Terrestrial vertebrate assemblage and species level patterns between major habitat types inside and outside a fenced exclosure in south-western Australia.

G.J. Yeatman, A.F. Wayne, and H. Mills

Understanding the processes that shape the distribution and abundance of species is necessary for effective wildlife conservation. This becomes more important for managers in areas where significant manipulation of the environment is undertaken. This study investigated patterns in the small vertebrate assemblage inside and outside a fully fenced exclosure in south-western Australia, and forms part of a larger monitoring project on the flora and fauna of the area. Three small mammal, thirteen reptile and nine frog species were recorded at 18 sites within three main habitat types. There were significant differences in the small vertebrate assemblage between habitat types. There was a decline in species richness and overall capture rates moving upwards in the landscape from creeks to ridge top sites. Eight of the small vertebrate species trapped accounted for 95% of the variation in the total species set. Creek sites were characterised by a greater relative abundance of frog species, particularly *Crinia glauerti*, *Pseudophryne guentheri* and *Heleioporus eyrei*. Slope sites had a greater abundance of some small mammal (*Sminthopsis griseoventer*) and reptile (*Morethia obscura*) species. Ridge sites were dominated by reptiles, in particular, *Lerista distinguenda*. The differences in the assemblage between habitats suggest that habitat level monitoring should continue as broader scale observation may lack sensitivity to changes within the different assemblages in the area. This study provides vital information for the management of this exclosure and as part of the wider monitoring project, has the potential to inform the management of other fenced areas.

Introduction

The use of exclusion fencing to conserve native fauna has become an increasingly necessary option for wildlife managers around the world (Whitehouse and Kerley, 2002, Ikuta and Blumstein, 2003, van Dyk and Slotow, 2003, Long and Robley, 2004, Hayward and Kerley, 2009, McGavin, 2009, Hameed *et al.*, 2012). The selective removal and exclusion of undesirable species and subsequent introduction of species of conservation concern is a radical form of ecological manipulation. A common objective of the managers of fenced areas is the conservation of focal species (Short *et al.*, 1994, Short and Turner, 2000, Whitehouse and Kerley, 2002, Richards and Short, 2003, Gross, 2009, de Tores and Marlow, 2012, Hayward, 2012). As a result of this type of single species management, it is often unclear whether human intervention has led to other profound ecosystem changes beyond those which were directly intended. Information about such changes has the potential to inform the managers of exclosures as well as broader species conservation. Particular areas of interest include identifying which species benefit and which are disadvantaged in fenced systems and whether exclosures can be self sustaining or require continued management intervention to prevent ecosystem collapse (Hayward and Kerley, 2009).

Currently in Australia there are 23 fenced exclosures designed to protect native fauna from the threat of introduced species (Long and Robley, 2004, Dickman, 2012). A common management aim for many of these exclosures is the conservation of medium-sized mammals (Moseby *et al.*, 2009, de Tores and Marlow, 2012, Dickman, 2012). As a result, where these areas are monitored, data collection is generally biased towards those target mammalian species (Moseby *et al.*, 2009, de Tores and Marlow, 2012). There is very little published literature regarding the potential impact that fencing may have on the ecology of vertebrate species in general, but a particular deficit in information relating to reptiles, amphibians and small, less mobile mammal species. Other than an increase in the density of medium-sized mammals in the absence of introduced predators (Kinnear *et al.*, 2002, Richards and Short, 2003, Dexter *et al.*, 2012), the consequences of establishing a fenced exclosure are not inherently obvious and the ecosystem-wide implications remain unclear.

Perup Sanctuary (PS) in the Upper Warren region of south-western Australia is one example of a fenced exclosure used as a management tool for the conservation of medium-sized mammals. The target species for conservation at Perup is the woylie (*Bettongia penicillata*) which has suffered a 95% decline in abundance in the Upper Warren region (Yeatman and Groom, 2012). The sanctuary was constructed with the aim of creating an insurance population of woylie which were protected from predation by introduced foxes (*Vulpes vulpes*) and cats (*Felis catus*). There are many elements to consider when investigating the potential impacts of the construction of a fenced area. These include, but are not limited to, the absence of introduced predators, restriction of movement of species for which the fence is a barrier, increased abundance of species limited by predation or competition with those species which have been removed, changed abundance of species (both plants and animals) which are responding to those released from predation/competition and changes in habitat structure in response to changes in the abundance of species which can modify habitats (Hayward and Kerley, 2009, Dickman, 2012, Hayward and Somers, 2012). At Perup it is also important to consider the additional removal of the native chuditch (*Dasyurus geoffroii*) which is a predator of small vertebrates, and the removal of large herbivorous macropods (*Macropus fuliginosus* and *Macropus irma*).

We set out to study the small terrestrial vertebrates as part of a wider initiative monitoring the flora and fauna in and outside the newly constructed PS. The aim of the wider project was to establish a baseline understanding of the ecology of the species in the area which could then be used as a reference point for future monitoring efforts. An essential foundation to the study of ecology is an understanding of the processes that shape the distribution and abundance of species (Odum, 1959, Krebs, 1985).With this in mind, this study investigated patterns in the small mammal, reptile and amphibian assemblage between three main habitat types. Specifically, we addressed the following questions:

1. Are there differences in the assemblage of small vertebrates found in each of the habitat types? In particular, are there differences in the structure of the assemblage and/or the abundance of species within the assemblage group?
2. Are there differences in the size of individuals of the same species that may relate to the habitat in which they are found? Are there characteristics of the habitats or the species themselves which may account for such differences?

Methods

Study area

Perup Sanctuary is a 423 ha fully fenced area approximately 50 km outside Manjimup (34.2506º S, 116.1425º E) in the Upper Warren region of south-western Australia (Figure 1). The sanctuary lies within Perup Nature Reserve in the southern jarrah forest and is managed by the Western Australian Department of Environment and Conservation (DEC). The region has a Mediterranean-type climate with warm dry summers and cool wet winters. The area is characterised by open dry sclerophyll forest in which the overstorey is dominated by a jarrah (*Eucalyptus marginata*) and marri (*Corymbia calophylla*), and in some places, wandoo (*Eucalyptus wandoo*).

The sanctuary fence was designed to withstand penetration by foxes, cats and rabbits (Oryctolagus cuniculus). Construction was completed in September 2010 and followed by an intensive program to completely remove foxes and cats. Western grey kangaroos (*Macropus fuliginosus*), brush wallabies (*Macropus irma*), all emu (*Dromaius novaehollandiae*), some brushtail possums (*Trichosurus vulpecula*) and all chuditch (*Dasyurus geoffroii*) were also removed as they were identified as potential problem species if left within the sanctuary. Forty one woylies were sourced from the surrounding Upper Warren population and released into the sanctuary in December 2010.

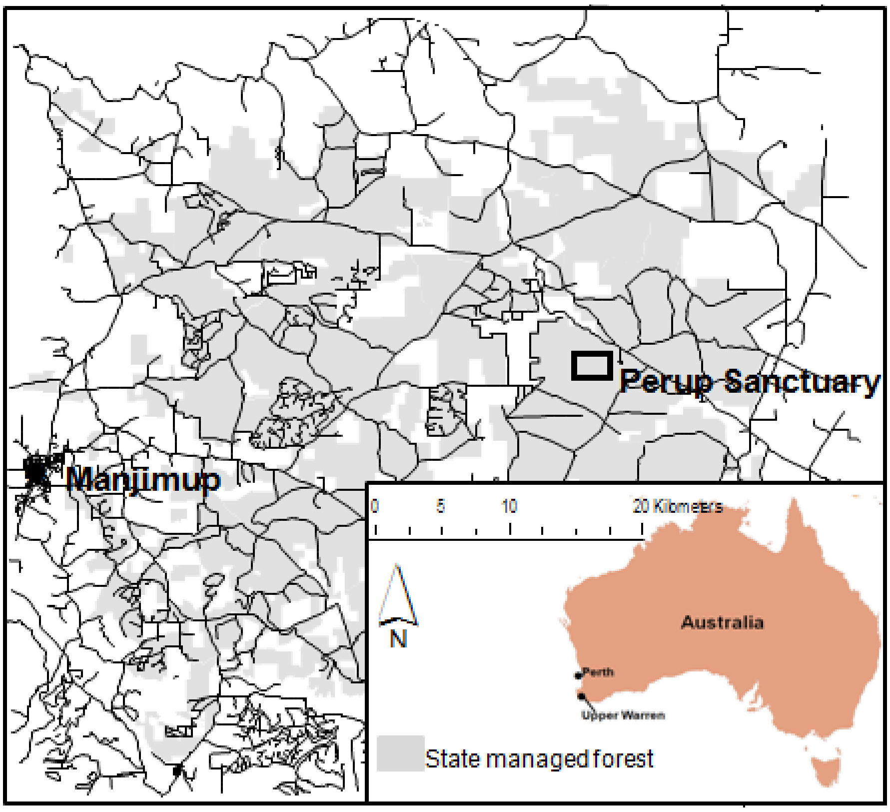

Figure 1: Location of Perup Sanctuary in relation to Manjimup and Perth in south-western Australia.

Small Vertebrate Trapping Surveys

Nine sites were selected inside the sanctuary and nine outside to the south west of PS (Figure 2). There were three replicates of each of three habitat types inside and outside the sanctuary. The three vegetation types were identified using Havel and Mattiske (2000) classifications. These were Yerraminnup flat, Yerraminnup and Bevan. These are broad and common vegetation types in Perup Nature Reserve and are the dominant types found within PS. Each of the types is characterised by position in the landscape, soil structure/hydrology and vegetation structure/composition.

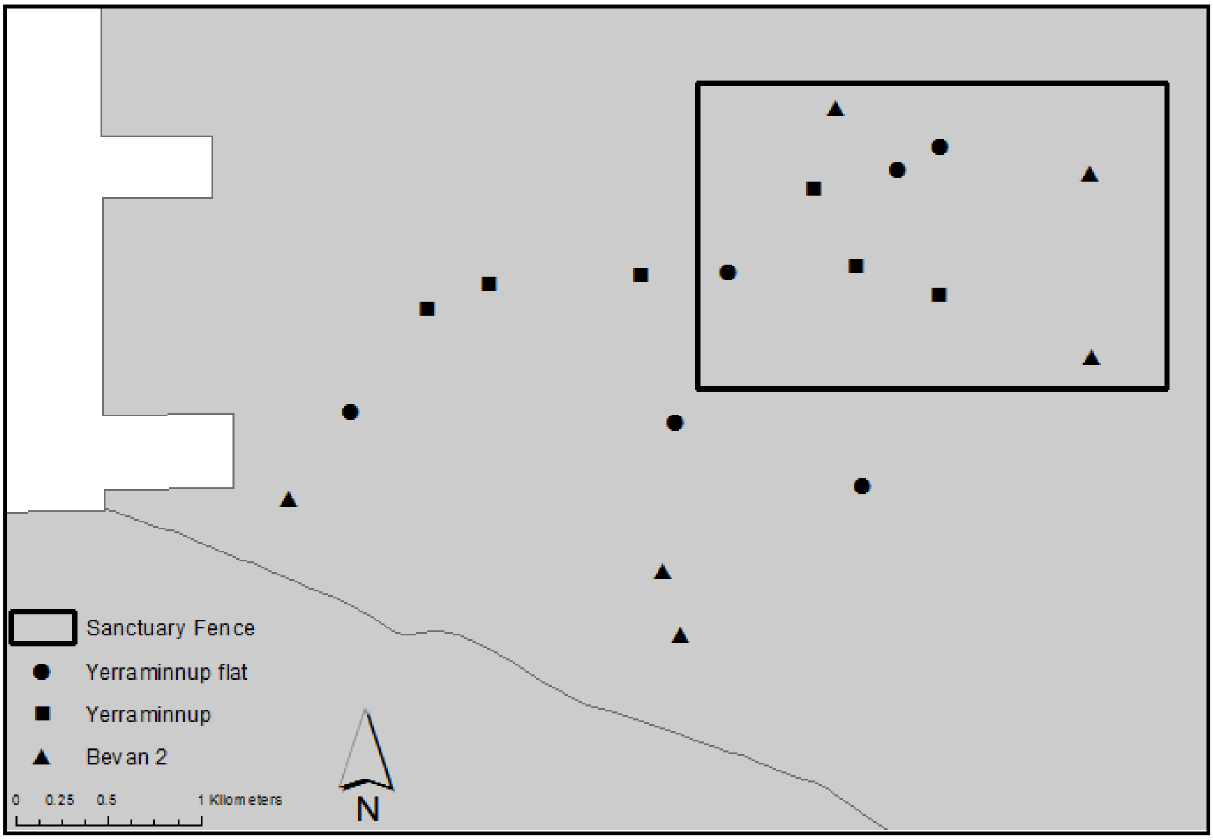

Figure 2: Map of 18 survey sites in Perup Nature Reserve. Symbols indicate habitat type of each site. There were three replicates of each of three habitat types inside and outside the sanctuary.

Yerraminnup flat describes the floor of a minor valley which has sandy loam topsoil over mottled clay. These sites are moderately fertile, water gaining areas which become seasonally waterlogged because of the lack of lateral drainage. There is less canopy cover at these sites and usually only scattered wandoo (*Eucalyptus wandoo*) and flooded gum (*Eucalyptus rudis*). The second storey is a mix of *Acacia* and *Hakea sp*. and the shrub/herb storey is dominated by thickets of *Melaleuca viminea* with some *Xanthorrhoea preissii*, *Hypocalymma angustifolium* and *Drosera bulbosa*. The six Yerraminnup flat study sites all had seasonal creeks flowing and were heavily water logged during winter months.

Yerraminnup describes the woodland slopes of a minor valley moving up away from the Yerraminnup flat sites. These areas are moderately fertile with gravelly sandy loam soil over sandy clay. Water is shed away from these sites and although the soil absorbs water well, it has a moderate to poor storage capacity. There is much greater canopy cover at Yerraminnup sites compared to Yerraminnup flat sites and the dominant species are marri and jarrah with some scattered wandoo. There is no clear second storey. The shrub/herb layer consists of *Acacia pulchella*, *Hakea lissocarpha*, *Hibbertia cunninghamii*, *Leucopogon capitelatus*, *Leucopogon propinqus*, *Macrozamia riedlei*, *Bossiaea linophylla* and *Trymalium ledifolium*.

Bevan describes the woodland and open forests of the ridges and upperslopes moving up further from the Yerraminnup sites. These areas have low fertility with gravelly sand topsoil and some lateritic duricrust. Water can infiltrate and be stored in these areas and there is some weak water shedding capacity. The overstorey is a mixture of jarrah and marri and the second storey is weakly developed with some *Persoonia longifolia* and *Banksia grandis*. The shrub and herb storey consists of *Bossiaea ornata*, *Astroloma pallidium*, *Macrozamia riedlei* and *Trymalium ledifolium*. For ease of reference, the site types will furthermore be referred to as creek (Yerraminnup flat), slope (Yerraminnup) and ridge (Bevan).

All sites except for PS7 and PS8 were >200 m apart with the average distance between one site and its nearest neighbour being >400 m. Sites PS7 and PS8 were 131 m apart. All sites except PS7 were at least 100 m away from tracks to reduce possible edge effects. Due to the reduced number of creek type habitats within the sanctuary at which sites could be established, site PS7 had to be established <50 m from the edge of a track. At each site, 25 trap points were arranged in a web formation (Figure 3). Traps were spaced 25 m apart along the eight arms of the web. Each trap point consisted of a 25 L bucket dug into the ground with a 7 m long, 30 cm high fly wire mesh running over the centre of the bucket. There were a total of 450 traps across the 18 sites.

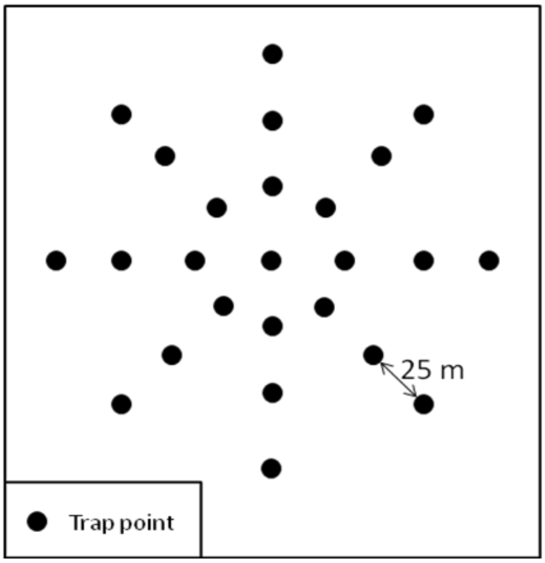

Figure 3: Web arrangement of the 25 trap points located at each of the 18 sites. Each trap point consisted of one 25 L pit trap and 7 m long drift fence erected over the centre of each pit. Trap points were spaced 25 m apart along each arm of the web.

A total of six trapping sessions were conducted in September and November 2011 and January/February, March, May and July of 2012. Each session consisted of four consecutive nights trapping. Traps were checked in both the morning and afternoon. When an animal was captured, it was weighed, marked (excluding frog species), sexed (if possible) and skeletal measurements were taken. Mammals were marked using individual ear notch numbers and reptiles were marked on the underside of the body with a non toxic permanent marker. The non toxic permanent marker allowed the identification of recaptures during that trapping session but would not last between sessions. Frogs were not marked.

Statistical Analyses

Small vertebrate assemblage

The number of captures at each site during each of the six sessions was combined to form a species by sample matrix. Each sample referred to a particular site and session, for example the animals captured at site PS1 in the September session were treated as one sample. Data were standardised according to trap effort. The September session had half the trap effort of the following five sessions as it was treated as a trial session. During each session there was slight variation in effort as some traps had to be closed due to water logging or ant activity.

Data were fourth root transformed prior to analysis to ensure that sites with a particularly large number of captures during a session did not dominate the data (Clarke and Warwick, 2001). Pair-wise similarities between samples were estimated by the Bray-Curtis similarity coefficient. The data were then analysed using a permutational repeated measures analysis of variance (PERMANOVA; (Anderson, 2001). All data analysis was conducted using the PRIMER-E software package and the PERMANOVA+ add-on (Anderson *et al.*, 2008).

Habitat type, month (session) of trapping and whether the site was inside or outside the sanctuary were all treated as fixed factors (Table 1). As there was only one sanctuary, this factor was not able to be properly replicated. The 'Sanctuary' factor was included as a blocking factor in order to help account for any variation between the two areas. This paper will only report differences found between inside and outside the sanctuary. This research does not intend to conclude on any impact of the sanctuary itself as this survey is intended as a baseline dataset to which future surveys can be compared. The individual site was treated as a random factor nested within habitat and sanctuary. Average maximum temperature and average rainfall during each session were included as covariates in the PERMANOVA analysis. Average rainfall was taken from the Bureau of Meteorology records from Deeside rain station approximately 25 km from the study site (BOM, 2012). Maximum temperature records were taken from Manjimup weather station (BOM, 2012). Significant relationships were tested using 9999 randomisations. Terms in the analysis were pooled when $p > 0.25$ (Anderson *et al.*, 2008). Where factors were found to be significant, principal coordinates analysis (PCO) was conducted to visualise the separation of samples.

SIMPER analysis within the PRIMER-E package was used to identify which species accounted for the dissimilarity between and similarities within habitats. BEST analysis within the PRIMER-E package was used to identify which subset of species best explained the patterns occurring in the 24 species captured.

Table 1: PERMANOVA design table of factors included in the small vertebrate assemblage and species level analyses.

Factor	Nested within	Fixed or Random
Sanctuary	-	Fixed
Month	-	Fixed
Habitat	-	Fixed
Site	Habitat and Sanctuary	Random

Species richness

Species richness was an estimate of the number of species that would be captured when 25 individuals had been collected. This was calculated using rarefaction in the PRIMER-E package. These data were analysed as a univariate PERMANOVA. The factors and covariates used in the species richness analyses were as described above (Table 1).

Species level responses

Individual species analyses were conducted on the eight most abundant species. These were *Sminthopsis griseoventer* (Class Marsupialia), *Lerista distinguenda*, *Hemiergis peronii*, *Morethia obscura* (Class Sauropsida), *Heleioporus eyrei*, *Limnodynastes dorsalis*, *Crinia sp.* complex and *Pseudophryne guentheri* (Class Amphibia). Comparisons of the size of individuals of the same species were made between the different habitat types. Both body mass and a skeletal measurement were used to compare the size of animals between sites. Head length was used as the skeletal measurement for the mammal species and snout-vent length (SVL) was used for reptile and frog species. For each species, a PERMANOVA analysis with mass and the skeletal measurement as variables was conducted to determine if there was a difference in the size of individuals based on the habitat in which they were found. The structure of the design and factors included are described in Table 1. No covariates were included in this analysis.

Results

A total of 751 individuals from 25 species were captured over 9625 trap nights (Table 2). Three mammal, 13 reptile and nine frog species were identified. Due to the similarity in morphology of *C. subinsignifera* and *C. pseudinsignifera* (Roberts, 2010), these species were difficult to identify in the field. As both species are also very similar in their ecology, captures of these species were combined and will be referred to as *Crinia sp*. complex. Captures of the introduced mouse (*Mus musculus*) were not included in any data analysis. Creek sites had the greatest number of captures over the six sessions at 0.12 (SE = 0.028) captures per trap, with slope and ridge sites both at 0.05 (SE = 0.009 and 0.016 respectively). The total number of captures for reptiles was greatest in the ridges and slopes at 0.03 captures per trap (SE = 0.016 and 0.012 respectively; Figure 4). The total number of captures for frogs was greatest at creek sites at 0.10 (SE = 0.053) captures per trap. Mammal captures per pit were greatest on the slopes at 0.01 (SE = 0.002).

The average number of captures per trap for all taxa across all sessions was 0.07 (SE = 0.11). The greatest number of captures was in January (0.23, SE = 0.043) and fewest in July (0.007, SE = 0.0016; Figure 5). All three major taxa had their highest capture rates in January (0.15 (SE = 0.047), 0.01 (SE = 0.002) and 0.06 (SE = 0.011) for frogs, mammals and reptiles respectively). The lowest capture rates were in September for frogs (0.001, SE = 0.0011) and July for mammals (0.003, SE = 0.0010) and reptiles (0).

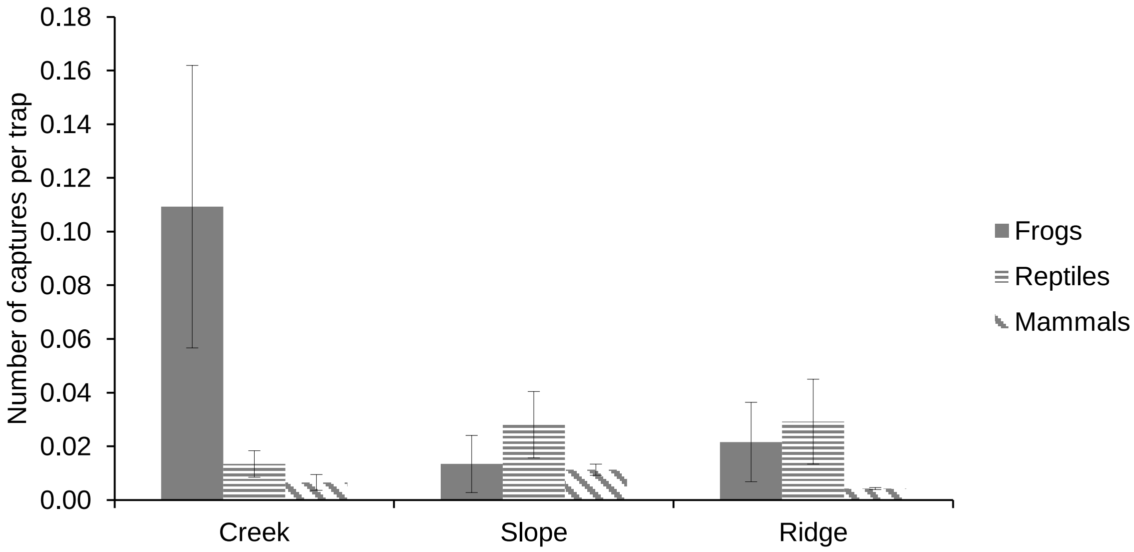

Figure 4: Total number of captures per trap during the six survey sessions according to habitat type. Bars indicate standard errors.

Table 2: Total number of amphibian, mammal and reptile captures and number of species trapped during the six survey sessions according to habitat. Number in brackets indicates the total number of species.

			Habitat	
Taxon	**Species**	**Creek**	**Slope**	**Ridge**
Amphibians	*Crinia georgiana*	6	7	-
	Crinia glauerti	18	3	-
	Crinia sp complex	79	10	4
	Heleioporus eyrei	75	15	5
	Heleioporus inornatus	-	-	1
	Heleioporus psammophilus	5	1	4
	Limnodynastes dorsalis	31	5	57
	Neobatrachus pelobatoides	2	-	-
	Pseudophryne guentheri	95	3	-
		311 (8)	**44 (7)**	**71 (5)**
Mammals	*Sminthopsis griseoventer*	16	37	13
	Mus musculus	17	3	1
	Cercartetus concinnus	4	1	1
		37 (3)	**41 (3)**	**15 (3)**
Reptiles	*Acritoscincus trilineatum*	3	1	2
	Christinus marmoratus	-	2	-
	Ctenotus catenifer	-	1	-
	Ctenotus labillardieri	-	1	3
	Egernia napoleonis	1	1	1
	Hemiergis peronii	9	19	6
	Lerista distinguenda	13	43	71
	Lerista microtis	-	1	-
	Menetia greyii	2	2	2
	Morethia lineoocellata	-	1	3
	Morethia obscura	11	23	6
	Parasuta gouldii	1	-	-
	Ramphotyphlops australis	1	-	2
		41 (8)	**95 (11)**	**96 (9)**

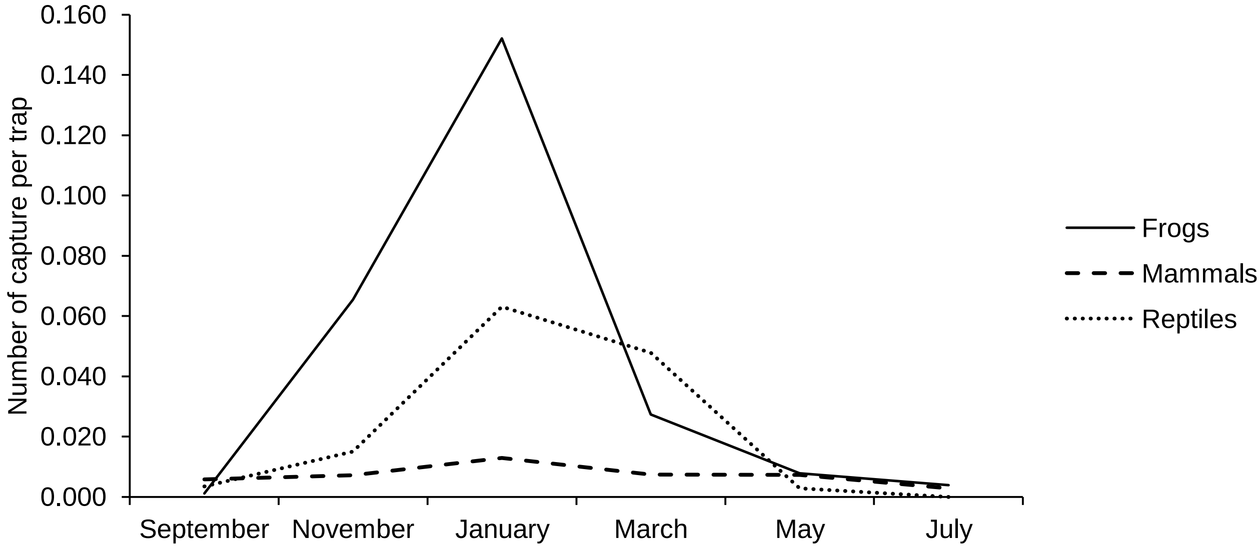

Figure 5: Total number of captures per trap of each taxonomic group according to month of survey. Bars indicate standard errors.

Small vertebrate assemblage responses

There was a significant difference in the composition of the small vertebrate assemblage between the three habitats (PERMANOVA, p=0.0001; Table 3). There was also a significant difference in the assemblage based on the month of sampling (p=0.0001), average rainfall (p=0.0001) and maximum temperature (p=0.0178). There was no evidence that there was a difference in the small vertebrate assemblage inside the sanctuary compared to outside (p=0.1027). There was a significant interaction between habitat and month (p = 0.0061). Habitat type accounted for 10.6% of the variation in the assemblage whereas month sampled, average rainfall and maximum temperature accounted for 28.8, 7.9 and 0.8 % of the variation respectively. The Habitat x Month interaction accounted for 5.4% of the variation in the assemblage. The terms Sanctuary x Habitat, Sanctuary x Month and Sanctuary x Habitat x Month were pooled for this analysis.

Table 3: PERMANOVA results of the effect of sanctuary, habitat, month and site on the structure of the small vertebrate assemblage at Perup. Maximum temperature and average rainfall were also included in the analysis as covariates. Stars indicate significant factors.

Source	**df**	**Mean Square**	**Pseudo-F**	**P value**	**# Permutations**
Maximum temperature	1	2885.2	2.9791	0.0178*	9959
Average rainfall	1	18925	19.539	0.0001*	9958
Sanctuary	1	2374.7	1.8441	0.1027	9950
Habitat	2	9478.9	6.6323	0.0001*	9941
Month	4	9534.5	9.7866	0.0001*	9908
HabitatxMonth	10	1666.8	1.721	0.0061*	9865
Pooled (Site(HabitatxSanctuary) + SanctuaryxHabitat +SanctuaryxMonth)	19	1080.5	1.1157	0.2504	9833
Pooled (Residual + SanctuaryxHabitatxMonth)	69	968.37			
Total	107				

Due to the significant Habitat x Month interaction, pair-wise comparisons were performed on the factor habitat within each month (Table 4a) and on the factor month within each habitat (Table 4b). There was a significant difference in the assemblage in the creek compared to the ridge and the creek compared to the slope in November, January and March (Table 4a). There was also a difference in the assemblage between the creek and the slope in May. There was no difference in between the slope and the ridge during any month.

Within creeks, the months of November, January and March had a significantly different assemblage to all other sampled months but not to each other (Table 4b). Within slopes, January had a significantly different assemblage to all sampled months except March. March had a significantly different assemblage to all months except January and November. Within ridges, January and March had a significantly different assemblage to all other sampled months but not to each other.

Table 4: Pair-wise tests comparing (a) habitat within month of sampling and (b) month within habitat. 'Yes' indicates a significant difference in the assemblage and 'No' indicates no evidence of a difference (PERMANOVA, a = 0.05).

(a)	**September**		**November**		**January**		**March**		**May**		**July**	
	Creek	**Slope**	**Creek**	**Slope**	**Creek**	**Slope**	**Creek**	**Slope**	**Creek**	**Slope**	**Creek**	**Slope**
Slope	No	-	Yes	-	Yes	-	Yes	-	Yes	-	No	-
Ridge	No	No	Yes	No	Yes	No	Yes	No	No	No	No	No

(b)	**Creek**					**Slope**					**Ridge**				
	Sept	**Nov**	**Jan**	**March**	**May**	**Sept**	**Nov**	**Jan**	**March**	**May**	**Sept**	**Nov**	**Jan**	**March**	**May**
Nov	Yes	-	-	-	-	No	-	-	-	-	No	-	-	-	-
Jan	Yes	No	-	-	-	Yes	Yes	-	-	-	Yes	Yes	-	-	-
March	Yes	No	No	-	-	Yes	No	No	-	-	Yes	Yes	No	-	-
May	No	Yes	Yes	Yes	-	No	Yes	Yes	Yes	-	No	No	Yes	Yes	-
July	No	Yes	Yes	Yes	No	No	Yes	Yes	Yes	No	No	No	Yes	Yes	No

CAP analysis based on the factor habitat (Figure 6a) correctly classified 65% of the samples into their correct habitat based on five principal coordinate axes. This was significant with a trace statistic of 0.51468 (p = 0.0001). The abundance of the *Crinia sp*. complex, *H. eyrei* and *P. guentheri* was positively correlated with the CAP1 axis but the abundance of *S. griseoventer* was negatively correlated with this axis. The CAP 2 axis was positively correlated with *S. griseoventer* and *M. obscura* abundance.

CAP analysis based on the factor Month (Figure 6b) correctly classified 55% of the samples into their month of sampling. This was significant, with a trace statistic of 1.30527 (p = 0.0001). The abundance of *H. eyrei*, *L. distinguenda* and *M. obscura* was positively correlated with the CAP 1 axis. The CAP 2 axis was negatively correlated with the abundance of *H. peronii*, *H. eyrei* and *M. obscura*.

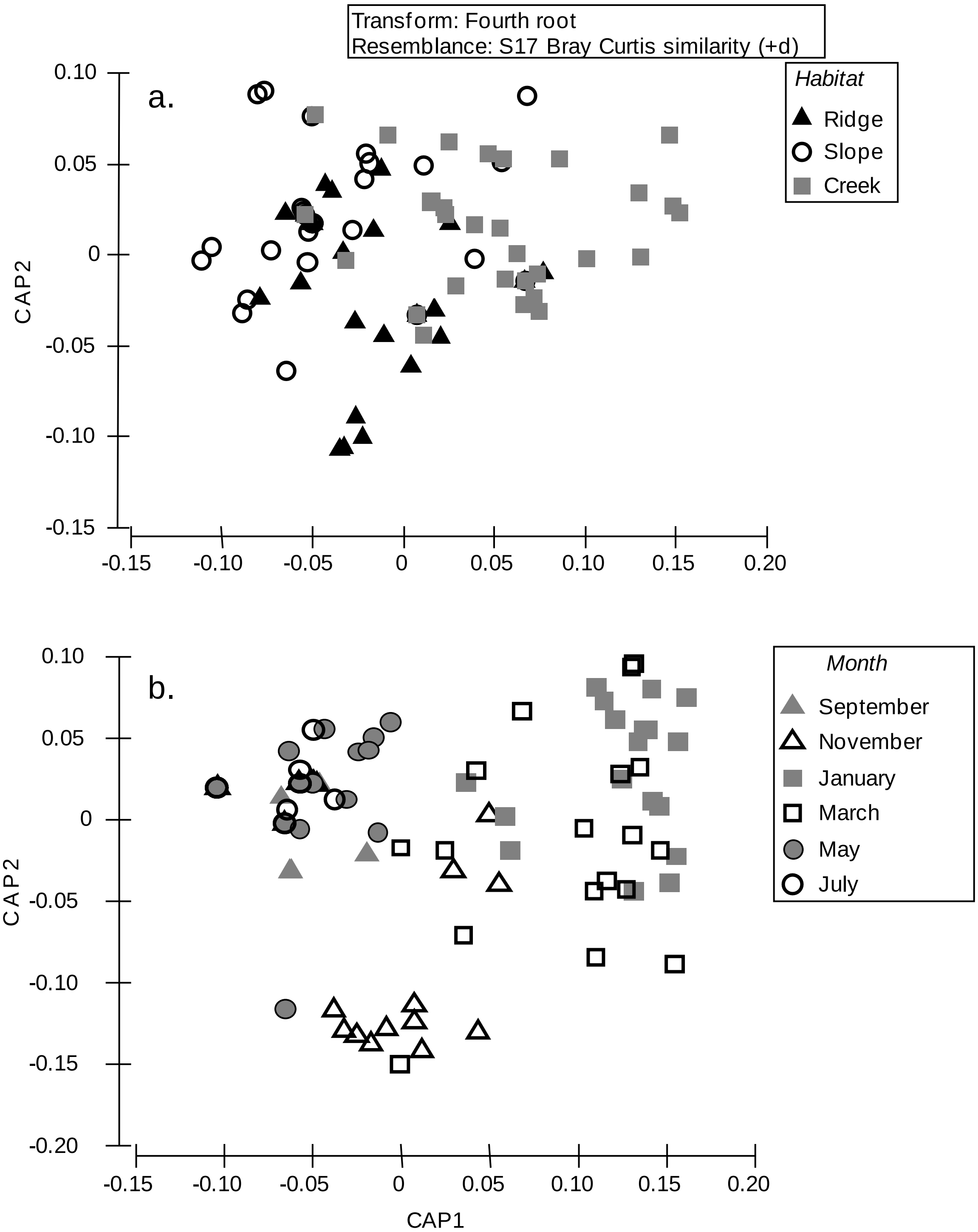

Figure 6: CAP analysis of the small vertebrate assemblage from 108 samples collected during six surveys based on (a) habitat type and (b) month of sampling. Data were fourth root transformed prior to analysis.

The distinctions between habitats in the ordination can be accounted for by differences in the relative abundances of individual species. SIMPER analysis reported an average dissimilarity of 83.8% between ridge and slope habitats, 91.5% between ridge and creek habitats and 87.6% between slope and creek habitats (Table 5). The dissimilarity between ridge and slope sites appeared to be the result of a higher average abundance of *S. griseoventer* at slope sites which contributed 28.8% to the dissimilarity between these habitat types. There was also a higher average abundance of *L. distinguenda* at ridge sites contributing to 15.0% difference, and greater abundance of *H. peronii*, *M. obscura* and *H. eyrei* at slope compared to ridge sites.

Creek sites had a greater average abundance of *H. eyrei* and *Crinia sp*. complex than ridge sites which contributed 12.3 and 11.0% to the dissimilarity between sites. Creek sites had a lower average abundance of *L. distinguenda* but higher average abundances for *P. guentheri* and *Crinia glauerti* when compared to ridge sites. Creek and ridge sites had similar average abundances for *L. dorsalis*.

Slope sites had a greater average abundance of *S. griseoventer*, *M. obscura*, *H. peronii* and *L. distinguenda* when compared to creek sites which contributed 22.7, 10.2, 8.8 and 7.8% respectively to the dissimilarity between these habitats. Creek sites had a greater average abundance of *H. eyrei*, *Crinia sp*. complex and *P. guentheri* contributing to 11.0, 9.3 and 9.2% of the dissimilarity between sites.

Within each habitat type, sites were not very similar. Ridge sites had an average similarity of 10.6%, slopes 30.5% and creeks 14.5%. The species contributing to the similarity of ridge sites were *L. distinguenda* and *S. griseoventer* at 36.0 and 35.8% contribution respectively. The species contributing to the similarity of slopes sites were *S. griseoventer*, *M. obscura* and *H. peronii* (77.1, 7.6 and 7.1% contribution respectively). The species contributing to the similarities between creek sites were *P. guentheri*, *H. eyrei* and *Crinia sp*. complex (24.3, 22.2 and 21.5% contribution respectively).

BEST analyses starting with a set of ten random species identified a subset of nine species sampled that best explained the patterns occurring in the overall small vertebrate assemblage (Spearman correlation coefficient = 0.95, $P<0.001$). These species were the *Crinia sp*. complex, *S. griseoventer*, *H. eyrei*, *H. psammophilus*, *H. peronii*, *L. distinguenda*, *L. dorsalis*, *M. obscura* and *P. guentheri*.

Table 5: Summary of the average similarity within and dissimilarity between habitats based on the small vertebrate assemblage. The shaded boxes contain the average similarity between sites of the same habitat type and the species which are common within that habitat. The unshaded boxes contain the average dissimilarity between sites of different habitat types and the species which contribute to those differences. Which habitat has the greater average abundance of the species is also described.

Habitat	**Ridge**	**Slope**	**Creek**
Ridge	**10.55** *Lerista distinguenda, Sminthopsis griseoventer*		
Slope	**83.79** *Sminthopsis griseoventer* Slope > Ridge *Lerista distinguenda* Ridge > Slope *Hemiergis peronii* Slope > Ridge *Morethia obscura* Slope > Ridge *Heleioporus eyrei* Slope > Ridge	**30.54** *Sminthopsis griseoventer, Morethia obscura, Hemiergis peronii*	
Creek	**91.51** *Heleioporus eyrei* Creek > Ridge *Crinia sp*. complex Creek > Ridge *Pseudophryne guentheri* Creek > Ridge *Lerista distinguenda* Ridge > Creek *Limnodynastes dorsalis* Creek ≈ Ridge	**87.59** *Sminthopsis griseoventer* Slope > Creek *Morethia obscura* Slope > Creek *Hemiergis peronii* Slope > Creek *Lerista distinguenda* Slope > Creek *Heleioporus eyrei* Creek > Slope *Crinia sp.* complex Creek > Slope *Pseudophryne guentheri* Creek > Slope	**14.45** *Pseudophryne guentheri, Heleioporus eyrei, Crinia sp*. complex

Species richness

PERMANOVA analysis showed a significant effect for habitat (p = 0.0066), month of sampling (p = 0.0001), maximum temperature (p = 0.0003) and average rainfall (p = 0.0001; Table 6). The terms Sanctuary, Sanctuary x Month and Habitat x Sanctuary x Month were pooled for this analysis. Pair-wise tests demonstrated a significant difference in species richness between ridges and slopes (p = 0.04) and ridges and creeks (p = 0.02). There was no evidence to suggest a difference in species richness between slopes and creeks (p = 0.11). Species richness was on average greatest at creek sites and lowest at ridge sites (Figure 8). Species richness was greatest in January and lowest in July (after accounting for the reduced trapping effort in September; Figure 8).

Table 6: PERMANOVA results of the effect of habitat, month and site on species richness at Perup. Maximum temperature and average rainfall were also included in the analysis as covariates. Stars indicate significant factors.

Source	df	MS	Pseudo-F	P(perm)	# Permutations
Maximum temperature	1	3317.2	11.015	0.0003*	9943
Average rainfall	1	17736	58.911	0.0001*	9955
Month	4	7434.6	25.105	0.0001*	9953
Habitat	2	1081.3	3.9848	0.0066*	9944
HabitatxSanctuary	2	205.27	No test		
HabitatxMonth	10	407.97	1.355	0.1656	9920
Pooled (1) Site(HabitatxSanctuary)+Sanctuary+SanctuaryxMonth	18	203.69	No test		
Pooled (2) Res + HabitatxSanctuaryxMonth	69	301.23			
Total	107				

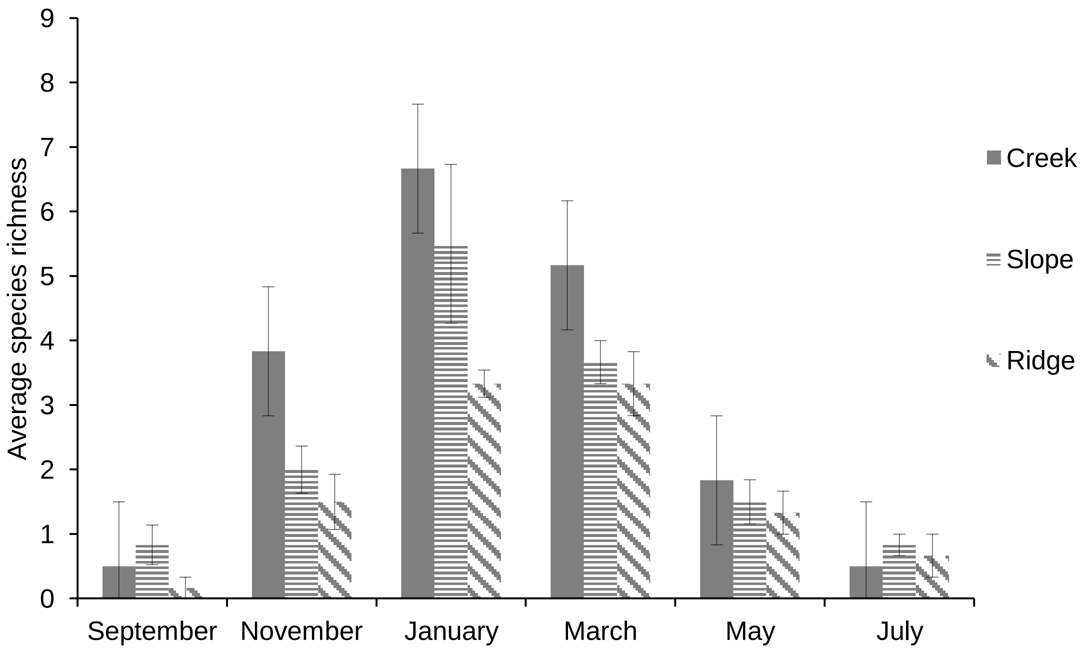

Figure 8: Average species richness of each habitat type according to month of sampling. Bars indicate standard errors.

Species level responses

Sminthopsis griseoventer

There was a significant difference in the size of *S. griseoventer* based on month of capture (PERMANOVA, $p = 0.0002$). The heaviest dunnarts were caught during July and September and the lightest during November. Head length was greatest in May and July and the smallest in November. On average, females were heavier than males and had shorter head lengths. All other terms were pooled in the analysis and there was no evidence to support a difference in size based on sanctuary, habitat or site.

Morethia obscura

There was a significant difference in the size of *M. obscura* based on month of capture (PERMANOVA, $p = 0.0163$). This species was captured during September, November, January and March. On average, the heaviest individuals were captured during November and the lightest in September. The greatest SVL on average was in November and January and smallest during March. All other terms were pooled during the analysis and there was no evidence to support a size difference based on habitat, sanctuary or site.

Hemiergis peronii

There was no evidence to support a difference in the size of *H. peronii* based on month, sanctuary, habitat or site. All terms except sanctuary were pooled during the analysis.

Lerista distinguenda

There was a significant difference in the size of *L. distinguenda* based on month (PERMANOVA, $p = 0.0001$). This species were captured during January and March. On average, individuals had a greater mass and SVL during January compared to March. There was no evidence to suggest a difference in the size of individual inside compared to outside the sanctuary. All other terms were pooled in the analysis and there was no evidence to support a difference in the size of individuals based on site or habitat.

Limnodynastes dorsalis

There was a significant difference in the size of *L. dorsalis* based on month (PERMANOVA, $p = 0.0001$) and sanctuary ($p = 0.0197$). On average the smallest individuals were captured during January. The largest individual was captured in July but there was only one capture during this month. Individuals were on average smaller outside the sanctuary compared to inside however there were only five captures inside the sanctuary compared to 88 captures outside the sanctuary. The data is heavily skewed because of one trapping night during January where 52 sub adult individuals were captured at a ridge site and another 27 at a creek site adjacent to the ridge site. There was a large movement of sub adults from the creek upwards towards the ridge site and this large activity event resulted in the highest capture rate outside the sanctuary and during the January session. There was also a significant interaction effect between month and sanctuary ($p = 0.0158$). There was a significant difference in the size of individuals between individual sites ($p = 0.0027$). All other terms were pooled in the analysis and there was no evidence to support a size difference based on habitat.

Heleioporus eyrei

There was a significant difference in the size of individuals based on month of capture (PERMANOVA, $p = 0.0159$). The smallest individuals on average were captured during November and the largest were captured during May. All other terms were pooled in the analysis and there was no evidence to support a size difference based on habitat, sanctuary or site.

Crinia sp. complex
There was a significant difference in the size of *Crinia sp*. complex based on month of sampling (PERMANOVA, $p = 0.0259$). The smallest individuals on average were captured during November and the largest in July. There was also a significant difference in the size of individuals based on site ($p = 0.004$). All other terms were pooled during analysis and there was no evidence to support a difference based on habitat or sanctuary.

Pseudophryne guentheri
There was a significant difference in the size of *P. guentheri* based on month of capture (PERMANOVA, $p = 0.0079$) and site ($p = 0.0256$). *P. guentheri* were captured in November, January, March and July (although there was only a single capture during July). There was an increase in the average body mass and SVL of individuals from November to March. All other terms were pooled during the analysis and there was no evidence to support a size difference based on sanctuary or habitat.

Discussion

There was a significant difference in the small vertebrate assemblage found in each of the three main habitat types. Creek habitats had the greatest number of captures and species richness overall but this was heavily influenced by the large seasonal influx of frogs after rainfall events in November, January and March. The creek habitat is unsurprisingly the most important site for the frog species in the area as it provides appropriate refuge and a breeding location for all frog species captured (Lee, 1967, Main, 1968, Silla, 2010). These sites were dominated by *P. guentheri*, *H. eyrei* and the *Crinia sp*. complex. The least mobile and the smaller of the frog species were found in the creek habitats only, with only a few records of those species higher up in the landscape. The more mobile and larger species such as *H. eyrei* and *L. dorsalis* were spread more evenly through the landscape. It was not uncommon to find *H. eyrei* in the sandy slope habitat as this is a suitable refuge location used after the breeding season (Lee, 1967, Main, 1968, Berry, 2001). *L. dorsalis* was as likely to be found in the ridge sites as the creek sites and did not show the apparent preference for areas lower in the landscape as with the other frog species. This species is the largest of the frog species captured and has a relatively smaller body surface from which evaporation takes place compared to smaller frogs (Bellis, 1962). This suggests that they may dehydrate at a slower rate and so are able to move away from the moister areas in the lower parts of the landscape (Bellis, 1962).

Slope sites were characterised by a greater presence of *S. griseoventer*, *H. peronii* and *M. obscura*. Slope locations appeared to be particularly important for *S. griseoventer* with over 56% of their captures in this habitat. *S. griseoventer* are opportunistic predators which feed on small arthropods which dwell in the leaf litter and coarse woody debris layer (Fisher and Dickman, 1993, McCaw, 2011). The slopes have greater leaf litter and debris cover as this habitat has the greatest canopy cover and lower storey vegetation density (Havel and Mattiske, 2000). This habitat may also provide better thermal insulation when *S. griseoventer* is resting and better cover from avian predators while foraging at night. The slope habitat also possesses characteristics suitable for *M. obscura* and *H. peronii* which are both terrestrial skinks which forage in dense leaf litter under low, dense shrubs (Smyth, 1974, Nichols and Bamford, 1985). Slope sites will also have patches of sunny and shady positions which reptiles can utilise for temperature regulation while maintaining cover from predators. The slopes sites may provide a more complex vegetation structure and therefore a greater number of microhabitats for these species to utilise.

Ridge sites were characterised by a greater relative abundance of *L. distinguenda* with over 55% of the captures of this species in this habitat. The association of this species with the ridge habitat supports the findings of other studies which show an association between *L. distinguenda* and areas with high leaf litter cover but a lower understorey density (Nichols and Bamford, 1985).

The patterns in seasonal activity and the significant effect of temperature and rainfall reflect the physiological and breeding characteristics of the species captured (Bellis, 1962, Main, 1968, Smyth, 1974, Friend, 1993). Frog species demonstrated the strongest seasonality with greatest relative abundance during January and March where it is likely that large rainfall events after extended periods of warm and dry conditions initiated mass breeding and/or feeding activities (Main, 1968). Reptiles also demonstrated seasonal patterns of activity with the greatest relative abundance during the warm summer months. Reptiles were less likely to be captured during or immediately after large rain events. Reptiles were not captured during the colder winter months as they are inactive during this time due to the cooler temperatures (Spellerberg, 1972). Female reptiles in temperate climates exhibit highly seasonal patterns in reproduction (Murphy *et al.*, 2006). Eggs are laid in late spring or early summer and offspring hatch or are born in summer (Murphy *et al.*, 2006). Summer is the only time of year where soil temperatures and insolation are high enough to permit rapid embryonic development (Murphy *et al.*, 2006). *S. griseoventer* which represented the majority of the mammal captures did have a slight increase in relative abundance during the warmer months but did not display a strong seasonal pattern of activity as in the frog and reptile species. This species is endothermic and is able to remain active year round (Friend, 1993), which probably explains the lack of a strong seasonal pattern in activity.

The size of the effect of habitat on the small vertebrate assemblage varied according to the month of sampling. This is shown by the significant interaction between month and habitat based on the relative abundances of each of the species captured. During months where fewer species were active, it was more difficult to distinguish the assemblages between each habitat. The seasonal change in the size of the effect of habitat highlights the importance of repeated sampling over many months (Mac Nally, 1997). Ideally this survey would have re-sampled for a following twelve months to verify the seasonal pattern in activity. The seasonal differences in the assemblage have implications for the monitoring of the area. Monitoring will need to be timed appropriately to ensure that a significant portion of the assemblage is detected. Ideally monitoring would occur every other month however this may be unachievable due to resource limitations. If sampling is restricted to only one part of the year, sampling in warmer conditions after rain will yield the most captures and therefore provide the best opportunity to survey species in the area.

Although species richness was different between habitats, this was not also reflected in species diversity. Creek sites had the greatest species richness but no greater diversity of species compared to the slope and ridge sites. This is because there were often one or two frog species which had very large relative abundances in the creek sites and so these species dominated the assemblage. Species diversity was however influenced by the month of sampling and this mirrored the pattern seen in species richness with the greatest diversity in the warm summer months and the least in the cooler winter months. This demonstrates again the seasonal behaviour of the species within the assemblage where many species become active and are breeding during the warmer months and after large rain events.

Of the 24 native species captured, a subset of nine species described over 95% of the variation in the complete assemblage (*Crinia sp*. complex, *S. griseoventer, H. eyrei, H. psammophilus, H. peronii, L. distinguenda*, *L. dorsalis*, *M. obscura* and *P. guentheri*). Most of these species are those which were shown to characterise each of the habitat types and this again demonstrates the distinction in the assemblage across the landscape. Each of these species (excluding *H. psammophilus*) are relatively abundant within the survey area and require less intensive trapping to be detected than rare or cryptic species. This survey used a relatively large number of pit traps (450) which required a significant effort in terms of labour by staff and volunteers as well as vehicle costs. If in the future only a portion of these traps are able to be monitored because of resource limitations, the smaller trap effort may not detect all 24 species. In this case, it is useful to know that data on just these more abundant and easily detectable species may aid in describing the patterns occurring in the wider assemblage.

There was no evidence to support a size difference of species between habitats. Of the eight most abundant species analysed, none of the species showed a significant difference in their body mass or skeletal length based on the habitat in which they were found. All of the species except *H. peronii* demonstrated an average size difference of individuals based on month of capture. This pattern is indicative of the life history and breeding biology of these species. With *S. griseoventer*, the largest individuals were captured in the winter months and the smallest in November. This reflects the winter breeding where the population is solely larger adults, followed by the emergence of juveniles from their natal nests in late October (Van Dyck and Strahan, 2008). *M. obscura* was largest in November as it moves into its breeding season and females are gravid and smallest in September when it becomes first active after the extended period of inactivity over the winter months (Chapman and Dell, 1985). The frog species *L. dorsalis*, *H. eyrei*, *C. pseudinsignifera* and *P. guentheri* are all winter breeders and the smallest individuals were captured during the spring and summer months. This is a result of juveniles and sub adult frogs entering the population following metamorphosis (Main, 1968).

The assemblage found inside the sanctuary was not significantly different to outside the sanctuary, however the p-value was borderline ($p=0.059$). When selecting survey sites we attempted to select similar sites outside the sanctuary as inside, but it became clear during trapping and as the season changed that one of the external creek sites (S9) was quite different to the other creek sites. This site was in a much broader valley than the other creek sites and held a greater amount of water. It also drained into a larger swamp area located on private land to the west of the survey area. This site had the greatest relative abundance of frog species and particularly the smaller *Crinia* sp. The characteristics of this site make it a particularly valuable breeding location for most frog species and also useful for the smaller species which require a moist environment year round. Smaller sub adult *L. dorsalis* captured at this site caused the significant size difference between inside and outside the sanctuary for this species. This was a result of one event where a large number of sub adult *L. dorsalis* (>70) moved out of the creek at site S9 and upwards toward the ridge line at site S8. As this site appears to be particularly valuable to many of the frog species in the area, it will be an important site to monitor and conserve in the future.

The association between species and habitat has important implications for how the sanctuary and surrounding area is managed. Sampling in only one habitat type or sampling without regard for the distinction between habitats is will compromise the monitoring of the species in the area. Future surveys should continue monitoring each of the habitats in order to be as sensitive as possible to changes in the assemblage. This approach is also relevant to the fire management of the sanctuary. As the movement of species across the landscape is potentially restricted because of the sanctuary fence, the use of prescribed burns has the potential to negatively impact populations. If a burn is conducted that impacts all of one particular habitat within the sanctuary, this could hinder the recovery of a fire sensitive species after the fire event if that habitat is of greater value than other areas in the sanctuary. One recommendation based on the results of this survey is the division of the sanctuary into smaller units based on habitat type which can then be monitored and burned at that scale. Any prescribed burn plan should ensure that if any one unit is burned, there are other units from the same habitat type which remain unburnt. This will assist in the recovery after fire of species which utilise that particular habitat.

Conclusion

The construction of fences can be viewed as a radical form of environmental manipulation particularly when the composition of species within the fenced area is also artificially modified. This strategy for the conservation of wildlife is becoming increasingly common in Australia and New Zealand as managers attempt to reduce the impact of introduced and invasive species (Burns *et al.*, 2012, de Tores and Marlow, 2012, Dickman, 2012). As the protection of vulnerable native species is time sensitive, few studies have been undertaken which deal with the potential unintended effects of fencing wildlife. The current study has established a baseline data set which can be used to monitor the changes in the extant small vertebrate population in Perup Nature Reserve in order to improve our knowledge of the processes occurring in a highly modified environment. In this situation, different habitats appear to hold different values for particular species and possess characteristics necessary for the completion of life history traits. The modification of the environment in the form of the construction of the fence, increasing medium sized mammal density and prescribed burning mean that close monitoring and consideration of the documented ecology of the small vertebrate species will be a necessary part of the management of the area if negative impacts are to be minimised. This information will be of interest to all managers of fenced exclosures as the success of management outcomes of target species within fenced areas may be linked to the health and viability of the wider assemblage.

References

ANDERSON, M. J. 2001. A new method for non-parametric multivariate analysis of variance. *Austral Ecology,* 26**,** 32-46.

ANDERSON, M. J., GORLEY, R. N. & CLARKE, K. R. 2008. *PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods,* PRIMER-E: Plymouth, UK.

BELLIS, E. D. 1962. The Influence of Humidity on Wood Frog Activity. *American Midland Naturalist,* 68**,** 139-148.

BERRY, O. 2001. Genetic evidence for wide dispersal by the sand frog *Heleioporus psammophilus* (Anura: Myobatrachidae), in Western Australia. *Journal of Herpetology,* 35**,** 136-141.

BURNS, B., INNES, J. & DAY, T. 2012. The Use and Potential of Pest-Proof Fencing for Ecosystem restoration and Fauna Conservation in New Zealand. *In:* SOMERS, M. J.

& HAYWARD, M. W. (eds.) *Fencing for Conservation: Restriction of Evolutionary Potential or a Riposte to Threatening Processes?* : Springer, New York.

CHAPMAN, A. & DELL, J. 1985. Biology and Zoogeography of the Amphibians and Reptiles ofthe Western Australian Wheatbelt. *Records of the Western Australian Museum,* 12 1-46.

CLARKE, K. R. & WARWICK, R. M. 2001. *Change in marine communities: an approach to statistical analysis and interpretation, 2nd edition*, PRIMER-E: Plymouth, UK.

DE TORES, P. J. & MARLOW, N. 2012. The Relative Merits of Predator-Exclusion Fencing and Repeated Fox Baiting for Protection of Native Fauna: Five Case Studies from Western Australia. *In:* SOMERS, M. J. & HAYWARD, M. W. (eds.) *Fencing for Conservation: Restriction of Evolutionary Potential or a Riposte to Threatening Processes?* : Springer New York.

DEXTER, N., RAMSEY, D. S. L., MACGREGOR, C. & LINDENMAYER, D. 2012. Predicting Ecosystem Wide Impacts of Wallaby Management Using a Fuzzy Cognitive Map. *Ecosystems,* 15**,** 1363-1379.

DICKMAN, C. R. 2012. Fences or Ferals? Benefits and Costs of Conservation Fencing in Australia. *In:* SOMERS, M. J. & HAYWARD, M. W. (eds.) *Fencing for Conservation: Restriction of Evolutionary Potential or a Riposte to Threatening Processes?* : Springer New York.

FISHER, D. O. & DICKMAN, C. R. 1993. The body size - prey size relationship in dasyurid marsupials: tests of three hypotheses. *Ecology,* 74**,** 1871-1883.

FRIEND, G. R. 1993. Impact of fire on small vertebrates in mallee woodlands and heathlands of temperate Australia: A review *Biological Conservation,* 65**,** 99-114.

GROSS, M. 2009. Fence protection progress. *Current Biology,* 19**,** R465-R465.

HAMEED, M., NAZ, N., ASHRAF, M., AHMAD, M. S. A., NAWAZ, T. & CHAUDHRY, A. A. 2012. Impact of fencing on the conservation of wildlife habitat in a sub-mountainous open scrub forest. *Acta Oecologica-International Journal of Ecology,* 45**,** 16-24.

HAVEL, J. J. & MATTISKE, E. M. 2000. Vegetation mapping of south west forest region of Western Australia. Department of Conservation and Land Management & Environment Australia.

HAYWARD, M. W. 2012. Perspectives on Fencing for Conservation Based on Four Case Studies: Marsupial Conservation in Australian Forests; Bushmeat Hunting in South Africa; Large Predator Reintoduction in South Africa; and Large Mammal Conservation in Poland. *In:* SOMERS, M. J. & HAYWARD, M. W. (eds.) *Fencing for Conservation: Restriction of Evolutionary Potential or a Riposte to Threatening Processes?* : Springer New York.

HAYWARD, M. W. & KERLEY, G. I. H. 2009. Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes? *Biological Conservation,* 142**,** 1-13.

HAYWARD, M. W. & SOMERS, M. J. 2012. An Introduction to Fencing for Conservation. *In:* SOMERS, M. J. & HAYWARD, M. W. (eds.) *Fencing for Conservation: Restriction of Evolutionary Potential or a Riposte to Treatening Processes?* : Springer New York.

IKUTA, L. A. & BLUMSTEIN, D. T. 2003. Do fences protect birds from human disturbance? *Biological Conservation,* 112**,** 447-452.

KINNEAR, J. E., SUMNER, N. R. & ONUS, M. L. 2002. The red fox in Australia - an exotic predator turned biocontrol agent. *Biological Conservation,* 108**,** 335-359.

KREBS, C. J. 1985. *Ecology. The Experimental Analysis of Distribution and Abundance,* Harper & Row, New York.

LEE, A. K. 1967. Studies in Australian Amphibia. II, taxonomy, ecology, and evolution of the genus Heleioporus Gray (Anura: Leptodactylidae). *Australian Journal of Zoology,* 15.

LONG, K. & ROBLEY, A. 2004. Cost effective feral animal exclusion fencing for areas of high conservation value in Australia. Melbourne: Arther Rylah Institute for Environmental Research, Department of Sustainability and Environment.

MAC NALLY, R. 1997. Monitoring forest bird communities for impact assessment: The influence of sampling intensity and spatial scale. *Biological Conservation,* 82**,** 355-367.

MAGURRAN, A. E. 1988. *Ecological Diversity and Its Measurement*, Princeton University Press, Princeton, New Jersey.

MAIN, A. R. 1968. Ecology, Systematics and Evolution of Australian Frogs. *In:* CRAGG, J. B. (ed.) *Advances in Ecological Research.* New York: Academic Press.

MCCAW, W. L. 2011. Characteristics of jarrah (Eucalyptus marginata) forest at FORESTCHECK monitoring sites in south-west Western Australia: stand structure, litter, woody debris, soil and foliar nutrients. *Australian Forestry,* 74**,** 254-265.

MCGAVIN, S. 2009. Density and pair fidelity in a translocated population of North Isalnd robin (*Petroica longipes*). *Notornis,* 56**,** 206-212.

MOSEBY, K. E., HILL, B. M. & READ, J. L. 2009. Arid Recovery - A comparison of reptile and small mammal populations inside and outside a large rabbit, cat and fox-proof exclosure in arid South Australia. *Austral Ecology,* 34**,** 156-169.

MURPHY, K., HUDSON, S. & SHEA, G. 2006. Reproductive Seasonality of Three Cold-Temperate Viviparous Skinks from Southeastern Australia *Journal of Herpetology,* 40**,** 454-464.

NICHOLS, O. G. & BAMFORD, M. J. 1985. Reptile and Frog Utilisation of Rehabilitated Bauxite Minesites and Dieback-Affected Sites in Western Australia's Jarrah *Eucalyptus marginata* Forest *Biological Conservation* 34**,** 227-249.

ODUM, E. P. 1959. *Fundamentals of Ecology,* W.B Saunders Company, Philadelphia.

RICHARDS, J. D. & SHORT, J. 2003. Reintroduction and establishment of the western barred bandicoot Perameles bougainville (Marsupialia : Peramelidae) at Shark Bay, Western Australia. *Biological Conservation,* 109**,** 181-195.

ROBERTS, J. D. 2010. Natural Hybrid between the Frogs *Crinia pseudinsignifera* and *Crinia subinsignifera* (Myobatrachidae) from Southwestern Australia Defined by Allozyme Phenotype and Call. *Journal of Herpetology,* 44**,** 654-657.

SHORT, J. & TURNER, B. 2000. Reintroduction of the burrowing bettong Bettongia lesueur (Marsupialia : Potoroidae) to mainland Australia. *Biological Conservation,* 96**,** 185-196.

SHORT, J., TURNER, B., PARKER, S. & TWISS, J. (eds.) 1994. *Reintroduction of endangered mammals to mainland Shark Bay: a progress report,* Chipping Norton: Surrey Beatty & Sons.

SILLA, A. J. 2010. Effects of luteinizing hormone-releasing hormone and arginine-vasotocin on the sperm-release response of Günther’s Toadlet, *Pseudophryne guentheri*. *Reproductive Biology and Endocrinology,* 8**,** 139-148.

SMYTH, M. 1974. Changes in the Fat Stores of the Skinks *Morethia boulengeri* and *Hemiergis peronii* (Lacertilia) *Australian Journal of Zoology,* 22**,** 135-145.

SPELLERBERG, I. F. 1972. Temperature tolerances of Southeast Australian reptiles examined in relation to reptile thermoregulatory behaviour and distribution. *Oecologia,* 9**,** 23-46.

VAN DYCK, S. & STRAHAN, R. 2008. *The Mammals of Australia,* Sydney, New Holland.

VAN DYK, G. & SLOTOW, R. 2003. The effects of fences and lions on the ecology of African wild dogs reintroduced to Pilanesberg National Park, South Africa. *African Zoology,* 38**,** 79-94.
WHITEHOUSE, A. M. & KERLEY, G. I. H. 2002. Retrospective assessment of long-term conservation management of elephants in Addo Elephant National Park, South Africa. *Oryx,* 36**,** 243-248.
YEATMAN, G. J. & GROOM, C. J. 2012. National Recovery Plan for the woylie *Bettongia penicillata*. *Wildife Management Program No. 51.* Department of Environment and Conservation, Perth.