Biodivers Conserv (2010) 19:725–743
DOI 10.1007/s10531-009-9730-3

ORIGINAL PAPER

Abundance and habitat preferences of the southernmost population of mink: implications for managing a recent island invasion

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Received: 1 April 2009 / Accepted: 30 September 2009 / Published online: 16 October 2009

Abstract Since 2001 invasive American mink has been known to populate Navarino Island, an island located in the pristine wilderness of the Cape Horn Biosphere Reserve, Chile, lacking native carnivorous mammals. As requested by scientists and managers, our study aims at understanding the population ecology of mink in order to respond to conservation concerns. We studied the abundance of mink in different semi-aquatic habitats using live trapping ($n = 1{,}320$ trap nights) and sign surveys ($n = 68$ sites). With generalized linear models we evaluated mink abundance in relation to small-scale habitat features including habitats engineered by invasive beavers (*Castor canadensis*). Mink have colonized the entire island and signs were found in 79% of the surveys in all types of semi-aquatic habitats. Yet, relative population abundance (0.75 mink/km of coastline) was still below densities measured in other invaded or native areas. The habitat model accuracies indicated that mink were generally less specific in habitat use, probably due to the missing

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limitations normally imposed by predators or competitors. The selected models predicted that mink prefer to use shrubland instead of open habitat, coastal areas with heterogeneous shores instead of flat beaches, and interestingly, that mink avoid habitats strongly modified by beavers. Our results indicate need for immediate mink control on Navarino Island. For this future management we suggest that rocky coastal shores should be considered as priority sites deserving special conservation efforts. Further research is needed with respect to the immigration of mink from adjacent islands and to examine facilitating or hampering relationships between the different invasive species present, especially if integrative management is sought.

Keywords Capture-mark-recapture · *Castor canadensis* · Chile · Exotic species · Management · *Neovison vison* · Population size · Sign surveys · Trapping · Wetlands

Abbreviations

GLMs	Generalized linear models
PCA	Principal component analysis

Introduction

In the past 200 years, the numbers of species that have entered new ranges through human agency have increased enormously. Although many exotic species are an integral component of our global economy, biotic invasions can cause fundamental changes in native biodiversity (Vitousek et al. 1997; Sala et al. 2000). Island ecosystems are more susceptible to loss of biodiversity because indigenous species have often evolved in the absence of competition, herbivory, parasitism or predation (Elton 1958; Quammen 1997; Courchamp et al. 2003). The introduction and establishment of carnivorous mammals on islands is considered a major factor in reducing populations of native species, and therefore provokes high conservation concern (Macdonald and Thom 2001; Krajik 2005).

The American Mink (*Neovison vison*) is a semi-aquatic carnivorous mustelid native of North America and was introduced to South America as a fur bearer, with feral populations still restricted to the southern parts of Chile and Argentina (Jaksic et al. 2002). The mink represents a recent invasion on Navarino Island within the Cape Horn Biosphere Reserve (Jaksic et al. 2002; Rozzi et al. 2006). Liberated or escaped animals from mink farms on the Argentine side of Tierra del Fuego might have swum across the Beagle Channel (ca. 5 km wide) probably reaching Navarino Island during the mid-1990s, and first registered by scientists in 2001 (complete record history in Rozzi and Sherriffs 2003). The native mammal assemblage on the island is extremely low in its number of species and lacks mustelids or other carnivores (Anderson et al. 2006a); sea otters (*Lontra felina*) are mainly associated with the Wollaston and Cape Horn Islands (Rozzi et al. 2006). This situation has two consequences: first, mink have no competitors on the island, and second mink have no main predators. Feral dogs and raptors (*Caracara plancus*) might represent potential predators, but the latter is less probable as it primarily feeds on carrion (Travaini et al. 2001).

Among the biotic factors regulating populations are direct (e.g. predation, interference competition) and indirect (e.g. trophic webs, exploitative competition) interactions. The lack of potential predators (with the possible exception of feral dogs) on Navarino Island has presumably facilitated the establishment of the mink population. In carnivores, interspecific competition plays an important role, often leading to direct aggressive interactions (Palomares and Caro 1999). These interactions are absent for mink on the island. One of the factors influencing habitat use by mink is interference from competing mustelids (Dunstone and Ireland 1989; Sidorovich et al. 1996; Bonesi and Macdonald 2004). Consequently, the absence of other mustelids on Navarino should determine its habitat requirements, i.e. allowing the mink to be less specific. The aim of this study was to quantify and discuss these two parameters, abundance and habitat preferences, in the initial phase (<10 years of presence) of the invasion of mink on Navarino Island, in a situation that is somewhat different from other invaded regions as mink represent a new guild of terrestrial mammalian predators.

Although the ecology of mink as an invasive species is well studied in some European countries, especially Britain (e.g. Yamaguchi et al. 2003; Reynolds et al. 2004; Bonesi et al. 2006), systematic data on its population ecology in the Cape Horn region are still wanted. The need for such basic data has been expressed by scientists and public agencies currently supporting the implementation of control strategies for invasive mammals in the Cape Horn Biosphere Reserve (Anderson et al. 2006a; Rozzi et al. 2006; Soto and Cabello 2007). Specific data requested includes the estimation of the population size on the island in order to define target numbers to be removed, data helping to improve the capture success (e.g. season-dependent effectiveness of trapping), habitat preferences, and relationships between different introduced mammal species (invasional meltdown hypothesis, i.e. invasive species aid new species to establish, Simberloff and Von Holle 1999). Among the latter relationships there is a particular interest in whether beaver engineering (*Castor canadensis*) would improve habitats for the mink, by creating slow-flowing ponds and burrows (as shown by Żurowski and Kammler 1987; Sidorovich et al. 1996). This interest also arises from the current plans for eradication of beavers from Chilean and Argentine Tierra del Fuego Island and adjacent islands within the Cape Horn Archipelago (Choi 2008).

Major conservation concerns are derived from studies on the impact of mink conducted in Europe (see review in Bonesi and Palazon 2007). The authors report reductions in populations of ground-nesting waterbirds (Craik 1997; Ferreras and Macdonald 1999; Nordström and Korpimäki 2004), fish and crustaceans (Delibes et al. 2004), amphibians (Ahola et al. 2006), and even the local extirpation of species, particularly in the case of water voles *Arvicola terrestris* (Jefferies 2003). In South America, studies on the impact of mink are still scarce, but mink are considered as detrimental to waterbirds (Lizarralde and Escobar 2000; Rozzi and Sherriffs 2003), and perhaps to southern river otters *Lontra provocax* (Previtali et al. 1998; but see Medina 1997; Fasola et al. 2009). Studies on the diet of mink on Navarino Island (Schüttler et al. 2008; Ibarra et al. 2009) demonstrated relatively high proportions of birds in the spring and summer diet of mink. Indeed, breeding failure has occurred in some aquatic bird species endemic to Patagonia (Schüttler et al. 2009), a possible result of prey naivety to the new terrestrial predator (e.g. Nordström et al. 2004).

In this paper, we describe (1) the relative abundance of *Neovison vison* in different semi-aquatic habitats of Navarino Island; (2) the relationship between mink abundance and small-scale habitat features; and (3) the relation between beaver habitats and mink abundance. The study will provide practical information much needed for the design of a management plan. Finally, the results can be used to predict habitats favouring the invasion of American mink in the Cape Horn Biosphere Reserve and elsewhere in southern South America, where this species is currently expanding its range.

Study area

The study was carried out on Navarino Island (2,528 km^2), Chile, located at the extreme southern tip of South America (Fig. 1). The island forms part of the Cape Horn Biosphere Reserve (54–56°S) and belongs to the Magellanic Sub-Antarctic Evergreen Rainforest ecoregion, recently identified as one of the 24 most pristine wilderness areas of the world (Mittermeier et al. 2003). The main habitats include (i) evergreen rainforests dominated by *Nothofagus betuloides* and *Drimys winteri*, (ii) Magellanic deciduous forests of *Nothofagus pumilio*, (iii) peatlands, moorlands and bogs, (iv) high-Andean vegetation communities dominated by cushion plants and lichens, (v) streams and lakes, and (vi) thickets or shrublands in naturally or anthropogenically disturbed areas (Pisano 1977; Rozzi et al. 2006). The climate type is oceanic, with a low annual thermal fluctuation (<5°C), a mean annual temperature of 6°C, and an annual precipitation of 467.3 mm (Pisano 1977). During winter, streams and lakes are ice-bound. The human population of approximately 2,300 people is concentrated in the settlement of Puerto Williams, capital city of the Chilean Antarctic Province, on the northern coast of Navarino Island. A small fishing village, Puerto Toro, exists on the eastern coast of the island. Outside these towns, human settlements are limited to rural houses, and some Navy stations. Access to the settlements and other areas relies mostly on boats; except for a dirt road that connects the northern coast of Navarino Island. Therefore, our research was concentrated in the northern part of the island.

As mink are semi-aquatic mustelids, our study sites comprised shorelines of marine coasts, river banks, lake and pond margins. The habitat adjacent to the water's edge included meadow communities, shrubland dominated by *Berberis buxifolia*, *Pernettya mucronata* and *Chiliotrichum diffusum* (Moore 1983), peatlands (*Sphagnum* spp.), evergreen and deciduous forests dominated by the genus *Nothofagus* (basically found in the northern part of the island), and habitats modified by beavers. Beaver foraging for both food and construction activities in *Nothofagus* forests clear trees and alters the riparian community structure (Anderson et al. 2006b). This results in greater understorey species richness, particularly of exotic plants, and productivity (Martínez Pastur et al. 2006; Anderson et al. 2009).

Methods

Capture-mark-recapture

We applied capture-mark-recapture with the aim of estimating the relative abundance of mink. Trapping took place at three sites on the northern coast of Navarino Island: Robalo at the Omora Ethnobotanical Park (54°56′S, 67°39′W), Guerrico (54°54′S, 67°51′W), and Mejillones (54°53′S, 67°58′W), where we selected 4 km of riverside and 4 km of rocky coastline within each study site ($n = 6$ sites). Each trapping session lasted 4–5 nights. Trapping was repeated during all four seasons for coastal sites (April 2005–September 2007), but at rivers we trapped only once during autumn and early winter in 2005 ($N = 1{,}320$ trap nights during 15 trapping sessions). For each trapping session, we used 20 camouflaged Tomahawk traps (20 × 20 × 70 cm) set at approximately 200 m intervals (Bonesi and Macdonald 2004). Traps were baited with fresh fish and placed at a maximal distance of 10 m from the water's edge (Dunstone 1993). Traps along rivers were placed on one side of the river facing downstream, because presumably animals are more likely to use an overland route when moving up river (Gerell 1970). Traps were checked every morning. Captured mink were lightly anaesthetized with Ketamine (Drag Pharma Chile), weighed, sexed, measured and marked with AEG-ID passive integrated transponders that were injected directly under the skin. Mink were classified as juveniles or adults by their body weight, wear of teeth, juvenile shaped facial characteristics and presence of grey hair following Halliwell and Macdonald (1996). However, only post-mortem determination of age is an objective method (Dunstone 1993). The animals were released after full recovery from anaesthesia at the spot where captured.

Sign surveys

Capture-mark-recapture is a time-consuming method of estimating mink abundance and when recapture rates are low sound analysis of the data is limited (Bonesi and Macdonald 2004). Therefore, we complemented abundance estimates with an indirect method, namely sign surveys. Sign surveys are an appropriate way to efficiently estimate distribution and relative abundance of carnivores, which are often cryptic, nocturnal and may have large home ranges (review in Wilson and Delahay 2001; Gruber et al. 2008). For American mink, sign surveys are recommended when carried out in comparable seasons and when the aim is to monitor mink populations over large areas (Bonesi and Macdonald 2004). Indirect survey methods based on scats, however, can be seriously problematic due to the possibility of misidentification (Davison et al. 2002; Harrington et al. 2008). For the identification of mink scats and tracks on Navarino Island, this does not represent a challenge as the mammal assemblage does not include other mustelids (Anderson et al. 2006a); and sea otters (*Lontra felina*) are restricted to the Wollaston and Cape Horn Islands (Rozzi et al. 2006) being only rarely seen in the southern parts of Navarino Island (Yaghan indigenous people, personal communication). We searched for scats and tracks in four different semi-aquatic habitats ($n = 68$ sites): along coastal shores ($n = 15$), river banks ($n = 9$), lake margins ($n = 31$) and pond margins ($n = 13$) (Fig. 1) 124 times (surveys were repeated during different seasons, see below). In the absence of depth measurements, we classified wetland habitats into lakes when the perimeter was >1 km and into ponds when ≤1 km. Coastal sites comprised 1.8–4 km shoreline (median 4 km), rivers 1.4–4 km (median 4 km), lakes 1.1–5.8 km (median 2 km) and ponds 0.3–1 km (median 0.8 km). The majority of sites ($n = 59$) were located in the northern part of Navarino Island. The southern part of the island (>54°06′S, beneath Lake Windhond) was accessed by boat. Lakes and ponds in the interior of the island (including Lake Windhond) were mainly reached through the three trekking trails on the island. Study sites were divided into 200 m contiguous sections (Bonesi and Macdonald 2004). Shores and river banks (one bank only) were surveyed up to 5 m from the water's edge. A team of three trained surveyors conducted the surveys. We repeated sign surveys during different seasons at ten lakes each (spring, summer, autumn, winter, 2006), and at twelve coastal sites (autumn 2005: $n = 3$, spring 2005: $n = 7$, spring 2006: $n = 9$, summer 2006: $n = 7$, summer 2007: $n = 12$). For rivers and ponds, we relied on summer surveys (2006/2007) only.

Systematic errors can arise from the probability of signs being detected by the surveyor in different habitats exhibiting either a consistent positive or negative influence on the results (Thompson et al. 1998; Bonesi and Macdonald 2004). These errors should be particularly avoided when coinciding with a research question. In our case we aimed to investigate habitat preferences of mink on Navarino Island. It is apparent that detectability of scats might correlate with the type of habitat. In order to quantify whether this represented a source of systematic error, we used ‘artificial scats’ in a small exemplary experimental design. We distinguished between coastal sites characterized by steep shorelines, cliffs and rocks (*rocky outcrop*, $n = 6$ sites) and sites characterized by a basically flat shore and presence of pebbles, sand or mud (*beaches*, $n = 6$ sites). We placed 50 artificial scats (pack-twine, 1 cm diameter, 10 cm long) at possible marking places in each 1.5 km rocky outcrop coastal habitat ($n = 6$) and along beaches ($n = 6$). A second trained surveyor then searched for artificial scats in the 12 study sites, annotating the number of scats found. Our results indicate that the surveyor detected scats independently of the coast type (Mann–Whitney Test: $U = 7.5$, $P = 0.1$).

During each sign survey, 9–15 habitat variables were recorded depending on the semi-aquatic habitat type (Appendix). Variables for all sites concerned habitat type, vegetation cover of three different strata, distance to the forest, coarseness of the shoreline and incline, presence of dogs and humans. For rivers, lakes and ponds, we also recorded the influence of beavers, elevation and distance to the coast; for rivers we additionally estimated water depth, water flow and river width.

Statistics

Relative abundance of mink was measured as the number of sections (200 m) containing signs per survey. During our sign surveys we searched for tracks and scats and recorded sightings. As the detection of tracks is rather dependent on ground composition and weather conditions (Wilson and Delahay 2001), we used Spearman's rank correlation coefficients to check whether the combined set of signs was different from surveys that relied on scats only. Seasonal differences were tested with Kruskal–Wallis rank sum tests and Spearman's rank correlations.

The effect of small scale habitat features on mink abundance was examined using generalized linear models (GLMs). Our response variable was mink abundance (presence/absence of scats in 200 m sections). This was a binary variable and thus presumed to follow a binomial distributional family with a logit link function in the GLMs. Based on literature, we designed three different candidate models (Table 1) guided by the following hypotheses: (1) mink should favour a high vegetation cover (Previtali et al. 1998; Yamaguchi et al. 2003); (2) they should be most commonly associated with heterogeneous rocky shorelines (Allen 1984; Bonesi et al. 2000; Moore et al. 2003; Fasola et al. 2009); (3) beavers should improve the habitat quality for mink (Żurowski and Kammler 1987; Sidorovich et al. 1996); and (4) the more suitable riverine habitats should be shallow, slow-moving bodies of water (Dunstone 1993). The first and second hypothesis could be tested

Table 1 Candidate models for predicting mink habitat preferences corresponding to different hypotheses

Candidate models GLM	Data set	n	Variables of fitted models
Null model			Intercept only
M1	Coast, lakes, rivers, ponds	611	HABITAT + STRATA1 + DIST_FOREST + COARSE + INCLINE + DOGS
M2	Lakes, rivers, ponds	333	HABITAT + COARSE + INCLINE + ELEVATION + DIST_COAST + INFL_BEAVERS
M3	Rivers	139	HABITAT + COARSE + INCLINE + INFL_BEAVERS + DEPTH + FLOW + WIDTH

An explanation of the habitat variables is given in Appendix by the full data set of the four semi-aquatic habitats, the third one excluded coasts from our data set, and the fourth hypothesis concerned merely river characteristics. In order to build models containing as few parameters as possible (Crawley 2007), in the second and third models (M2, M3) we only included variables that had proved significant in the previous models (M1, M2). We primarily performed principal component analysis (PCA) as an explanatory tool to identify suitable variables. Thus, we excluded variables with a strong correlation (Spearman's rho > 0.6) (Fielding and Haworth 1995) and less biological relevance for mink. To perform the PCA we used the ade4 package rewritten for the R environment (R Development Core Team 2008). The model selection procedure was based on Akaike's Information Criterion (AIC), a statistical method that rewards parsimony by penalizing the maximum likelihood for the number of model parameters (Akaike 1973). The predictive model accuracy was assessed by constructing relative operating characteristics (ROC) curves (e.g. Mason and Graham 2002). The area under the ROC curve is expressed as an Area Under the ROC Curve (AUC) value that characterizes the quality of a forecast system by describing the system's ability to anticipate correctly the occurrence or non-occurrence of pre-defined events. When the model (forecast system) has some accuracy, the AUC value will exceed 0.5. For the interpretation of AUC values, we can use the following categories (Hosmer and Lemeshow 2000): $0.7 \leq \text{AUC} < 0.8 = \text{acceptable}$; $0.8 \leq \text{AUC} < 0.9 = \text{excellent}$; $0.9 \leq \text{AUC} \leq 1.0 = \text{outstanding}$. Analyses of variance tables were used to present significant model terms.

To evaluate which of the categories of the significant habitat variables was preferred by mink, we calculated the ratio of mink presence and the availability of the habitat feature. For example, the availability of shrubland (the 'simple' habitat type, see Appendix) was 209 out of 611 cases with a mink presence of 59 out of 124 cases (presence only) yielding a ratio of 1.39. We tested for significance with two-sample tests for equality of proportions with Bonferroni corrections. All statistical analyses were performed using the R statistical software 2.7.1. (R Development Core Team 2008); *P*-values were considered as significant when <0.05.

Results

In total, 21 individual mink were trapped 25 times during 15 trapping sessions ($N = 1{,}320$ trap nights) in six study sites. All captured mink were of the normal wild type, i.e. dark-brown in colour. Adults ($n = 10$) were mainly captured during autumn ($n = 8$) and juveniles ($n = 11$) during summer ($n = 8$). The overall sex ratio of the catches was 2.5 males ($n = 15$) to one female ($n = 6$). Mean weight of females was 600 ± 110 g (range 500–700 g), and of males 970 ± 210 g (500–1,300 g).

Trappability in rocky coastal sites was highest during autumn and summer yielding a median relative mink abundance of 0.75 individuals/km (ranges 0.5–1.25 for autumn, and 0.0–1.25 for summer) (Table 2). Although we set traps at rivers during autumn when mink were frequently trapped in coastal sites, the relative abundance of mink was low with 0.0 and 0.25 individuals/km along riparian shores. Due to the presence of raptors (e.g. *Milvago chimango*, *Caracara plancus*) in our study sites, traps were frequently disturbed (e.g. bait missing, closed door). When excluding disturbed traps from our analysis, median trapping success was 4.0 (2.17–6.25) captures/100 trap nights compared to 3.0 (2.0–5.0) (all traps set) at coastal habitat during autumn, and 3.9 (0.0–8.47) compared to 3.75 (0.0–6.25) at coastal habitats in summer. For rivers, the exclusion of disturbed traps made no big difference (0.52, range 0.0–1.06 with disturbed traps excluded versus 0.5, range 0.0–1.0 with all traps set).

Table 2 Results of live trapping in three sites of costal habitat with seasonal repetitions, and three sites along river banks during autumn/winter in the northern part of Navarino Island ($n = 15$ trapping sessions)

Site	Total length (km)	Habitat	Year	Season	Month	Trap nights	Males	Females	Recaptures	Mink/km
ROB	4	Coast	2005	Autumn	April	100	3	0	1	0.75
GUE	4	Coast	2005	Autumn	April	100	4	1	2	1.25
MEJ	4	Coast	2005	Autumn	May	100	1	1	1	0.5
ROB	4	River	2005	Autumn	May	100	1	0	0	0.25
GUE	4	River	2005	Autumn	June	100	0	0	0	0
MEJ	4	River	2005	Winter	July	100	0	0	0	0
ROB	4	Coast	2005	Spring	December	80	0	0	0	0
GUE	4	Coast	2005	Spring	December	80	0	0	0	0
MEJ	4	Coast	2005	Spring	December	80	1	0	0	0.25
ROB	4	Coast	2007	Summer	March	80	0	0	0	0
GUE	4	Coast	2007	Summer	February	80	2	1	0*	0.75
MEJ	4	Coast	2007	Summer	February	80	3	2	0*	1.25
ROB	4	Coast	2007	Winter	August	80	0	0	0	0
GUE	4	Coast	2007	Winter	September	80	0	1	0	0.25
MEJ	4	Coast	2007	Winter	September	80	0	0	0	0
Total	60					1,320	15	6	4	

ROB Robalo, *GUE* Guerrico, *MEJ* Mejillones

* Four dead juvenile mink in traps lowered the recapture probability (GUE: 1, MEJ: 3). Death probably occurred due to adverse climatic conditions in combination with a low body weight

Seventy-nine per cent of all surveys conducted once during summer (2006/2007, $n = 68$) contained signs of mink ($n = 403$) (Fig. 1). The results show that mink have been able to colonize the whole island starting from their theoretical arrival point in the northern part of the island and reaching the very South of Navarino. From the eastern part of the island where we lacked sign surveys, we included a record of a captured mink in the fishing town of Puerto Toro. Scats were also found at high elevations (577 m) at the lower edge of the high-Andean zone. Surveys relying on scats and surveys recording scats, tracks and sightings highly correlated with each other, for coastal habitat ($S = 272.7$, $n = 41$, $P < 0.001$, rho = 0.98), lake shores ($S = 2{,}413.4$, $n = 61$, $P < 0.001$, rho = 0.94) and pond margins ($S = 0$, $n = 13$, $P < 0.001$, rho = 1). However, for rivers it made a difference whether surveyors were collecting only scats or additionally tracks ($S = 73.9$, $n = 9$, $P = 0.31$, rho = 0.38). Mink sightings occurred only five times in 124 surveys and can therefore be neglected. For further analyses, we refer to scat surveys only, assuming that this method is more reliable for comparisons between different types of semi-aquatic habitats (Bonesi and Macdonald 2004).

Relative abundance of mink did not differ significantly between the four semi-aquatic habitats (Kruskal–Wallis rank sum test, $n = 68$, $\chi^2 = 1.06$, df = 3, $P = 0.79$) (Fig. 2). For coastal habitats, we estimated a median of 10% sections (200 m) with scats (1st Qu. = 7.5, 3rd Qu. = 40), for rivers 14.3% (5.6–25), for lakes 15.4% (3.3–25) and for ponds 20% (0–50). Ponds and coastal sites showed an especially high variance between different sites of the same semi-aquatic habitat ranging from min. 0 to max. 75% sections found positive by searching for scats.

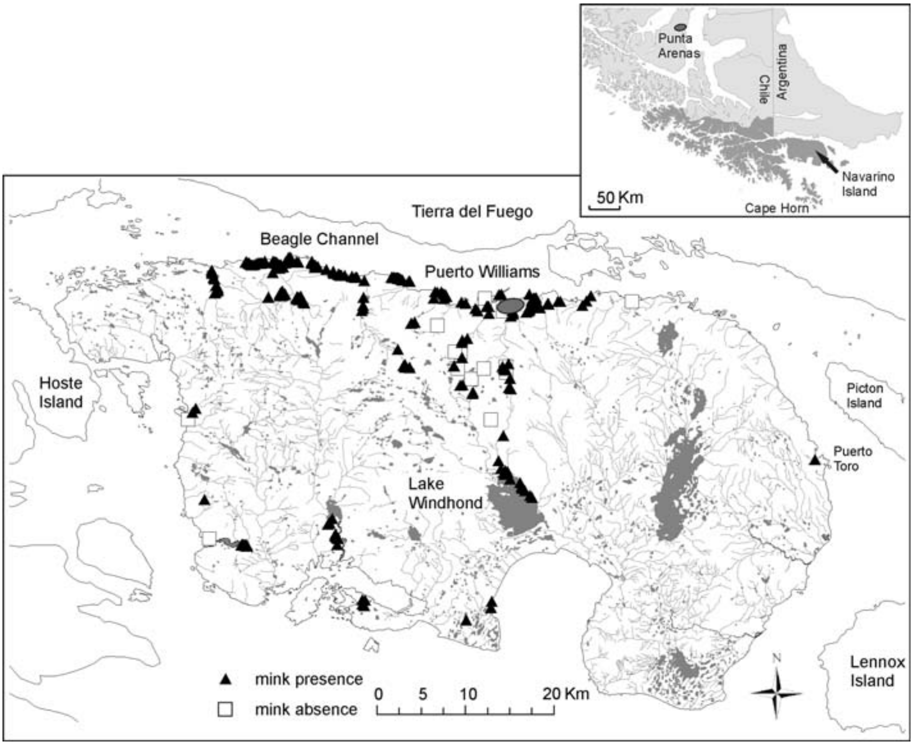
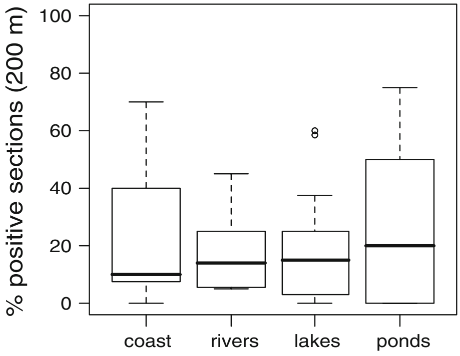


Fig. 1 Sign surveys of mink (scats, tracks, sightings, $n = 68$) in different semi-aquatic habitats (coast, rivers, lakes, ponds) during summer 2006/2007 on Navarino Island. Navarino is located within the Cape Horn Biosphere Reserve (54–56°S, shaded in *dark grey*, *top right*). Mink presence refers to each sign found during the sign surveys ($n = 403$, overlapping triangles due to enlarged size). Minimum transect length was 300 m, maximum transect length 5.8 km. Mink presence in the eastern part of the island refers to a captured mink in Puerto Toro

Fig. 2 Percentage of sections (200 m) with mink scats (summer 2006/2007) in semi-aquatic habitats: coastal ($n = 15$), riparian ($n = 9$), lake shores ($n = 31$) and pond margins ($n = 13$). The boxplot indicates the sample minimum, the lower quartile, the median, the upper quartile and the sample maximum



We conducted repetitive sign surveys at 10 lakes located in the northern part of Navarino Island and checked whether the proportion of sections with signs of mink varied between seasons. No significant differences between seasons were found (Kruskal–Wallis rank sum test, $n = 40$, $\chi^2 = 1.14$, df $= 3$, $P = 0.77$) (Fig. 3). Neither did we find evidence

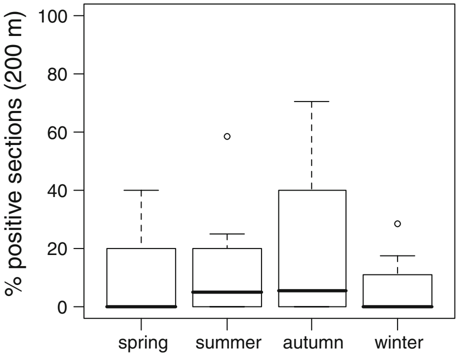


Fig. 3 Percentage of sections (200 m) with mink scats at lakes ($n = 10$) for each season in 2006. The boxplot indicates the sample minimum, the lower quartile, the median, the upper quartile and the sample maximum

for variation between spring and summer surveys at nine coastal sites (North) during 2006/07 (Spearman correlations, $S = 38.0$, $n = 9$, $P = 0.04$, rho = 0.68), nor during 2005/2006, where correlations were not significant, but showed a positive trend (S = 14.8, $n = 7$, $P = 0.06$, rho = 0.74).

In order to evaluate small-scale habitat preferences of mink, we built a first Model M1 (Table 1), which covered summer presence/absence data ($n = 611$) from all semi-aquatic habitats studied (coast, rivers, lakes, ponds). The first axis of the PCA explained 30.2% of the variance, the second axis 19.9%. Variables with high loadings (>|0.7|) on the first principal component were habitat type (0.85), vegetation cover of strata two (0.79) and three (0.76), and distance to forest (−0.71), whereas dogs (0.87) and humans (0.86) had high loadings on the second component. We excluded two variables from the model procedure: humans correlating with dogs, which are possible direct predators and therefore the more proximate variable for mink, and vegetation cover of strata two and three correlating with habitat, the biologically more relevant variable for mink as prey availability depends on the type of habitat. Model accuracy of the most parsimonious model was at the limit of acceptance (AUC = 0.65). It included three significant variables: incline, habitat and coarseness of shoreline (Table 3, M1). Dogs, distance to forest and vegetation cover of strata one (0–1 m) did not have a significant effect on mink abundance and were removed by stepwise AIC selection. Thus, among the habitat types available, the simple habitat (shrubs, grasses, but no mature trees) was preferred. Two-sample tests for equality of proportions (all df = 1) proved this to be significant (simple versus uniform: $\chi^2 = 6.24$, $P < 0.05$; simple versus complex: $\chi^2 = 9.56$, $P < 0.01$). Mink favoured steeper shorelines, a difference we found significant (flat versus medium: $\chi^2 = 10.73$, $P < 0.01$; flat versus steep: $\chi^2 = 7.28$, $P < 0.05$). Mink also preferred shorelines characterized by a higher degree of coarseness, i.e. a higher percentage of cliffs and rocks, in contrast to beaches with sand, mud or vegetation as the main substrate ($\chi^2 = 5.34$, $P = 0.02$) (Fig. 4).

Model M2 (Table 1) was fitted with the three significant variables of the first model and the three additional variables: influence of beavers, elevation, and distance to the coast, valid for rivers, lakes and ponds only ($n = 333$ cases). The first principal component explained 39.1% of the variance, the second 24.0%. High loading variables for the first component were elevation (−0.85) and distance to coast (−0.78). All variables were entered in this model. The best model reached an AUC value of 0.67. Significant variables of this model were influence of beavers and elevation (Table 3, M2). The significant variables from M1 did not contribute to explain the presence of mink in this scenario; neither was distance to coast of importance. Interestingly, mink preferred habitats without beaver influence. Thus, mink presence was significantly higher where beavers were absent (none versus medium

Table 3 Analysis of variance table for the effects of habitat variables on the abundance of mink modelling summer presence/absence data for three candidate models M1, M2 and M3 (Table 1)

Model	df	Deviance Resid.	df	Resid. Dev	$P(>\|Chi\|)$
M1					
Intercept			610	616.45	
INCLINE	1	9.66	609	606.79	0.002**
HABITAT	3	12.55	606	594.23	0.01**
COARSE	1	4.45	605	589.79	0.03*
M2					
Intercept			332	295.20	
DIST_COAST	1	0.01	331	295.19	0.92
ELEVATION	1	4.47	330	290.72	0.04*
INFL_BEAVERS	1	12.84	329	277.88	<0.001***
M3					
Intercept			138	121.43	
INFL_BEAVERS	1	11.76	137	109.67	0.001***
WIDTH	1	1.03	136	108.64	0.29
DEPTH	1	5.37	135	103.27	0.02*

* Significant at 0.05, ** significant at 0.01, *** significant at 0.001

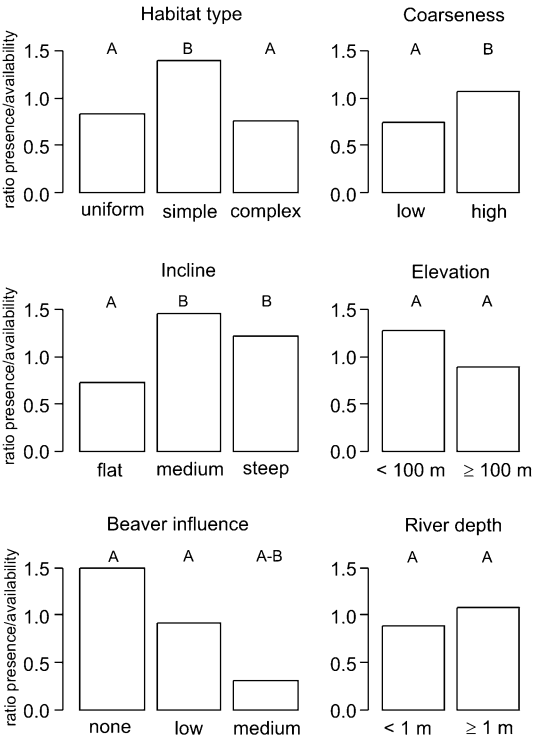


Fig. 4 Preferences of habitat categories using significant habitat features according to models M1, M2, and M3. For each graph letters above the bars indicate significant differences ($P < 0.05$ with 2-sample tests for equality of proportions) of the ratios of presence of mink versus the habitat availability. Categories of coarseness are pooled; low coarseness refers to index values of 1–3 and high coarseness to index values of 4–6. The categories 'bare' habitat type and 'high' beaver influence had to be excluded from this analysis due to low sample sizes ($\leq$5% available)

influence: $\chi^2 = 8.61$, $P < 0.05$). Mink presence was slightly greater in habitats at lower elevations (<100 m), but not significantly ($\chi^2 = 1.52$, $P = 0.2$) (Fig. 4).

The last model M3 was built for river data only ($n = 139$) containing the significant variables of models M1 and M2, and the additional variables river depth, river flow and river width. The first axis of the PCA explained 34.1% of the variance, the second component 17.6%. Variables with high loadings on the first axis were coarseness (−0.85), incline (−0.77) and river flow (−0.77); on the second axis river depth (−0.71). As neither of the variables strongly correlated with each other, we included all variables in the model (Table 1, M3). The accuracy of the most parsimonious model was very acceptable (AUC = 0.79). M3 again revealed the effect of beaver influence on mink abundance, and river depth as one of the new river variables (Table 3, M3). Although mink tended to prefer deeper rivers (Fig. 4), this difference was not significant ($\chi^2 = 0.06$, $P = 0.81$).

Discussion

Abundance estimates

Our results show that mink have colonized a high proportion of semi-aquatic habitats throughout the island only 10 years after the first mink was recorded. We used trapping and sign surveys to answer two questions: what type of habitats do mink prefer and how many of them are there? Our data on relative mink abundances along rocky coastal shores (median: 0.75 mink/km for both, autumn and summer, range 0.0–1.25) were lower than studies in Canada, Scotland or Argentina revealed (summary in Table 4). Repetitive trapping also lowered the abundance of our preliminary trapping data (0.79–1.32 mink/km at coasts, and 0.26 at rivers) given in Anderson et al. (2006a). Our low relative abundances of maximum 0.25 mink/km along rivers trapped in autumn somehow seem in line with the generally lower densities of mink along rivers and inland lakes in comparison to coastal shores measured in other regions. Here, abundances usually did not exceed one mink/km (Table 4), with some exceptions (Smal 1991). As Dunstone and Ireland (1989) argued, the density of mink appears to vary with the productivity of the habitat, i.e. mink occupy larger home ranges when associated with prey-impoverished habitats. This was shown for oligotrophic rivers where home range lengths for male mink reached 2.5 km, in comparison to 1.5 km in coastal habitat (Dunstone and Birks 1985). Our estimations of relative mink abundances based on sign surveys, however, did not reveal significant differences between rivers and coastline; on the contrary, mink were shown to be equally present in different semi-aquatic habitat types. On the one hand, these contradicting results may be based on the low trapping efforts performed at rivers. On the other hand, studies have shown that the proportion of sections with mink signs were only loosely correlated with mink abundance estimated from live trapping (Bonesi and Macdonald 2004; Harrington et al. 2008). Bonesi and Macdonald (2004) recommended that sign surveys at low mink densities might be a better way to estimate the relative density of mink than trapping. In this context, sign surveys might also control seasonal differences as our seasonal surveys at 10 lakes were not significantly different in terms of percentages of positive sections (also shown by Harrington et al. 2008). Yet, trapping success in the same study sites was highly variable and was dependent on seasonal factors, i.e. reproduction. Maximum abundance was measured during autumn, the mating season, when adult males prevailed, and during late summer, when juveniles predominated (see also Dunstone 1993; Bartoszewicz and Zalewski 2003; Moore et al. 2003). A male bias as in our case is commonly found in mustelid

Table 4 Summary of relative mink densities measured as individuals/km by trapping in different semi-aquatic habitats in invaded and native (Canada, USA) areas sorted by ascending mink densities

Mink density (mink/km)	Habitat	Study site	Source
1.35–2.27	Coast	Canada	Hatler (1976)
2.0	Coast	Scotland	Birks and Dunstone (1991)
1.5	Coast	Argentina	Previtali et al. (1998)
1.1	Coast	Scotland	Moore et al. (2003)
0.75	Coast	Chile	Present work
0.48–1.37	Rivers	Ireland	Smal (1991)
0.57–0.92	Swamps, marshes	USA	Mitchell (1961)
0.1–0.7	Rivers	England	Halliwell and Macdonald (1996)
0.26–0.53	Rivers	England	Birks and Dunstone (1991)
0.15–0.51	Rivers	England	Harrington et al. (2009)
0.46	Rivers	Belarus	Sidorovich et al. (1996)
0.35–0.43	Lakes	Ireland	Smal (1991)
0.17–0.33	Rivers	England	Bonesi and Macdonald (2004)
0.18	Lakes	Scotland	Moore et al. (2003)

trapping studies (Craik 2008). This bias probably arises from the greater home range size (Yamaguchi et al. 2003 measured a male's home range to be 1.73 times larger than that of a female) and increased mobility of males, especially during the mating period (Dunstone 1993). In general, live trapping of American mink likely underestimates population densities as authors report trap avoidance (e.g. Smal 1991; Yamaguchi et al. 2003). In conclusion, considering the low abundances in comparison to other areas (see Table 4), the lack of predators and competitors, and presuming high habitat productivity, our results suggest that the population of mink on Navarino Island seemed not to be saturated yet.

Habitat requirements

Patterns of habitat use of mink have been mainly related to the availability and distribution of prey and dens, to the risk of predation or to the interference from competitors (e.g. Halliwell and Macdonald 1996; Bonesi et al. 2000; McDonald 2002; Yamaguchi et al. 2003; Bonesi et al. 2006). The absence of potential predators or competitors of mink on Navarino Island should influence its habitat preferences in terms of being less specific. Our three habitat models had forecast quality, but the AUC-values of 0.65 (model M1) and 0.67 (model M2) were at the limits of acceptability ($0.7 \leq \text{AUC} < 0.8 =$ acceptable following Hosmer and Lemeshow 2000). This can be interpreted as the generalist habitat preferences we assumed. When an invasive species experiences release from natural enemies in its new environment, it can undergo a niche-shift (e.g. Fitzpatrick et al. 2007), i.e. it can extend its realized niche toward its fundamental niche (Hutchinson 1957). The fundamental niche refers to the genetically and physiologically determined requirements of a species to maintain a positive population growth rate, while the realized niche is a portion of the fundamental niche that includes the constraints arising from biotic interactions (review in Pearman et al. 2008).

As assumed in hypothesis (1), mink avoided open habitats and instead used shrubland, a pattern previously described (Allen 1984; Previtali et al. 1998; Yamaguchi et al. 2003). This can be explained by the higher availability of dens and hiding places provided in heterogeneous landscapes (Dunstone 1993; Halliwell and Macdonald 1996). The abundance and diversity of prey also explains this pattern. Small mammals in Chile have been shown to use shrub microhabitat based on its profitability in terms of seed and arthropod availability (Simonetti 1989). The yellow nosed grass mouse (*Abrothrix xanthorhinus*), one of the most important mammal prey of mink together with introduced muskrats (*Ondatra zibethicus*) (Schüttler et al. 2008), was predominantly associated with dense shrubland (*Berberis buxifolia*, *Chiliotrichum diffusum*, *Pernettya mucronata*) or reed (*Marsippospermum grandiflorum*) (Gañan et al. unpublished data). Muskrat foods and feeding habits vary widely with habitat and season (Perry 1982) and remain to be investigated in our study region in order to better understand their importance for mink.

Following hypothesis (2), we expected mink to be more abundant in rocky areas and boulder fields of coastlines, in contrast to beaches with pebbles or sand as the main substrate. Our results confirmed this hypothesis. As shown in previous studies, rock pools and boulder fields represented important feeding areas for mink in Alaska and Scotland (Ben-David et al. 1996; Bonesi et al. 2000). Although mink on Navarino Island consumed significantly more fish in marine habitats than along lake shores and rivers (Schüttler et al. 2008), fish as a prey group (with 17.4% dominant items in the diet) was still less important than were birds or mammals (40.7 and 37.6%, respectively). Therefore, we suggest that the availability of birds and their offspring might also play a role in mink's preferences for rocky outcrops. Raya Rey and Schiavini (2002) have shown that the presence of kelp beds presented the highest abundance of seabird species breeding in the Beagle Channel, coinciding with rocky coasts, where kelp forests typically monopolize (Steinberg and Kendrick 1999). The fact that mink were significantly associated with steeper shorelines might coincide with the fact that cliffs or rocks normally constitute steeper shorelines, though our indices for coarseness and incline revealed only a moderate positive correlation (Spearman's rho = 0.43). Again, the association of bird species with steep shorelines might provide an explanation, as is the case for some species of Pelecaniformes (a bird order found in the diet of mink on Navarino Island, Schüttler et al. 2008), like for example rock or imperial shags (*Phalacrocorax magellanicus*, *P. atriceps*) (Couve and Vidal 2003).

Contradictory to hypothesis (3), we found mink to be more abundant in habitats without beaver presence or with low beaver influence, i.e. signs of beaver activity, but no dams. As in other invaded areas (Belarus, Poland) mink were shown to profit from the ice-free access to water around beaver lodges and even to use them as dens (Żurowski and Kammler 1987; Sidorovich et al. 1996), it is questionable why mink should not do this on Navarino Island. As beavers' impacts on habitat, community and ecosystem variables have been shown to have similar directions in their native range and in South America (Anderson et al. 2009), diverging conditions seem not to be a reason. One explanation might be the methodological challenge of surveying for mink signs around the beaver lodges where plenty of fallen trunks and the muddy substrate might decrease the detectability of scats. Further studies including a mix of methodologies (e.g. search for mink dens, trapping, detecting probability experiments etc.) are needed to concentrate on this relationship in order to further comment on the invasional meltdown hypothesis (Simberloff and Von Holle 1999; Simberloff 2006). The planned beaver eradication campaign (Choi 2008) might represent a good possibility to design a study on mink abundance in beaver removal areas.

Regarding rivers, we predicted that suitable habitats would be large, slow-moving rivers, hypothesis (4). Although river depth was a significant variable in our habitat model, mink preference for deeper rivers had no significant relevance when tested with univariate statistics. Neither did river width and river flow play a role in the choice of riverine habitats. Although Dunstone (1993) suggested that mink's adaptation suit it for hunting in slow-moving rivers, Strachan and Jefferies (1993) reported its preference for fast-flowing rocky water courses. In Belarus, mink occurs in higher densities on larger, deeper rivers than on small rivers less than one metre deep (Sidorovich et al. 1996). Also in Argentinian Patagonia, mink favoured deeper water because of the availability of crustacean prey (Previtali et al. 1998). Thus habitat requirements of mink for rivers seem less consistent, while prey availability might be of special importance (see also Clode et al. 1995).

Conclusions for management

American mink are currently expanding their range in southern South America (Jaksic et al. 2002), but the local conditions determining their dispersal are virtually unknown. Our data on habitat use of mink in southern Chile together with data available from Argentinean Patagonia (Previtali et al. 1998; Fasola et al. 2009) can be used to model future distribution maps. Knowledge of where mink are and where they could be is a first step towards an evaluation of its management. Despite the fact that studies on the impact of mink on the native austral fauna in South America are still much needed, decision-making on its management also raise practical questions. Is a control program cost-effective and feasible? Control programs in Belarus and Britain have shown that trapping can effectively reduce the local abundance of mink (Sidorovich and Polozov 2002; Moore et al. 2003; Harrington et al. 2009), and successful eradications have been carried out on relatively small islands (<1,500 km^2) remote from reinvasion in Finland, Estonia and the Western Isles in the UK (reviewed by Bonesi and Palazon 2007). Given the limited knowledge of distribution and impacts of this invasive species in southern South America and the infrastructural conditions met in this region, answers cannot quickly be provided. At the moment, the management of mink populations in South America rather requires selective decisions depending on the specific local conditions.

For the case of Navarino Island, we can say that mink have been able to colonize the entire island (>2,500 km^2) over a decade. They were found in all types of semi-aquatic habitats, as well as under extreme conditions like in the high-Andean habitat. Our trapping data indicate that carrying capacity might not yet have been reached; waiting for a response can thus make control efforts more difficult in the future (e.g. Simberloff 2003). We were able to predict that mink should live at higher abundances in coastal areas with heterogeneous shores. Thus, for the design of a management plan, steep rocky coasts might represent priority sites with intensive trapping efforts. At the same time those habitats that harbour most vulnerable bird species (Schüttler et al. 2009) will be protected. The possibility of immigration from adjacent islands and landscape barriers like mountain ranges (Zalewski et al. 2009) should be taken into consideration. Dispersal might be measured in areas where control is going on in separated zones (see Sleeman et al. 2009). If management is planned in an integrative manner, i.e. considering various introduced mammal species (Soto and Cabello 2007), then the relationships between those species must be considered. As shown here, the removal of beavers will probably not necessarily result in a decrease of mink habitat quality. Ideally, control efforts will follow a conjoint strategy together with scientists and the local community in order to guarantee the most accepted and cost-effective management implications.

Acknowledgements We thank all the field assistants and in particular Melisa Gañan, José Llaipén, Julia and Germán Gonzáles, and Francesca Pischedda. We are grateful to Fidel and Patricio Quelín for their hospitality during field work. Reinhard Klenke supported our project with scientific advice. We thank Tina Heger, who made valuable comments on the manuscript. We also appreciate the advice of two anonymous reviewers to improve an earlier version of this paper. This research was conducted under trapping permit N° 1,192 (16 March 2005) issued by the Chilean Agriculture and Livestock Service (SAG). Concar S.A. kindly sponsored the fish bait for our traps. Financial support was provided by the German Academic Exchange Service (DAAD), the Chilean Millennium Institute of Ecology and Biodiversity Contract (ICM, PO2-051-F), the Magallanes University, the Omora Foundation, and by the German-Chilean Research Project BIOKONCHIL funded by the German Ministry of Education and Research (FKZ 01LM0208).

Appendix

Table 5 Habitat variables recorded in all sites (coast, rivers, lakes, ponds) and in selected habitats

Habitat variables	Recorded in	Description and *categories*
HABITAT	All sites	Predominant habitat type 10 × 5 m^2: *1. Bare: earth/rock* *2. Uniform: pasture, peatland, wetland* *3. Simple: shrubs, grasses, but no mature trees* *4. Complex: evergreen, mixed, deciduous forest*
STRATA 1 STRATA 2 STRATA 3	All sites	Vegetation cover for vegetation strata 1,2 and 3 (<1 m, 1–5 m, >5 m): *<20%, 20–40%, >40%*
DIST_FOREST	All sites	Distance to the forest: *<10 m, 10–50 m, >50–100 m, >100 m*
COARSE	All sites	Percentage of cliff, rock, pebbles, sand, mud, vegetation within 10 × 1 m^2 of the shoreline, merged into one continuous index of coarseness (range 1–6)
INCLINE	All sites	Incline of shore within 10 m from water shed: *Flat, medium, steep*
DOGS	All sites	Presence/absence of dogs
HUMANS	All sites	Presence/absence of human settlement within 500 m
INFL_BEAVERS	Rivers Lakes Ponds	Influence of beavers within 100 m *1. None (absence of beavers)* *2. Low (signs of beaver activity)* *3. Medium (signs of beaver activity and destroyed dams)* *4. Strong (signs of beaver activity and intact dams)*
ELEVATION	Rivers, Lakes, Ponds	Elevation above sea level [m] measured with GPS
DIST_COAST	Rivers, Lakes, Ponds	Direct distance to coast [m] measured with GPS
DEPTH	Rivers	Water depth at 1 m from river bank *<1 m, ≥1 m*
FLOW	Rivers	Water flow of rivers *None, low, medium, strong*
WIDTH	Rivers	River width *0.5–2 m, >2–5 m, >5 m*

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