

The influence of habitat and landscape on small mammals in Estonian coastal wetlands

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Received 5 June 2008, in revised form 22 October 2008

Abstract. We investigated the influence of habitat type and landscape composition on small mammal relative abundance and diversity in coastal wetlands in western Estonia. Seventy live-trap lines in eight representative habitats, across six wetlands revealed seven species. The most diverse habitats were reed bed and scrub woodland, whereas tall grasslands had the highest biomass. Short grass was low in species richness and relative abundance. Small mammal relative abundance, richness, and biomass were positively correlated with tall grass and mosaic habitats and negatively correlated with lower shore habitats and bare ground. Our study indicates potential nature conservation conflicts between small mammals and other biodiversity priorities in wetlands.

Key words: diversity, Estonia, habitat, landscape, small mammals, wet grasslands.

INTRODUCTION

Wetlands are of international conservation importance for the biological diversity they support, including wet grasslands, which were historically managed for low-intensity agricultural production using extensive grazing or hay-cutting (Joyce & Wade, 1998). Flood alleviation, drainage, and agricultural intensification have reduced wetland extent and quality, while abandonment of management threatens wetland biodiversity (Strijker, 2005) through reed (*Phragmites australis*) and scrub encroachment (Leibak & Lutsar, 1996; Joyce & Wade, 1998; Luhamaa et al., 2001; Burnside et al., 2007). Therefore initiatives to reinstate appropriate management to wetlands in Europe and North America have been developed. Wetlands comprise more than 30% of Estonia (Paal, 1998) but coastal wetlands have declined from an estimated 28 750 ha in the 1950s to 8000 ha in 2000 (Luhamaa et al., 2001; Kuresoo & Mägi, 2004), many being abandoned due to political and economic changes (Unwin, 1997).

Wetlands are important habitats for small mammals (Bowland & Perrin, 1993; Krištofik, 2001) but research into factors affecting their distribution and relative abundance within wetland landscapes is relatively scanty. Small mammals play a fundamental role in ecosystem functioning as they constitute the prey base for many predators (Schmidt et al., 2002) and may potentially influence vegetation

composition via selective foraging (Brown & Heske, 1990). Generally, habitats with increased structural heterogeneity positively influence small mammal abundance and richness (Ecke et al., 2002). Mammal diversity tends to be lower in open habitats, where cover providing food and resources (Silva et al., 2005) is reduced, leading to lower fecundity (Grant et al., 1982) as well as increased predation risk (Kotler, 1997; Andreassen & Ims, 1998). Changes in landscape use and management can lead to a reduction in suitable habitats for small mammals (Raoul et al., 2001; Van Apeldoorn et al., 1992), which is compounded by habitat fragmentation causing isolation of populations (Kozakiewicz, 1993; Fitzgibbon, 1997), and consequentially gene flow (Witt & Huntly, 2001). Animal movement and dispersal are affected by vegetation structure, habitat type, landscape composition, and connectivity (Szacki et al., 1993; Fitzgibbon, 1997; Mazerolle & Villard, 1999). Hence landscape mosaic and characteristics, as well as habitat type availability, are potentially influential in determining species presence and persistence (Kupfer et al., 2006).

In this study we investigated the impact of habitat type and landscape composition on small mammal relative abundance and diversity in coastal wetlands in western Estonia. This region has some of the most biologically rich wetlands in Europe supporting internationally important numbers of birds and rare plants (Paal, 1998; Puurmann & Ratas, 1998; Luhamaa et al., 2001; Rannap et al., 2004), although comparatively little is known about their small mammal communities and the extent to which they utilize different habitats within the wetland landscape. The analysis of scale is important within landscape ecological studies as scale relates to the spatial and temporal dimensions of a landscape or habitat system (Farina, 1998). Forman (1997) defines the landscape as a mosaic where the mix of land use or ecosystems occurs over kilometre-wide areas, and habitats are local in scale, relatively homogeneous and distinct by their boundary. We surveyed small mammals in six different wetland sites with representative management intensities, habitat types, and landscape composition. Our objectives were to (a) assess species abundance, composition, and structure of small mammal assemblages in representative wetland habitats; (b) assess landscape characteristics of the wetland sites in relation to small mammal assemblages; (c) determine which habitat features locally influence mammal abundance at the species level. Results will enable better integration of the requirements of small mammals into wetland conservation and management, such as agri-environment schemes and restoration initiatives.

MATERIALS AND METHODS

Study area and site selection

Western Estonia is characterized by limestone geology and a flat topography, with an average altitude of 50 m a.s.l. The climate is continental-temperate; temperatures can reach 30°C in summer and fall below -5°C in winter with an annual precipitation of approximately 550 mm (Peterson, 1994; Puurmann & Ratas, 1998).

The study was conducted in the summers of 2002 and 2003 at six coastal wetlands in west Estonia (Fig. 1). Four sites were located in Vormsi Island landscape reserve, a protected area of over 10 km² established in 2000. Two of these sites (Sviby (58°58'31.9" N, 23°17'42.6" E) and Rumpo Peninsula (58°57'47.4" N, 23°16'32.5" E)) had recent (<5 years) re-establishment of management in the form of extensive sheep grazing and periodic hay cutting, while the other two sites (Hullo Bay (58°59'5.93" N, 23°13'37.1" E) and Hosby (58°58'49.0" N, 23°21'24.6" E)) had been abandoned and were not managed. The final two wetlands were situated on the mainland and had been managed for at least 5 years with cattle grazing at <1 livestock unit per hectare. These were Keemu (58°43'31.1" N, 23°40'22.8" E) in Matsalu National Park and Tahu (58°58'57.8" N, 23°34'3.87" E) in Silma National Nature Reserve. For further site descriptions see Burnside et al. (2007).

Aerial photographs and GIS maps were used to select sampling locations of representative habitats within the wetland sites based on vegetation composition, mapped at a resolution of 10 m × 10 m (Burnside et al., 2007). Wetland habitat types used for small mammal sampling were reed beds, short grass, tall grass, and scrub and developing woodland. Neighbouring habitats were also sampled to enable comparisons: coniferous woodlands, deciduous woodlands, juniper scrub,



Fig. 1. Map showing locations of the six coastal wetland study sites in west Estonia. 1 = Hullo Bay; 2 = Rumpo Peninsula; 3 = Sviby; 4 = Hosby; 5 = Tahu, Silma; and 6 = Keemu, Matsalu.

agricultural field margins, and hedgerows. Trap-line locations within a site were restricted by the habitat types available, a patch size of greater than 100 m to fit a trap line within a single habitat type and a minimum distance of 100 m between samples. The aim was to maximize the diversity of wetland habitats sampled within a site irrespective of contribution to the overall wetland area. Wetland areas were also prioritized over non-wetland areas. Where the same habitat type was sampled within a site a minimum distance of 300 m was applied to aid independence. All these contributory factors led to differential sampling effort between habitats. To investigate annual variation and avoid temporal bias in communities, 23 locations were replicated annually, 15 at Sviby and the Rumpo Peninsula and eight at Matsalu. In total 70 sampling occasions were undertaken, but with 23 sites being repeated annually and averaged across the two years, this resulted in 47 locations with data.

Vegetation and habitat assessment

At each mammal sampling location vegetation structure, composition and abundance (using the DAFOR scale, Kent & Coker, 1992) were assessed using 11 variables selected considering their importance in determining small mammal communities (see Grant et al., 1982; Chętnicki & Mazurkiewicz, 1994; Ecke et al., 2002). For woodlands and hedgerows, habitat structure was separated into three distinct vegetation layers: field layer (<1 m), understorey (≥ 1 <5 m) and canopy (≥ 5 m). The percentage cover of tree and shrub canopy was recorded in four random 10 m \times 10 m plots. Understorey and field layer structure and composition were determined from nine habitat variables recorded in four random 2 m \times 2 m quadrats: grass cover and average height in centimetres, forb cover and average height in centimetres, reed cover and average height in centimetres, bare ground cover, litter cover, and litter depth.

Small mammal trapping

Multiple survey methods for small mammals were initially piloted, namely field sign counts (Wilkinson et al., 2004), bait tubes (Churchfield et al., 2000), and pitfall traps (Francl et al., 2002). Due to either poor returns or impracticality (i.e. non-detection, flooding of pitfall buckets), only live-trapping was eventually employed across all study areas. Small mammals were surveyed between July and August to maximize trapping yields (Masing, 1987) and because low temperatures and snow cover outside the summer period can render live-trapping unfeasible due to logistics and animal welfare. As some small mammal species have cyclic populations (Sundell et al., 2004), sampling over two years reduced temporal bias. A mammal 'sample' comprised three consecutive nights trapping with 10 Sherman live-traps (23 cm \times 9 cm \times 8 cm, H. B. Sherman Traps Inc., Orlando, Florida). The minimum number of trap nights was determined by plotting species accumulation curves from six trap nights, which showed that three nights trapping revealed the

same species as six. Rarefaction curves were also plotted for all habitats (Gotelli & Colwell, 2001). Sherman traps were used because they have been shown to attain higher species richness than pitfalls or snap-traps (Francl et al., 2002). Traps were arranged along a line transect with one trap per station spaced at 10 m intervals. Trap lines can be a better rapid method of assessing community diversity than grid trapping (Read et al., 1988). Traps were baited with a mixture of oats, peanut butter, carrots, and apple and checked twice daily at first light and again at dusk, when they were re-baited. Captures were identified to species, individually marked with a unique hair clip (Gurnell & Flowerdew, 2006), weighed, and external evidence of sexual status recorded to determine age category. All individuals were then released at their point of capture.

GIS and landscape assessment

Habitats GIS (Arc GIS 8.x ESRI) was used to quantify landscape structure at each study site. The habitat type and patch characteristics were assessed along with the surrounding landscape mosaic within each 50 m, 100 m, 150 m, and 200 m 'buffer' zone from the central GPS point of a trap line. Each increase in buffer zones included the previous area and, although not independent, gave a more realistic description of habitat use by small mammals. Each buffer zone was converted to a raster grid and the proportion of each habitat within this was calculated. Buffer sizes were selected as appropriate scales at which the landscape matrix may influence small mammal movement and dispersal (MacDonald & Barrett, 1993). The habitat patch types recorded in the buffer zones included lower shore, club-rush beds, and 'mosaic', which is defined as combinations of different vegetation types (see Burnside et al., 2007). A 'landscape mosaic' describes the spatial heterogeneity present in the landscape (Forman, 1997). FRAGSTATS 2.0 (McGarigal & Marks, 1994) was used to quantify habitat and landscape characteristics, which included total area (ha), number of patches and mean patch size (ha), edge length (m), nearest-neighbour distance (m), shape index (SI), Shannon diversity (H'), and evenness (E) (Waite, 2000).

Statistical analyses

Relative abundance was calculated as Catch Per Unit Effort (CPUE), excluding recaptures, with 0.5 trap nights removed per falsely triggered trap (Beauvais & Buskirk, 1999). Average relative biomass was calculated as total body weight trapped per 100 trap nights. Small mammal diversity was calculated using Simpson's ($1-D'$) index (Waite, 2000). To compare habitat types the average of two years trapping was calculated for repeated sites, giving 47 samples in total. For landscape analysis only data from 2003 were used. All data were checked for normality prior to analysis. Where data were non-normal non-parametric tests were applied. Spearman rank correlation was used to relate small mammal species

relative abundance to habitat and landscape characteristics. The 11 habitat variables were checked for inter-correlation and consequently reduced to 6 variables (with Bonferroni correction $p < 0.003$), which were selected based on their likely ecological importance. As species are mainly ground dwelling percentage cover was selected over height when the two variables were inter-correlated. SPSS version 12 was used for analysis (Norusis, 1998). To further investigate patterns in communities and the influence of environmental parameters a Canonical Correspondence Analysis (CCA) was run using PcOrd (Version 4).

RESULTS

Species–habitat associations

A total of 2208 trap nights resulted in 381 captures (overall CPUE = 17%), revealing 286 individuals of seven different species: 74 yellow necked mice (*Apodemus flavicollis*), 66 field voles (*Microtus agrestis*), 52 striped field mice (*Apodemus agrarius*), 47 bank voles (*Myodes glareolus*), 44 common shrews (*Sorex araneus*), 2 harvest mice (*Micromys minutus*), and 1 water vole (*Arvicola terrestris*). No animals were found moving between trap lines, therefore they were assumed to be spatially independent. Use of all habitat data revealed no significant difference in the overall relative abundance and species richness, or individual species abundance, between samples taken on the Vormsi Island and mainland sites (t -test, $p > 0.05$ for relative abundance and richness; Mann–Whitney U, $p > 0.05$, for all other). In addition there was no significant difference between mainland and island individual species abundance in reed bed, short grass, or tall grass habitats (Mann–Whitney U, $p > 0.05$). Only these habitats had sufficient samples to enable comparisons. As there was no difference between mainland and island samples, data were pooled. Species habitat associations, richness, biomass, and diversity index data are summarized in Table 1. Rarefaction curves showed a species richness plateau in the majority of habitats. Too few individuals were trapped in coniferous woodland and field margin to reveal a plateau and so further trapping in these habitats may reveal more species. *Apodemus flavicollis* was the most widespread species, found in 89% of the samples and all habitats, with greatest relative abundance in deciduous woodland and hedgerows. *Sorex araneus*, *A. agrarius*, *M. agrestis*, and *M. glareolus* were also present in a wide range (67%) of samples and at least six habitat types. *Apodemus agrarius* appeared to be most common in reed bed and tall grass habitats. The relative abundance of *M. agrestis* was greatest in tall grass, whereas *M. glareolus* was found most abundantly in deciduous woodland and hedgerows. *Micromys minutus* was only trapped in two locations, a reed bed and scrub woodland, and a solitary *A. terrestris* was trapped in a hedgerow. There was no significant difference in the overall small mammal relative abundance between 2002 and 2003 (Mann–Whitney U, $U = 261$, $n = 23$, $p = 0.937$); however, the community composition differed between the two years (Chi-squared test, $X^2 = 678$, d.f. = 5, $p < 0.05$).

Table 1. Summary of small mammal trapping results for each habitat. RB = reed bed, SG = short grass, TG = tall grass, SW = scrub and developing woodland, CW = coniferous woodland, DW = deciduous woodland, H = hedgerow, and M = field margin. Species data are shown as average CPUE with associated standard error in italics

	Habitat															
	RB	SG	TG	SW	CW	DW	H	M								
No. of samples (Total 70)	15	19	10	5	4	8	4	3								
Species richness (Total 7)	6	4	4	6	3	4	5	4								
Average biomass, g	328.7	99.0	51.0	18.3	439	116.8	199.9	84.0	90.8	66.9	610.8	229.8	745.2	371.2	451.1	322.4
<i>Sorex araneus</i>	3.28	1.46	1.01	0.72	4.00	2.37	0.50	0.50	0	0.21	0.21	0.21	7.50	6.44	1.11	1.11
<i>Apodemus flavicollis</i>	2.11	1.29	1.05	0.63	1.67	0.90	1.33	0.82	0.83	0.83	10.71	4.65	14.71	10.92	8.89	8.89
<i>Apodemus agrarius</i>	6.11	2.19	0.35	0.24	4.67	1.81	3.33	2.58	0	0.42	0.42	3.33	2.36	0	0	0
<i>Micromys minutus</i>	0.22	0.22	0	0	0	0.67	0.67	0	0	0	0	0	0	0	0	0
<i>Microtus agrestis</i>	4.67	1.95	0.88	0.43	11.33	3.38	1.33	1.33	0.83	0.83	0	0	0	0	3.33	1.93
<i>Myodes glareolus</i>	0.44	0.44	0	0	0	0	2	2.00	1.67	1.67	12.05	6.05	5.83	3.44	3.33	3.33
<i>Arvicola terrestris</i>	0	0	0	0	0	0	0	0	0	0	0	0	0.83	0.83	0	0
Simpson's diversity (1-D')	0.74	0.72	0.64	0.77	0.62	0.53	0.70	0.63	0.63	0.63	0.53	0.70	0.70	0.63	0.63	0.63

Out of the 70 samples taken 22 revealed no captures, including both samples made from the juniper scrub. Hedgerow, field margin, and tall grass trapping sessions were always successful, whereas coniferous woodland and short grass had $\geq 50\%$ of samples resulting in no captures. Highest CPUE was recorded in reed bed (57%), hedgerow (57%), and deciduous woodland (70%). Overall relative abundance and species richness significantly differed between habitat types (ANOVA, $F = 2.54$, d.f. = 8, $p < 0.05$; $F = 2.18$, d.f. = 8, $p = 0.05$, respectively). However, only *M. agrestis*, *M. glareolus*, and *A. terrestris* showed significant difference in CPUE between habitat types (Kruskal–Wallis, $X^2 = 22.09$, d.f. = 8, $p < 0.01$; $X^2 = 18.52$, d.f. = 8, $p < 0.05$; $X^2 = 22.50$, d.f. = 8, $p < 0.01$, respectively). Average relative biomass also showed a statistically significant difference between habitat types (Kruskal–Wallis, $X^2 = 20.45$, d.f. = 8, $p < 0.01$) with the highest average biomass recorded in hedgerows followed by deciduous woodland. There was no significant difference in diversity between habitats. No habitat had all seven species present. Highest species richness and diversity index values were recorded in reed bed and scrub woodland. Wetland habitats had greater average species richness and average diversity, but lower average abundance compared to neighbouring habitats.

Table 2 shows the relative age and sex composition, percentage breeding, and recapture rates of animals trapped within four different wetland habitats. Other habitats were excluded as they were not deemed ‘wetland’. All habitats had similar sex ratios. Short grass had the highest proportions of sub-adults and non-breeding individuals as well as the lowest recapture rates. Reed bed had the highest proportion of adult females breeding, whereas scrub and developing woodland, and tall grass, had high proportions of breeding males. Data were insufficient to allow comparisons of individual species between habitats (i.e. no species had >10 individuals in all four habitats).

Table 2. The percentage of adults, females, breeding individuals, and re-capture rates from small mammal captures in four different wetland habitats. RB = reed bed, SG = short grass, TG = tall grass, SW = scrub and developing woodland

Habitat		% Adults	% Females	% Breeding	% Recaptures
RB	Male	57	44	48	19.7
	Female	78		78	
	Total	66		61	
SG	Male	50	54	20	11.3
	Female	57		42	
	Total	54		33	
TG	Male	81	60	75	20.0
	Female	58		54	
	Total	68		63	
SW	Male	83	50	83	20.6
	Female	50		33	
	Total	67		62	

Effects of habitat and landscape characteristics

Habitat variables were tested for correlations with small mammal species relative abundance (Table 3). As vegetation cover differed annually, each of the 70 samples was kept separate. Due to the low number of captures *A. terrestris* and *M. minutus* were excluded from the analysis. The relative abundance of *A. flavicollis* and *M. glareolus* was significantly negatively correlated with grass cover and significantly positively related to tree density. *Apodemus agrarius* was significantly positively correlated with litter depth and reed cover, whereas *M. agrestis* was significantly positively correlated with grass cover and showed a significant negative association with bare ground and tree density. There were also three significant relationships between species, with the relative abundance of *M. agrestis* and *A. agrarius*, *M. glareolus* and *A. flavicollis*, and *A. terrestris* and *S. araneus* all positively correlated (Table 3). The ordinations of environmental variables in the first two canonical axes were plotted using bi-plot scores from CCA (Fig. 2).

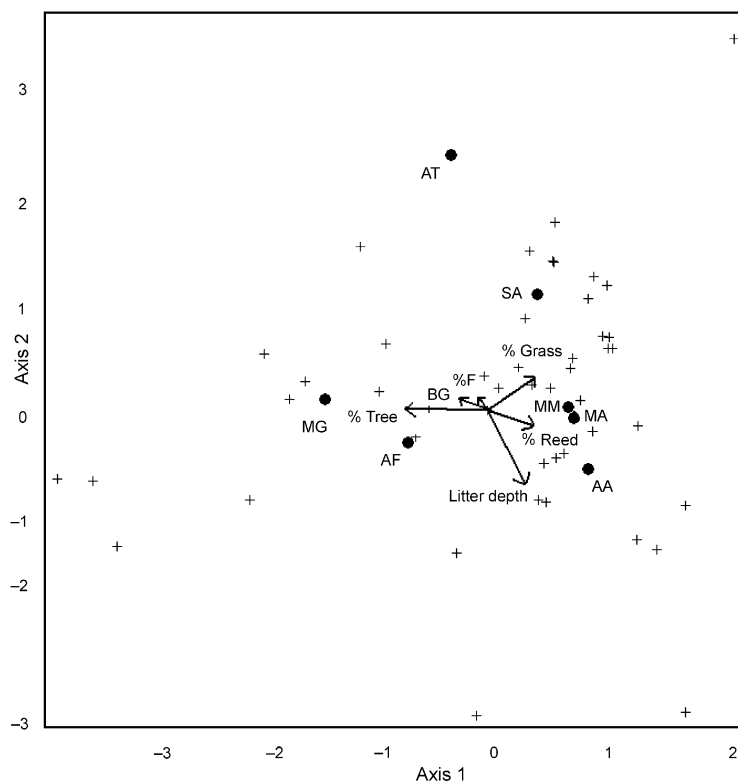


Fig. 2. Bi-plot and LC scores from Canonical Correspondence Analysis plotted for species, sample, and habitat variables (No. of samples = 47) in the first two canonical axes. Here ● represents species, + represents sites, and arrows represent environmental variables. AF = *Apodemus flavicollis*, SA = *Sorex araneus*, AA = *Apodemus agrarius*, MA = *Microtus agrestis*, MG = *Myodes glareolus*, MM = *Micomys minutus*, and AT = *Arvicola terrestris*. % is percentage cover, BG = bare ground, F = forbs and herbs.

Table 3. Spearman rank correlation coefficient between the relative abundance of small mammal species and key habitat characteristics. SA = *Sorex araneus*, AF = *Apodemus flavicollis*, AA = *Apodemus agrarius*, MA = *Microtus agrestis*, MG = *Myodes glareolus*, AT = *Arvicola terrestris*, and MM = *Micromys minutus*. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$

Species	Habitat variables							Species				
	% Forbs	% Grass	Bare ground	Litter depth	Tree density	% Reed	SA	AF	AA	MA	MG	AT
SA	-0.108	0.150	-0.044	-0.161	-0.135	0.104						
AF	0.037	-0.272*	0.127	0.014	0.316**	-0.165	0.109					
AA	-0.102	-0.118	-0.221	0.486***	-0.124	0.400***	0.122	0.206				
MA	-0.102	0.304*	-0.243*	0.087	-0.306**	0.147	0.162	-0.113	0.366**			
MG	0.021	-0.281*	0.209	-0.113	0.461***	-0.174	0.082	0.519***	-0.035	-0.212		
AT	0.200	-0.080	0.003	-0.116	0.161	-0.075	0.268*	0.174	-0.072	-0.080	-0.052	
MM	-0.132	0.093	-0.079	0.189	0.013	0.098	-0.100	0.031	0.136	0.116	0.164	-0.021

The first canonical axis correlated most highly with % tree cover to % reed cover. However, this axis accounted for only 20.3% of the variance (eigenvalue = 0.424, $R = 0.76$). The combined first, second, and third axes only accounted for 36.3% of the variance. A Monte Carlo permutation test showed that the first canonical axis was significantly correlated to species distribution patterns ($p = 0.02$).

Only 29 trap locations had sufficient spatial mapping extending to 200 m zones to enable landscape analysis. The presence of small mammals was not related to the size of the wetland habitat patch in which trapping took place, as the extent of tall grass, reed bed, short grass, or scrub and developing woodland surrounding the trap lines was not correlated with mammal relative abundance, richness, or biomass. However, small mammal parameters tended to be significantly positively correlated with landscapes containing tall grass and mosaic habitats, and significantly negatively correlated with larger areas of short grass, club rush swamp, lower shore, and bare ground (Table 4). Mammal species richness and diversity were particularly sensitive to landscape composition, being significantly correlated to the extent of each habitat type at nearly all spatial scales (Table 4).

Table 4. Correlation coefficients showing the relationship between small mammal parameters and landscape composition (% cover) at four different zone sizes ($n = 29$). R.A. = relative abundance, Diversity = Simpson's diversity ($1-D'$). TG = tall grass, LS = lower shore, SG = short grass, BG = bare ground, CB = club rush swamp. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$. Spearman rank correlation was used, gaps represent no significant correlation

Zone size, m	Habitat	Small mammal parameter			
		R.A.	Species richness	Diversity	Biomass
50	Mosaic		0.428**	0.428**	
	TG	0.660***	0.632***	0.544**	0.652***
	LS		-0.384**	-0.458*	0.404**
	SG		-0.380**		
100	Mosaic	0.663***	0.708***	0.657***	0.643***
	TG				0.399*
	CB	-0.574**	-0.599**	-0.508**	-0.520**
	BG		0.441*	0.502**	
150	Mosaic	0.562**	0.692***	0.606***	0.550**
	TG	0.643***	0.621***	0.629***	0.556**
	CB	-0.461*	-0.509**		-0.419*
	LS	-0.529**	-0.592**	-0.657***	-0.516**
	BG	-0.442*	-0.447*		-0.441*
200	Mosaic	0.538**	0.678***	0.607***	0.552**
	TG	0.610***	0.623***	0.620***	0.550**
	CB	-0.506**	-0.567**	-0.397*	-0.462*
	LS	-0.573**	-0.649***	-0.657***	-0.556**
	SG	-0.458*	-0.464*	-0.380*	-0.394*
	BG	-0.503**	-0.509**		-0.502**

All other recorded landscape variables ($n = 20$) did not show any correlation with mammal relative abundance, richness, or biomass (Pearson product correlation $p > 0.05$ for all).

DISCUSSION

Although seven representative species (e.g. see Masing, 1987) were recorded, rarer species may have been under-represented due to survey length, timing, or method. In comparison to data collected by Masing (1987) in Matsalu, we did not capture common voles (*Microtus arvalis*) or Northern birch mice (*Sicista betulina*). However, Masing's study encompassed agricultural and urban habitats as well as coastal. Sherman traps can bias for species >10 g (Francel et al., 2002), this bias may have affected *S. betulina*, which has a body mass range of 5–13 g (MacDonald & Barrett, 1993). Additionally water shrews (*Neomys fodiens*) were expected to be found, but were not captured. As trap success of water shrews tends to be low, an alternative method to effectively survey them is to set bait tubes and identify faecal remains (Churchfield et al., 2000). It is therefore recommended that in future surveys a variety of methods be employed to detect a wider range of species. While standardized trap lines provide a rapid assessment of presence and relative abundance, they do not yield density estimates (Gurnell & Flowerdew, 2006). Whilst the design of this survey enabled comparisons across a range of habitats, variation in capture rates suggests modifications in future surveys are needed to account for density variations. Also, although 70 transect lines were sampled in six different wetland sites with over 2200 trap nights, the number of small mammal captures was low (381, 17% CPUE) and high variability was found within habitats. Whilst the value of this study is partially derived from the paucity of information on small mammals in wetland landscapes, particularly in Estonia, greater capture effort and returns would clearly help verify findings. Further studies should also investigate spatial and temporal utilization of the coastal wetland landscape by small mammals to determine the relative importance of different habitats at different times of the year, as this study was limited to summer sampling. Community assemblages will vary spatially and temporally. Populations of *Microtus agrestis* in Finland have been recorded to fluctuate on a three year cycle (Huitu et al., 2003), hence the timing of the study in relation to population cycles would affect results. In addition, Huitu et al. (2003) found landscape composition to influence cycles, therefore this also needs to be taken into account. Habitat quality is an important factor determining mammal presence and abundance (Poulin et al., 2002) and may explain the large differences within samples recorded in this survey. As flood frequency and duration have direct impacts on small mammals (Jacob, 2003), water level monitoring may also help to understand temporal and spatial patterns in small mammal habitat associations.

Habitat and landscape effects

The results of this study in west Estonia support research indicating the importance of wetlands for small mammals (e.g. Krištofik, 2001). For example, reed beds had the highest species diversity of all habitats surveyed. Large variations in capture rates (0–90%) may have been due to reed bed quality and/or hydrological regime, which was found to determine reed bed dwelling birds in France (Poulin et al., 2002). A study of small mammals in reed stands in Slovakia resulted in 16 species, with the community dominated by *S. araneus*, *M. glareolus*, and *A. sylvaticus* (Krištofik, 2001). Reed beds in west Estonian sites were dominated by *A. agrarius* followed by *M. agrestis* and then *S. araneus*. Differences in dominance may be due to the geographic species pool (MacDonald & Barrett, 1993), shifts in habitat preferences in relation to competition, or reed age, type, or quality (Poulin et al., 2002). Reed beds supported the highest proportion of breeding female small mammals, possibly due to a combination of cover and inaccessibility providing protection from predators, and plentiful resources for both nest construction and meeting the increased metabolic demands during weaning. In a study in Austria, Haberl & Kryštufek (2003) found harvest mice (*M. minutus*) at densities of up to 93 individuals per hectare in reed beds, suggesting that this could be an important habitat for this internationally ‘near threatened’ species (Amori, 1996). Tall grass had the same mammal species composition as short grass but in greater relative abundance, probably because short, highly grazed or cut grass is commonly avoided by small mammals (Grant et al., 1982; Eccard et al., 2000; Tattersall et al., 2001; Schmidt et al., 2005). Short grass had the highest proportions of sub-adults and non-breeding individuals as well as the lowest recapture rates, indicating that this is less likely to be a breeding habitat and that animals captured here are mostly transitory, dispersing sub-adults or non-breeding adults. However with sufficient data, species specific analysis should be undertaken to elucidate the role of each habitat to each species. Short grass habitats seem more likely to be ‘sink’ as opposed to ‘source’ habitats for small mammals. Low recapture rates also indicate that this habitat is probably used during dispersal rather than maintaining a high resident population. Scrub and developing woodland habitats had relatively high species richness, although most species were in low relative abundance. This habitat is typically positioned on higher ground at the wetland periphery and may offer temporary refuge during flood events. Agricultural field margins and hedgerows were utilized by a range of small mammals. Hedgerows provide protection from predators, food sources, and can act as corridors for dispersal, especially when adjacent land use is lower in cover and higher in predation risk (Chętnicki & Mazurkiewicz, 1994; Tattersall et al., 2001).

Mammal relative abundance, diversity, biomass, and richness were positively correlated with the amount of adjacent mosaic and tall grass habitat within 200 m of the trapping site and negatively correlated with more open landscapes supporting habitats such as short grass, bare ground, and lower shore herbaceous vegetation.

Tall grass provides cover and protection from predators for small mammals (Tattersall et al., 2001) whereas short grass and open patches provide little protection and tend to be avoided (Eccard et al., 2000; Tattersall et al., 2001).

Implications for conservation

Management of wetlands through grazing and cutting for agriculture or conservation often leads to a reduction of those wetland habitats such as reed bed and tall grass that in this study were found to support high diversity and relative abundance of small mammals. Such management favours short grass and lower shore habitats (see Burnside et al., 2007), which contained relatively low small mammal richness and relative abundance in this study. Estonian wetlands are of international conservation concern due in part to the wetland birds that utilize them (Joyce & Wade, 1998). Many of these birds, such as waders, require medium-short grass for feeding and successful breeding (Kuresoo & Mägi, 2004) and large open areas reduce the risk of predation (Butler et al., 2005), both of which are less suitable for small mammals. At the landscape scale, management for specific conservation or agricultural targets may result in greater uniformity with less mosaic vegetation, which this study suggests would also be detrimental to small mammals. Furthermore, isolation of suitable habitat patches may prevent movement between habitats and consequently more isolated patches tend to have lower small mammal populations (Marsh et al., 2001). This illustrates a conflict of interest in nature conservation and highlights the need for a holistic approach to restoration and conservation management of wetlands, including a better understanding of the impacts of conservation management practices on different (e.g. non-target) components of the ecosystem. Such a perspective for wetland landscapes may seek to provide a mosaic of habitat types for a range of different species, although in practice reconciling the needs of different biodiversity components is difficult. For small mammals, however, this study has indicated that biodiversity management should aim to incorporate refuge habitats such as reed beds, tall grassland and scrub within the wetland landscape.

ACKNOWLEDGEMENTS

We are grateful to Steve Waite for useful comments on the manuscript. We would like to thank NGO Läänerannik, the Estonian Ministry of Environment, and the administrations of Matsalu National Park and Silma Nature Reserve who allowed us to undertake the study. Thanks to Elle Puurmann, Kaja Lotman, Meelis Mägi, and Ivar Ojaste for their support. Thanks also to all the Earthwatch Volunteers, Fran Southgate and Sarah Toogood, who aided in the fieldwork. The study was funded by the University of Brighton and the Earthwatch Institute.

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Elupaiga tüübi ja maastiku mitmekesisuse mõju pisiimetajatele Eesti ranniku-märgaladel

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On uuritud elupaiga tüübi ja maastiku struktuuri mõju pisiimetajate suhtelisele arvukusele ning liigilisele mitmekesisusele Lääne-Eesti ranniku-märgaladel. Kuuelt märgalalt, kokku kaheksast biotoobist, õnnestus tabada seitsmekümne eluspüügilõksuliiniga seitse liiki pisiimetajaid. Liigiline mitmekesisus oli kõrgeim roostikes ja põdsastikes ning pisiimetajate biomass oli kõrgeim niitmata rohumaadel. Madal-murustel rohumaadel olid madal nii liigirohkus kui isendite hulk. Pisiimetajate arvukus, liigirohkus ja biomass seostusid positiivselt niitmata rohumaaga ning mosaiikelupaiga hulgaga ümbruses ja negatiivselt vahetu mere kalda ning avatud pinnasega alade hulgaga. Uuringu tulemused viitavad võimalikule looduskaitselele vastuolule pisiimetajate ja ülejäänud biomitmekesisuse kaitsel märgaladel.