

Groundwater - Biodiversity - Land use

PATTERNS OF GROUND-DWELLING VERTEBRATE BIODIVERSITY IN THE GNANGARA SUSTAINABILITY STRATEGY STUDY AREA



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Draft Report to the Department of Environment and Conservation and the Gnangara Sustainability Strategy

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Front cover photos (clockwise from top right): Rankinia adelaidensis adelaidensis, Tarsipes rostratus, Neelaps calonotos and Myobatrachus gouldii. Photo credit: L. Valentine.



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Patterns of ground-dwelling vertebrate biodiversity in the Gnangara Sustainability Strategy study area

Introduction

Human-induced disturbances are major causes of global environmental change and substantially modify ecosystems and alter natural disturbance patterns (Sala *et al.* 2000; 1997). Common contributors to environmental change include habitat destruction, altered fire regimes and climate change. Understanding how fauna respond to these human-mediated disturbances is critical for effective conservation planning and management of biodiversity assets, and may determine future management actions. However, understanding the influence of disturbances on biodiversity assets is often difficult to assess without basic knowledge of the occurrence, distribution and relative abundance of species in relation to local landscape features.

A case study example includes the biodiversity assets of the northern Swan Coastal Plain, north of Perth, Western Australia. The region is close to a capital city, and has undergone extensive habitat modification and alterations to fire regimes. In addition, the region is currently experiencing declines in water supply caused by reduced rainfall and increased abstraction of the underground aquifers. Yet the area contains one of the largest remnant vegetation patches on the Swan Coastal Plain, and has high biodiversity values, particularly of the ground-dwelling vertebrates (e.g. Kitchener *et al.* 1978; Storr *et al.* 1978). To aid future conservation planning and management activities, it is imperative to broaden our understanding of the potential influence of landscape features and human-mediated disturbances on biodiversity assets of the region.

The Gnangara groundwater system

The Gnangara groundwater system is located on the Swan Coastal Plain (SWA2) IBRA sub-region, north of the Swan River, Perth, Western Australia and covers an area of approximately 220 000 ha. The Gnangara groundwater system consists of an unconfined, superficial aquifer known as the Gnangara Mound that overlies the confined Leederville

and Yarragadee aquifers, as well as the smaller Mirrabooka and Kings Park aquifers (Government of Western Australia 2009b). The area covered by the Gnangara groundwater system represents a distinct water catchment that extends from Perth (Swan River) in the south, to the Moore River and Gingin Brook in the north, and from the Darling Scarp in the east to the Indian Ocean in the west (Government of Western Australia 2009b). The Gnangara Mound is directly recharged by rainfall (Allen 1981; Government of Western Australia 2009b) and provides the city of Perth with ~ 60 % of its drinking water. It supports numerous significant biodiversity assets, including the largest patch of remnant vegetation south of the Moore River, a number of Bush Forever sites, threatened species and ecological communities, and ~ 600 wetlands. However, declining rainfall and runoff levels in the past 30 years have heavily impacted on water availability and the ecosystems in the region.

The impacts of a drying climate and declining groundwater levels strongly influence the water levels of the Gnangara groundwater system (Froend *et al.* 2004; Horwitz *et al.* 2008). Since the late 1960s, monthly rainfall has generally been below average (Yesertener 2007), resulting in decreased flows to public water supply dams and declining groundwater levels in the aquifers (Vogwill *et al.* 2008). Indeed, groundwater levels have decreased by up to 4 m in the centre of the Gnangara Mound and the eastern, north-eastern and coastal mound areas have experienced declines in the water table of 1 - 2 m (Yesertener 2007). In addition, other threatening processes are impacting on biodiversity in the region, including habitat clearing, fragmentation, altered fire regimes and impacts of *Phytophthora cinnamomi* (Government of Western Australia 2000; Mitchell *et al.* 2003).

The Gnangara Sustainability Strategy

Maintaining biodiversity is fundamental to maintaining ecosystem processes and is an environmental policy and priority of both Commonwealth and State Governments in Australia. To tackle the impending water crisis, the Gnangara Sustainability Strategy (GSS) was initiated to provide a framework for balancing water, land and environmental issues; and to develop a water management regime that is socially, economically and environmentally sustainable for the Gnangara groundwater system (DOW 2008). A multiagency taskforce was established in 2007 to undertake the GSS project, which incorporates existing land and water use policies, studies on the ecosystem assets and processes, and the

development of a decision-making process to integrate values, risks and planning processes (DOW 2008).

Current understanding of biodiversity values, ecosystem processes and the dynamics of the Gnangara groundwater system, particularly at landscapes scales, is inadequate (Government of Western Australia 2009b). Gaps in our capacity to measure impacts on biodiversity, landscape condition and ecosystem processes as a result of disturbances (e.g. climate change, changed water regimes, fire, and plant pathogens) are likely to result in ineffective management actions and low quality outcomes. The ability to develop successful planning relies on the quality of the biodiversity information (Pressey 1999; Wilson *et al.* 2005). Indeed, unless an adequate understanding of these issues is accomplished, justification of changed management actions in the face of potentially degrading impacts on biodiversity is difficult.

Species richness and endemicity

The southwest of Western Australia – a globally significant region

The GSS study area is located centrally in the biodiversity hotspot of southwest Western Australia – a region of globally significant biodiversity (Mittermeier *et al.* 2004). This area is internationally recognised because of its high levels of biodiversity and endemicity and the high degree of threatening processes (Mittermeier *et al.* 2004). The high level of plant biodiversity is generally attributed to the extreme climatic shifts and poor soils that have promoted local specialisations (Hopper and Gioia 2004). The floristic diversity of the area is particularly well renowned (Beard *et al.* 2000; Hopper and Gioia 2004), containing over 7 000 vascular plant taxa, of which approximately 50 % are endemic (Coates and Atkins 2001; Hopper and Gioia 2004). Similarly, southwest Western Australia provides habitat for > 500 vertebrates, of which nearly $1/6^{th}$ are endemic (Mittermeier *et al.* 2004). The high degree of endemicity is particularly well reflected among the rich reptile and frog taxa with a total of approximately 177 species of reptiles and 32 species of frogs, of which 15 % and 80 % respectively are endemic to southwest Western Australia (Mittermeier *et al.* 2004). Similarly, the mammal fauna is historically diverse with 59 species recorded, including 12 endemics (Mittermeier *et al.* 2004). However, the mammal fauna in southwest Western Australia has undergone a number of species extinctions and declines (Abbott 2008; Kitchener *et al.* 1978).

The occurrence and distribution of ground-dwelling vertebrates are often influenced by landscape features, including structural complexity and succession age, and are thus likely to respond to disturbance induced changes in habitat (e.g. Pianka 1989; Rosenzweig and Winakur 1969; Wilson *et al.* 1986). Indeed the mammal fauna of southwest Western Australia has already undergone a series of extinction and range contractions, indicating a susceptibility to human-mediated disturbances (Kitchener *et al.* 1978). Reptiles are strongly dependent on habitat structure (Pianka 1989), typically have small home ranges and are therefore often used as a surrogate measure of faunal diversity in response to disturbances (e.g. Cunningham *et al.* 2002; Valentine and Schwarzkopf 2009). Frogs are primarily influenced by changes in water cycling and quality, which is strongly related to water balance physiology and reproduction (White and Burgin 2004). Although there is scant information regarding the relative importance of habitat factors for herpetofauna assemblages, it is likely that the importance of factors varies between reptiles and amphibians, as well as among species, and response is based on individual life history attributes (e.g. How and Dell 2000; Jellinek *et al.* 2004).

Diversity and endemicity in the GSS

The GSS study area is located north of the Swan River (Figure 1), in the northern Swan Coastal Plain, in an area that is well known for its richness of terrestrial vertebrate taxa (Kitchener *et al.* 1978; Storr *et al.* 1978). Part of the Swan Coastal Plain's significance for biodiversity lies in its geographic location, with several species occurring at the limits of their distribution (How 1978). A number of parallel reviews on the diversity of the ground-dwelling vertebrates of the Swan Coastal Plain were conducted as part of the GSS process (see Bamford and Huang 2009; Huang 2009; Reaveley 2009) and these provide a detailed review of the biological diversity and history of vertebrates in the GSS study area.

Herpetofauna

All 13 species of frogs (Bamford and Huang 2009) historically known to occur in the GSS study area still occur on the Swan Coastal Plain, with the possible exception of

Heleioporous barycragus, which may occur only on the south eastern edge, and the burrowing frog *Neobatrachus pelobatoides*, formerly widespread but now known only from populations south of Perth (How and Dell 2000). There are approximately six frog species that are restricted to wetlands, including the Motorbike Frog *Litoria moorei*, the Slender Tree Frog *Litoria adelaidensis, Crinia georgiana, Crinia pseudinsignifera, Crinia glauerti* and possibly Günther's Toadlet (*Pseudophryne guentheri*) (Bamford and Huang 2009). Additional frog species include the terrestrial Turtle Frog *Myobatrachus gouldii*, the Pobblebonk Frog *Limnodynastes dorsalis* and the Moaning Frog *Heleioporus eyrei*, which are dependent on upland woodland habitats during the non-breeding season (Bamford and Huang 2009).

The reptilian fauna on the northern Swan Coastal Plain is speciose (How and Dell 1993; 1994; 2000; Storr et al. 1978). Its species richness (in the GSS) is currently estimated to comprise of 39 genera and 64 species including 2 turtles, 8 geckoes, 8 pygopodids, 2 dragons, 3 goannas, 21 skinks, 2 turtles, 4 blind snakes, 2 pythons and 14 elapid snakes (Huang 2009). The study area has 17 taxa at or close to their geographical limit. Of these species, 27 taxa are restricted to southwest Western Australia. Of these, seven taxa are local endemics to the SCP, including *Pseudemydura umbrina*, *Delma concinna concinna*, Pletholax gracilis gracilis, Hemiergis quadrilineata, Lerista christinae, Rankinia adelaidensis adelaidensis and Neelaps calonotos. Only one species, the critically endangered Western Swamp Tortoise Pseudemydura umbrina, is endemic to the GSS study area, with populations restricted to Ellen Brook Nature Reserve and Twin Swamps Nature Reserve on the eastern boundary of the GSS (Burbidge and Kuchling 2004). The Western Swamp Tortoise is Australia's most endangered reptile species (Cogger 2000). Several species may have experienced range and abundance contraction since European settlement, with only one species believed to have become extinct from the GSS study area: Stimson's Python (Antaresia stimsoni stimsoni) (How and Dell 1994).

The legless lizards, skinks and elapid snakes are well represented in the northern Swan Coastal Plain (Storr *et al.* 1978). In particular, nearly half of the pygopods in the northern SCP are endemic to the west coast and coastal plains of WA, representing a stronghold for this reptilian taxa (Storr *et al.* 1978). In addition, the fossorial skinks and snakes are well represented on the deep siliceous and calcareous sands of the western part of the SCP (How and Dell 2000).

Mammals

Historically, up to 33 native mammal species have been recorded on the northern Swan Coastal Plain (Kitchener *et al.* 1978), although by the late 1970s only 12 species were recorded during extensive surveys conducted by the Western Australian Museum (Kitchener *et al.* 1978). A recent review (Reaveley 2009) estimated that approximately 10 non-bat mammal species are currently extant within the GSS. Species such as the Woylie (*Bettongia penicillata*) and Numbat (*Myrmecobius fasciatus*) are presumed extinct in the northern SCP while extant species such as the Bush Rat (*Rattus fuscipes*), Water Rat (*Hydromys chrysogaster*), Western Brush Wallaby (*Macropus irma*) and Honey Possum (*Tarsipes rostratus*) occur only in restricted or isolated populations (How and Dell 2000; Kitchener *et al.* 1978). The mammals have thus experienced very high extinction rates and all species are considered to have declined in distribution and abundance (Kitchener *et al.* 1978).

Disturbances to species diversity in the GSS

A number of threatening processes for terrestrial vertebrates have been identified previously in the study area including declining groundwater levels, habitat fragmentation, inappropriate fire regimes and *Phytophthora* infection (Government of Western Australia 2000; Wilson *et al.* 2007). The impacts of such processes on the reptiles and mammals of the study area were also explored in parallel literature reviews (Huang 2009; Reaveley 2009) and some of the threatening processes are examined here.

Recently, declining rainfall and increased aquifer abstraction has led to declining groundwater levels (Yesertener 2007). This has emerged as a significant threat to wetland-associated vertebrate fauna in the GSS study area. There is evidence that lake systems are being converted to swampy flats, and seasonal wetlands are drying, with some becoming acidic (Froend *et al.* 2004; Pettit *et al.* 2007). Long-term changes in groundwater may also have a negative impact on *Banksia* woodland ecosystems that are not directly linked to the groundwater. Declines in wetlands and *Banksia* woodlands should have significant impacts on vertebrate fauna and their habitats, as most of the fauna of the northern Swan Coastal Plain has adapted to the predictability of the climate (How 1978). However, there

is little knowledge of both the habitat components that are most affected, and the fauna communities or taxa that are most susceptible to declining groundwater levels. Mammals (e.g. the Water Rat, Bush Rat and Quenda), reptiles (e.g. the Tiger Snake and the Glossy Swamp Skink *Egernia luctuosa*) and frogs that are dependent on lakes, wetlands, and damp and thick vegetation are likely to be highly susceptible to declines in rainfall and groundwater levels. Indeed, it is suspected that the Mardo (*Antechinus flavipes*) have disappeared locally as a consequence of wetland losses (How 1978; Kitchener *et al.* 1978).

Located within close proximity to Western Australia's capital city, Perth, the GSS study area has undergone substantial clearing for urban development and agriculture. Numerous studies have examined the relationship of faunal response to urban expansion, habitat loss and fragmentation (How and Dell 1993; 1994; 2000). Reptiles have shown a marked decline in Perth and its suburbs, with very few species persisting after development occurs, and there is a rapid decline as soon as native vegetation is cleared for farming (How and Dell 1993; 1994; 2000). Native mammals are typically the most disadvantaged, with few native mammals surviving in urban areas (How and Dell 2000). Congruently, the success of introduced species associated with urban expansion (e.g. cats) is correlated with native mammal declines (Kitchener *et al.* 1978). However, in the GSS study area DEC manages $> 70\ 000$ ha of remnant vegetation, including the largest contiguous remnant vegetation patch south of the Moore River. In this habitat, the persisting fauna may be strongly influenced by fire regimes and climate change, although data to this effect are scarce.

Fire often occurs as a human-mediated disturbance and is an important contemporary land management tool (Russell-Smith *et al.* 2003; Whelan 1995). Fire plays a pivotal role in determining environmental and biological heterogeneity (Brawn *et al.* 2001; Whelan 1995), and is instrumental in structuring numerous communities in southwest Western Australia (Burrows and Abbott 2003). Ecosystems on the Swan Coastal Plain are considered some of the most flammable in southwest Western Australia, due to the lengthy period of the year that the vegetation is combustible, and plant growth adaptations that result in rapid accumulation of vegetation after fire (Burrows and Abbott 2003). However, inappropriate fire regimes have been recognised as a major threatening process on the Swan Coastal Plain at the landscape level, for protected areas, wetlands, riparian zones, ecosystems and species at risk.

A potential management measure that has been suggested to reduce declining groundwater levels of the Gnangara groundwater system is the use of 10 year rotational burning strategies (Government of Western Australia 2009a), where fire management may be used to promote groundwater recharge. However, burning to increase groundwater recharge is still only a modelled concept (Vogwill et al. 2008; Yesertener 2007). If burning for groundwater recharge is implemented across the Gnangara groundwater systems, empirical data on the responses of fauna to fire is essential. Previous studies in other habitats have shown that reptile and mammal fauna communities are strongly influenced by fire regimes (Cunningham et al. 2002; Letnic et al. 2004; Valentine and Schwarzkopf 2009; Wilson and Aberton 2006; Wilson et al. 2001). Urban remnants in the Perth region have also shown that lizard diversity was greatest in areas that have remained unburnt the longest (Dell and How 1995), indicating a susceptibility of some species to frequent burning. Understanding how fauna in larger remnant vegetation patches respond to fire is important for developing appropriate fire management strategies. Within the GSS study area, fauna groups of particular interest with regard to fire regimes include the diverse reptile community and targeted mammal species such as the Honey Possum (Tarsipes rostratus). Although the strong association of frog species with wetland vegetation obviously make this taxon relevant, we do not focus on responses by frogs to fire regimes in this report.

Aims

There has been a range of studies conducted at various locations and with a range of objectives on the GSS study area over the past 40 years. Previous assessments of patterns of vertebrate biodiversity have been restricted to single phylogenetic groups, and/or land classes or geographically localised areas. There is little understanding of the habitat requirements of taxa and communities, and of the impacts of threatening processes.

In our report, we assess the current occurrence and distribution of terrestrial vertebrate fauna across the GSS study area, examine patterns in biodiversity with landscape features and assess the susceptibility of taxa and communities to threatening processes such as declining groundwater levels and fire. The project was designed with the main objectives to:

 Determine the current species richness and abundance of ground-dwelling vertebrate fauna persisting in the region;

- 2. Investigate patterns of biodiversity with broad landscape features including landforms, vegetation types and time since fire;
- 3. Examine patterns in biodiversity and specific species responses with site characteristics including habitat and microhabitat structure and floristics;
- 4. Relate findings to the impacts of fire, and predictions on the potential impacts of declining groundwater levels.

Methods

Study area

The GSS study area is situated on the Swan Coastal Plain and extends from the Swan River in the south, to the Moore River and Gingin Brook in the north, and from the Ellen Brook in the east to the Indian Ocean in the west (DOW 2008) Figure 1). The Swan Coastal Plain is built up from foothill, aeolian, lake, river and estuarine deposits laid down to the west of the scarp (Davidson 1995). Three main dune systems underlie most of the groundwater system – the younger Quindalup Dunes close to the coast, the Spearwood Dunes associated with Tamala Limestone ridges within about ten kilometres of the coast and the inland Bassendean Dunes which are older and flatter, and contain leached and slightly acidic sands (Figure 1).

The three main dune systems are dominated by a *Banksia* overstorey with sporadic stands of *Eucalyptus* and *Allocasuarina*, and an understorey consisting mainly of low shrubs from the Myrtaceae, Fabaceae and Proteaceae families. There are many seasonal damplands, swamps and permanent wetlands, fringed by *Banksia littoralis* and *Melaleuca* trees with a variable understorey of species from the Cyperaceae, Juncaceae and Myrtaceae (Semeniuk *et al.* 1990). The distribution of vegetation on the northern Swan Coastal Plain is predominantly determined by the underlying landforms, soils, depth to water table and climatic conditions (Cresswell and Bridgewater 1985; Heddle *et al.* 1980). Heddle *et al.* (1980) defined broad vegetation complexes across the Swan Coastal Plain in relation to these landform–soil units (Churchward and McArthur 1980) and the varying climatic conditions.

Although there have been large amounts of clearing for urbanisation and agriculture, the total remnant native woodland in the GSS study area covers more than 100 000 ha (72 447 ha of which is managed as DEC-estate), and includes the largest continuous area of remnant vegetation on the Swan Coastal Plain, south of the Moore River. The remnant woodland within the GSS study area has significant biodiversity values, containing threatened species and ecological communities, and numerous wetlands. The study area also includes approximately 25 000 ha of pine plantation (Figure 1).



Figure 1. The remnant vegetation extent and DEC managed lands located within the GSS study area. Fauna survey sites, including targeted mammal trapping areas are also indicated.

The GSS study area experiences a dry Mediterranean-type climate (Beard 1984), with hot dry summers (December – March) and cool wet winters (June – August), and an average of 870 mm annual rainfall recorded at the Perth meteorological station. Rainfall and runoff declines in the last 30 years have been significant, with approx 21 % less rainfall and 64 % less runoff for the 1997 - 2003 compared to 1911 - 1974 (Yesertener 2007).

Fauna surveys

The GSS fauna survey included a general ground-dwelling vertebrate trapping program across the GSS study area and a targeted program for particular wetland-associated species (quenda and rakali) in specific wetlands and damplands within the GSS study area. The fauna trapping was designed to cover a range of remnant bush land in the northern and eastern part of the GSS study area, predominantly within DEC-managed estate. The trap sites were selected to represent a range of landform units, vegetation associations and fuel ages, were initially examined by desktop spatial analysis and maps and subsequently validated in the field. Landform unit was classified according to Churchill and McArthur (1980) with potential sites selected by spatial analysis. Vegetation associations were identified initially by spatial analysis of the Mattiske dataset (Mattiske Consulting Pty Ltd 2003) and field validation confirmed the dominant plant species at each site. Time since last fire was obtained from the 2007 Corporate Data fuel age dataset provided by Fire Management Services of DEC, with time since last fire at each site validated using VegMachine.

General survey trapping

Trapping design

The sampling regime was designed to assess 40 sites in the major areas of continuous remnant bush land in the northern and eastern areas of the GSS study area (Figure 1). Sites were selected to represent the major landform units (Quindalup, Spearwood and Bassendean), vegetation communities (*Banksia* woodland, coastal scrub, jarrah forest, tuart forest and *Melaleuca* wet or dampland; Figure 2) and time since last fire. As the time since last fire varied considerably among sites (3 – 36 years since last fire, YSLF), we grouped sites into two major categories: Young, those recently burnt (< 11 YSLF); or Old, those long unburnt (> 16 YSLF) (Figure 1; Appendix 1). Where possible, the 40 sites were

grouped into 20 paired sites, located between 300 – 500 m from one another. Some categories, such as the Quindalup landform were not well represented due to their limited occurrence and/or poor accessibility in the study area.



Figure 2. Vegetation types surveyed, including a) tuart forest, b) *Melaleuca* wetland c) *Banksia* woodland and d) jarrah forests.

The trapping layouts, sampling design and methodology employed in previous vertebrate surveys on the GSS study area has been variable. We used a trapping design that included an array of pitfall traps (20 L buckets), and a transect line of small aluminium box traps (elliotts) and cage traps (sheffields). Several studies have indicated that the large pitfall traps in conjunction with drift fences have superior trap success (Friend *et al.* 1989; Morton *et al.* 1988; Ryan *et al.* 2002) and hence were utilised in our study. Each of our sites contained one pitfall trap array, with 10 pitfall traps located in a Y shape, with three pits placed along each arm radiating out from a central pit and placed at approximately 7 metre intervals along each arm. The pitfall traps were connected with 30 cm high aluminium fly wire drift fence and extended out one metre past the end of each arm. Each arm measured 22 metres long, with a total fence length of 66 metres in an array (see Appendix 2 for pitfall trap layout).

Where sites were located in pairs (paired sites were at least more than 300 m apart), twenty elliott and ten sheffield traps were positioned along a 300 m-long transect between the two sites. Elliott traps were located at 15 m intervals, and a Sheffield cage trap was set at 30 m intervals corresponding with every second elliott trap point. Both trap types were baited with universal bait comprising a mixture of peanut paste, rolled oats and tinned sardines.

Survey effort

Sites were opened for 12 - 20 nights in spring 2007, autumn 2008 and spring 2008. While open, all traps were checked once per day in the early morning. Captured animals were identified, processed and released onsite. Measurements taken included: weight (g), snoutvent length (mm), total length (mm) for reptiles; and, weight (g), head length (mm), short pes length (mm), gender, and presence of pouch young for mammals. Mammals were ear notched for recapture purposes (quenda received a unique ear tag), whilst reptiles were marked with a non-toxic permanent marker pen under the throat. Amphibians were not marked. Taxonomic nomenclature, including common names, followed the Western Australian Museum. Incidental observations of species were also recorded and are included in the tally of species totals.

Species accumulation curves were examined for the initial two trapping periods (spring 2007) to establish optimal trapping nights. Curve asymptotes indicated that 5 to 8 nights was adequate.

Targeted trapping surveys

Targeted trapping surveys for quenda (south brown bandicoot; *Isoodon obesulus fusciventer*) and rakali (water rats; *Hydromys chrysogaster*) were undertaken in May 2008. Quenda were targeted following the lack of captures during the general spring 2007 terrestrial vertebrate survey, and to confirm recent information indicating that the species appears to be linked to wetland-associated vegetation (T. Friend pers. comm.). Rakali were targeted to determine if the species is persisting within the study area and if so, their distribution. As the only known aquatic mammal in southwest Western Australia, their presence in a wetland or waterway could be a useful indicator of the health of wetland systems.

Quenda sites were selected in areas that contained a permanent wetland or were swampy or damp for part of the year. Four sites were selected based on historical records; these included Little Badgerup Swamp, Neaves Road Nature Reserve, Maralla Road Nature Reserve and Nowergup Nature Reserve (Figure 1). The five remaining sites were selected based on presence of suitable habitat for quenda. Permanent water bodies are believed to be suitable rakali habitat (Olsen in Van Dyck and Strahan 2008), and rakali sites were selected to reflect this habitat preference. Sites were chosen at three lakes within the study area: Lake Joondalup in the suburb of Joondalup, Lake Goolellal in the suburb of Kingsley and Loch Mc Ness in Yanchep National Park (Figure 1). Trap transects were located on the north western section of Lake Joondalup, the eastern side of Lake Goolellal and 4-5 different locations around Lake Loch McNess. Traps for rakali were established on the water's edge, with the open door facing the lake.

A continuous transect of Sheffield cage traps was established amongst vegetation considered to be suitable habitat for quenda or rakali at 50 m intervals through the site. Depending on the size of the site, between 10 and 30 traps were set. Universal bait was used in all traps set to capture quenda and a sardine was hung on a hook in the cages set for rakali. All sites were trapped over 4 nights, except for Lake Goolellal, which was trapped for 3 nights due to logistic issues. Captured animals were identified, measured and released onsite. Measurements included: weight (g), head length (mm), short pes length (mm) and gender. Captured quenda were ear-tagged in each ear with unique identification tags. Rakali were ear-notched in one ear.

Plant species richness and habitat parameters

Floristic surveys

A number of 10 m x 10 m floristic survey quadrats were established in the GSS study area, located as close as possible to the centre point of the established fauna pit traps. The location data of each quadrat was recorded on a GPS at the centre of the quadrat, using GDA94 as the datum, and each quadrat was physically demarcated using four galvanised fence droppers, one at each corner. In total, floristic surveys were conducted at 36 of the 40 fauna sites. Due to logistical constraints, floristic surveys were not conducted at the two Quindalup Dune (coastal scrub vegetation) sites or two of the northern *Banksia*

woodland sites (sites 3A and 3B). All vascular plants were recorded in spring 2008, and in our study we focus on the number of plant species per site. Plant specimen identifications are currently being confirmed with the Western Australian Herbarium. A detailed examination of floristic diversity patterns will be the focus of a separate report (Mickle and Swinburn in prep).

Habitat parameters – vegetation structure

Within close proximity to the floristic plots (within 20 m) and the pit-fall trap array, 20 x 1 m² quadrats were established to examine microhabitat attributes. Attributes assessed included vegetation structure and ground substrate composition. To provide an index of vegetation complexity within a 2 m height range, vegetation contact (both live and dead) was recorded (for height classes 0-20cm, 20-40cm, 40-60cm, 60-80cm, 80-00cm, 100-150cm, 150-200cm) using a graduated pole, placed at the point in the centre of the quadrat. At the same point, canopy cover was measured using a densitometer, which calculated an approximate percentage canopy cover. Ground cover within the 1m² quadrat was estimated as a percentage of vegetation (live and dead), soil (bare ground) and litter (including leaf and woody debris). Litter depth (cm) was measured using a ruler that was pressed through the litter (where relevant) until it touched a firm soil surface.

The vegetation structure and cover were recorded using Keighery (1994). The crown cover is estimated to as close to 5 % as per the National Vegetation Inventory System as well as the class rating as per Muir (1977). Up to three dominant species were recorded as part of each layer and any more that three dominants were classed as mixed. Vegetation condition was recorded using the Keighery (1994) Vegetation Condition Scale.

Data collation and analyses

General trapping survey

Abundance, species richness and diversity

Data from pit-fall trapping of sites, and the trapping using elliott and cage traps between paired sites is described and collated as species presence/absence tables. Most data were collected using pit-fall trapping, and further exploration and analyses were conducted on

the pit-fall trapping data, excluding data from the two sites on Quindalup dunes (due to limited number of replicates in this dune system). Pit-fall trapping intensity varied among sites, and we have performed analyses on the data using average abundance and species richness estimates, based on the relative abundance for each species and species richness from 10 trap nights. Diversity of the reptile assemblage at each site was calculated from these measures using Simpson's Diversity of Index (1-D), which ranges from 0 (low diversity) to 1 (high diversity). In addition, the Evenness of the reptile assemblage at each site (E, derived from Shannon-Wiener Index) was calculated. Evenness represents the degree of variation in a community, with higher values of E representing a less variable community.

To explore patterns in data, one-way ANOVAs (using SPSS 2008, version 17.0) were used to examine differences in reptile abundance, reptile species richness, reptile diversity and evenness among the 38 sites with respect to landform unit (Spearwood and Bassendean), vegetation type (*Banksia, Melaleuca*, jarrah and tuart) and time since last fire (Old, > 16 YSLF and Young, < 11 YSLF). Post-hoc Tukey HSD tests were used to determine differences among vegetation types where applicable. The abundance of dominant species (reptiles: \geq 10% of observed individuals; mammals and frog: the two most abundant species) were also compared among factors using ANOVAs. In addition, the number of plant species recorded at each site (n = 36) were also explored in relation to landform, vegetation type and fuel age using ANOVAs.

Patterns in species composition

Community composition, defined as the average abundance (per 10 trap nights) of each species per site, was compared among factors (landform unit, vegetation type and fuel age) using Multi-Response Permutation Procedure (MRPP, Mielke 1984), based on a rank-transformed Sorensen distance matrix in the statistical package, PC-ORD (McCune and Mefford 1999). Rare species (species that were observed in less than three sites) were not included in the analysis. MRPP is a type of nonparametric multivariate procedure for testing differences between groups and provides an A statistic, which is the chance-corrected within group agreement, and an associated p-value (McCune *et al.* 2002). Posthoc pair-wise comparisons were used to examine differences in reptile assemblages among vegetation types. Where community composition differed significantly among factors ($\alpha <$

0.05), non-metric multidimensional scaling (NMDS, Kruskal 1964) was used to graphically depict the site assemblage relationships using PC-ORD (McCune and Mefford 1999). Where 3-dimension ordinations were deemed more appropriate (determined using scree plots and Monte Carlo tests), the two axes that represent the highest proportion of variance in the ordination are displayed. Reptile species and microhabitat and vegetation structure variables that were correlated with the NMS axes ($r^2 > 0.2$) are graphically depicted on the ordinations.

Habitat parameters and correlations with fauna

Habitat parameters examined included percentage estimates of ground vegetation cover, soil cover, litter cover and canopy cover, a measurement of litter depth and data collected on understorey structure (touch pole data). Habitat variables for each site were correlated, using Pearson's correlation coefficient (SPSS, version 17), with reptile abundance, reptile species richness, reptile diversity, reptile evenness, plant species number and time since last fire (recorded in years since last fire or YSLF).

Succession with time since fire in Banksia woodlands

Initial groupings of sites into time since fire categories were fairly broad, with recently burnt sites < 11 YSLF and long unburnt sites > 16 YSLF. However, sites located in *Banksia* woodlands (n = 16) were from a range of time since fire (4 – 36 YSLF), and could be sorted into five different categories of time since fire, including: Young, 4 YSLF (n = 4); Young-Intermediate, 6-7 YSLF (n = 3); Intermediate, 17 YSLF (n = 4); Old, 22-26 YSLF (n = 2); and, Very Old, 36 YSLF (n = 3). This range in time since fire allowed us to examine possible patterns of succession in different stages of post-fire *Banksia* woodlands. We calculated average abundance per 10 trap nights for each species in the five different time since fire categories, and then relativised abundance estimates for each species and graphically interpreted patterns in relative abundance of dominant reptile and mammal species. Rare reptile species (those species observed in less than 3 sites) were excluded from analyses.

Data transformations

Habitat parameters measured using percentages were adjusted by arcsine transformation of the square-root proportional data (Zar 1999). Count data (reptile abundance, species number, individual species abundance, and vegetation touch pole data) and litter depth were examined for normality and heteroscedasticity using box plots, Q-Q plots and residual plots. Individual species abundances, vegetation touch pole data and litter depth were square-root transformed to meet assumptions of ANOVA and Pearson's correlations.

Targeted trapping survey

The results of the targeted trapping surveys, including the capture of non-target species, is described and graphically interpreted. As trapping intensity varied among sites, mammal capture rates are presented as abundance per 100 trap nights.

Results

Occurrence and distribution of fauna

Trapping effort in the general fauna survey involved 5 600 pitfall trap nights, 5 600 elliott trapping nights and 2 800 cage trapping nights, and 930 cage trap nights in the targeted surveys for quenda and rakali. In total, 38 reptile, 16 mammal and 6 frog species were trapped and/or recorded during surveys (see Appendix 3 for complete species list). The reptile and frog families represented include: Myobatrachidae (ground frogs; 5 genera, 5 species), Hylidae (tree frogs; 1 genus, 1 species), Gekkonidae (geckos; 3 genera, 3 species), Pygopodidae (legless lizards; 6 genera, 7 species), Agamidae (dragons; 2 genera, 2 species), Varanidae (goannas; 1 genus, 1 species) and Elapidae (front-fanged snakes; 7 genera, 8 species). Nine native mammal species and seven introduced mammal species were recorded: Tachyglossidae (1 species), Dasyuridae (1 species), Peramelidae (1 species), Muridae (4 species), Canidae (1 species), Felidae (1 species), Leporidae (1 species), Suidae (1 species), and Bovidae (1 species).

Trapping for the general fauna surveys yielded 2 021 individuals, representing 43 species of frogs, reptiles and mammals (Table 1, Figure 3). Skinks were the most commonly caught animal during surveys (65 %). The majority of species (36 species) were captured in the pit-fall traps, however, six species were only captured in the cage and/or elliott traps (Table 1). All reptiles were recorded in one or more trap type, excluding *Varanus gouldii*, which was recorded only by incidental observation. Six mammals were captured by one or more trap type (Table 1), while ten mammal species were recorded only by incidental observation. The priority reptile species, *Neelaps calonotos*, was captured on two sites: one in the north in *Melaleuca* wetlands on long unburnt (22 YSLF) Bassendean soils; second in the east in *Banksia* woodland on long unburnt (36 YSLF) Bassendean soils.

Table 1. Species captured during pit-fall trapping or cage and elliott trapping during general fauna surveys throughout the GSS study area, excluding targeted mammal surveys. The presence or absence of each species is represented based on landform unit, vegetation type and fuel age (presence of species indicated by a '1', and the number of sites in each category is provided in brackets). The number of site a species was observed at, and the abundance of each species in either pit-fall traps or cage and elliott traps is included. Summary data on general abundance and family-level representation in different landform units, vegetation types and time since fire category is also provided.

	Landform Unit ^a			Vegetation Type ^b			Years Si	nce Fire ^c	# of sites ^d	Abun	dance ^e		
	Q (2)	Sp (20)	Bn (18)	Cs (2)	Tt (4)	Jh (4)	Me (14)	Bk (16)	<11 (19)	>16 (21)	Total (40)	PF	CE
					Amph	ibia							
Myobatrachidae					•								
Crinia insignifera	0	0	1	0	0	0	0	1	0	1	1	1	0
Heleioporus eyrei	1	1	1	1	1	1	1	1	1	1	17	84	0
Limnodynastes dorsalis	1	1	1	1	1	1	1	1	1	1	24	52	1
Myobatrachus gouldii	0	1	1	0	0	1	1	1	1	1	19	75	0
Pseudophryne guentheri	0	0	1	0	0	0	0	1	1	0	1	3	0
Hylidae													
Litoria moorei*	0	1	0	0	0	0	1	0	1	0	1	0	1
					Renti	ilia							
Gekkonidae													
Christinus marmoratus	0	1	1	0	1	1	1	1	1	1	10	17	0
Diplodactylus polyphthalmus	0	1	0	0	0	0	0	1	1	0	1	1	0
Strophurus spinigerus spinigerus	1	1	1	1	0	1	1	1	1	1	15	42	0
Pygopodidae													
Aprasia repens	0	1	1	0	0	0	1	1	1	1	12	31	0
Delma concinna concinna	0	0	1	0	0	0	0	1	1	0	1	1	0
Delma fraseri fraseri	0	1	1	0	0	0	1	1	1	1	3	5	0
Delma grayii	1	1	1	1	0	0	1	1	1	1	3	3	0
Lialis burtonis	1	1	1	1	0	1	1	1	1	1	15	27	0
Pletholax gracilis gracilis	0	1	1	0	0	1	0	1	1	1	3	3	0
Pygopus lepidopodus	0	0	1	0	0	0	1	1	1	1	7	9	0
Agamidae													
Pogona minor minor	1	1	1	1	0	1	1	1	1	1	21	50	0

	La	ndform U	J nit ^a		Veg	getation	Type ^b		Years Since Fire ^c		# of sites ^d	Abundance ^e	
	Q (2)	Sp (20)	Bn (18)	Cs (2)	Tt (4)	Jh (4)	Me (14)	Bk (16)	<11 (19)	>16 (21)	Total (40)	PF	CE
Rankinia adelaidensis adelaidensis	1	1	1	1	0	1	1	1	1	1	24	165	0
Scincidae													
Acritoscincus trilineatum	0	1	1	0	1	0	1	0	1	1	5	10	0
Cryptoblepharus buchananii ^f	1	1	1	1	1	1	1	1	1	1	31	149	0
Ctenotus australis	1	1	1	1	1	1	1	1	1	1	9	17	1
Ctenotus fallens	1	1	1	1	1	1	1	1	1	1	31	119	6
Ctenotus impar	0	0	1	0	0	0	1	0	1	0	1	1	0
Egernia kingii	0	1	0	0	0	0	1	0	1	0	1	2	1
Egernia napoleonis	0	1	1	0	0	1	1	1	1	1	5	7	4
Hemiergis quadrilineata	1	1	1	1	1	1	1	1	1	1	26	167	0
Lerista elegans	1	1	1	1	1	1	1	1	1	1	35	362	0
Lerista lineopunctulata	1	0	0	1	0	0	0	0	0	1	1	1	0
Lerista praepedita	1	1	1	1	0	0	1	1	1	1	13	17	0
Menetia greyii	1	1	1	1	1	1	1	1	1	1	33	278	0
Morethia lineoocellata	0	0	1	0	0	0	0	1	1	1	4	13	0
Morethia obscura	1	1	1	1	1	1	1	1	1	1	29	168	0
Tiliqua occipitalis	1	0	1	1	0	0	0	1	0	1	2	2	3
Tiliqua rugosa rugosa	1	1	0	1	1	1	1	0	1	1	7	10	82
Typhlopidae													
Ramphotyphlops australis	0	1	1	0	0	0	0	1	0	1	2	4	0
Elapidae													
Brachyurophis semifasciata	0	0	1	0	0	0	1	0	0	1	1	1	0
Demansia psammophis reticulata	1	0	1	1	0	0	1	0	0	1	2	3	0
Neelaps calonotos	0	0	1	0	0	0	1	1	0	1	2	2	0
Notechis scutatus*	0	0	1	0	0	0	1	0	0	1	1	0	1
Parasuta gouldii	0	0	1	0	0	0	1	1	1	1	3	4	0
Pseudonaja affinis affinis*	0	1	0	0	0	0	0	1	1	0	1	0	1
Pseudonaja nuchalis*	0	0	1	0	0	0	0	1	1	0	1	0	1
Simoselaps bertholdi	1	1	1	1	1	0	0	1	1	1	6	8	0
-					Mamn	nalia							
Tachyglossidae					1710/////	inttu							
Tachyglossus aculeatus*	0	0	1	0	0	0	1	1	1	1	3	0	3
Dasyuridae													
Sminthopsis sp	0	0	1	0	0	0	0	1	1	0	1	1	0
Tarsipedidae													

	Landform Unit ^a			Veg	Vegetation Type ^b			Years Since Fire ^c		# of sites ^d	Abun	dance ^e	
	Q (2)	Sp (20)	Bn (18)	Cs (2)	Tt (4)	Jh (4)	Me (14)	Bk (16)	<11 (19)	>16 (21)	Total (40)	PF	CE
Tarsipes rostratus	0	1	1	0	0	1	1	1	1	1	14	28	0
Muridae													
Mus musculus	1	1	1	1	1	1	1	1	1	1	27	77	39
Rattus fuscipes	0	1	0	0	0	0	1	0	1	0	1	1	1
Rattus rattus*	0	1	0	0	1	0	1	0	1	0	2	0	2
					Summar	y Data							
Abundance												Total	Total
Total Number of Individuals	628	99	554	420	99	163	351	248	382	899		2021	147
Relative Average Abundance per													
site (10 trap nights)	4.025	3.4115	3.1676	4.025	5.8854	3.755	2.5253	3.2081	2.90132	3.72242			
Species Richness													
Number of Frog Species	2	3	5	2	2	3	3	5	4	4		5	1
Number of Reptile Species	18	23	32	18	11	15	27	28	29	31		33	9
Number of Mammal Species	1	4	4	1	2	2	5	4	6	3		4	4
Total Number of Species	21	30	41	21	15	20	35	37	39	38		42	14
Relative Average Richness per site													
(10 trap nights)	0.95	0.8479	0.812	0.95	0.651	0.839	0.7545	0.9409	0.86294	0.81329			

^a Landform Unit: Q = Quindalup, SP = Spearwood Dunes, Bn = Bassendean Dunes ^b Vegetation Type: Cs = Coastal Scrub, Tt = tuart forest, Jh = jarrah forest, Me = *Melaleuca* wetlands, Bk = *Banksia* woodlands ^c Years Since Fire: < 11 = more than 11 years since last fire, > 16 = more than 16 years since last fire ^d # of sites: refers to the number of sites a species was captured at out of a total of 40 sites ^e Abundance: refers to the total abundance of captured species in either the pit-fall traps (PF) or cage and elliott traps (CE)

^f Formerly considered to be *C. plagiocephalus*, recent revision indicates the species in the GSS is *C. buchananii* (Horner 2007).

* Indicates species that were exclusively captured in the cage or elliot traps



Figure 3. Representative fauna captured during fauna surveys in the GSS study area. Species include a) *Strophurus spinigerus spinigerus*, b) *Pygopus lepidopodus*, c) *Aprasia repens*, d) *Ctenotus fallens* e) *Hemiergis quadrilieata*, f) *Lerista elegans*, g) *Tiliqua rugosa*, h) *Pogona minor*, i) *Hydromys chrysogaster*, j) *Tarsipes rostratus*, and k) *Isoodon obesulus fusciventer*. Photo credit: a-j) L. Valentine, k) DEC.

Total species richness varied among sites, from a minimum of two to a maximum of 18 species, with the highest number of species recorded at two long unburnt (> 20 YSLF) Bassendean sites in *Banksia* woodland. The two sites located on the Quindalup landform in coastal scrub vegetation were also species rich, with 15 species captured at both sites. One of these species, *Lerista lineopunctulata*, was only captured in the coastal scrub vegetation. The lowest species richness was recorded at a Spearwood site in *Melaleuca* dampland with young fuel age (3 YSLF). Total species richness also varied among landform units and vegetation type, with the highest species richness on Bassendean sites, and in *Melaleuca* and *Banksia* sites (Table 1). There were few differences in total species richness between recently burnt and long unburnt sites. However, sites were not trapped equally, and the relative species richness variable provides a more equitable measure. Based on this, Quindalup Dunes contains the highest species richness of landform units, with coastal scrub vegetation type and *Banksia* woodlands being the most species rich sites (Table 1).

The most widespread or commonly-occurring reptile species across all sites were *Lerista elegans*, (85 % of sites), *Cryptoblepharus buchananii* (75 % of sites) and *Menetia greyii* (68 % of sites) (Table 1). A total of 1 163 reptiles were captured in pitfall traps. Of these, the most abundantly captured species were *Lerista elegans* and *Menetia greyii*, making up 21 % and 17 % of total reptile captures respectively. There were 7 reptile species and 60 individuals caught by cages and elliots, with a vast majority of these (80 %) being *Tiliqua rugosa* (Bobtail). Three reptile species were caught exclusively by cages and elliots (Table 1). Several species of snake were only captured in sites with older fuel ages, including the blind snake *Ramphotyphlops australis* and the two small elapids *Brachyurophis semifasciata* and the Priority listed species *Neelaps calonotos*.

The most widely distributed frog species was *Limnodynastes dorsalis* (pobblebonk frog), occurring at 30 % of sites. Of the 74 frogs recorded, the most abundantly captured frog was *Helioperous eyrei* (moaning frog) making up 42 % of total frog captures. Six species of frog and reptile were captured at only one of the 40 sites (*Crinia insignifera, Pseudophryne guentheri, Brachyurophis semifasciata, Demansia psammophis reticulata, Delma concinna concinna* and *Lerista lineopunctulata*). For four of these species only one individual was captured (*C. insignifera, B. semifasciata, D. concinna concinna* and *L. lineopunctulata*). Eleven frog and reptile species were found exclusively in Bassendean sands, while two species were found exclusively in Quindalup Dunes.

Mammal capture rates were fairly low with the introduced *Mus musculus* (house mouse) occurring at 45 % of sites and the native *Tarsipes rostratus* (honey possum, noolbenger) trapped at 30 % of sites. In addition, one *Rattus fuscipes* (bush rat, moodit) and three

Tachyglossuss aculeatus (echidna) were captured during general fauna trapping. During general fauna surveys, the most consistently captured native mammal species was the honey possum. A total of 18 individuals were captured at 12 sites which were widely distributed across the study area from the west to the east. Honey possums only occurred on Bassendean and Spearwood soils and were associated with *Banksia*, jarrah and *Melaleuca* vegetation types. During the targeted rakali and quenda trapping, a number of non-target species were captured (see results section below).

General fauna surveys

Abundance, species richness and diversity

In total, 37 species of reptiles were captured at the 38 sites on Spearwood and Bassendean Dune systems. Of the three factors examined, vegetation type influenced the greatest number of response variables, including reptile abundance, species richness, diversity, evenness, plant species number and a number of individual species (Table 2). Mean reptile abundance was similar between both landform units but varied among vegetation types (Table 2), with greater abundance of reptiles observed within tuart-dominated vegetation (Figure 4). In contrast, reptile species richness was highest in *Banksia* woodland sites (Table 2; Figure 4). Although the vegetation type coastal scrub was not included in any analyses, we have presented the standardised average abundance of reptiles in coastal scrub appears slightly higher than *Banksia* or *Melaleuca* sites, while the species richness of coastal scrub sites appears variable (Figure 4).



Figure 4. Differences in the mean (\pm 95%CI) a) reptile abundance and b) species richness of different vegetation types. Letters above error bars indicate significant differences of means between vegetation types (Tukey HSD, $\alpha < 0.05$). Coastal Scrub vegetation type was not included in analyses.

Both reptile and plant species richness were highest in *Banksia* woodlands (Table 2; Figure 5) however reptile diversity did not significantly vary among vegetation types. Evenness was significantly lower in tuart sites, suggesting that the sites were dominated by a small number of species. Interestingly, sites in jarrah-dominated vegetation tended to represent a composite of *Banksia* and tuart sites, with intermediate abundance levels, reptile species number and plant species number. A number of individual species responded to vegetation type, with several species, including the frog *Heleioporus eyrie*, and the skinks *Hemiergis quadrilineata*, *Lerista elegans*, *Menetia greyii* and *Morethia obscura* observed in higher abundances in tuart vegetation (Table 2; Figure 5). In contrast, the heath dragon *Rankinia adelaidensis* was most abundant in *Banksia* woodlands (Figure 5).

Reptile abundance was higher in long unburnt sites (Table 2; Figure 6). In addition, the skinks *M. greyii* and *M. lineoocellata* had higher abundances in long unburnt sites (Table 2; Figure 6). The introduced house mouse *Mus musculus* had higher abundances in recently burnt sites. Landform unit influenced a few species, including the dragon *R. adelaidensis*, found in higher numbers on Bassendean sites, and the skink *H. quadrilineata* and introduced house mouse *M. musculus*, observed in higher numbers on Spearwood sites.



Figure 5. Significant differences in mean (\pm 95%CI) a) reptile evenness, b) plant species richness, and c-d) selected individual species abundances among different vegetation types. Letters above error bars indicate significant differences of means between vegetation types (Tukey HSD, $\alpha < 0.05$).

Table 2. ANOVA F-values for reptile abundance, species richness, diversity and evenness and plant species richness and individual species abundances showing responses to landform unit, vegetation type and time since fire.

	Landform $_{df = 1,36}$	Vegetation Type $_{df = 3,34}$	Time since fire $_{df=1,36}$
Reptile Abundance	0.119	3.874* T > M, B^	4.225* O > Y
Reptile Species Richness	0.082	3.688* B > M, T^	0.155
Reptile Diversity	0.100	$2.708^{B} > T$	2.453
Reptile Evenness	0.726	4.092* B,M, J > T	0.360
Flora Species Richness	0.991	6.499** B > M	1.445
	Select Species Ab	oundance	
Frogs			
Heleioporus eyrei	0.011	6.887** T > J,M,B	0.151
Myobatrachus gouldii	1.265	1.924	0.742
Reptiles			
Rankinia adelaidensis ~	4.357* B > S	5.096** B > T,M	1.063
Cryptoblepharus buchananii	0.838	1.998	1.595
Hemiergis quadrilineata	10.373** S > B	7.731*** T,J > M,B	1.093
Lerista elegans	0.223	3.309* T > M	0.863
Menetia greyii	0.051	3.028* T > B	6.422* O > Y
Morethia obscura	0.467	4.187* T > M,B	7.614** O > Y
Mammals			
Tarsipes rostratus ~	0.591	1.523	0.002
Mus musculus	18.164*** S > B	0.599	8.502** Y > O

Significant values are in bold (* P < 0.5, ** P < 0.01) and values approaching significance are identified (^ $0.06 > P \ge 0.05$). Letters beside significant values indicate results from post-hoc Tukey HSD tests for vegetation type (B = *Banksia*, M = *Melaleuca*, T = tuart and J = jarrah) or which landform unit (B = Bassendean, S = Spearwood) or time since fire (O = Old, > 16 YSLF; Y = Young, < 11 YSLF) had higher abundances. ~ indicates that variable did not meet Levene's test of Equality for Equal Variances.



Figure 6. Significant differences in the mean $(\pm 95\%$ CI) of a) reptile abundance and b) abundance of *M. obscura* between long unburnt (Old) and recently burnt (Young) sites.

Patterns in species composition

Of the 37 reptile species observed during pit-fall trapping surveys, 23 were detected in more than two sites and were included in community analyses. MRPP detected differences in community structure between landform unit (MRPP: A = 0.025, P = 0.006), vegetation type (MRPP: A = 0.106, P = 0.001) and time since fire (MRPP: A = 0.025, P = 0.009). Pair-wise comparisons indicated that *Banksia* sites were typically different to *Melaleuca* (P = 0.001), tuart (P < 0.001) and jarrah (P = 0.001) sites; *Melaleuca* sites were also different to tuart (P = 0.007) and jarrah (P = 0.009) sites.

NMDS ordination found a stable 3-dimensional solution representing 83 % variance, with axes one and two representing 66 % of the community variation (Figure 7). Different landform units separated mostly along axis 1, although there was substantial overlap of sites. Vegetation types separated along both axes 1 and 2, with *Banksia* and *Melaleuca* sites clearly separating apart, with tuart sites more similar to melaleuca sites, and jarrah sites representing a mixture between *Banksia* and *Melaleuca* sites, and *Banksia* and tuart sites (Figure 7). Separation of time since fire for sites was also along both axes 1 and 2. Species associated with *Banksia* sites include the skink *Lerista praepedita*, the dragon *R. adelaidensis* and the pygopod *Aprasia repens*, whereas species associated with *Melaleuca* sites include the skinks *Tiliqua rugosa* and *H. quadrilineata*, although this species was also associated with tuart sites. These three species, as well as *A. repens* were also associated with long unburnt sites, while *T. rugosa* was associated with recently burnt sites (Figure 7).

Litter cover, litter depth and canopy cover were associated with long unburnt sites and tuart sites, while touch pole counts at 60-80 cm and 80-100 cm were associated with recently burnt sites, mostly of *Melaleuca* vegetation. Soil cover and touch pole counts at 20-40 cm were associated with *Banksia* woodland sites.



Figure 7. NMDS ordination (Sorensen distance measure) on the assemblage of reptiles (n = 23 species) at 38 sites coded by a) landform unit, b) vegetation type and c) time since fire. The ordination is in three dimensions (stress = 0.135), with axis 1 and 2 plotted ($r^2 = 0.346$ and 0.306 respectively). (d) Correlations of species and habitat variables ($r^2 > 0.2$) with NMDS ordination.

Fauna and habitat parameters

The number of plant species was correlated with reptile species richness (Pearson's r = 0.452, P = 0.006, Figure 8), but not reptile abundance. However, reptile abundance was correlated with times since fire (Pearson's r = 0.327, P = 0.045; Figure 8), unlike reptile species richness. In addition, reptile abundance was correlated with a number of habitat parameters (Table 3), including positive associations with litter cover, canopy cover and litter depth, and negative associations with soil cover and a number of the touch pole count

intervals (Table 3; Figure 8). The association of reptile abundance with litter depth indicated that the pattern was only significant in long unburnt sites (Figure 8), with trend lines indicating r^2 values for the subset groups time since fire. Of the touch pole counts, the interval 20 - 40 cm was positively correlated with all the reptile diversity measures, and the number of plant species, but negatively correlated with the abundance of reptiles (Table 3; Figure 8). The number of plant species was negatively correlated with other touch pole count intervals (intervals > 40 cm; Table 3). Time since fire was only significantly correlated with litter depth (Table 3).

	Reptile Abundance	Reptile Species Richness	Reptile Diversity	Reptile Evenness	Flora Species	Time since Fire
Vegetation cover	-0.017	0.396*	0.373*	0.192	0.276	0.170
Litter cover	0.475**	-0.267	-0.173	-0.192	0.128	0.170
Soil cover	-0.466**	0.133	0.011	0.085	-0.159	-0.219
Canopy Cover	0.447**	-0.380*	-0.179	-0.310	-0.173	0.153
Litter Depth	0.512**	-0.241	-0.019	-0.084	0.046	0.433**
Touch pole counts						
0 - 20 cm	-0.077	0.249	0.289	0.209	0.180	0.098
20 - 40 cm	-0.336*	0.504**	0.361*	0.331*	0.375*	-0.206
40 - 60 cm	-0.430**	-0.016	-0.033	0.243	-0.346*	-0.286
60 - 80 cm	-0.346*	-0.115	-0.170	0.082	-0.623**	-0.221
80 - 100 cm	-0.353*	-0.124	-0.209	0.039	-0.634**	-0.264
100 - 150 cm	-0.208	-0.286	-0.277	-0.072	-0.626**	-0.249
150 - 200 cm	-0.274	-0.390*	-0.255	0.020	-0.469**	-0.298

Table 3. Pearson's correlations (r) of reptile abundance, species number, diversity, evenness, plant species number and fuel age with habitat variables at each site.

Significant values are in bold (* P < 0.5, ** P < 0.01) and values approaching significance are identified (^

 $0.06 > P \ge 0.05).$



Figure 8. Associations of a-b) reptile species and c-e) reptile abundance with various habitat parameters, and, f) reptile abundance with litter depth, with times since fire categories identified. Linear trend lines are plotted for each graph, and an r^2 value provided. Trend lines and r^2 values are for all data, with the exception of f) where they identified the subset groups based on time since fire.

Succession with time since fire in Banksia woodlands

In the *Banksia* woodland sites, 17 reptile species were detected in more than two sites and were included in describing patterns of succession. Reptile species responded in numerous ways to time since fire, with relative abundance estimates peaking at every time since fire category for at least one species of reptile (Figure 9). The relationship with time since for a number of species could be described as 'early' successional, with species such as *Rankinia adelaidensis* and *Ctenotus australis* having peak relative abundances in sites of 4 YSLF (Figure 9). Interestingly, most of the species described as being early successional respondents showed a second, lower 'peak' in relative abundance, often in the very old sites. This pattern was particularly pronounced for the dragon *R. adelaidensis* (Figure 9), indicating that this species was abundant in both young and very old sites. Some species, such as *Ctenotus fallens* and *Stropherus spinigerus*, displayed an 'intermediate, intermediate or old categories. Finally, a number of species (n = 7), including *M. greyii*, *M. obscura* and *Aprasia repens*, displayed 'late' successional responses, with peak abundances in the old and very old categories (Figure 9).

Only two mammal species, *Mus musculus* and *Tarsipes rostratus*, were captured frequently enough to examine patterns in succession with time since fire. The introduced *Mus musculus* peaked in relative abundance in the young-intermediate category, and then progressively declined with time since fire (Figure 10). In contrast, the native *Tarsipes rostratus* had low relative abundance in the young category and progressively increased in relative abundance to peak in the old category (Figure 10). However, the relative abundance dropped markedly in the oldest unburnt sites (very old category).



Figure 9. Successional responses of reptiles (using relative abundance estimates) to time since fire. Responses are separated into a) early, b) intermediate, and c) late.



Figure 10. Successional responses of *Mus musculus* and *Tarsipes rostratus* (using relative abundance estimates) to time since fire.

Targeted quenda and rakali trapping

Quenda (*Isoodon obesulus fusciventer*) were recorded from 5 of the 9 selected sites where trapping for quenda was conducted. At two sites, Little Badgerup Swamp and Nowergup Nature Reserve, only one individual was captured during trapping. However, their abundance was relatively high (\geq 5 individuals per 100 trap nights) at the three other sites (Figure 11). The highest numbers of quenda were recorded at Twin Swamps Nature Reserve, which was the only site that is fenced and baited against predators. In addition to quenda, non-target native captures included *Rattus fuscipes* (bush rat), *R. rattus* (black rat), *M. musculus* (house mouse), *Notechis scutatus* (tiger snake) and *Tachyglossus aculeatus* (echidna). *R. fuscipes* were only captured at two sites (Figure 11), but were recorded in high density at Loch McNess. The introduced *R. rattus* and *M. musculus* were observed at most sites (Figure 11).



Figure 11. Abundance of four mammal species at sites targeted for trapping quenda. The abundance of quenda was highest at Twin Swamps Nature Reserve, which is fenced and baited to reduce predators. (M'burra NR = Muckenburra Nature Reserve).

Rakali (*Hydromys chrysogaster*) were captured at all three of the selected sites. Although Lake Goolellal had the least survey effort, this site produced the most number of rakali (6 individuals from 30 trap nights), indicating that a sizable population may reside at this lake (Figure 12). Rakali tended to be captured at sites where vegetated islands occurred within 50 – 100 m of the trap site. In contrast to Lake Goolellal and Loch McNess, only one individual was captured at Lake Joondalup, which is substantially larger than the two other lakes. However, the vegetation surrounding Lake Joondalup had recently been burnt (< 3 YSLF), and hence visibly differed in structure to the vegetation at Lake Goolellal and Loch McNess. In addition, Lake Joondalup lacked islands close to the lake edge. Non target captures included *R. fuscipes*, *R. rattus* and *M. musculus* although captures of these species were fairly low (Figure 12).



Figure 12. Abundance of three mammal species at sites targeted for trapping Rakali. The abundance of Rakali was highest at Lake Goolellal.

Discussion

General observations

The fauna survey results indicate a diverse and rich assemblage of fauna within the GSS study area. Of the 64 reptile species believed to occur in the GSS region (Huang 2009), the fauna surveys recorded nearly 60 % of species. Although our study did not target frog captures, we also recorded 6 species of the expected 13 frog species recorded within the GSS study area (Bamford and Huang 2009). Furthermore, our fauna trapping surveys, targeted surveys and general observations recorded 9 native mammal species. Historically, up to 33 mammal species have been recorded on the northern Swan Coastal Plain, however, only 11 - 12 species are considered to be currently extant (Kitchener *et al.* 1978; Reaveley 2009). The results from our comprehensive survey strongly support earlier conclusions that the northern SCP is a species rich area (Western Australian Museum 1978).

The most widespread species included the skinks *M. obscura, M. greyii* and *L. elegans*. The skink family is very widespread across the SCP as they are highly diverse and adaptable to a range of habitats (How and Dell 2000). Previous work has shown that this group of lizards will inhabit a range of vegetation types, and are one of the only groups that maintain populations in small isolated remnant vegetation patches (How and Dell 1993; 1994). In general, fauna were fairly evenly distributed across landform, vegetation and time since fire age sites. However there were some obvious exceptions. Eleven frog and reptile species were found exclusively in Bassendean sands. Of these, 5 species were elapids, potentially indicating a preference of elapids for Bassendean dunes. In addition, the majority of the burrowing snake species, including the Priority listed elapid *N. calonotos* were captured at long unburnt sites. Further trapping for the Priority-listed species (*N. calonotos*) may identify more individuals of this little-known species. In addition, the inclusion of funnel traps that target small elapids and large skinks, may increase capture rates for this and other species.

The coastal scrub vegetation of the Quindalup Dunes was not a target of this study, however the preliminary results indicate that this vegetation type is fairly species rich in reptile fauna in comparison to the *Melaleuca* and tuart vegetation types. In addition, one

species, *L. lineopunctulata* was only observed in the coastal scrub sites. How and Dell (1993) similarly observed the coastal scrub vegetation to be very rich in reptile species. Further survey effort within the Quindalup Dune system is crucial to completely document the reptile fauna of this habitat, particularly given that the coastal scrub remnant vegetation is threatened by encroaching urban development projects.

In general, mammal capture rates were very low, with the exception of *M. musculus* and *T. rostratus*, which were the most frequently captured mammal species. Mammal species not recorded in our study that were present in the WAM study in 1977 – 1978 include *Pseudomys albocinereus* (ash grey mouse). Since this survey, the ash grey mouse has only been recorded once at Melaleuca Park (in 1987) and there have been no further records in the GSS study area. Their absence from any fauna studies spanning the 22 years since 1987, suggest this species has either disappeared from this part of the northern Swan Coastal Plain, or is persisting in low density or at specific sites not surveyed. In addition, our study did not capture *Dasyurus geoffroi* (chuditch) or (*Cercartetus concinnus*) western pygmy possums.. Western pygmy possums were trapped in 2005 in the Lexia Wetlands (Rob Davis pers. comm.) and there are occasional public reports of chuditch (Brent Johnson pers. comm.). More intensive targeted survey effort is required to validate their persistence in the GSS study area.

Fauna trapping surveys

Patterns in diversity across vegetation types and landform units

Habitat structure and composition strongly influences faunal assemblage, as differences in these attributes invariably lead to differences in the resources available for fauna. Reptiles in particular are strongly dependent on habitat structure for their survival (Pianka 1989). In our study, the dominant type of vegetation at a site was a major factor influencing faunal abundance, species richness, and community structure. Each vegetation type varied considerably in the composition of their broad habitat characteristics as well as within their microhabitat and vegetation variables. When a species was associated with a vegetation type, it was also often correlated with microhabitat attributes common in that habitat. For example, tuart sites contained the highest abundance of reptiles, but were dominated by only a few species, typically those that were litter dwellers and associated with litter depth,

including *H. quadrilineata* and *M. greyii*. Correspondingly, tuart sites were associated with deep piles of litter, potentially providing the required resources for these species.

Banksia woodland sites were clearly the most species rich for reptiles and plants. *Banksia* woodlands are floristically diverse (Dodd and Griffin 1989), especially in the lower strata (< 0.5m) and this floristic diversity may reflect a structural diversity that provides a range of habitat resources for a suite of reptile species. Indeed, reptile species richness was correlated with both plant species richness, and the number of touch pole counts below 40 cm. Several species, with a variety of habitat preferences, were associated with *Banksia* woodlands sites, including the sand swimming skinks *L. elegans* and *L. praepedita*, the burrowing worm lizard *A. repens*, the relatively open-living heath dragon *R. a. adelaidensis* and ground-living skink *M. obscura*, a species which was only captured in *Banksia* woodland.

In contrast to *Banksia* woodland sites, *Melaleuca* sites contained fewer reptile species and plant species. This habitat type was more similar in vegetation composition to tuart sites, with high amounts of canopy cover and deeper litter piles. Consequently, the litter dwelling skinks common in tuart, *H. quadrilineata* and *M. greyii* were also associated with *Melaleuca* habitat. The jarrah-dominated sites tended to represent a combination of tuart/*Melaleuca* and *Banksia* sites, with intermediate levels of reptile and plant species richness, and only reptile one species *H. quadrilineata* observed in high abundances. This may reflect the potential intermediate levels of habitat structure.

The diversity of reptiles in *Banksia* suggests that this habitat provides a correspondingly diverse array of habitat opportunities for reptile species. The biological and ecological functions of reptiles are dependent upon body temperature (Heatwole and Taylor 1987). As ectotherms, the body temperatures of small reptiles are largely dependent upon habitat temperatures (Heatwole and Taylor 1987), and thermal preferences of reptiles may influence habitat selection (e.g. Valentine *et al.* 2007). The diverse lower vegetation strata of *Banksia* woodlands may provide a range of thermal habitat, suitable for numerous species. In contrast, *Melaleuca* sites tended to have high amounts of canopy cover and high amounts of structure in the 1 - 2 m height category that may limit the thermal range at ground level. Skinks of the genus *Hemiergis* are often associated with cool, moist locations (Bush *et al.* 2007), and the high abundances of these lizards in the tuart, jarrah

and to a lesser degree *Melaleuca* sites, suggests that these areas tend to stay cooler than *Banksia* woodlands.

Landform types did not drive patterns in species composition, and where differences in abundance between landform types were detected, they were generally reflecting a preference for a specific habitat types. For example, *H. quadrilineata* was more abundant in Spearwood versus Bassendean dune systems, however this is likely to be because the tuart and jarrah sites that this species preferred, were only located on Spearwood dunes. Only the introduced *M. musculus* displayed a clear preference for the Spearwood dune systems without showing a clear vegetation preference. This may be because the Spearwood dune sites are in closer proximity to habituated areas, and the introduced *M. musculus* are typically more abundant in urbanised areas.

Previous work in the northern SCP has recorded a distinct decrease in species richness in landform units, heading eastwards (How and Dell 1993). Although we did not observe this pattern, we only intensively sampled two distinct landform units (Bassendean and Spearwood), in contrast to the five landform units that How and Dell (1993) surveyed. The substantial work on reptile diversity patterns conducted by How and Dell (1993; 1994; 2000) have been principally located in urban remnant, while our research targeted larger tracts of remnant vegetation.

Patterns in diversity with time since fire

Our results indicate that the response of reptile communities to time since fire varied among different combinations of vegetation type and time since fire. In this study we have focussed on examining differences in reptile abundances between recently burnt and long-unburnt sites, across several vegetation types. Although vegetation type strongly influenced community composition, time since fire also influenced reptile assemblages. Our surveys show that overall reptile abundance, as well as the abundance of some specific species, was higher in long unburnt sites. In addition, the majority of the burrowing snake species, including the Priority listed elapid *N. calonotos* were captured at sites of old fuel age. This perhaps reflects some difference in resource availability between recently burnt and long unburnt sites. In previous studies, skink-consuming elapids tend to be absent, or in lower abundances in recently burnt habitat (Valentine and Schwarzkopf 2009).

Changes in the abundance of reptiles following burning is often linked to fire-induced changes in the resource availability of the post-fire environment (Friend 1993; Masters 1996). Because reptiles tend to occupy sites with suitable thermal, shelter, and food resources (Friend 1993; Letnic *et al.* 2004; Masters 1996), burning may have modified elements of the habitat in a manner undesirable to some species. The recently burnt sites contained deeper piles of litter, and those species with a preference for deeper litter, were observed in high abundances in the long unburnt sites. Typically, litter-associated lizards, such as *M. greyii* in our study, respond strongly to the removal of vegetation and are usually observed in high abundance in the least-disturbed sites, and their density is often correlated with variables of vegetation cover (e.g. litter cover; Greenberg *et al.* 1994; Masters 1996).

Reptile species tend to have species-specific habitat preferences (Letnic *et al.* 2004), and the preferred attributes (e.g. deep leaf litter) are likely to be in dissimilar supply among different vegetation types. In addition, the affect of burning on these attributes is also likely to differ among vegetation types. Further analysis, focussing on the interaction of time since fire and vegetation type, is currently underway and will be presented in separate documents (Valentine and Wilson, unpublished data). It is very likely that the responses of reptiles to fire will be dictated by the type of vegetation that is burnt.

Succession with time since fire in Banksia woodlands

Understanding succession of fauna with time since fire is often desirable, as this understanding can lead to more informed fire management strategies, and highlights habitat preferences of target species. With reptiles, succession following fire tends to be habitat dependent. Long-term studies of reptile assemblages in arid regions (Letnic *et al.* 2004; Masters 1996; Pianka 1989), forests (Caughley 1985; Taylor and Fox 2001), and sand-pine scrub (Greenberg *et al.* 1994) consistently indicate a reptile succession with time since fire as different species dominate when appropriate habitat presides. However, this pattern is not obvious in tropical savannas, possibly due to the high frequency of fires (Braithwaite 1987; Valentine and Schwarzkopf 2009; Woinarski *et al.* 1999). Our study highlights that the responses of reptiles to fire in *Banksia* woodlands are fairly complex. Several species preferred recently burnt sites, whilst others were most abundant in intermediate fuel age sites, and still other species were abundant in long unburnt sites. Furthermore, several species displayed a cyclic response to time since fire, with relative abundances peaking in both recently burnt and long unburnt sites. This indicates that a diverse range of post-fire habitat is necessary to cater for the species rich reptile fauna in the GSS study area.

The responses of mammals to fire were fairly clear. The introduced *M. musculus* preferred more recently burnt sites, and is often associated with habitat that is disturbed in some form. In contrast, honey possums were more abundant in older sites, with peaks in relative abundance at sites 20 - 26 YSLF. Although honey possums are know to return to burnt areas within 2 – 4 years since fire (Bamford 1986; Everaardt 2003; Richardson and Wooller 1991), higher densities are typically recorded in longer unburnt sites sites, with peaks in abundance in the 20 - 30 years since last fire (Bradshaw *et al.* 2007; Everaardt 2003). Our results are very similar, with low abundance in recently burnt sites (<7YSLF), followed by an increase in abundance as time since fire increases. However, in the Banksia woodlands in the GSS study area, we also noticed lower abundances in sites that have remained unburnt for a very long time (> 36 YSLF). Honey possums are dependent on nectar and pollen, particularly from plants of the Proteaceae, Myrtaceae and Epacridaceae families (Wooller et al. 1984). Capture rates of honey possums are closely linked to food sources (Bradshaw et al. 2007) and have been correlated with the densities of flowers and the flowering periods of Banksias (Everaardt 2003). Hence, the impact of fire on honey possums will be related to the post-fire responses of target food species (Bradshaw et al. 2007).

Targeted Trapping Surveys

Quenda

Quenda were chosen as a species to target for survey effort as their distribution in the GSS study area appears to be linked to wetland-associated vegetation (Bamford and Bamford 1994). In the GSS study area, quenda were originally recorded occurring as far north as Moore River, which was the northern end of its distribution in Western Australia. It was

historically considered plentiful near Perth (Glauert 1933; Kitchener *et al.* 1978). However, Kitchener et al. (1978) claimed that by the time of the Western Australian Museum (WAM) 1977-78 survey, this species was only sparsely distributed in the GSS study area, in thick vegetated damp areas and only two specimens were captured during that comprehensive survey. Prior to the GSS study, Whiteman Park and Ellenbrook Nature Reserve were the only two known areas within the GSS study area that could confirm a strong secure population (Bamford and Bamford 1994). Given that most other mammal species have declined or disappeared from the GSS study area, the occurrence and distribution of one persisting species such as the quenda is quite significant, and the habitat in which it is persisting of high conservation value.

The combined results of the targeted field survey in May 2008 (this study), the 2008 community survey and recent existing data from Whiteman Park and Ellenbrook Nature Reserve (Reaveley 2009), indicate that quenda are widespread in their distribution from west to east across the GSS study area. A community survey (Reaveley 2009) found that populations are persisting almost continuously in the coastal vegetation from Burns Beach to Two Rocks. Inland there is evidence of populations inhabiting a narrow west-east strip from Nowergup in the north-west to Whiteman Park in the south east. Quenda were not recorded from the northern portion of the GSS study area from either actual survey effort nor community feedback.

The quenda populations living in the coastal regions are largely occupying vegetation on private property that is proposed for development (Reaveley 2009). The majority of this vegetation will be lost to urbanisation and subsequently habitat for quenda will either be greatly diminished or fragmented. The habitat that quenda are occupying in the west-east band of vegetation from Nowergup across to Ellen Brook and south to the northern edge of Perth metropolitan suburbia is often associated with swamps or lakes. Quenda were typically found in moist low-lying areas with dense mid-storey vegetation. Similarly, studies at Whiteman Park have indicated that quenda are persisting in the dense mid-storey level heath associated with wetlands (Bamford and Bamford 1994).

The association of quenda with wetland-associated vegetation is possibly linked to the potential protection this habitat provides from fox predation (Bamford and Bamford 1994). Indeed, when fox baiting was introduced at Whiteman Park, quenda both increased in

number and began using upland areas as the threat of predation diminished (Bamford and Bamford 1994). In our study, quenda were only observed in high densities at Twin Swamps Nature Reserves, indicating a suppression of quenda populations from fox predation in unbaited habitat. In other parts of their range where baiting occurs, such as the jarrah forest on the Darling Scarp, quenda inhabit a variety of habitats including open woodland and upland areas. In the GSS study area, quenda may be favouring the dense wetland-associated vegetation habitat type to a greater extent because of the presence of foxes. Hence, the persistence of quenda in unbaited areas in the GSS study area is strongly reliant on dense wetland-associated vegetation.

Rakali

The number of rakali trapped at Lake Goolellal and Loch McNess, with minimal survey effort was surprising, indicating that these two lakes support reasonable populations of this species. Typically, rakali tend to be easily captured in areas where they are numerous. Given the lack of previous records for rakali within the GSS study area, it is likely that small survey efforts have previously overlooked this important wetland species. Further surveys may establish that Rakali persisting in most of the permanent wetlands within the GSS study area.

Rakali are mostly carnivorous, foraging in the water and also hunting on land. Their diet includes freshwater fish and crustaceans, mussels, aquatic insects and frogs, lizards, water birds, small mammals and turtles (Strahan 2004). The survival of rakali is critically linked to the persistence of wetland eco-systems. They are a species that will require consideration and careful management as climate change impacts the viability of current wetlands persisting today.

The future for biodiversity in a drying climate

This project has focused on identifying some of the biodiversity values of the Gnangara groundwater system, and will provide a baseline for future work in the area. The reptiles of the GSS study area are particularly species-rich. Long-term reductions in groundwater levels and rainfall are likely to alter lake systems by drying wetland areas (Froend *et al.* 2004), and will ultimately impact the diversity of habitat in the GSS study area. Declines

in the extent and condition of wetlands and *Banksia* woodlands are predicted to impact on vertebrate biodiversity (Bamford and Bamford 2003; Government of Western Australia 2009b). However, the extent to which species will be affected is still uncertain. The diverse array of reptile species occurring in the GSS study area likely to respond in a variety of ways, and greater understanding of the ecological requirements of species is necessary. Clearly wetland associated species, such as rakali and quenda, are threatened from declines in groundwater and rainfall levels (Bamford and Bamford 2003). Identifying potential strongholds for these species (and others) and managing these areas as refugia will be critical in the future, especially as wetlands dry and connectivity decreases between wetland communities.

Burning to increase groundwater recharge is a potential mitigation measure in the face of declining rainfall levels and increased abstraction (Government of Western Australia 2009a). Our study provides some empirical data on how biodiversity may be affected by changes in fire regimes. Firstly, the relationship of fauna with fire varies depending on the vegetation type, and fire regimes should ideally be structured on the specific vegetation type at a local scale. Secondly, understanding species-specific relationships with fire is crucial. Developing fire management strategies suitable for groundwater recharge, which are also compatible with biodiversity, requires detailed information on faunal responses to fire. Our study highlights that different species are influenced by fire in different ways, with some species preferring recently burnt habitat, while other preferred intermediateolder or long unburnt habitat. Thirdly, the retention of unburnt refugia of each vegetation type will be crucial for successful recolonisation of recently burnt areas. At a landscape level, the remnant vegetation in the GSS currently has a mosaic of fuel ages, which may facilitate in dispersal and recolonisation following burning. A change in the fire regime (either increase frequency or intensity), aimed at increasing groundwater recharge, may alter the current mosaic, potentially removing unburnt refugia that are critical habitat for some species (e.g. honey possums). As with other fire-management practices, incorporating a range of fire regimes and maintaining unburned refugia when planning burning for groundwater recharge, may help maintain overall biodiversity (Woinarski et al. 1999).

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Appendix

Appendix 1. Number of sites across each landform unit, fuel age group and vegetation type.

		Landform Unit and Vegetation Type								
	Bass	sendean		Spearwo	Quindalup					
Fuel Age	Banksia	Melaleuca	Banksia	Melaleuca	Jarrah	Tuart	Coastal			
							scrub			
<11 YSLB	4	2	4	6	2	2	NIL			
>16 YSLB	6	6	2	NIL	2	2	2			

Appendix 2. Pit-fall trap design layout



Appendix 3. Species captured or observed during trapping sessions in the GSS study area, including targeted mammal trapping. Species nomenclature primarily follows the Westralian Australian Museum database (October 2008); ^ denotes common names of reptiles obtained from Bush *et al.* (2007).

Family	Species	Common Name						
U U	AMPHIBIANS							
Frogs: Class Amphibia, Order Salientia								
Myobatrachidae	Crinia insignifera	Squelching Froglet						
5	Heleioporus evrei	Moaning Frog						
	Limnodvnastes dorsalis	Banjo Frog						
	Myobatrachus gouldii	Turtle Frog						
	Pseudophryne guentheri	Günther's Toadlet						
Hylidae	Litoria moorei	Motorbike Frog						
	DEDTH ES							
	KEFTILES Lizards: Class Reptilia. Order Sauamat	ta. Suborder Sauria						
Galdonidaa	Christians mannonatus	Marbled Geaka						
Gerkomdae	Diplodactalus polyophthalmus	Speekled Stope Geeke						
	Steenhumus aniai e emus aniai e emus	Speckled Stolle Gecko						
Decemential	Strophurus spinigerus spinigerus	South-western Spiny-tailed Gecko						
Pygopodidae	Aprasia repens	West Coast Isualin Lizard						
	Delma concinna concinna	west Coast Javenn Lizard						
	Delma fraseri	Fraser's Legless Lizard						
	Delma grayii Lialia huntania	Gray's Legless Lizard						
	Lians durionis $D_{1,\alpha}$ and $D_{2,\alpha}$	West Coast Keeled Lealers Lizerd						
	Pietnolax gracilis gracilis	Common Social East						
A somidos	Pygopus lepidopodus	Common Scaly Foot						
Agamidae	Pogona minor minor Regulinia a delaidencia a delaidencia ^{a}	Dwarf Bearded Dragon						
Vananidaa	Kankinia adelalaensis adelalaensis	Western Heath Dragon						
Varanidae Sainaidae	Varanus goulait*	Gould's Monitor						
Scincidae	Acritoscincus trilineatum	Southwestern Cool Skink ^A						
	Cryptoblepharus buchananii	Fence Skink						
	Ctenotus australis	West Coast Long-tailed Ctenotus						
	Ctenotus fallens	West Coast Ctenotus ^A						
	Ctenotus impar	South-western Odd-striped Ctenotus						
	Egernia kingu	King's Skink						
	Egernia napoleonis	Southwestern Crevice Skink						
	Hemiergis quadrilieata"	Two-toed Mulch Skink [^]						
	Lerista elegans	West Coast Four-toed Lerista						
	Lerista lineopunctulata	Line-spotted Robust Lerista ^A						
	Lerista praepedita	West Coast Worm Lerista^						
	Menetia greyii	Common Dwarf Skink						
	Moretnia lineoocellata	west Coast Pale-flecked Morethia						
	Morethia obscura	Southern Pale-flecked Morethia						
	Tiliqua occipitalis	Western Bluetongue						
	Tiliqua rugosa rugosa	Bobtail						
	snakes: Sudoraer Serpe	mues						
Typhlopidae	Ramphotyphlops australis	Southern Blind Snake						
Elapidae	Brachyurophis semifasciata ^b	Southern Shovel-nosed Snake						
	Demanisa psammophis reticulata ^b	Yellow-faced Whip Snake						
	Neelaps calonotos ^{a, c}	Black-striped Snake						
	Notechis scutatus	Tiger Snake						
	Parasuta gouldii	Gould's Snake						
	Pseudonaja affinis affinis	Dugite						
	Pseudonaja nuchalis	Gwardar						
	Simoselans hertholdi	Ian's Banded Snake						

MAMMALS

Tachyglossidae	Tachyglossus aculeatus	Echidna							
	Marsupial Mammals: Subclass Marsupialia, Order Dasyuromorphia								
Dasyuridae	Sminthopsis sp.	Dunnart							
Order Peramelemorphia									
Peramelidae	Isoodon obesulus fusciventer ^e	Southern Brown Bandicoot (Quenda)							
Order Diprotodontia, Suborder Phalangerida									
Tarsipedidae	Tarsipes rostratus	Honey Possum (Noolbenger)							
Phalangeridae	Trichosurus vulpecula	Common Brushtail Possum							
Macropodidae	Macropus fuliginosus*	Western Grey Kangaroo							
	Macropus irma*	Western Brush Wallaby							
Eutherian Mammals: Subclass Eutheria, Order Rodentia									
Muridae	Hydromys chrysogaster ^e	Water Rat (Rakali)							
	Mus musculus ^d	House Mouse							
	Rattus fuscipes	Bush Rat (Mootit)							
	Rattus rattus ^d	Black Rat							
	Order Carnivo	ra							
Canidae	Vulpes vulpes*	Fox							
Felidae	Felis catus ^e	Cat							
	Order Lagomor	<i>vha</i>							
Leporidae	Oryctolagus cuniculus ^e	Rabbit							
	Order Artiodact	tyla							
Suidae	Sus scrofa*	Pig							
Bovidae	Capra hicrus*	Feral Goat							
* Species not traj	pped - observed only by incidental observation	ations							

Monotremes: Subclass Prototheria, Order Monotremata

^a Species endemic to Swan Coastal Plain

^b Reptile species only captured at one site

^c Priority-listed species

^d Introduced species

^e Species only captured during targeted trapping for wetland-associated mammals

^f Formerly *C. plagiocephalus* as per Horner (2007).