a more thorough definition of fossorial herpetofauna should explicitly include species that inhabit other substrates such as alluvial sand. Measey (2006) demonstrates that while seemingly ecologically distinct, many taxa fit onto an ecological continuum, ranging from species that spend almost all their time underground, to species that only reside underground infrequently, and that a species' position on that continuum may be affected by numerous factors such as life history and habitat quality (Measey, 2006).

Since different species show differing degrees of fossorial habits, it is useful to define two groups of fossorial species as Measey (2006) has done. While this distinction is particularly useful for separating classically fossorial taxa such as amphisbaenids from taxa that simply take refuge below the surface either for short periods of time or extended periods of aestivation, a few problems still remain. Firstly, intermediate groups are likely to occur, making designation to a particular group difficult. Secondly, this definition requires a basic understanding of the biology of the relevant organisms, which is not always available given the cryptic nature of these animals.

I define herpetofauna as being fossorial if there is evidence that the species utilises any substrate (including sand, soil, leaf litter) below the surface of the terrestrial environment. Thus, fossorial species may have a range of lifestyles from strictly fossorial organisms that spend nearly all their active time below ground, to species that construct burrows for the purposes of shelter, sand-swimming species, and those that simply shuffle into the substrate for ambush or thermoregulatory purposes. I also subjectively define a subset of these species as being “strictly fossorial” and included in this group species that show strong fossorial affinities such as morphological, physiological or behavioural adaptations.

The paucity of data for almost all aspects of the biology and ecology of herpetofauna is a cause for concern. Most ecological available data for fossorial herpetofauna have been inferred from morphology and the examination and dissection of museum specimens. As a result, there is a bias toward information on feeding preferences and reproductive biology, inferred from gut contents and gonad condition of voucher specimens respectively (e.g., Shine and Webb, 1990; Webb et al., 2000; Webb et al., 2001). Patterns of diversity and abundance remain very poorly documented, largely because quantitative data are very difficult to collect due to the exceedingly cryptic nature of fossorial animals. Without even a rudimentary understanding of patterns of abundance and diversity, and the factors driving these patterns, the function of such organisms in community ecology remains entirely speculative. As a first step, development and testing of appropriate quantitative survey methods is crucial (Measey et al., 2003). Secondly, these must be applied at multiple scales so that an understanding of the nature of fossorial herpetofauna begins to emerge.

Recently, Measey et al. (2003) and Measey (2006) have described two methods of surveying fossorial herpetofauna, and has applied them in the measurement of densities for fossorial herpetofauna from a number of regions, albeit at fairly localised scales. This work has targeted particular taxa and has not been aimed at estimating diversity of fossorial herpetofauna. In southern Africa, such surveys are truly scarce. Pooley et al. (1973) excavated pits in northern KwaZulu-Natal, South Africa and recorded the density and diversity of fossorial herpetofauna. Measey (2006) reports on surveys conducted in the same area, but the investigations suffer from small sample size and poor capture rates. Few other anecdotal observations of fossorial herpetofaunal densities have been published (e.g., Burger, 1993), and these are rarely quantitative and are often published in inaccessible journals.

The paucity of previous quantitative fossorial herpetofaunal surveys provides a strong indication of the difficulties involved in performing such surveys. These can be broadly classed into two categories: those problems arising from the ecology and behaviour of fossorial herpetofauna, and those problems arising from the difficulties associated with the physical movement of soil.

Certain biological traits exhibited by some fossorial species make collecting specimens and ecological data difficult. Escape behaviour and locomotion of fossorial herpetofauna need to be considered when surveying these animals as they can have major implications for detection probability and accuracy of the estimates derived from the data. Most techniques employed to survey organisms assume very high detection probabilities. Yet, in general, escape behaviour for many herpetofaunal species is poorly known with most studies focusing on abundant terrestrial species (e.g., Diego-Rasilla, 2003; Downes and Hoefer, 2004; Losos et al., 2002; Whiting et al., 2003). General locomotion in fossorial herpetofauna has not been extensively studied either (but see Gans, 1985; Leonard, 1989; Navas et al., 2004) and thus mechanisms of escape are poorly known. In the case of snakes and apodal lizards, escape mechanisms are likely to represent a serpentine undulation through the substrate (Leonard, 1989).

Anecdotal evidence suggests that several species of southern African herpetofauna have the potential to move rapidly though the soil (e.g., *Scelotes* spp.: J.J. Marais, Pers. Comm.). Several fossorial species are known to construct a network of burrows, through which they can move rapidly, resulting in easy escape (*Breviceps* spp. – Minter, 2004a; *Ptenopus* sp. – Branch, 1998). Limiting fossorial herpetofauna from escaping detection by moving away from the site of disturbance is thus particularly difficult to quantify.

Some fossorial herpetofaunal species may reside deep underground, making accessing such species very difficult. Branch (1998) suggests that the snake *Rhinotyphlops lalandii* burrows to great depths but does not provide any details. Cowles (1941) reports *Chionactis occipitalis* from depths of up to 600 mm and Barbour et al. (1969) inferred that *Carphiophis amoenus* burrowed to depths of over 450 mm. These reports indicate that fossorial herpetofauna may be able to attain depths that current survey methods do not, with obvious implications for species detection.

Measey (2006) states that excavation is the most efficient way of surveying fossorial herpetofauna. However, such techniques often have logistical drawbacks, especially at larger scales and greater depths. The first and most obvious of these is the amount of work required to process adequate samples of soil. Soil on the Fairbreeze C Ext site weighs approximately 1650 $kg.m^{-3}$. The result is that soil excavated from a small plot of 1 m^2, to a depth of approximately 300 mm weighs more than 0.5 metric tons. The calorimetric implications to the herpetologist excavating by hand are obvious.

Here, I attempt to advance the study of fossorial herpetofaunal ecology in southern Africa. I have several objectives: Firstly I introduce a new quantitative method for surveying fossorial herpetofauna with heavy-duty earthmoving machinery. I compare my novel method with a previously described method in an attempt to make my data comparable to previously published results. I produce density estimates at both the landscape scale (the entire study site) and under different land uses. I attempt to tease apart some of the factors that may be driving any observed patterns and discuss how some of the difficulties involved in surveying fossorial herpetofauna may be overcome so as to advance fossorial herpetofaunal ecology.

3.2 Methods

3.2.1 Distribution mapping

In order to clarify the underlying trends in geographic distribution of fossorial herpetofauna, I mapped the distributions of South African reptiles in South Africa. By digitizing and georeferencing distribution data from Branch (1998), and summing all reptile distribution data in South Africa, I produced a reptile species richness map at Quarter Degree Square (QDS) resolution. Similarly, by summing all the distribution data for all fossorial reptile species in South Africa, I produced a fossorial reptile species richness map. Finally, by dividing the number of fossorial reptile species in each QDS by the number of reptile species in that QDS, I produced a map showing the proportion of the reptile community made up of fossorial species. The resultant maps provided an indication of the proportion of reptile species at the study site that show fossorial habits and allowed me to place the data collected from the study site into a South African context.

3.2.2 Fossorial herpetofaunal surveys

I quantitatively surveyed fossorial herpetofauna by excavating 218.6 m^3 of soil, covering an area of 311 m^2 and weighing approximately 360.7 metric tons. The soil was thoroughly sieved and searched, and all herpetofauna were capture and identified.

Method 1 entails digging large-scale excavations with earthmoving machinery and passing the excavated soil through a custom built sieve to expose any buried reptiles or amphibians. Excavations involved the digging of four trenches approximately 1.5 m deep and 0.75 m wide to form a “soil island” measuring 3 m x 3 m in area (initial plots of 5 m x 5 m proved to be too large and time consuming to sample). The top meter of the soil island was then systematically scooped and placed onto a custom built sieve. The sieve (Fig. 3.1), a table-like structure, measured 1 m x 0.75 m that stood approximately 1.2 m, was constructed from two sheets of expanded metal, each with diamond shape apertures measuring approximately 25 mm x 15 mm, overlaid on each other. The apertures of the resultant grid varied in size and shape because of the imperfect overlay, but were approximately half the size of the apertures in the original grids. Two or more people carefully sifted the soil through the sieve so that all soil was thoroughly examined for the presence of reptiles and amphibians. The efficacy of sieving was proven by the fact that even small invertebrates such as isopterans, coleopterans (adults and larvae), blattodeans, isopods, arachnids and annelids, many no bigger than 15 mm in length, were easily recovered. Collected reptiles and amphibians were identified, counted and released at point of capture.

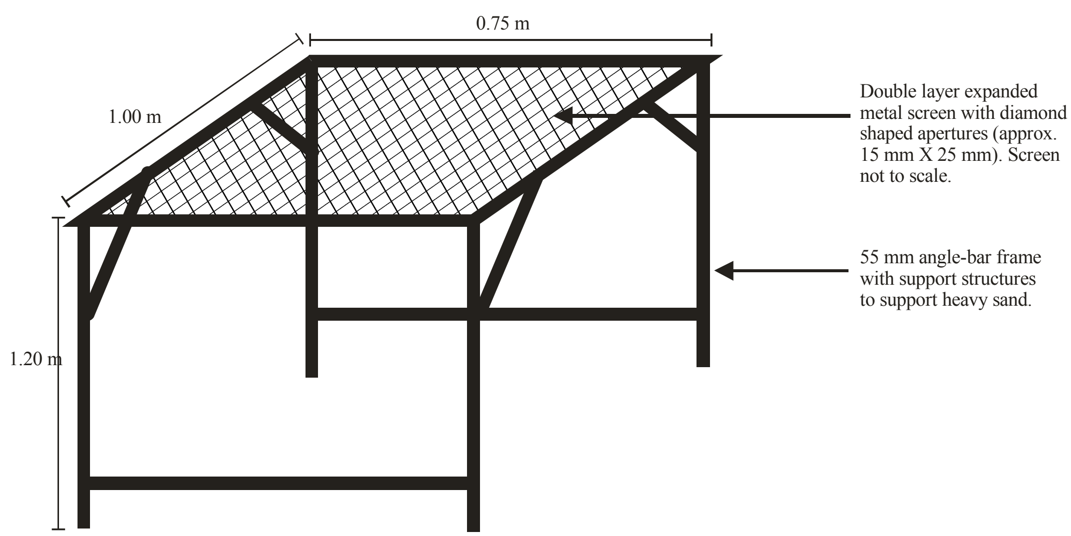

Figure 3.1: Table-like, custom-built sieve used to remove fossorial herpetofauna from sampled sand.

Method 2 is based on the method developed by Measey et al. (2003). Each survey comprised five pits randomly distributed within a 100 m^2 site. Holes measuring 1 m x 1 m, and 0.3 m deep were excavated rapidly by two people using shovels. All excavated soil was placed onto a plastic sheet.

Both people then sieved through the excavated soil using their hands and removing any reptiles or amphibians. Collected animals were identified, counted and released at point of capture.

The habitat at each site was classified according to its land use (categories: *Eucalyptus* plantation, Sugarcane, Forest, or Grassland). Longitude and latitude were recorded using a GPS. A soil sample, comprising three sub-samples from the immediate area (within 2 m of the point of excavation), was taken from each site for analysis of particle size distribution. Particle size distribution within a soil sample can be used to assess soil texture (Oberthür et al., 1999), a physical characteristic that may influence the occurrence of organisms (Rietkerk, 2002). Particle size distribution was assessed by passing each soil sample through sieves with screens sizes ranging from 800 ųm to 45 ųm. Because particle size distribution did not vary extensively over the study site, I developed an index of soil texture by subtracting the proportion of the sample falling above the mean of particle size for all samples, from the proportion falling below this size. This normally-distributed index provided a measure of whether soil at a particular site was more or less coarse than soil from other sites. I also measured soil compaction at each site by measuring the depth to which a Dynamic Cone Penetrometer penetrated from three standardised impacts. Measures were repeated at three random positions around the site after excavating the soil. Soil type at each site was classified according to Golder Associates (2005), but the limited extent of coverage of their maps forced me to exclude soil type as a determinant of fossorial herpetofaunal density during statistical analyses since several of the excavation sites fell outside of classified areas.

3.2.3 Data analysis

All statistical analyses were performed using Statistica ver. 6 (2002). I used the Generalized Linear/Nonlinear Model (GLZ) function to determine which factors, if any, predicted fossorial herpetofaunal density. This non-parametric analysis was performed because the distribution of the response variable (fossorial herpetofaunal density) matched a Poisson distribution rather than a normal distribution, as is assumed by a parametric General Linear Model (GLM). Continuous predictive variables include soil texture (from particle size distribution) and mean soil compaction, while land use was included as a categorical predictive variable. I used a Mann-Whitney U-Test to test for differences in mean estimated fossorial herpetofaunal density between survey methods and a Kruskal-Wallis ANOVA to test for differences in estimated fossorial herpetofaunal density between sites under different land uses. Non-parametric analyses were preferred of parametric equivalents because of the skewed data distribution and poor capture rates.

3.3 Results

3.3.1 Distribution mapping

A large proportion of South African herpetofauna show fossorial characteristics. More than 110 species (± 33 %) of South African reptile species live fossorial lifestyles to some degree and 73 (± 21 % of total) of those species being classed as "strictly fossorial" (Table 1). Of the 116 amphibian species known from South Africa (Minter et al., 2004), approximately 32 species (± 28 %) could be classed as fossorial with 26 of those species (± 22 % of total) classed as "strictly fossorial" (Table 3.1)

Reptile species richness in South Africa is not uniformly distributed over the country and ranged from 25 - 97 species per QDS. Higher species richness is evident from the north-eastern Mpumulanga and eastern Limpopo Provinces (Fig. 3.2). "Strictly fossorial" reptile species richness ranged from 0 – 25 species and showed a similar pattern of distribution to the entire South African reptile fauna (Fig. 3.3). The central grassland regions, the Limpopo valley, and areas bordering the Kalahari showed the greatest proportion of fossorial reptiles (Fig. 3.4). Proportional richness in northern KwaZulu-Natal was also high but decreased with increasing latitude.

Table 3.1: Fossorial Herpetofauna of South Africa. Bold typeface indicates species considered to be “strictly fossorial”.

Reptiles

Rhinotyphlops lalandei
Rhinotyphlops shinzi
Rhinotyphlops schlegelii
Typhlops fornasinii
Typhlops bibronii
Leptotyphlops longicaudus
Leptotyphlops nigircans
Leptotyphlops incognitus
Leptotyphlops scutifrons
Leptotyphlops telloi
Leptotyphlops distanti
Leptotyphlops sylvicolus
Atractaspis bibronii
Atractaspis duerdeni
Aparallactus lunulatus
Aparallactus capensis
Macrelaps microlepidotus
Amblyodipsas concolor
Amblyodipsas polylepis
Amblyodipsas micropthalma
Xenocalamus sabiensis
Xenocalamus transvaalensis
Xenocalamus bicolor
Lamprophis fiskii
Lamprophis fuscus
Lamprophis inornatus
Lycophidion pygmaeum
Pseudaspis cana
Dipsina multimaculata
Rhamphiophis rostratus
Prosymna bivittata
Prosymna frontalis
Prosymna janii
Prosymna stuhlmannii
Prosymna sundevallii
Aspidelaps scutatus
Elapsoidea boulengeri
Elapsoidea sundevalli
Homoroselaps dorsalis
Homoroselaps lacteus
Bitis schneideri
Chirindia langi
Dalophia pistillum
Monopeltis capensis
Monopeltis decosteri
Monopeltis infuscata
Monopeltis leonhardi
Monopeltis rhodesiana
Monopeltis sphenorhynchus
Zygaspis quadrifrons
Zygaspis vandami
Acontias breviceps
Acontias gracilicauda
Acontias meleagris
Acontias percivali
Acontias plumbeus
Acontias poecilus
Acontiophops lineatus
Microacontias lineatus
Microacontias litoralis
Typhlosaurus aurantiacus
Typhlosaurus cregoi
Typhlosaurus gariepensis
Typhlosaurus lineatus
Typhlosaurus lomii
Typhlosaurus meyeri
Typhlosaurus vermis
Lygosoma sundevalli
Scelotes anguineus
Scelotes arenicolus
Scelotes bidigittatus
Scelotes bipes
Scelotes bourquini
Scelotes caffer
Scelotes capensis
Scelotes fitzsimonsi
Scelotes gronovii
Scelotes guentheri
Scelotes inornatus
Scelotes kasneri
Scelotes limpopoensis
Scelotes mirus
Scelotes mossambicus
Scelotes sexlineatus
Scelotes vestigifer
Trachylepis capensis
Trachylepis depressa
Trachylepis homalocephala
Trachylepis occidentalis
Trachylepis variegata
Ichnotropis squamulosa
Meroles ctenodactylus
Meroles cuneirostris
Meroles knoxii
Nucras caesicaudata
Nucras holubi
Nucras livida
Nucras tessellata
Pedioplanis burchelli
Pedioplanis lineooccellata
Pedioplanis laticeps
Pedioplanis namaquensis
Tropidosaura cottrelli
Tropidosaura gularis
Cordylus giganteus
Gerhosaurus flavigularis
Gerhosaurus nigrolineatus
Gerhosaurus typicus
Agama aculeata
Agama armata
Agama hispida
Chondrodactylus angulifer
Colopus wahlbergii
Ptenopus garrulus

Amphibians

Arthroleptis stenodactylus
Amietophrynus garmani
Amietophrynus gutturalis
Poyntonophrynus vertebralis
Schismaderma carens
Vandijkophrynus angusticeps
Breviceps acutirostris
Breviceps adspersus
Breviceps bagginsi
Breviceps fuscus
Breviceps gibbosus
Breviceps macrops
Breviceps maculates
Breviceps montanus
Breviceps mossambicus
Breviceps namaquensis
Breviceps rosei
Breviceps sopranus
Breviceps sylvestris
Breviceps verrucosus
Hemisus guineensi
Hemisus guttatus
Hemisus marmoratus
Hildebrandtia ornate
Pyxicehalus adspersus
Pyxicephalus edulis
Tomopterna cryptotis
Tomopterna krugerensis
Tomopterna marmorata
Tomopterna natalensis
Tomopterna tandyi
Tompoterna delalandii

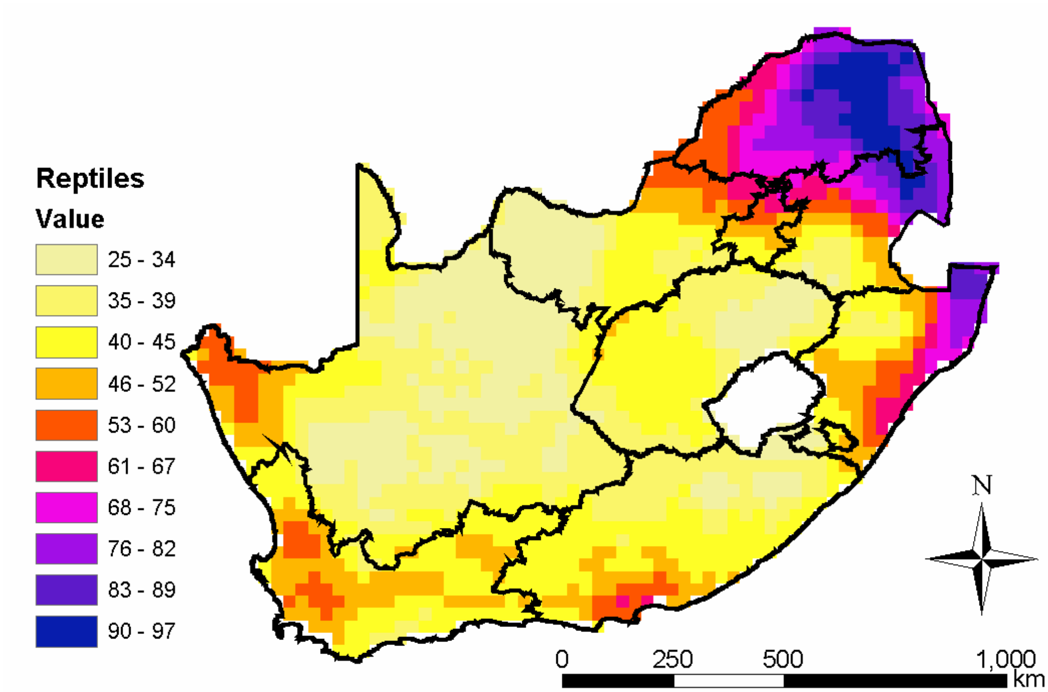

Figure 3.2: Predicted reptile species richness in South Africa at QDS resolution

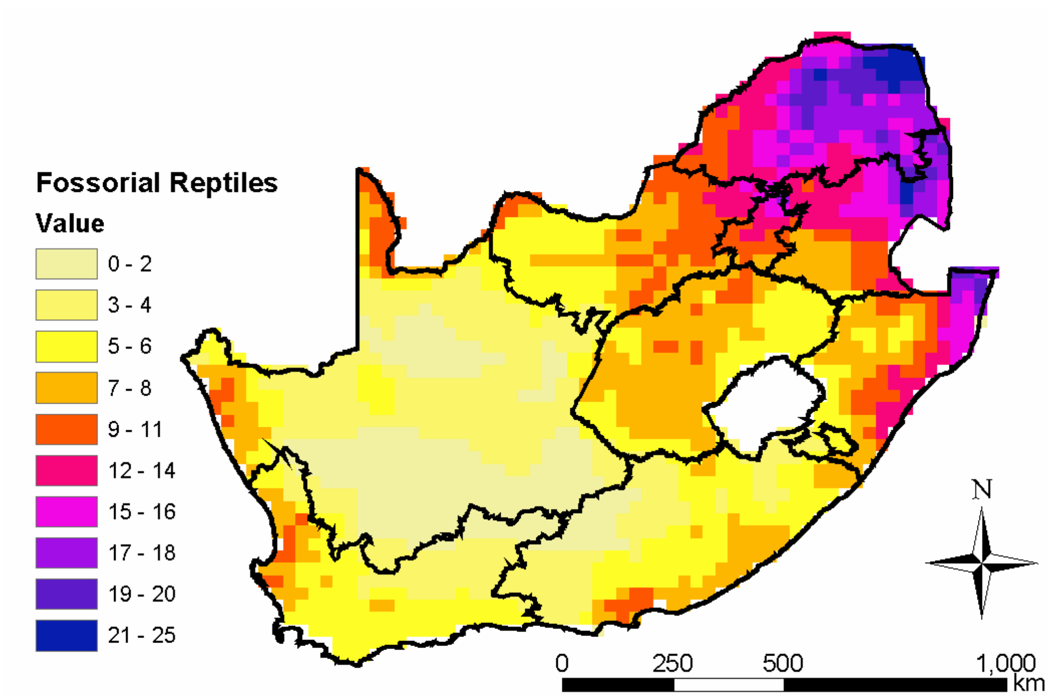

Figure 3.3: Predicted fossorial reptile species richness in South Africa at QDS resolution.

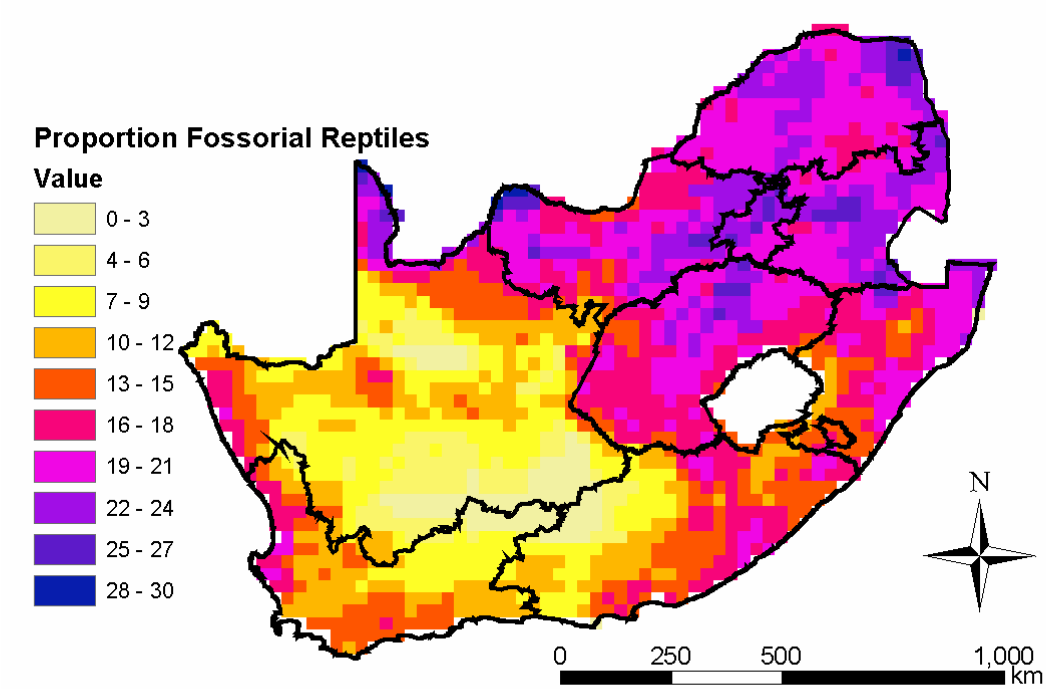

Figure 3.4: Predicted percentage of reptile community in each grid square showing fossorial habits in South Africa at QDS resolution.

The Mtunzini area is predicted to host approximately 70 reptile species of which 13 species (18.6 %) are fossorial in their habits, corresponding with the known reptile distribution records in the literature and from my field surveys (Chapter 2).

3.3.2 Fossorial herpetofaunal surveys

Surveys yielded very low capture rates, suggesting low population densities of fossorial herpetofauna in the sampled area. A total of only seven individual animals were captured despite 360.7 metric tons of soil being processed from 47 sites. These represented three species, namely the lizard *Scelotes mossambicus* (2 individuals), and the frogs *Amietophrynus gutturalis* (2 individuals) and *Breviceps mossambicus* (3 individuals) (Table 3.2). Mean fossorial herpetofaunal density across the study site was 0.019 ± 0.010 individuals.m^{-2} (mean ± SE). The estimated fossorial herpetofaunal density across the study area showed frequency distribution that differed significantly from normal (Kolmogorov-Smirnov: $d = 0.50$, $p < 0.01$) (Fig. 3.5). All individuals were captured within approximately 100 mm the surface.

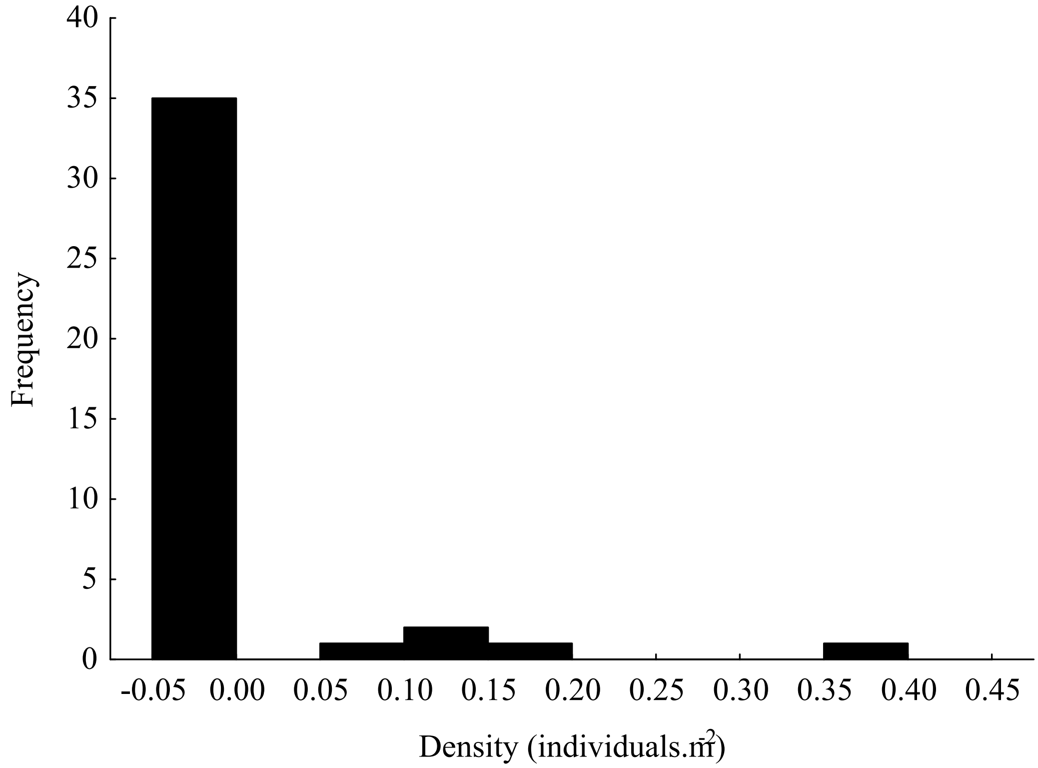

Figure 3.5: Frequency distribution of estimated densities across all sites (n = 47).

Density measures from the different survey methods did not differ significantly with regards to capture rates per unit area (Mann-Whitney U Test: $U = 246.0$, $p = 0.66$, Fig. 3.6). The 19 sites surveyed using Method 1 produced only three specimens yielding a density of 0.016 ± 0.009 individuals.m^{-2} (mean ± SE). Similarly, Method 2 only produced specimens at two of the 28 sites at a density of 0.021 ± 0.016 individuals.m^{-2} (mean ± SE).

Table 3.2: Quantitative fossorial herpetofaunal survey results collected from 47 excavations, using two survey methods in Zululand, KwaZulu-Natal.

Method	Area (m^2)	Volume (m^3)	Mass (tons)	Land use	No. of sites	Specimens
1	99	99	163.35	Secondary Grassland	11	*Breviceps mossambicus*
1	27	27	44.55	Sugarcane	3	*Amietophrynus gutturalis* x 2
1	27	27	44.55	Forest	3	*Scelotes mossambicus*
1	18	18	29.70	Eucalyptus	2	
Sub-total	**171**	**171**	**282.15**		**19**	**4 specimens (3 species)**
2	35	11.9	19.64	Secondary Grassland	7	
2	30	10.2	16.83	Sugarcane	6	
2	45	15.3	25.25	Forest	9	*Breviceps mossambicus*, *Scelotes mossambicus*
2	30	10.2	16.83	Eucalyptus	6	*Breviceps mossambicus*
Sub-total	**140**	**47.6**	**78.55**		**28**	**3 specimens (2 species)**
Total	**311**	**218.6**	**360.70**		**47**	**7 specimens (3 species)**

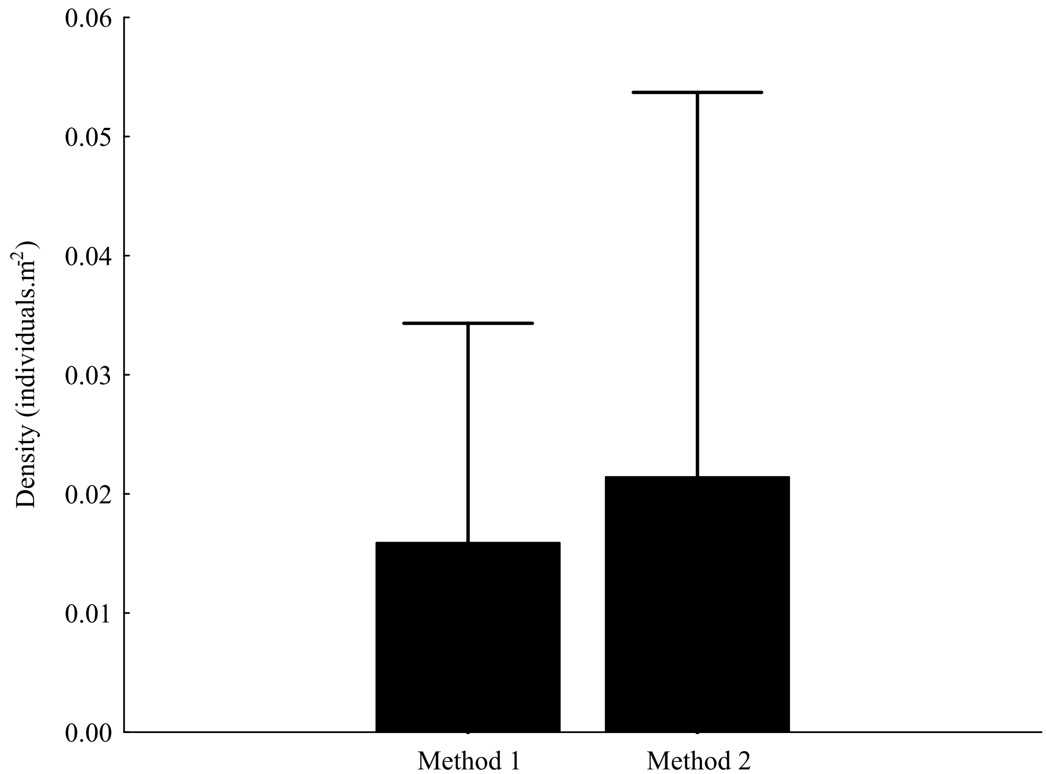

Figure 3.6: Comparison of mean density estimates produced from Method 1 (n = 19) and Method 2 (n = 28) used to survey fossorial herpetofauna. Error bars indicate 95 % confidence limits.

Since no difference was detected between density estimates from the two survey methods, I pooled the survey data to investigate whether land use influenced fossorial herpetofaunal density in a detectable manner. There was no difference between fossorial herpetofaunal density estimates from the four categories of land use (Kruskal-Wallis ANOVA: $H_{(3,47)} = 1.079$, $p = 0.78$, Fig. 3.7). Fossorial herpetofaunal density estimates ranged from 0.006 ± 0.006 individuals.m^{-2} (mean ± SE) for Grasslands to 0.043 ± 0.034 individuals.m^{-2} (mean ± SE) for Forests.

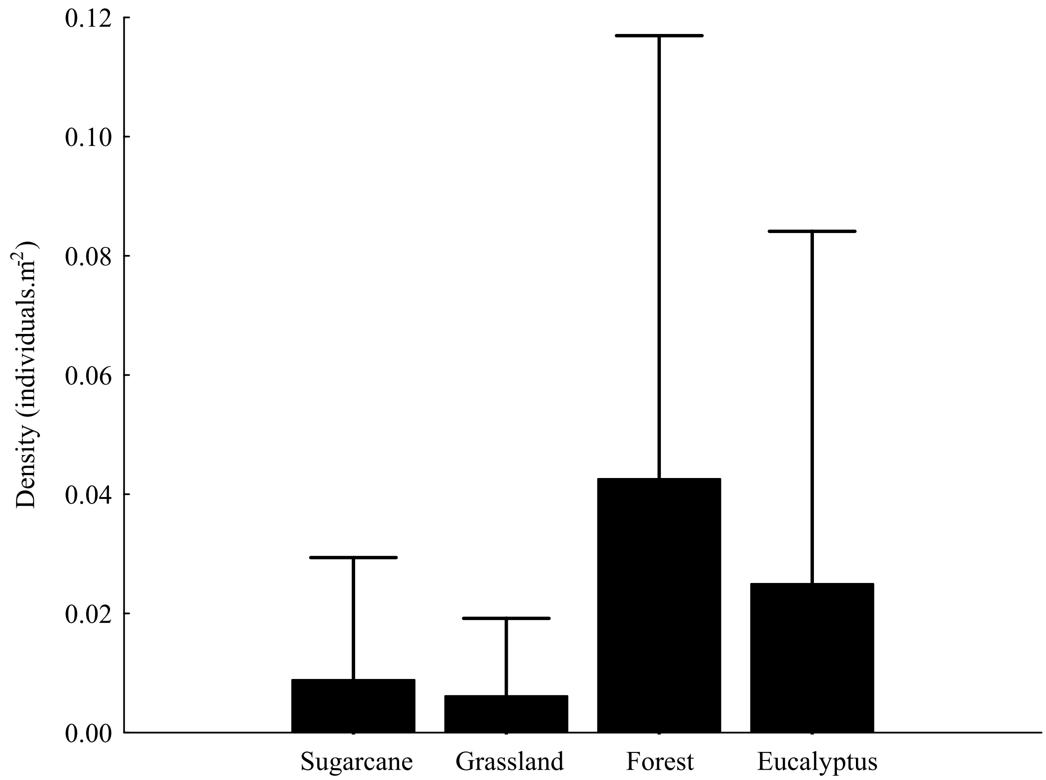

Figure 3.7: Mean estimated fossorial herpetofaunal density from four categories of land use. Error bars indicate 95 % confidence limits.

None of the selected factors (soil texture, mean soil compaction or land use) used in the Generalised Linear/Nonlinear Model successfully predicted fossorial herpetofaunal density (Table 3.3).

Table 3.3: Results from the Generalised Linear/Nonlinear Model (GLZ) showing the effect of Texture (from soil particle size distribution), mean soil compaction, and land use on fossorial herpetofaunal density.

	Degrees of freedom	**Log-Likelihood**	**Chi ^{2}**	**P**
Texture	1	- 4.03	0.01	0.91
Compaction	1	- 4.02	0.00	0.98
Land use	3	- 4.05	0.46	0.93

3.4 Discussion

Fossorial herpetofaunal abundance (0.019 ± 0.010 individuals.m^{-2}) and diversity (three species) were lower than I expected. Measey et al. (2003) estimated *Gegeneophis ramaswamii* density at between 0.51 and 0.63 individuals.m^{-2}, depending on season. I calculated mean fossorial herpetofaunal density for Measey's surveys (Measey et al. 2003, Table 1, Pg 47) to be 0.62 individuals.m^{-2}. Pooley et al. (1973) found fossorial herpetofaunal density to be 0.23 individuals.m^{-2}. Kuhnz et al. (2005) estimated *Anniella pulchra* density at 0.23 individuals.m^{-2} and Marais (unpublished data) estimated *Scelotes inornatus* density at approximately 0.02 individuals.m^{-2} although these estimates are taxa specific and representative of optimal habitat. It is clear that my density estimates are much lower than most other published estimates.

The difference between density estimates recorded during my study and those published (Kuhnz, 2005; Measey, 2006; Measey et al., 2003; Pooley, 1973) could result from several causes that may not be mutually exclusive. Actual densities across the sites could vary greatly, or minor discrepancies in the survey methods (such as surveying different microhabitats) could produce incomparable results. It is likely that the differences in this case are a combination of both of these factors. Measey (2006) states that "quantitative surveys always followed semi-quantitative surveys", a factor that has important implications for the density estimates published. Evidently, Measey (2006) pre-selected some sites for quantitative searches on the basis that they hosted target fossorial taxa. If fossorial taxa are likely to co-exist in patches (as my data indicate), then Measey's measures probably represent the average density of fossorial herpetofauna in optimal microhabitat across the study site, not average fossorial herpetofaunal density for the whole site. Alternatively, if one performs quantitative surveys randomly (or in an evenly stratified design) across the entire site, the resultant density estimate is likely to be closer to the average density at the landscape scale.

Figure 8 shows the potential impacts of different sampling regimes on density estimates. In the graphic Block A approximates regular plot location, Block B approximates a random plot placement, as used in this study, and Block C shows the effect of only sampling in areas perceived to be optimal microhabitat. Outlined areas represent actual optimal microhabitats and individuals are represented by the crosses. Notice that while none can claim to accurately represent landscape density, Block C in particular is likely to produce an overestimate. Additionally, the magnitude of this error is unknown unless the investigator is aware of the proportion of sub-optimal microhabitat to optimal microhabitat and the density of fossorial herpetofauna in each.

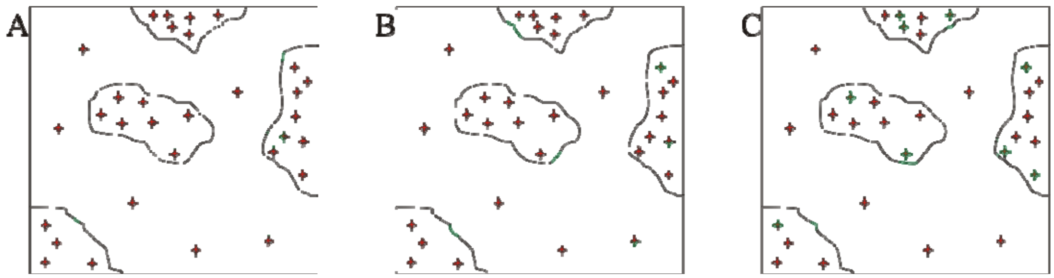

Figure 3.8: Schematic representation of the potential effects of sampling regime on fossorial herpetofaunal density estimates. Block A represents a regular sampling regime, Block B represents a random sampling regime, and Block C represents a sampling regime based on surveying perceived optimal fossorial herpetofaunal habitat.

Ultimately the choice of sampling regime is dependent on the objectives and spatial scale of the survey. If the researcher intends to investigate density related aspects of ecology relative to the organisms themselves or density at fine spatial scales, then sampling in areas perceived to represent optimal habitat as proposed by Measey (2006) is more appropriate. At the landscape level however, such density estimates lose value as they over-estimate density by an unknown magnitude.

Data collected during my study did not show differences in fossorial herpetofaunal density between sites under different land uses. Unfortunately, this is probably the result of the poor capture rate achieved, which resulted in low statistical sensitivity. Despite the lack of statistical significance in this analysis, there does appear to be a trend towards higher densities in more closed habitats such as forests. It should also be noted that most of the secondary grasslands on the site have, at some stage been under sugarcane, and so may share a common factor that act to depress fossorial herpetofaunal abundance (see Chapter 4).

Kuhnz et al. (2005) showed that the presence of grasses, forbs and exotic vegetation and the degree of soil disturbance negatively influence the distribution of *Anneilla pulchra*. Additionally several authors (Hinde et al., 2001; James and M'Closkey, 2003; Masterson et al., *in prep*.) have shown that habitat structure can be an important driver of terrestrial herpetofaunal diversity and abundance. While changes in surface structure may influence fossorial herpetofauna less than it does their terrestrial counterparts, anecdotal evidence suggests that subsurface structure (rocks, roots etc.) which is often removed by agricultural practices, may influence the occurrence of fossorial herpetofauna. Food availability may vary with land use, particularly if certain land uses employ pesticides (e.g., sugarcane: Johnston, 1989), and this may drive changes in fossorial herpetofaunal diversity and density. Finally, the management of tracts of land under different land uses may result in changes in fossorial herpetofaunal diversity or abundance. Numerous authors have shown that management, through the alteration of habitat structure or the addition of chemicals can alter diversity or abundance of herpetofauna (e.g., Ford et al., 1999; Hailey, 2000; James and M'Closkey, 2003; Jones et al., 2000).

The data suggest that fossorial herpetofauna may be patchy in their occurrence. A frequency plot of fossorial herpetofaunal density across the study site (Fig. 3.5) indicates a non-uniform or highly aggregated distribution of animals (Zar, 1996). Of the five excavation sites that yielded specimens, two (40 %) produced more than one individual which would not be expected for a low density, uniformly distributed pattern of occurrence. Kuhnz et al. (2005) found non-uniform distribution of *Anniella pulchra* providing further evidence of a non-uniform distribution of fossorial herpetofauna.

At small spatial scales the distribution of any organism is determined by how that organism interacts with its micro-environment. Characteristics of a micro-environment will interact with the biology of an organism to limit its occurrence in an area with sub-optimal conditions. Since fossorial organisms are closely associated with substrate in which they occur, substrate characteristics, both biotic and abiotic, may influence the occurrence of fossorial herpetofauna. Kuhnz et al. (2005) state that soil characteristics such as organic content and particle size distribution may be important in determining the abundance of fossorial herpetofauna but do not explicitly test this relationship. Marais (unpublished data) has shown than the fossorial scincid lizard, *Scelotes inornatus*, is largely limited to Berea Red soil deposits in the greater Durban area of KwaZulu-Natal, South Africa. This may be due to the aeolian nature of the soil, and its effect on soil texture and chemistry, but remains untested.

Unfortunately, because of the low capture rates achieved in this survey, my analysis of the factors that may influence fossorial herpetofaunal density is not very sensitive and thus the GLZ result is not surprising. Potential explanations for the non-significant result achieved may lie in the factors I chose to measure or the low capture rates achieved. Alternatively soil characteristics may not actually influence fossorial herpetofaunal density in this area, in which case the question remains as to what predicts fossorial herpetofaunal density? I recommend further standardised sampling from multiple sites as a means to address this question.

The two methods compared in this investigation produced very similar estimates of fossorial herpetofaunal density despite Method 1 surveying a greater area ($31 m^2$ more) than Method 2. However, each method has strengths and weaknesses. These include the resulting environmental impact, time and effort requirements, as well as financial costs. While Method 1 (large scale excavations) has the advantage of allowing high volumes of soil to be processed in a relatively short period of time, it also presents some drawbacks. The "ecological footprint" left by the earth moving machinery is large. Although I did not explicitly measure the area impacted by the machinery during a single $9\ m^2$ excavation, I estimate that approximately $225\ m^2$ of land is scarred per site (in this instance, this impact was acceptable because the area was already earmarked for mining). Conversely, the Method 2 (small scale excavations) had a much more restricted impact, which was limited to the immediate vicinity of area being excavated.

Selection of survey sites for Method 1 surveys was also limited by the size of the machinery. Sites hosting suitable micro-habitat such as those along forest edges or under leaf litter in wooded areas can not always be accessed with earthmoving machinery without the complete destruction of the habitat, whereas people digging pits with shovels can easily access and survey these areas. Importantly, excluding such areas will produce underestimates of fossorial herpetofaunal density and can have major implications for survey results and their subsequent application. The earth-moving machinery required for Method 1 also limits the areas that can be surveyed. Such machinery is not always available in remote locations. Alternatively, the equipment required for conducting surveys using Method 2 can be easily transported to remote locations.

Method 2 offers a financial advantage over Method 1 as the earthmoving machinery used in Method 1 is costly to hire as the machinery has high running costs and requires skilled labour.

Thus the financial aspect of this method may place this technique beyond the financial reach of many interested researchers. Alternatively, Method 2 requires only inexpensive equipment and intensive labour.

The relative advantages and disadvantages make the choice of technique situation dependent. While Method 1 gives investigators piece of mind in terms of the completeness of the sampling procedure through reduction of escape rates and the opportunity to sample to greater depths (although my data suggests that most organisms occur superficially in the soil profile), it carries major financial, environmental and logistic drawbacks. I recommend the application of Method 2 for surveying fossorial herpetofauna but urge researchers to be explicit about the sampling regime used.

3.5 Conclusions

Accurately classifying fossorial herpetofauna into discreet groups is difficult if not impossible given our current lack of understanding of their biology. Nonetheless, a subjective distinction can be made between fossorial herpetofauna and the subset "strictly fossorial" herpetofauna. Strictly fossorial herpetofauna taxa are not uniformly distributed across South Africa, showing disproportionately high occurrence in the central grassland, Limpopo Valley, Zululand and Kalahari areas.

Fossorial herpetofauna are difficult to survey because of problems associated with the biology and ecology of the animals themselves, and the logistic problems associated with moving large amounts of soil. Accordingly, fossorial herpetofauna are particularly poorly understood, with most ecological information regarding such taxa being inferred from museum data.

In northern KwaZulu-Natal, South Africa, fossorial herpetofauna can occur at very low densities. While sampling technique did not significantly influence measures of fossorial herpetofaunal density, evidence suggests that fossorial herpetofauna are likely to occur in an aggregated pattern and thus sampling regime could have a critical effect on density estimates.

Neither soil texture, nor soil compaction nor land use significantly affected fossorial herpetofaunal density, although statistical sensitivity for this analysis is likely to be low, warranting further surveys. My data showed that land use did not significantly affect fossorial herpetofaunal density, although a trend towards higher densities in closed habitats (forest and *Eucalyptus* plantation) was observed. The data suggest that fossorial herpetofauna occur at very low densities on the study site, despite the high regional fossorial herpetofaunal richness and apparent suitability.

I recommend that quantitative fossorial herpetofaunal surveys become part of all herpetofaunal surveys. Resultant data will improve our understanding of how patterns of distribution and abundance change on spatial and temporal scales vastly improving our ability to predict the occurrence of fossorial species and perform accurate conservation assessments. Studies should be explicit about the scale at which they predict fossorial herpetofaunal density as small scale surveys or biased sampling regimes may overestimate landscape scale fossorial herpetofaunal density.

Chapter 4: Herpetofaunal utilisation of areas of different land use and the potential of riparian buffers as mitigatory tools

4.1 Introduction

Habitat transformation represents one of the largest threats to global biodiversity (Myers et al., 2000). This holds true for South African biodiversity (Driver et al., 2005) and logically for many faunal groups within the country, including certain amphibians (Branch and Harrison, 2004) and reptiles (Branch, 1988). Few investigations have attempted to detect the effects of habitat transformation on most taxa, particularly cryptic taxa such as the herpetofauna. Knowledge of which herpetofaunal species utilise areas under different land uses and the degree to which those land uses may affect herpetofaunal diversity allows for the development of conservation appropriate management of those areas.

Mining activities on the Exxaro KZN Sands Fairbreeze C Ext mine are likely to result in local habitat transformation in two main ways. Direct habitat transformation through the removal of mineral-rich substrate will undoubtedly result in habitat loss (Lubke and Avis, 1999). Such transformation, at least at the local scale is likely to negatively influence herpetofaunal populations in the area and may lead to the extirpation of species with localized distributions. Secondly, mining activities could indirectly influence local faunal populations through alteration of local hydrology (Shepherd et al., 2004).

While mining activities generally produce public outcries because of the resultant habitat transformation (Fahn, 2002), comparatively little is said regarding widespread habitat transformation resulting from agricultural activities. Mining activities will result in habitat transformation but agricultural practices could potentially have already reduced local levels of diversity and abundance to low levels, resulting in areas with greatly reduced conservation value. Currently, the study area is in a transformed state, dominated by sugarcane plantation, *Eucalyptus* plantation with areas of secondary grassland (hereafter grassland) and “semi-natural” forest (hereafter forest). Anecdotal evidence suggests that the use of pesticides, harvesting regimes and habitat homogeneity in the sugarcane and *Eucalyptus* plantations have depressed herpetofaunal diversity and abundance in the study area but this remains untested.

Riparian woodlands occur on sections throughout the study site. These riparian areas are protected and will not be mined (R. Hattingh, Pers. Comm.). As a result, these areas can potentially perform important functions, not only as refugia for animals during mining activities, but also as source areas for faunal recolonisation post-mining, and as corridors that can facilitate re-colonisation. Yet the suite of species and the herpetofaunal abundance that these areas host are largely unknown along with the potential of these areas to act as refugia or corridors.

Differential habitat use by species in herpetofaunal communities is not uncommon (Reinert, 1993; Pianka and Vitt, 2003), and one would expect that certain species would be limited to certain habitats ("habitat specialists") while others would occur in various habitats ("habitat generalists"). Several factors could be interacting to produce such differential habitat use. These include the thermal properties of the habitat, structural features and importantly the animal's perception of these factors (Reinert, 1993).

By investigating the patterns of herpetofaunal occurrence within areas under different land uses, mitigatory measures can be developed that either reduce the impact of mining activities or facilitate effective recovery of disturbed lands after mine closure. Accordingly, I compared the herpetofaunal communities of areas under sugarcane plantation, grassland, forest, and *Eucalyptus* plantation to quantify herpetofaunal diversity in each. I also investigated the importance of riparian areas as habitat for herpetofauna under the current land use regime by comparing herpetofaunal diversity and abundance of sites in and outside of riparian areas.

4.2 Methods

I surveyed herpetofaunal communities on the study site using terrestrial trap arrays (Campbell and Christman, 1982; Gibbons and Semlitsch, 1981; Maritz et al., In Press). Trapping took place at 21 locations, covering the four main land uses, on and adjacent to the Exxaro KZN Sands Fairbreeze mining area for various periods of time (Chapter 2, Fig. 2.4). Each trap array consisted of eight funnel traps, five pitfall traps and approximately 28 m of plastic drift fencing as described in Chapter 2. Traps were checked daily and all captured herpetofauna were removed, identified and released at point of capture.

I used sample-based rarefaction (Gotelli and Colwell, 2001) and the Bray-Curtis Similarity Index as calculated by PRIMER (Clarke and Gorley, 2001) to compare the communities of each land use. I used these techniques as sample effort was different for each land use making traditional empirical comparisons inappropriate. Sample-based rarefaction curves are used to compare the species richness of two or more communities. They are read from right to left and the comparison is made at the highest common sample size. Should the curve of a particular community fall outside of the 95 % Confidence Limit of the most thoroughly sampled community, then those two communities have different predicted species richness (Magurran, 2004).

I also assessed the importance of riparian areas as habitat for herpetofauna under the current land use regime by comparing herpetofaunal diversity and abundance from a sub-set of array traps set in riparian and non-riparian areas. Riparian areas were defined as being under riparian woodland vegetation, always within 10 m of the stream channel whereas non-riparian areas were always further than 50 m away from the river channel (Fig. 4.1).

Figure 4.1: Aerial view of study site showing the placement of trap arrays used in the comparison of the herpetofaunal communities of riparian (red markers) and non-riparian areas (yellow markers). Image courtesy of Google Earth.

I compared the species assemblages of these two habitat types to assess similarity in the resident suites of species using the Analysis of Simialrity (ANOSIM) function in PRIMER (Clarke and Gorley, 2001). ANOSIM is a non-parametric technique that compares variation in specific diversity within a defined group of sites (e.g., those in riparian areas) to the variation between two defined groups of sites (e.g., riparian areas and non-riparian areas). A dendrogram was used to illustrate clustering of sites, defined by their Bray-Curtis similarity, relative to one-another. Additionally I compared herpetofaunal abundance and species richness from the six sites in riparian areas and the six sites in non-riparian areas. Since the data were collected during two consecutive trapping sessions, I used analysis of covariance (ANCOVA) to test for differences in mean species richness (total number of species trapped at each site) and abundance (total number of captured specimens) between the two categories of sites whilst coding for the effect trapping session.

4.3 Results

In total 308 specimens were trapped, representing 16 snake, six lizard and eight frog species (Table 4.1). Areas under riparian woodlands and sugarcane produced the greatest number of specimens but this is likely a sampling effect given the uneven distribution of trapping effort in each habitat.

Table 4.1: Capture frequency of herpetofaunal species from areas under different land uses

Species	Eucalyptus	Forest	Grassland	Sugarcane	All Habitats
Amblyodipsas concolor		2			2
Amblyodipsas polylepis		1			1
Aparallactus capensis		3			3
Arthroleptis wahlbergi	13	22		2	37
Atractaspis bibronii		1			1
Breviceps mossambicus	13		4	17	34
Amietophrynus gutturalis	10	14	15	22	61
Causus rhombeatus		2		1	3
Crotaphopeltis hotamboeia	1	12		2	15
Dasypeltis scabra				1	1
Duberria lutrix	1	5	2		8
Gerrhosaurus flavigularis		1	4	1	6
Hemidactylus maboiua	6	5			11
Hemisus guttatus			1	1	2
Lamprophis capensis			3	1	4
Lycodonomorphus rufulus		4			4
Lycophidion capense				3	3
Mehelya nyassae		2	2		4
Panaspis walbergi	2	13	20	39	74
Philothamnus hoplogaster		1			1
Philothamnus semivariegatus	1				1
Phrynobatrachus mababiensis		2		1	3
Phrynobatrachus natalensis		1			1
Psammophis brevirostris				1	1
Psammophis mossambicus		1	1	2	4
Scelotes mossambicus		1			1
Schismaderma carens	1	4	6	4	15
Tomopterna natalensis	1	2			3
Trachylepis striata	1	1	1		3
Trachylepis varia				1	1
All species	50	100	59	99	308

Sample-based rarefaction curves showed “forest” areas to be the most diverse with the remaining three land uses producing similar curves (Fig. 4.2).

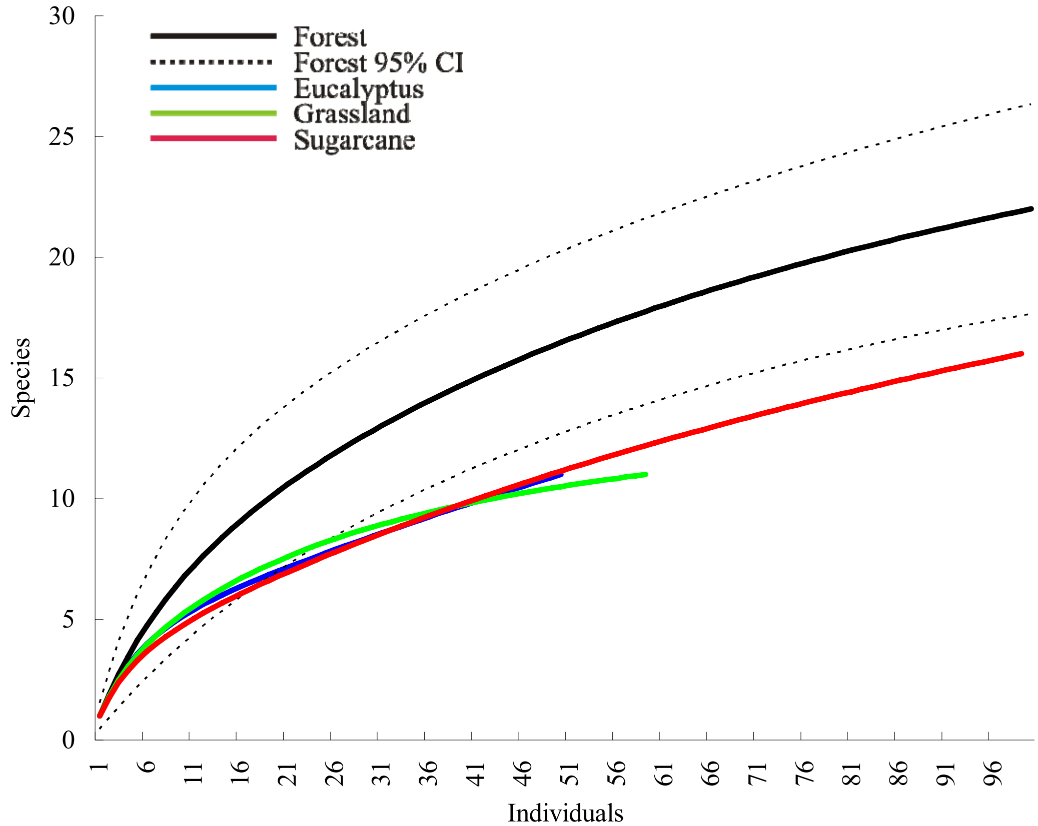

Figure 4.2: Sample-based rarefaction curves for herpetofaunal communities of areas under four land uses. 95 % Confidence limits are shown for the forest community only as this is the most adequately sampled.

The grasslands and sugarcane plantations produced a relatively high Bray-Curtis Similarity Index (59.5 %). Forest areas, as a result of their high species richness were moderately similar to *Eucalyptus* (46.7 %) and Grassland areas (47.8 %), while all other pair-wise comparisons yielded similarities of less than 40 %.

Mean species richness varied significantly between sites in riparian areas and non-riparian areas (ANCOVA: $F_{(1,9)} = 9.93$, $p = 0.01$). Sites in riparian areas hosted more species (mean ± SE: 5.83 ± 0.60 species) than sites in non-riparian areas (mean ± SE: 3.17 ± 0.60 species) (Fig 4.3 - left). Herpetofaunal abundance (total herpetofaunal captures per site) also varied significantly between sites in riparian and non-riparian areas (ANCOVA: $F_{(1,9)} = 9.48$, $p = 0.01$), with higher capture rates in riparian areas (mean ± SE: 12.17 ± 1.72 specimens) than non-riparian areas (mean ± SE: 4.67 ± 1.72 specimens) (Fig. 4.3 - right). While specimens were not marked, the inclusion of potential re-captures in this analysis is unlikely to affect the results. Characteristics of individuals of rarely captured species were noted and compared with subsequent captures, and indicated that very few specimens, if any, are re-captured. Re-captures in more commonly captured species are similarly likely to be rare (probability of re-capture is dependent on home range size and activity levels). This finding is supported by the very low re-capture rates in grasslands (Masterson et al., Submitted). It is thus likely that re-captures had little effect on the interpretations of abundance. The wide variation in size class in commonly captured species alone would indicate multiple individuals have been captured and that those species are indeed common in such areas.

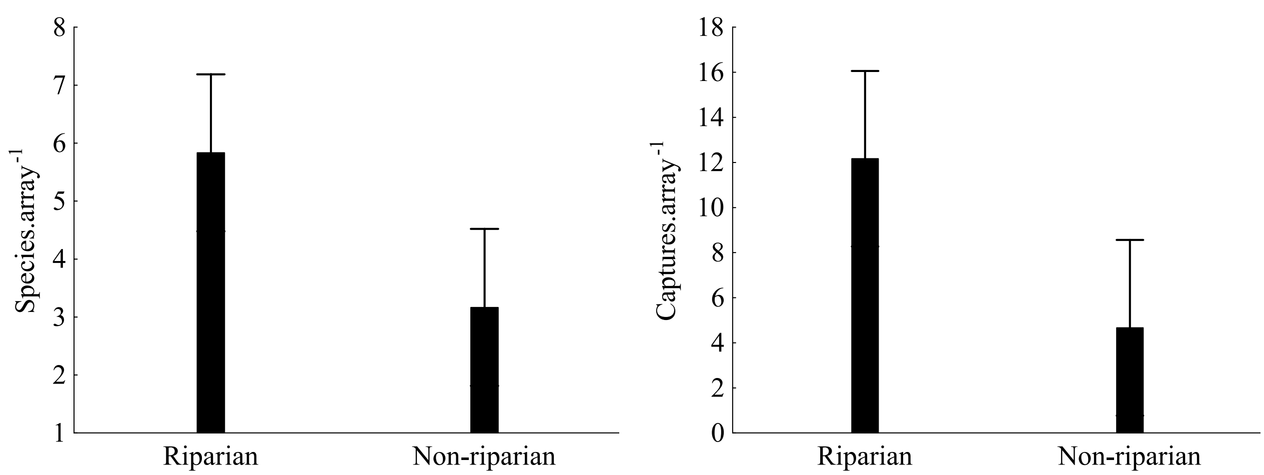

Figure 4.3: Mean species richness (left) and mean herpetofaunal abundance (right) from riparian and non-riparian areas.

Bray-Curtis similarity varied significantly between sites (ANOSIM: Global $R = 0.537$; $p < 0.004$) indicating that riparian and non-riparian sites host significantly different suites of species (Fig. 4.4).

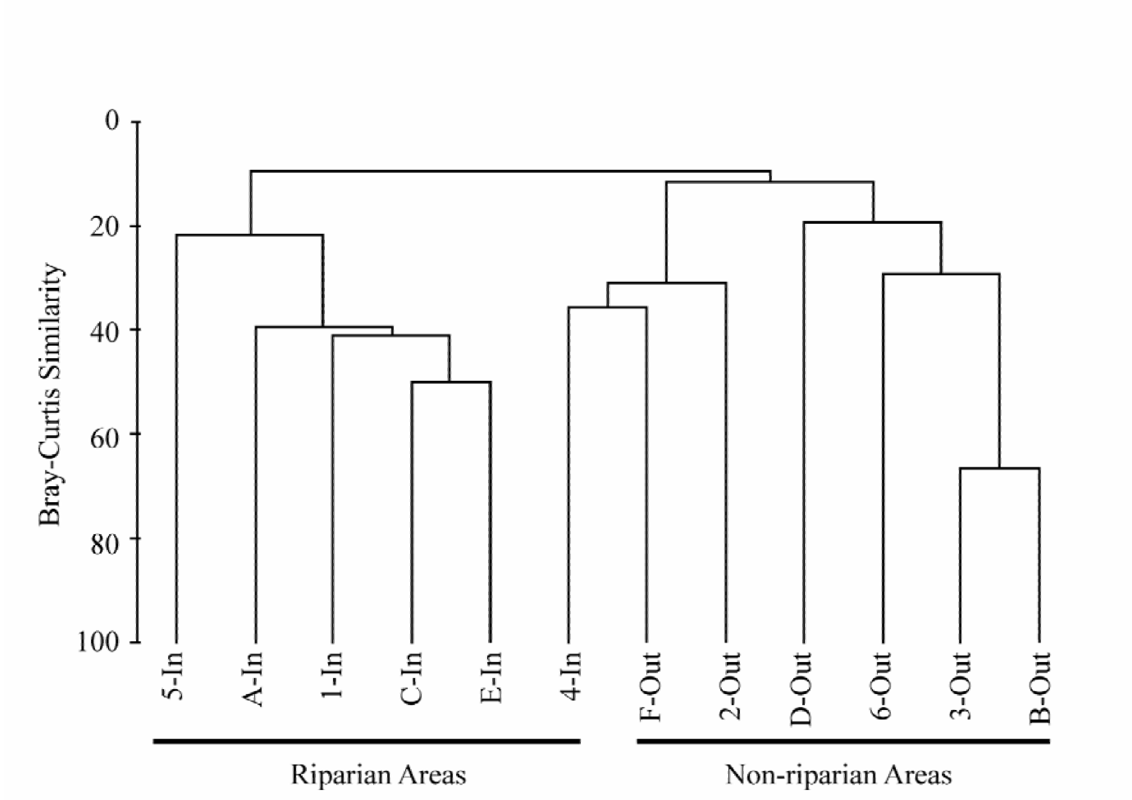

Figure 4.5: Bray-Curtis Similarity between sites in riparian and non-riparian areas.

4.4 Discussion

My data indicate that herpetofaunal diversity is significantly higher in forest areas than it is in areas under the other three, more transformed land uses on the study site, supporting the hypothesis that some of the current land uses may be a factor in suppressing herpetofaunal diversity. The significantly higher diversity in forest areas may result because forest areas represent a heterogeneous habitat relative to the homogenous habitats represented by the monocultures of sugarcane and *Eucalyptus* plantations, and to a lesser degree, grasslands. Maisonneuve and Rioux (2001) and Masterson et al. (Submitted) found that herpetofaunal abundance increased with increasing habitat complexity supporting this hypothesis.

One would expect that the susceptibility of riparian areas to flooding would make protection of such areas unsuitable as mitigatory measures. However, the major streams in the area (the Amanzinyama and Siyayi) have relatively small catchments and are represented by low stream orders on the study site, making flooding unlikely. On inspection, sections of both stream channels appear scoured, but there is little evidence of extensive flooding events.

Additionally, this investigation has demonstrated that riparian areas currently host greater species richness and herpetofaunal abundance than areas outside of riparian areas, suggesting that conservation of such areas could help to mitigate the effects of habitat removal by mining activities.

Herpetofaunal diversity was highest in the forest areas, followed equally by the two monocultures (sugarcane and *Eucalyptus*) and grassland. Grasslands on the study site appear secondary in nature, having previously been planted with sugarcane at some stage in the recent past (Douglas Saint, Pers. Comm.). Accordingly, the strong similarity between the herpetofaunal assemblages of grasslands and sugarcane plantations is not surprising. Johnson (1987) and Johnson and Raw (1989) report that sugarcane plantations on the KwaZulu-Natal north coast host surprisingly high herpetofaunal diversity (33 species), with a bias towards species usually associated with grasslands. Johnson’s (1987) and Johnson and Raw’s (1989) higher diversity estimates can be explained by the larger area over which they sampled. Little work has been conducted on the herpetofauna of *Eucalyptus* plantations in southern Africa, but *Pinus radiata* plantations in Australia have been shown to significantly reduce amphibian species richness (Parris and Lindenmayer, 2004), supporting the contention that exotic monocultures have the potential to reduce faunal diversity. Shepherd et al. (2004) have shown a reduction in groundwater levels in the study area, which is likely to have arisen as a result of afforestation. This hydrological change may significantly influence local amphibian populations.

Harvesting regimes may negatively influence herpetofaunal populations in agricultural areas. Sugarcane on the study site is burned prior to harvesting, a common practice in KwaZulu-Natal (Johnson, 1987). Frequent, intense fires may kill individual reptiles and amphibians but the effects on herpetofaunal communities remains poorly understood (Masterson, 2004; Parr and Chown, 2003). Both sugarcane and timber harvesting drastically alter habitat structure which may result in sub-optimal habitat and have a direct effect on mortality rates of herpetofauna (G. Alexander, Pers. Comm.).

Reductions in herpetofaunal diversity could have resulted from several causes. A primary cause could be the management of agricultural areas, specifically the application of pesticides and herbicides. Several authors have demonstrated how the application of such chemicals can result in the direct mortality of amphibians (Johnson, 1989; Relyea, 2005b) and reptiles (Alexander et al., 2002; Johnson, 1989). Although several of these studies suffer from application of higher concentrations of chemicals than usually applied in practice, they do highlight the potential danger that such pesticides pose to local fauna. Reylea (2005a) describes how synergistic factors can also contribute to the susceptibility of certain species to such pesticides. The land uses with the lowest diversity estimates in this investigation (grassland, sugarcane and *Eucalyptus*) have all been treated with pesticides and/or herbicides in recent years (D. Saint, Pers. Comm.) while chemical applications in forest areas have been avoided, indicating that such chemicals may have directly or indirectly reduced herpetofaunal diversity.

Only three "generalist" species (10 % of all species) were detected in all four habitats, namely the frogs *Amietophrynus gutturalis* and *Schismaderma carens*, and the lizard *Panaspis walbergi*. Several species were detected from only a single land use however most of these are represented by only one capture. Four "specialist" species were recorded on multiple occasions from only a single land use namely *Lycophidion capense* (3 individuals – sugarcane), *Lycodonomorphus rufulus* (4 individuals – forest), *Aparallactus capensis* (3 individuals – forest) and *Amblyodipsas concolor* (2 individuals – forest). These multiple captures of species from single land uses strongly influence the rarefaction prediction of high species diversity in the forest sites.

Several studies have indicated that wooded riparian buffer areas play integral roles in both bird (Whitaker and Montevecchi, 1997; Stauffer and Best, 1980) and mammal (Geier and Best, 1980; Doyle, 1990) ecology in North American forest systems. Additionally, Machtans et al. (1996) indicate that these buffers may act as corridors. The relative importance of riparian buffer strips for herpetofauna has been less thoroughly investigated.

I detected nine herpetofaunal species that were unique to riparian areas providing strong support for the preservation of such areas. These include the snakes *Amblyodipsas concolor*, *Amblyodipsas polylepis*, *Aparallactus capensis*, *Atractaspis bibronii*, *Lycodonomorphus rufulus* and *Philothamnus hoplogaster*, the lizard *Hemidactylus mabouia*, and the frogs *Phrynobatrachus natalensis*, and *Tomopterna natalensis*. The two *Amblyodipsas* spp., *Aparallactus capensis* and *Atractaspis bibronii* are fossorial snakes that spend much of their time underground or foraging in leaf litter (Branch, 1998; Shine et al., 2006). Thus it is likely that they are more abundant, if not unique to the riparian areas as these areas offer suitable microhabitat in the form of leaf litter and its associated fauna. *Lycodonomorphus rufulus* is a semi-aquatic snake (Broadley, 1983; Marais, 2004) and thus its presence in sites close to water is not surprising. Similarly, *Philothamnus hoplogaster* is a species that frequents riparian areas (Branch, 1998; Broadley, 1983). *Hemidactylus mabouia* is a wide ranging species capable of inhabiting many microhabitats. In the study area, this species tends to occur in wooded areas (including *Eucalyptus* plantations) and on buildings. Since sampling in non-riparian areas during this study did not include *Eucalyptus* plantations, its apparent absence from non-riparian areas is misleading. The frogs *Phrynobatrachus natalensis*, and *Tomopterna natalensis* are both savanna species (Channing, 2001) and may occur in non-riparian areas, but are likely to occur at greater abundances in riparian areas because of the presence of water.

Four species were only detected in non-riparian areas. These include the snakes *Psammophis brevirostris* and *Psammophis mossambicus*, the lizard *Trachylepis varia*, and the frog *Breviceps mossambicus*. *Psammophis brevirostris* and *P. mossambicus* are both grassland species that are known to occur at high densities in sugarcane plantations (Johnson and Raw, 1989) indicating that these two species would likely recolonise the area after mining from surrounding agricultural areas. *Psammophis mossambicus* is also known to occur in wetland areas (Branch, 1998) and so may well occur in the riparian areas on the study site. *Trachylepis varia* occurs in grasslands (Branch, 1998) and so its presence in only non-riparian areas is not surprising. *Breviceps mossambicus* is a fossorial frog that inhabits well-drained, sandy areas in coastal KwaZulu-Natal (Minter, 2004b) and so its absence in riparian areas is not unexpected.

The relatively high levels of both herpetofaunal species richness and abundance in riparian areas, as well as the relatively high numbers of species unique to riparian areas indicates that the protection of riparian buffer zones in the area should provide an effective conservation action, mitigating the effect of the mine on local herpetofaunal populations. Additionally, the protected riparian areas are likely to act as corridors to these and other species, facilitating recolonisation of re-vegetated areas. Of concern are the four species detected only in non-riparian habitats. Populations of these species are likely to be negatively affected to a greater degree by the mining than species that will utilise the protected buffer zones. The two *Psammophis* spp. are widely foraging species (Branch, 1998; Marais, 2004) and could thus recolonise the area post-mining from surrounding cane fields. *Psammophis mossambicus* is also known to inhabit wetland areas (Branch, 1998) and could take refuge in, and/or recolonise the area using the protected riparian buffer zones. *Breviceps mossambicus* and *Trachylepis varia* are likely to be poorer dispersers than the two *Psammophis* spp. and thus their rates of recolonisation are likely to be slower, and influenced by habitat suitability: sub-optimal habitat will take longer to be recolonised than more suitable habitat. Perhaps the most effective manner of mitigating the effect of the mine on these species would be to protect a portion of the non-riparian areas on the Fairbreeze C Ext. These areas could be rehabilitated to semi-natural grass-shrubland and additionally serve to buffer water quality in the Siyayi River (Correll, 2005; Dorioz et al., 2006).

The measured difference in species richness between the riparian and non-riparian habitats is likely to be an underestimate given the differences in structural complexity of the habitats. Terrestrial trap arrays only intercept specimens moving on the surface, reducing the probability of capturing individuals of species that frequent the fossorial, arboreal or aquatic environments. Accordingly, trapping in a complex habitat that includes woodlands and/or water bodies is likely to represent a comparatively smaller proportion of actual species richness than trapping in a simple habitat (Maritz et al., In Press). Thus the difference in species richness between the riparian and non-riparian habitats is likely to be greater than the trapping data indicate.

4.5 Conclusion

Riparian areas and forest areas host greater herpetofaunal abundance and diversity than other habitats on the Fairbreeze C Ext. Historical alteration of habitat as well as alteration through mining is likely to result in minimal faunal utilisation of the area post-mining, even after revegetation as recolonisation of mined lands by many taxa is slow (Ferreira and Van Aarde, 1996; Ferreira and Van Aarde, 1997; Kritzinger and Van Aarde, 1998; Majer and De Kock, 1992; Van Aarde et al., 1996; Vogt, 1993). The network of corridors represented by the riparian buffer zones on the Fairbreeze C Ext would potentially, should they maintain their integrity, facilitate the faunal recolonisation of the area post-mining, mitigating the effects of the resultant loss of habitat. I found that riparian areas hosted more unique species than non-riparian areas and may thus act as source areas from which numerous species can recolonise rehabilitated areas.

Chapter 5: Threatened herpetofauna of the Exxaro KZN Sands Fairbreeze C Extension mining area

5.1 Introduction

The South African National Management Biodiversity Act (2004) was promulgated with the objective of providing for the management and conservation of biodiversity in the country within the framework of the National Environmental Management Act (NEMA) (1998). In accordance with this legislation, environmental management plans must provide information on the activities proposed for the mitigation of the impacts that development could have on the environment. In order to accurately develop adequate mitigatory measures, knowledge on the presence of local fauna and flora is required, as well as the status of "threatened" taxa that may be influenced in that area. While conservation assessments of various taxa provide useful insight into such problems, not all taxa have been adequately assessed. This is particularly true for cryptic species such as reptiles and amphibians.

The first conservation assessment of South African herpetofauna was performed by McLachlan (1978). This document was superseded by the most recent South African Red Data Book for Reptiles and Amphibians (Branch, 1988), which is now somewhat obsolete. Published 18 years ago, many of the criteria used in this assessment are no longer recognised by the IUCN. Branch (1988) nonetheless acted as the major conservation tool for the herpetofauna of South Africa. Recently, the Atlas and Red Data Book of the Frogs of South Africa, Lesotho and Swaziland (Minter et al., 2004) has replaced Branch (1988), for the amphibians. The South African Reptile Conservation Assessment initiated in 2004 (www.saherps.net/sarca) is tasked with assessing the conservation status of the reptiles of South Africa, Swaziland and Lesotho, although outputs are not expected for some time yet.

Under the current scenario, a habitat suitability assessment for the different amphibian species occurring at a particular site is now a relatively simple task, but remains comparatively difficult for reptiles. Despite this obstacle, I have attempted to provide an objective assessment of the conservation status of all herpetofaunal species occurring, or likely to occur on the Exxaro KZN Sands Fairbreeze mining site, several of which are discussed below.

The primary objective of this chapter is to highlight species that could potentially be significantly negatively impacted by planned mining activities. Additionally, this chapter provides a critical assessment of the Herpetofaunal Specialist Report for the Ticor South Africa Fairbreeze C Ext ore body (Alexander, 2004a). Specifically, this critical assessment reviews Alexander's list of "Conservation Needy" species in the light of the extensive field work that I performed. Alexander (2004a) is an important report as a previous specialist report (Everard and van Wyk, 1996) proved to hold many inaccuracies and thus did not adequately assess the potential impacts of mining activities on local herpetofaunal populations. Finally, this chapter aims to provide recommendations for mitigatory measures that could be implemented to ameliorate the negative impacts of mining activities on sensitive taxa.

5.2 Methods

Relevant literature was consulted in order to determine which herpetofaunal species should be considered as being of conservation concern (Table 5.1). This task was relatively simple for amphibians given the short time that has lapsed since the completion of the South African Frog Atlas Project and its constituent conservation assessment (Harrison et al., 2001) and publication in Minter et al. (2004) by Branch and Harrison (2004). Species that were afforded a "Threatened" status (i.e., "Vulnerable", "Endangered", "Critically Endangered" or "Extinct") or "Data Deficient" status (IUCN, 2001) by either of these publications are discussed. Additionally, species listed in Alexander (2004a) are discussed. Distribution data for each species was drawn from Minter et al. (2004) and supplemented with my field observations.

The conservation status of reptile species that may occur on or near to the mining area was more difficult to determine because of the paucity of reptile distribution data and the lack of an up-to-date assessment for South African reptiles. Species included in my assessment include all those listed by Branch (1988), Hilton-Taylor (2000), and those subsequently assessed by IUCN affiliates (data from www.iucnredlist.org). Distribution data used in this assessment were taken from Broadley (1983) and Bourquin (2004). Additionally, species listed as "Conservation Needy" in Alexander (2004a) were included in my assessment.

Table 5.1: Current and previous conservation status of selected amphibian and reptile taxa occurring on or near to the Exxaro KZN Sands Fairbreeze Mine. [1]Additional conservation status proposed in specialist report; [2]Current IUCN Red List status (year indicates criteria used during assessment).

Species	Branch 1988	Hilton-Taylor 2000	Harrison *et al* 2001	Branch and Harrison 2004	Alexander 2004[1]	IUCN Redlist[2]
Amphibians						
Arthroleptis wahlbergi	-	Not Assessed	Not Assessed	Least concern	KZN Endemic	Least Concern 2001
Arthroleptella hewitti	-	Not Assessed	Not Assessed	Least concern	KZN Endemic	Least Concern 2001
Hemisus guttatus	-	Not Assessed	Near threatened	Vulnerable	KZN Endemic	Vulnerable 2001
Hyperolius pickersgilli	Rare	Vulnerable	Endangered	Endangered	KZN Endemic	Endangered 2001
Afrixalus aureus	Rare	Not Assessed	Not Assessed	Least concern	-	Least Concern 2001
Afrixalus spinifrons	-	Not Assessed	Not Assessed	Vulnerable	-	Vulnerable 2001
Leptopelis natalensis	-	Not Assessed	Not Assessed	Least concern	KZN Endemic	Least concern 2001
Breviceps sopranus	*Undescribed*	*Undescribed*	*Undescribed*	Data Deficient	-	Data Deficient 2001
Natalobatrachus bonebergi	-	Not Assessed	Endangered	Endangered	KZN Endemic	Endangered 2001
Pyxicephalus adspersus	-	Not Assessed	Near Threatened	Near Threatened	-	Least Concern 2001
Reptiles						
Pelusios rhodesianus	Peripheral	Not Assessed	-	-	-	Lower Risk: Least Concern 1994
Pelusios castanoides	Peripheral	Not Assessed	-	-	-	Lower Risk: Least Concern 1994
Python natalensis	Vulnerable	Not Assessed	-	-	-	Not Evaluated
Lamprophis aurora	-	Not Assessed	-	-	Rare	Not Evaluated
Naja melanoleuca	Peripheral	Not Assessed	-	-	-	Not Evaluated
Bitis gabonica	Vulnerable	Not Assessed	-	-	-	Not Evaluated
Bradypodion setaroi	Restricted	Endangered	-	-	-	Endangered 1994
Crocodylus niloticus	Vulnerable	Not Assessed	-	-	-	Lower Risk: Least Concern 1994

5.3 Herpetofauna of conservation concern

5.3.1 Amphibians

Arthroleptis wahlbergi Smith, 1849

Arthroleptis wahlbergi is common in many of the habitats on and adjacent to the ore body, including sugarcane, riparian forest and *Eucalyptus* plantations. This species is not considered as threatened (listed as "Least Concern" (Branch and Harrison, 2004)) and is included here because of its limited distribution (Alexander, 2004a). Mining activities will undoubtedly negatively influence local populations of *A. wahlbergi* although this species is known to rapidly recolonise disturbed areas (Channing, 2001; 2004). Maintenance of ideal habitat (riparian woodland and coastal forest) close to the ore body will facilitate such recolonisation and should be considered an important mitigatory measure for this species. *Arthroleptis wahlbergi* are "direct developers" (Channing, 2000) and thus changes in the hydrology and water quality in the Siyayi catchment are unlikely to affect this species.

Arthroleptella hewitti FitzSimons, 1947

Arthroleptella hewitti was not detected on or near to the Fairbreeze C Ext ore body, nor was any suitable habitat detected. Alexander (2004a) included this species as it is seemingly endemic to KwaZulu-Natal and has been recorded from the grid square 2831DC. This record is undoubtedly from the Ngoye area as this area represents, based on available information, the closest suitable *A. hewitti* habitat. Mining activities at the Exxaro KZN Sands Fairbreeze mine will not influence this species and thus no mitigatory actions are required.

Hemisus guttatus Rapp, 1842

Hemisus guttatus was listed as was listed as "Near Threatened" by Harrison et al. (2001) based on the species' limited area of occupancy, fragmented distribution, habitat loss and degradation, and predicted population declines. This status was changed to "Vulnerable" in Branch and Harrison (2004). Threats to this species include urban sprawl and agricultural developments (Alexander, 2004b). *Hemisus guttatus* breeds adjacent to wetlands and rivers with alluvial deposits (Alexander, 1990).

Hemisus guttatus was predicted to occur in wetland habitats adjacent to the ore body by Alexander (2004a), who also noted that individuals may also occur on the ore body whilst not breeding. I captured *H. guttatus* at two locations on the ore body, both in sugarcane plantations. It is thus evident that local *H. guttatus* populations will be directly impacted by mining activities as individuals are likely to be killed during the mechanical removal of topsoil. Alexander (2004a) suggests that this impact might be ameliorated by the removal of unearthed specimens but I do not feel that this is feasible as the probability of collecting individuals during the earthmoving activities is slim given the cryptic nature of these fossorial animals. Instead, I feel that the only feasible mitigatory action would be the protection of sites, preferably adjacent to riparian corridors. The floodplain of the Siyayi River that runs through the Fairbreeze C Ext ore body would be well suited to this. Water quality should also be monitored and controlled to minimise the impact of mining activities on populations adjacent to the ore body.

Hyperolius pickersgilli Raw, 1982

Hyperolius pickersgilli was listed as “Rare” by Branch (1988), a category no longer recognised by the IUCN (IUCN, 2001). Subsequently, the species was listed as “Vulnerable” by Hilton-Taylor (2000) and more recently its status was elevated to Endangered (Branch and Harrison, 2004; Harrison et al., 2001). The current listing was assigned to this species based on its small area of occupancy (< 500 km^2), the extent, quality and high levels of fragmentation of suitable habitat, the number of known localities, and evidence of decline in area of occupancy (Branch and Harrison, 2004; Harrison et al., 2001). Additionally, Bishop (2004) lists pollution and alien vegetation as threats to *H. pickersgilli*. Armstrong (2000) showed, using a predictive model, that this species is inadequately protected, with only 0.89 % of its predicted range falling inside protected areas.

Hyperolius pickersgilli is a small (SVL: < 30 mm) frog, endemic to the KwaZulu-Natal coast of South Africa (Channing, 2001) and is known from only nine quarter degree squares (Bishop, 2004). *Hyperolius pickersgilli* breeds in shallow (rarely deeper than 0.5 m), stagnant pans, and can be found in dense stands of Saw Grass (*Cyperus immensus*) at the edges of such water bodies (Bishop, 2004; Channing, 2001; Raw, 1982). Males appear to call between August and March (Bishop, 2004; Raw, 1982) but it is unclear if breeding takes place throughout this period.

Approximately 50 eggs are laid in a gelatinous mass on vegetation above the water (Bishop, 2004; Raw, 1982). Tadpoles emerge approximately a week later and drop into the water (Bishop, 2004). Emerging froglets (SVL: 11 – 12 mm) have been recorded in late January and early March (Raw, 1982).

Raw (1982) notes that while several other amphibian species (*Hyperolius tuberilinguis*, *Afrixalus fornasinii*, *A. aureus* (listed as *A. brachynemis*), *Leptopelis natalensis*, *Cacosternum nanum* and *Phrynobatrachus natalensis*) were found to share water bodies with *H. pickersgilli*, *H. marmoratus* "appeared to avoid the areas preferred by *H. pickersgilli*". Raw (1982) speculates that this is a result of the fact that eggs of *H. marmoratus* are submerged in water while the eggs of other syntopic Hyperoliids are not. While this finding may be coincidental and based on a small number of observations, a strong presence-absence relationship between these two species could be utilised as an indicator of habitat suitability. However, a better understanding of the habitat requirements of *H. pickersgilli* is required in order for this to be useful.

I did not detect the presence of *H. pickersgilli* in or around the Exxaro KZN Sands Fairbreeze C Ext ore body, despite the fact that it has been recorded from the Mtunzini area (Bishop, 2004). However, *H. pickersgilli* is known to be cryptic and capable of escaping detection for long periods of time (Bishop, 2004). Like Alexander (2004a), I did not detect any suitable *H. pickersgilli* habitat on the ore body. Additionally, very little suitable habitat was detected nearby. It is my opinion that a reduction in the water table as a result of agricultural activities in the area (Shepherd et al., 2004; Van der Elst et al., 1999) has reduced the number of dune slacks that have the potential to host *H. pickersgilli*.

Should *H. pickersgilli* occur adjacent to the ore body, mining activities could have negative or positive impacts. Increased water run-off resulting from mining activities (Shepherd et al., 2004) could re-establish suitable habitat by raising the water table so that wetlands form in dune slacks. This positive impact is likely to be ephemeral as the mining is only scheduled for a period of less than 15 years (R. Hattingh, Pers. Comm.), after which agricultural practices will undoubtedly lower the water table again (Shepherd et al., 2004). Reduction in water quality may not influence this species significantly as it breeds in stagnant water and is thus clearly capable of tolerating low oxygen levels. The degree to which *H. pickersgilli* can tolerate reduction in water quality remains to be tested.

I propose a three-tiered conservation plan for the re-establishment/ enhancement of *H. pickersgilli* in the Mtunzini area. Each step of the plan (Fig. 5.1) should be followed by a monitoring period of predetermined duration to assess the success of that step and inform decision making regarding further steps.

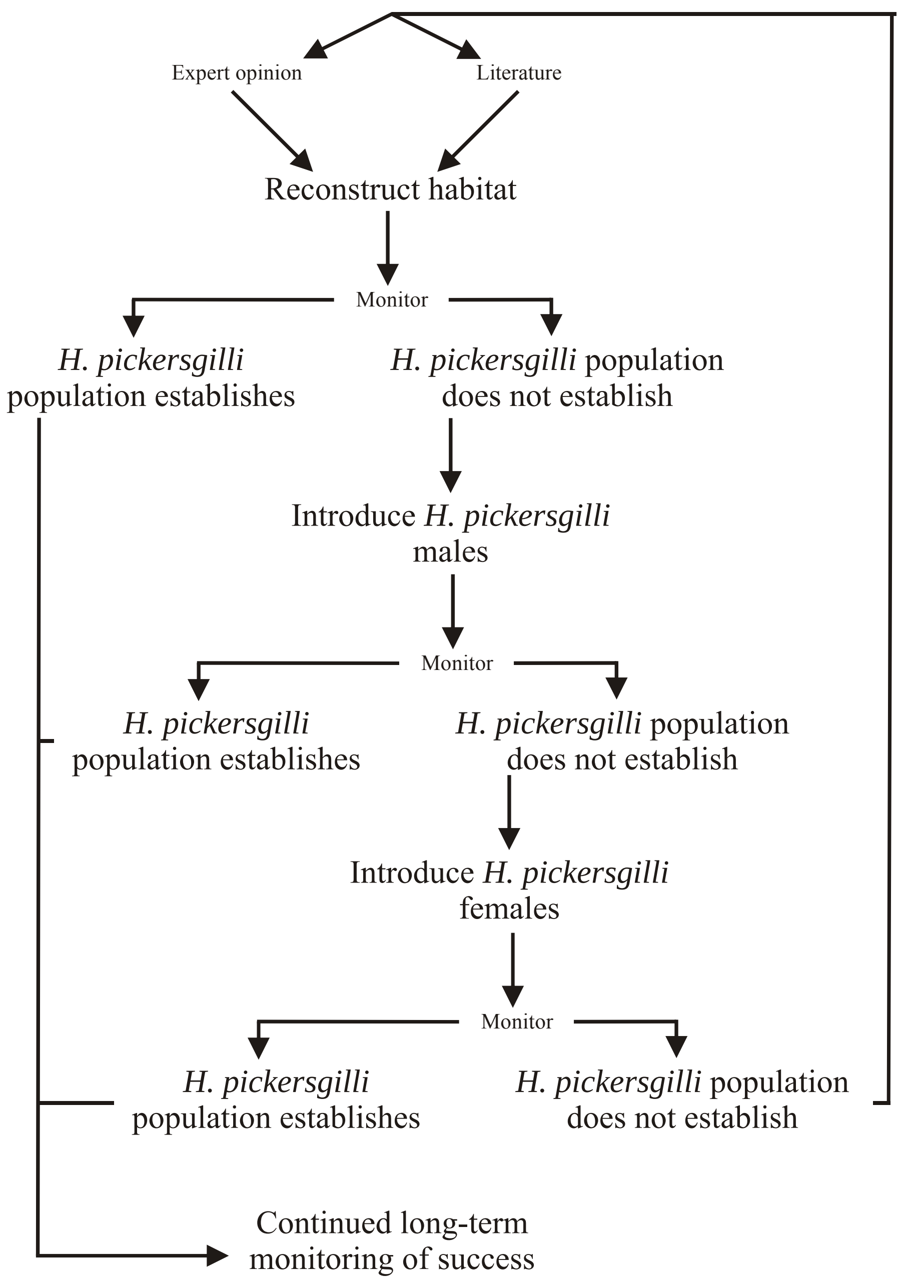

Figure 5.1: Conservation plan for the re-establishment of *Hyperolius pickersgilli* in the Mtunzini area.

Re-establishment of anuran populations through translocation has been attempted for several species with varying success (Dodd and Siegel, 1991; Fischer and Lindenmayer, 2000). Dodd and Siegel (1991) indicate that herpetofauna are poor candidates for translocations as there is little reported evidence for success in such projects. Dodd and Siegel (1991) also state that "…no RRT (Relocation, Repatriation and Translocation) project has yet established a self-sustaining population of snakes, turtles, frog or salamanders" but Burke (1991) lists several herpetofaunal taxa that have successfully colonised areas, often as invasive exotics. Alexander (1990) notes the successful reintroduction of *Hyperolius pusillus* and *H. tuberilinguis* into suitable habitat at Pigeon Valley, Durban, indicating that Hyperoliids may be suitable candidates for translocations.

Initial action involves the development of suitable *H. pickersgilli* habitat based on the available literature and consultation with experienced herpetologists (Fig. 5.1). The proposed wetland restoration was initiated by Exxaro KZN Sands and has, to my knowledge, been agreed to in principal. Steps are in place to restore a section of the Fairbreeze C Ext property (currently under sugarcane production) for this purpose. In order for this wetland reserve to meet its objectives (provide suitable habitat for wetland associated fauna, particularly *H. pickersgilli*), habitat reconstruction should focus on recreating *H. pickersgilli* habitat as described in the available literature on the species (discussed above), the literature on creating wetlands for amphibians (Babbit, 2005; Porej and Hetherington, 2005) and with the consultation of experienced herpetologists familiar with the species. Action 1 is to be followed by a monitoring period to determine if *H. pickersgilli* has colonised the wetland. The monitoring period should be approximately two breeding seasons in duration and have the primary objective of detecting the presence of *H. pickersgilli*. Should *H. pickersgilli* be detected, a long-term monitoring project should be established to ensure the continued survival of the population. Should no *H. pickersgilli* individuals be detected during this period, Action 2 should be initiated.

Action 2 involves the translocation of male *H. pickersgilli* individuals to the wetland from a suitable source population. Males are easier to collect than female because their locations can be betrayed by their vocalisations. Translocation should follow all IUCN guidelines for introducing species (IUCN, 1998) and should be cleared with all relevant authorities. Theoretically, introduced males can "call" local females (if any exist) to the wetland and thus establish a breeding population. Success at this stage would be advantageous as it would potentially reduce genetic mixing between introduced individuals and the local population (should one exist). Again, this action should be followed by a similar monitoring period, aiming to monitor male *H. pickersgilli* numbers and detect female *H. pickersgilli* individuals. Detection of female *H. pickersgilli* and maintenance of male *H. pickersgilli* numbers should initiate a long-term monitoring project to ensure the continued survival of the population. Failure to detect *H. pickersgilli* females should initiate Action 3.

Action 3 requires the translocation of both male and female from a suitable source population, following all IUCN guidelines (IUCN, 1998) and informing all relevant authorities. Again, a long-term monitoring project should be established to ensure population survival.

Afrixalus aureus Pickersgill, 1984

Afrixalus aureus (misspelled as *Afrixalus aurens* in Alexander (2004a)) was relegated to "Least Concern" (Branch and Harrison, 2004) after being listed as "Rare" by Branch (1988). This species has never historically been recorded in the Mtunzini area (Pickersgill and Bishop, 2004), nor was it detected during my fieldwork. Additionally, no suitable habitat was detected on the ore body (Pers. Obs.; Alexander, 2004). In the unlikely event that this species does in fact occur in wetlands adjacent to the ore body, maintenance of wetland habitats and water quality control, as required for mitigation of impacts against other local amphibian species, would suffice.

Afrixalus spinifrons (Cope, 1862)

Afrixalus spinifrons is a small frog that is known from the Mtunzini area (Pickersgill et al., 2004) and is currently listed as "Vulnerable" (Branch and Harrison, 2004) despite being excluded from assessment in Harrison et al. (2001). The major threat to this species, as with many amphibians, is habitat degradation and fragmentation resulting from anthropogenic activities including sugarcane production, timber production and urban sprawl, as well as invasive exotic vegetation (Pickersgill et al., 2004).

Afrixalus spinifrons breeds in standing water with abundant surface vegetation (Pickersgill, 2004). No such habitat was found on the ore body, but suitable habitat is available close-by. Although *A. spinifrons* was not detected in suitable habitat on the periphery of the ore body, *A. delicatus*, which is known to be syntopic with *A. spinifrons* in areas of sympatry such as Mtunzini

(Pickersgill and Bishop, 2004) was detected. These species have similar vocalizations, a “zip” followed by a “trill”, as well as a “rather quiet call” (Pickersgill et al., 2004) and so *A. spinifrons* may have been overlooked in the mentioned wetlands adjacent to the ore body.

Mining activities could negatively influence local *A. spinifrons* populations through reduction in water quality and loss of habitat. Mitigation should include the maintenance of wetland areas adjacent to the ore body and strict control of water quality. The proposed wetland targeting *Hyperolius pickersgilli* could be engineered to include components of suitable *A. spinifrons* habitat.

Leptopelis natalensis (Smith, 1849)

Alexander (2004a) listed *Leptopelis natalensis* as “conservation needy” based on the fact that it is “largely” endemic to the province of KwaZulu-Natal. I detected *L. natalensis* along the northern and eastern margins of the Fairbreeze C Ext ore body, as well as in riparian forest along the upper sections of the Siyayi River.

I agree with Alexander (2004a) that maintenance of wetland habitats along the margin of the ore body would largely mitigate the effects of mining activities on local populations of this species. I additionally recommend that riparian woodlands along the Siyayi River be protected. Water quality maintenance may not be as important to this members of this genus as it is to many other species as tadpoles have been collected from “…water so dirty that it might be described as thin mud” (Channing, 2001). *Leptopelis natalensis* is a species that is likely to gain from the creation of a wetland targeting *Hyperolius pickersgilli* establishment.

Breviceps sopranus Minter, 2003

Breviceps sopranus is the only species known from the area that is afforded the IUCN status of “Data Deficient” (Branch and Harrison, 2004). Recently described, very little is known about the distribution, ecology and conservation status of this species. Currently, Mtunzini represents the southern limit of the range, but the current distribution data are likely to be incomplete (Minter, 2004c). Minter (2003) states that advertisement call is the only reliable character that can be used to distinguish this species from sympatric *Breviceps* species.

While *B. sopranos* has been recorded from Mtunzini (Minter, 2003; 2004) I did not hear this species calling on or adjacent to the Fairbreeze C Ext ore body (although *B. mossambicus* was heard). It is unlikely that this species occurs on or immediately adjacent to the Fairbreeze C Ext ore body and is thus unlikely to be directly affected by mining activities. Additionally, *Breviceps* spp. are “direct developers”, not requiring water in which to breed (Channing, 2001). Accordingly, changes in flow regimes of the Siyayi River are unlikely to negatively influence *Breviceps sopranos.*

Natalobatrachus bonebergi Hewitt and Methuen, 1913

Natalobatrachus bonebergi was not detected on or near to the Fairbreeze C Ext ore body, nor was any suitable habitat detected. This species is listed as “Endangered” (Harrison et al., 2001, Branch and Harrison, 2004) based on its restricted area of occupancy, the fragmented nature of that habitat, and evidence for reduction in area of occupancy (du Preez, 2004). *Natalobatrachus bonebergi* occupies fast flowing forest streams, often in ravines (du Preez, 2004). Like *Arthroleptella hewitti*, the nearest suitable habitat is likely to be in the Ngoye area. Exxaro KZN Sands Fairbreeze mining activities will not influence *N. bonebergi* populations.

Pyxicephalus adspersus Tschudi, 1838

Alexander (2004a) included commentary on this species as Everard and van Wyk (1996) incorrectly included it in their assessment. Alexander (2004a) correctly states that *Pyxicephalus adspersus* does not occur in KwaZulu-Natal (Du Preez and Cook, 2004) and that northern KwaZulu-Natal records of *Pyxicephalus* sp. are referable to *P. edulis* which is not of conservation concern. *P. adspersus* will in no way be impacted by the Exxaro KZN Sands Fairbreeze mine.

5.3.2 Reptiles

Pelusios rhodesianus Hewitt, 1927

Pelusios rhodesianus was listed as “Peripheral” by Branch (1998) but de-listed by Hilton-Taylor (2000) and is thus currently listed as Lower Risk (Least Concern) (Tortoise and Freshwater Turtle Specialist Group, 1996b). This species inhabits a wide variety of habitats in southern Africa including temporary pans and vleis and more permanent dams and lakes (Boycott, 1988b).

I did not detect *P. rhodesianus* during my fieldwork although search effort was limited. Like other *Pelusios* spp., *P. rhodesianus* is likely to bask in exposed sites during the day (Spawls et al., 2002) limiting occurrence to water bodies with basking sites, of which very few occur on or adjacent to the ore body. Since no habitat is available, *P. rhodesianus* is unlikely to be negatively impacted by Exxaro KZN Sands Fairbreeze mining activities.

Pelusios castanoides Hewitt, 1931

Although the subspecies *P. c. intergularis* is listed as "Critically Endangered" (Gerlach, 2003), the nominate subspecies *P. c. castanoides*, known from southern Africa is listed as Lower Risk (Least Concern) (Tortoise and Freshwater Turtle Specialist Group, 1996a). The peripheral and discontinuous nature of the distribution of *P. c. castanoides* in South Africa may warrant regional protection for the species. Alexander (2004a) included this species on the basis of its inclusion by Everard and van Wyk (1996), but notes that the Fairbreeze C Ext ore body probably lies outside the geographic range of the species (Boycott, 1988a) and is thus unlikely to be impacted by mining activities. I did not detect *P. castanoides* during my fieldwork although search effort was limited. Nonetheless, I agree with Alexander (2004a) that the Fairbreeze C Ext occurs south of *P. castanoides* range and that mining is unlikely to impact on this species.

Python natalensis (A. Smith, 1840)

Southern African Python (*Python natalensis*), although listed as "Vulnerable" in Branch (1988), has not been assessed using current IUCN international criteria. As a result, it is not Red Data listed by Hilton-Taylor (2000). *Python natalensis* occurs from east Africa to southern Africa (Broadley, 1983; Broadley, 1999) including the eastern parts of South Africa (Branch, 1998, Marais, 2004). Over much of its range, *P. natalensis* occurs at high densities (Pers. Obs.; Alexander, 2004a). *P. natalensis* is unlikely to maintain its "Vulnerable" status under current IUCN criteria.

Specimens were observed on and adjacent the Fairbreeze C Ext ore body and appear to be common in the Mtunzini area despite local opinion (Pers. Obs.). The local population of *P. natalensis* is likely to be negatively influenced by mining activities largely through habitat transformation. That said, there is adequate habitat surrounding the ore body for the local population to persist during mining activities, and given that these animals are capable of moving many kilometres in a few days (G. Alexander, Pers. Comm.), unassisted recolonisation of the site post-mining is likely.

These large predators are charismatic and their needless killing should be discouraged. Education of mine employees as to the inherent value of these snakes would help to ensure that encountered specimens are properly dealt with. Since they are large (possibly up to 5 m in length: Branch, 1998), encountered specimens can be easily identified, facilitating removal and release into suitable habitat. Additionally, the development of an appropriate "problem animal" removal protocol would help to resolve animal – person conflicts. It is important to bear in mind that Southern African Pythons can have extensive home ranges (greater 500 ha: Alexander, unpubl. data) and that removal of a snake does not ensure that it will not return to the site at some later stage.

Lamprophis aurora (Linnaeus, 1754)

Alexander (2004a) included this species in his assessment based on its rarity and its dependence on grasslands, a highly transformed biome in South Africa (le Roux, 2002). While this species has been recorded nearby (2831DB; 2831CD) there is little evidence to suggest that they should occur to the coastal areas of KwaZulu-Natal as it is a temperate species restricted to cooler altitudes but reaching the coast in the Western Cape and Eastern Cape provinces (Broadley, 1983). Additionally, this species has never been formally listed in previous conservation assessments (Branch, 1988; Hilton-Taylor, 2000; McLachlan, 1978) nor is it likely to be listed in a threatened category in the upcoming Southern African Reptile Conservation Assessment (SARCA). I did not detect this species on or adjacent to the Fairbreeze C Ext ore body, and while grasslands do occur on the ore body, there is little biogeographic reason to suggest that they occur on the site. It is highly unlikely that this species will be influenced by the Fairbreeze mining activities.

Naja melanoleuca Howell, 1857

Naja melanoleuca was listed as "Peripheral" by Branch (1988) as this widespread species reaches its southern limit in northern KwaZulu-Natal (Broadley, 1983; Spawls and Branch, 1995; Spawls et al., 2002). The peripheral nature of *N. melanoleuca* distribution in South Africa may result in the species being afforded regional protection. It is unclear whether *N. melanoleuca* represents a single species or a species complex (Spawls and Branch, 1995), however, it is unlikely that changes in the taxonomy will affect its conservation status as this hardy species has broad dietary preferences and can inhabit various habitats (Spawls and Branch, 1995; Spawls et al., 2002). Alexander (2004a) intimates that this species is not likely to be found as far inland as the Fairbreeze C Ext ore body and is thus not likely to be influenced by mining activities. However, I found *N. melanoleuca* to be fairly common in forested areas on and around the ore body. Additionally, I removed a specimen from a farm house surrounded by sugarcane plantations near Gingindlovu (29°02’ S, 31°38’E). This record indicates that the species occurs both away from forested areas and a substantial distance inland.

Naja melanoleuca is an active and alert snake that is likely to move out of areas before they are mined. While mining activities will undoubtedly impact on local populations through habitat loss, the most severe impact is likely to come through persecution of specimens coming into contact with people. Additionally, specimens could be killed by increased vehicle traffic associated with mining activities. Mitigatory actions to reduce the impacts of mining activities on *N. melanoleuca* include the preservation of riparian woodlands and forested areas adjacent to the ore body. Additionally, the relocation of specimens encountered in buildings to preserved forested areas would reduce the impact that mining activities would have on local *N. melanoleuca* populations.

Bitis gabonica (Duméril and Bibron, 1854)

Bitis gabonica is currently Red-listed as “Vulnerable” in South Africa (Branch, 1988). Although the species is not listed internationally, two characteristics of the South African populations results in it being afforded this protection. Firstly, South African populations are peripheral to the main species distribution. Secondly, local populations appear to be threatened by illegal collecting and habitat modification (Branch, 1988). The peripheral characteristic of the species’ distribution however, does not hold weight under the current global IUCN Red List criteria (IUCN, 2001), and thus *B. gabonica* is not likely to regain its “Vulnerable” status upon international re-evaluation. However, given the peripheral and isolated nature of this species’ distribution in South Africa, *B. gabonica* may be afforded regional protection.

Bitis gabonica occurs throughout most of sub-Saharan Africa, southwards to Angola, Zambia and Mozambique with limited occurrence in southern Africa (Broadley, 1983) where it is restricted to the eastern highlands of Zimbabwe, western central Mozambique and northern Zululand, KwaZulu-Natal, South Africa (Branch, 1998). In South Africa, *B. gabonica* has historically been recorded along the coast from the Mozambique border, southwards to 2832CB. It should be noted that neither the historical nor current distribution of the species is known with any degree of certainty as low population densities, exceedingly cryptic behaviour, and illegal collecting have resulted in poor locality records for the species.

Between March 1995 and October 1998 a total of 97 Gaboon Adders were translocated from the Dukuduku area of northern Zululand to the Umlalazi (Mtunzini) area (A. Armstrong, Pers. Comm.; Alexander, 2004a; Bodbijl, 1994). Translocations were implemented as a reactionary conservation measure following perceived, broad-scale habitat modification in the Dukuduku area (Bodbijl, 1994). The Umlalazi area was selected as the destination for these translocations as it offered suitable habitat in terms of structure and prey availability (Bodbijl, 1996; Perrin and Bodbijl, 2001). Translocations were stopped in 1998 as the translocation protocols did not comply with IUCN guidelines (A. Armstrong, Pers. Comm.).

Given the that the translocation of the Dukuduku snakes was to a site outside of the historical distribution of the species, the conservation value of the translocated population becomes questionable as such populations often have reduced fitness and rarely survive (Dodd and Siegel, 1991; Fischer and Lindenmayer, 2000). Additionally, available evidence suggests that South African populations are not distinct from other African populations despite their apparent geographical isolation. Wüster (unpubl. data) compared the cytochrome *b* sequence from a South African *B. gabonica* and a published sequence from a snake from the eastern Democratic Republic of Congo. Sequence divergence was 0.8 %, which is very low given the distance between the localities. This evidence suggests recent genetic exchanges between the populations (W. Wüster, Pers. Comm.).

During and subsequent to my fieldwork, Gaboon Adders were detected in Mtunzini (Pers. Obs.), near the main entrance of the Umlalazi Nature Reserve (Boughey, 2006), and on the Twinstreams Road (J. Cromhout, Pers. Comm. and verified photo record). All of these records were of adult snakes, offering little information regarding the breeding status of the local population. Discovery of gravid females or neonate snakes would produce useful insight into the status of the "Umlalazi" population as it would indicate breeding. Until such time I think we should assume that the "Umlalazi" population is not sustainable.

Perrin and Bodbijl (1994) indicate that Gaboon Adders require specific habitat characterised by prey availability for survival. Warner (unpubl. data) alternatively proposes that *B. gabonica* is an opportunistic predator and that habitat selection is based on thermal preferences. As discussed in previous chapters, habitat on the Fairbreeze C Ext ore body has been modified by agricultural practices (specifically sugarcane production and forestry). However, seemingly suitable habitat is available adjacent to the Fairbreeze C Ext ore body and thus mining activities could indirectly affect *B. gabonica*.

Given the reduced conservation value of the "Umlalazi" population and the likely population status, it is difficult to justify intensive habitat preservation specifically for *B. gabonica*. I propose the development of the "problem animal" relocation protocol that will be suitable for dealing with Gaboon Adders that may be encountered. I also suggest that a *B. gabonica* monitoring program should be set up to catalogue all *B. gabonica* records from the area. This information will be useful in monitoring the status of the local *B. gabonica* population and inform management decisions around this species.

Bradypodion setaroi Raw, 1976

Bradypodion setaroi is the only reptile species presented in Alexander (2004a) that has been afforded international protection under current IUCN criteria. This small chameleon is listed as "Endangered" by Hilton-Taylor (2000) based on its limited geographic distribution and habitat degradation within that range (World Conservation Monitoring Centre, 1996). Branch (2002) suggests that this status is inflated, with Least Risk (Near Threatened) being a more appropriate status. The southern most distribution record for this species is Richard's Bay (2832CC) and thus it is not likely to occur in on or adjacent to the Fairbreeze C Ext ore body. Neither my search efforts nor Alexander's (Alexander, 2004a) search efforts yielded any specimens. Additionally, very little (if any) suitable habitat was detected on the ore body. *B. setaroi* is thus not likely to be influenced by the Fairbreeze mining activities.

Crocodylus niloticus Laurenti, 1768

Crocodylus niloticus is listed as “Vulnerable” in Branch (1998) but not Red Listed by Hilton-Taylor (2000) or The Crocodile Specialist Group (1996). While the species may be facing anthropogenic pressure throughout much of its South African range (Jacobsen, 1988), it does occur in large numbers in numerous protected areas in Limpopo Province, Mpumulanga and KwaZulu-Natal.

I did not observe any Crocodiles in water bodies adjacent to the ore body although a few individuals have recently been observed in the Umlalazi River (C. Beattie, Pers. Comm.). Alexander (2004a) is of the opinion that the Siyayi River probably only hosts Crocodiles on a transient basis and does not form important conservation habitat. While I agree with this assertion, I think that increased flow resulting from mining activities (Shepherd et al., 2004) could improve habitat suitability and thus suitable habitat should be monitored for the presence of crocodiles.

5.4 Discussion

In all, I have discussed eight reptile and ten amphibian species that have been flagged as being of conservation concern. The information and recommendations made in the preceding text can help to mitigate the effect of mining activities in the area of concern and provide a platform for the persistence of those species in that area, and should therefore be included in the development of any environmental management strategies.

Everard and Van Wyk (1996) produced a general ecology report for the Hillendale and Fairbreeze. The broad scope of the report resulted in several inaccuracies of varying relevance to the herpetofauna. Several names are misspelled or outdated (at time of publication), while other species are included without much evidence (either museum or literature records) for their presence in the area. In contrast, Alexander (2004a) presented a far more thorough representation of the potential impacts of mining. Alexander’s predictions of occurrence and assessment of potential impacts of mining activities are far more accurate than previous assessments.

I have outlined several mitigatory measures that could facilitate the preservation of herpetofaunal species in the study area during mining operations. These include the rehabilitation of a wetland adjacent to the Fairbreeze C Ext ore body, the preservation of riparian buffer zones, the development of a "problem animal" relocation protocol, and a Gaboon Adder monitoring forum.

Rehabilitation of the wetland environment, if achieved, will provide suitable habitat for numerous species of animals, particularly amphibians that are dependent on wetland habitats. Since a key motivation for the development of the wetland is to provide suitable habitat for the endangered frog, *Hyperolius pickersgilli*, wetland structure should aim to recreate the habitat requirements of this species. While I have discussed the known habitat requirements for this species, very little is known in this regard and the habitat requirements of this species certainly warrant further investigation and quantification. It is not within the scope of this report to describe the structure of the proposed wetland but I do suggest that the development of the wetland incorporates both biological and sound environmental engineering practices, failing which the project could result in further damage to the surrounding environment.

Preservation of riparian buffer zones along watercourses on and around the Fairbreeze C Ext ore body will have a dual function. Buffer zones have the potential to both provide habitat for numerous taxa, as well as removal of hazardous wastes from surface water before it enters the main river system (Correll, 2005). Buffer zones often represent a trade-off between effective function and cost and thus should aim to be as wide as feasibly possible, following all relevant environmental legislation. Buffer zones should be restored to "natural" states and closely monitored for the presence of both invasive exotic vegetation, and indigenous fauna.

During the course of the mining operations "problem animals" will be occasionally encountered either on the ore body or even in buildings on the ore body. While many smaller herpetofauna, especially small snakes lizards and frogs will pass unnoticed, larger potentially dangerous animals may need to be translocated, especially where animals enter buildings. Such animals need to be safely captured and released nearby in suitable habitat. The development of this protocol should coincide with an education program that informs all relevant employees of the animals that they are likely to encounter and how to react in such a situation.

Chapter 6: Discussion of outcomes

This study has multiple aims. These include: the provision information needed for the development of proactive mitigatory measures of the effects of mining on species of herpetofauna; the provision of conservation protocols to protect selected species and habitats in and around the Fairbreeze C Ext site; and provide insight into the biology of fossorial herpetofauna and the techniques best suited to surveying them.

6.1 Major outcomes

Field surveys, opportunistic captures, and a review of the literature have allowed me to develop a comprehensive assessment of the herpetofaunal assemblage and possible impacts associated with the Fairbreeze C Ext mining activities. Such data, along with knowledge gathered regarding the habitat requirements and biogeographic patterns, has also allowed me to critically review and correct existing inventories for Fairbreeze C Ext. For example, Everard and Van Wyk (1996) include the lizards *Trachylepis homalocephala* and *Bradypodion setaroi* in their inventory, neither of which are likely to occur on the site. Conversely, Alexander (2004a) excluded *Naja melanoleuca* from his inventory, but extensive fieldwork confirmed its presence on the study site. Everard and Van Wyk (1996) and Alexander (2004a) excluded *Naja annulifera* from their inventories, but both of these species were detected on the study site following intensive surveys.

Survey efforts have also confirmed the presence of several threatened species (Branch, 1988; Branch and Harrison, 2004, www.iucnredlist.org). These include the reptiles *Python natalensis* (Vulnerable), *Naja melanoleuca* (Peripheral) and *Bitis gabonica* (Vulnerable), as well as the amphibian *Hemisus guttatus* (Vulnerable). These species require special conservation attention and proactive planning to mitigate the effects of the mining activities. The threatened chameleon *Bradypodion setaroi* (Endangered) was not detected despite intensive search. Chameleons are readily detectable at night time, and failure to find any individuals on the site indicates that it is highly likely that this species does not occur on the site and will thus not be affected by mining activities. The threatened amphibians *Hyperolius pickersgilli* (Endangered) and *Afrixalus spinifrons* (Vulnerable) were also not detected in suitable habitat on the periphery of the mining areas, but these two species are exceedingly cryptic and can go undetected for long periods of time.

The difficulty of detecting species remains a major hurdle in herpetofaunal conservation. Once a species is detected, its presence in an area is highly probable, but the absence of a species in an area is far more problematic to ascertain. This is particularly true for cryptic species like many reptiles and amphibians that often form part of diverse communities, but are not abundant. The result is that despite intensive survey efforts, new species may continue to be discovered at a survey site as survey effort increases (i.e., the collecting curve asymptotes exceedingly slowly). For example, Pickersgill's Reed Frog (*Hyperolius pickersgilli*) was only detected in the Twinstreams area (near the study site) after several years of amphibian surveys (Bishop, 2004). Unfortunately these species, because of their cryptic nature (and accordingly the fact that little is known of their biology) often require some form of protection, yet are also simultaneously overlooked during surveys. Currently, the only way of minimising this problem is by increasing survey intensity and the employment of numerous techniques, at various times of the year.

In this particular survey, sufficient data were collected to make inferences as to the habitat preferences of most of the herpetofaunal species whose presence have been confirmed at Fairbreeze C Ext. Several species were only encountered or collected in riparian woodlands, supporting the motivation for protection of these habitats. The data also indicate that agricultural monocultures in the areas have low herpetofaunal diversity and abundance, and thus generally have low conservation value. All three of the 'Threatened' snake species on Fairbreeze C Ext, occur primarily in forested habitats, supporting proposals to conserve these habitats.

The apparent reduction in herpetofaunal diversity and abundance from the sugarcane and *Eucalyptus* monocultures is concerning. I have proposed that the synergistic effects of chemical treatments, habitat homogeneity and harvesting regimes could be suppressing diversity. The widespread occurrence of these agricultural practices (Driver et al., 2005) suggests that these negative impacts on diversity are of conservation importance.

Reduction in diversity and abundance of herpetofauna in agricultural monocultures has implications for the post-mining rehabilitation of Fairbreeze C Ext. Planned restoration activities include the replacement of sand and sludge mix, the replacement of stockpiled topsoil, dune shaping, and revegetation (Lubke and Avis, 1999) with sugarcane. Thus, it is likely that recolonisation from surrounding areas will be slower than recolonisation into optimal habitat. Galan (1997) showed that herpetofaunal recolonisation of mine spoils is Spain was closely correlated with habitat development, suggesting that herpetofauna were more likely to recolonise areas with habitats that approximated natural conditions. Thus rehabilitation of mined areas back to natural habitat is preferable to returning the land to sugarcane production.

Regardless of the restoration endpoint chosen by the mine, recolonisation will undoubtedly depend on herpetofaunal diversity and abundance of adjoining areas: if adjacent habitats host few species and at low abundances, recolonisation of restored areas will be slower. Thus protection of suitable habitats adjacent to mining areas, as well as corridors (such as those represented by the riparian areas) is important in facilitating rehabilitation of mined areas. This is particularly true as the majority of the herpetofaunal species occurring on the site inhabit riparian areas.

Other major findings of this investigation are that fossorial herpetofauna are not uniformly distributed across the landscape, and thus that they can occur a very low densities at the landscape level. Many fossorial species are assumed to form a uniformly abundant, if very cryptic component of the herpetofaunal community. My data show this not be a valid assumption. Additionally, I have shown logically that surveying fossorial herpetofauna in an area of optimal habitat (as evidenced by the presence of fossorial herpetofauna) and extrapolating the measured densities across a landscape is likely to produce overestimation of abundance of fossorial species.

The differences in patterns between total herpetofaunal diversity in South Africa and fossorial herpetofaunal diversity, particularly when viewed in a biogeographic framework, highlight the need for further fossorial herpetofaunal surveys. Of particular interest is the degree to which fossorial herpetofauna and indeed fossorial vertebrates, are protected by current conservation protocols and protected areas?

I was unable to show unequivocally which fossorial herpetofaunal survey technique is most appropriate. Indeed, chapter 3 discussed the advantages and disadvantages of the two techniques I employed. An additional consideration is that traditional terrestrial trap arrays may adequately detect fossorial herpetofauna. Terrestrial trap arrays are certainly easier and cheaper to install than the excavation techniques described in chapter 3. The major drawback to the use of terrestrial trap arrays when surveying fossorial herpetofauna is that such traps explicitly capture only those organisms moving on the surface and may thus exclude species that are not active at the time of trapping.

6.2 Recommendations

This section aims to outline the recommendations that I believe Exxaro KZN Sands should implement in order to ameliorate the effects of mining activities on the local herpetofauna of the Fairbreeze C Ext mining area. While some of these have been dealt with extensively in Chapter 5, others are novel, and based on overriding themes of conservation planning, rather than as reactionary mitigation efforts.

6.2.1 Preservation of riparian areas

Chapter 4 demonstrated that riparian areas along the Siyayi River host the greatest diversity and abundance of reptiles and amphibians on the study site. Included in the species assemblage of riparian areas are numerous species that were not detected anywhere else on Fairbreeze C Ext site. Conservation of these riparian areas will serve to protect species associated with the riparian areas, providing refuge for animals during mining activities and corridors to facilitate recolonisation of revegetated areas post-mining. Since certain species were found only outside of riparian areas, preserved areas should include non-riparian buffer that could be rehabilitated to grassland.

6.2.2 Active restoration of wetland

Chapter 5 describes a protocol for the establishment of a wetland targeting, the re-establishment of the threatened frog *Hyperolius pickersgilli*. The protocol involves active creation of suitable habitat, followed by monitoring and active introduction if necessary. All actions need to follow the relevant guidelines on wetland creation (e.g., Thompson and Luthin, 2004; Wyatt, 1997) and translocation of animals (IUCN, 1998) and inform the relevant authorities. The wetland is likely to benefit numerous other species of amphibians as well as other taxa such as mammals and birds.

6.2.3 Development of a problem animal removal protocol

During the course of mining, human-animal conflicts are likely to arise. Of particular concern are human-snake encounters that often arise when snakes enter buildings. A snake removal protocol should be developed that would allow for snakes to be safely (for the snake and humans) removed from buildings and released in suitable nearby habitat. Long distance relocations (species specific though generally > 1 km) are not recommended as translocations can result in undesirable genetic mixing, the transfer of pathogens, and increased mortality in translocated animals (Edgar et al., 2005; Plummer and Mills, 2000; Seigel and Dodd, 2002; Shine and Koenig, 2001; Whiting, 1997). Removals should be performed by trained individuals who are familiar with local species and that are able to capture and transport snakes without causing them undue stress.

6.2.4 Monitoring of water quality

Water quality in the Siyayi catchment should be monitored at regular intervals to ensure that quality levels are maintained. While the mining procedure does not result in the addition of toxic chemicals to effluent (Shepherd et al., 2004), suspended solids can reduce oxygen content of water and potentially reduce amphibian population viability (Alford and Richards, 1999; Dallas and Day, 1993). Additionally, siltation of the river system could have negative impacts on amphibian populations through the alteration of breeding habitat structure (Fredricksen and Fredricksen, 2004).

6.2.5 Monitor herpetofaunal recolonisation of revegetated areas

Post-mining restoration of the Fairbreeze C Ext mining area is likely to involve a return to agricultural land use, specifically sugarcane (R. Hattingh, Pers. Comm.). While I maintain that this protocol is undesirable and will likely slow herpetofaunal recolonisation of the area, it presents an opportunity to investigate rates of herpetofaunal recolonisation of “non-natural” habitats. The resultant findings would be valuable in conservation planning given the widespread nature of agricultural monocultures as “restoration” endpoints (e.g., Strzyszcz, 1996). A monitoring programme could be developed that surveyed herpetofaunal diversity on restored sites on an annual basis and compared changes in diversity to the baseline herpetofaunal diversity collected during the course of my MSc degree.

6.2.6 Development of Gaboon Adder monitoring forum

A Gaboon Adder monitoring forum should be developed that could keep records of Gaboon Adder sightings in the Mtunzini area. The forum should be advertised in the local newspaper and produce regular reports. The detection of neonates and gravid females should be priority. All spatial data should be made available for the upcoming IUCN Red-listing workshop that forms part of the South African Reptile Conservation Assessment.

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