



# New records of invasive mammals from the sub-Antarctic Cape Horn Archipelago

Elke Schüttler<sup>1</sup> · Ramiro D. Crego<sup>1,2</sup> · Lorena Saavedra-Aracena<sup>1</sup> · Eduardo A. Silva-Rodríguez<sup>3</sup> · Ricardo Rozzi<sup>1,2,4,5</sup> · Nicolás Soto<sup>6</sup> · Jaime E. Jiménez<sup>1,4</sup>

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## Abstract

The southernmost archipelago of the Americas is dominated by invasive mammals that outnumber their native counterparts. Despite the relatively low ability of most invasive mammals to cross cold sea water channels, invaders are apparently colonizing new islands. Our objective was to provide an assessment of the expansion of invasive mammals within these sub-Antarctic ecosystems, determine whether human-mediated movement of invasive species is a plausible dispersal mechanism, and identify areas likely to be colonized in the near future. We report a decade of fieldwork (2006–2017) in 44 sites on 13 islands within the Cape Horn Biosphere Reserve including opportunistic and systematic camera trapping, carnivore diet, questionnaires, small mammal trapping, and walks/transects. We found new records of invasive mammals on seven islands, particularly for American mink (*Neovison vison*) and American beaver (*Castor canadensis*). Interviews with fishermen showed that mink, dogs (*Canis familiaris*), and small rodents are likely passengers in vessels. Finally, species distribution models revealed that the putative invasive-free Cape Horn National Park (55°S) is suitable for several invasive species, suggesting a high risk of invasion if species are introduced. We conclude that it is urgent to implement barriers to dispersal to prevent further invasion. In the case of dogs and cats (*Felis catus*), the first step should be control actions that target pet owners. Finally, we highlight the need of systematic, long-term biodiversity monitoring and citizen science in the Cape Horn Archipelago and common conservation guidelines for the terrestrial sub-Antarctic ecosystems.

**Keywords** Assisted dispersal · Conservation · Free-ranging domestic animals · Islands · Monitoring · Range expansion · Vertebrates

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✉ Elke Schüttler  
elke.schuttler@umag.cl

<sup>1</sup> Sub-Antarctic Biocultural Conservation Program, Universidad de Magallanes, Teniente Muñoz 166, Puerto Williams, Región de Magallanes y Antártica Chilena, Chile

<sup>2</sup> Department of Ecological Sciences, Institute of Ecology and Biodiversity, Casilla 653, Santiago, Chile

<sup>3</sup> Facultad de Ciencias Forestales y Recursos Naturales, Instituto de Conservación, Biodiversidad y Territorio, Universidad Austral de Chile, Campus Isla Teja, Casilla 567, Valdivia, Chile

## Introduction

In the era of globalization, invasive species are increasing in abundance and diversity (Seebens et al. 2017), mainly as a consequence of human trade and transport (Hulme 2009). Currently, almost all ecosystems, even in remote places,

<sup>4</sup> Department of Biological Sciences, University of North Texas, 1511W Sycamore, Denton, TX, USA

<sup>5</sup> Department of Philosophy and Religion, University of North Texas, 1155 Union Circle, Denton, TX, USA

<sup>6</sup> Agriculture and Livestock Service, Chilean Ministry of Agriculture, Avenida Bulnes, 0309 Punta Arenas, Chile

include multiple invasive species (Hobbs et al. 2006) with significant ecological and socio-economic consequences (Simberloff et al. 2013; Bellard et al. 2016, but see Jeschke et al. 2014). Vertebrates, once introduced to a new range, have a high potential to establish themselves and spread (~50% success rate in Europe and North America, Jeschke and Strayer 2005), with invasive mammal predators (in particular cats, rodents, dogs, and pigs) mostly contributing to local species decline and extinction (Doherty et al. 2016). The impacts of mammalian invaders have been more severe on islands (Sax and Gaines 2008; Doherty et al. 2016, but see Quillfeldt et al. 2008), likely related to the absence of prey responses to predation risk due to the lack of co-evolution (Shea and Chesson 2002; Sih et al. 2010).

Once a non-native species becomes established, eradication and even control is extremely difficult. A few small islands offer the only examples of effective eradication of invasive species (e.g., Howald et al. 2007; Jones et al. 2016; Amos 2018), but eradication costs increase with island size (Martins et al. 2006; Holmes et al. 2015). Therefore, even though eradication may be technically feasible, funding becomes a key challenge (Howald et al. 2007). In addition, new and growing challenges for invasive species management are emerging. For example, conflicts associated with invasive species management are becoming increasingly common (Estévez et al. 2015; Crowley et al. 2017) and animal welfare needs to be considered when planning control actions (Dubois et al. 2017). Consequently, prevention is a fundamental and cost-effective way to cope with biological invasions.

In this context, understanding the mechanisms of invasive species dispersion is critical to prevent future introductions and range expansions (Simberloff et al. 2013). This is particularly true for Antarctic and sub-Antarctic ecosystems which have been little exposed to species introductions until recently, compared to other regions of the planet (Frenot et al. 2005; Hughes et al. 2015). During the last centuries, the terrestrial sub-Antarctic ecosystems experienced several introductions of plant and domestic animals resulting from human incursions (Russ 2007). Nowadays, the importance of preventing the introduction of non-native species to the Southern Ocean Islands is recognized, and partly set in practice via quarantine procedures, boot and clothing decontamination, and restricted access, but with significant room for improvement (de Villiers et al. 2006). In archipelagic environments, identifying areas of potential expansion of invasive species is crucial for prevention measures. In this respect, species distribution models can help predict where a species will spread next (Barbet-Massin et al. 2018, but see Václavík and Meentemeyer 2012), and that way serve as a tool to inform conservation planning (Guisan et al. 2013).

Fifteen years ago, the archipelagic Cape Horn Biosphere Reserve (CHBR), located at the extreme of South America

within the sub-Antarctic Magellanic forest ecoregion, was identified as one of the world's last wilderness areas remaining in the twenty-first century (Mittermeier et al. 2003). Today, this region faces several growing pressures (Rozzi et al. 2012), including biological invasions (as defined by Valéry et al. 2008). Wild invasive and free-ranging domestic mammals (i.e., owned, abandoned, feral), outnumber their native counterparts in the CHBR (12 vs. 10 spp., Anderson et al. 2006), and many ground-nesting birds are impacted by introduced predators, particularly on islands that lack native terrestrial predators (e.g., Schüttler et al. 2009). Roads and other human-made infrastructures, as well as tourism activities are increasing in the CHBR (Sernatur 2014). This creates new potential pathways for the arrival and spread of non-native species (Hulme et al. 2008; Anderson et al. 2015).

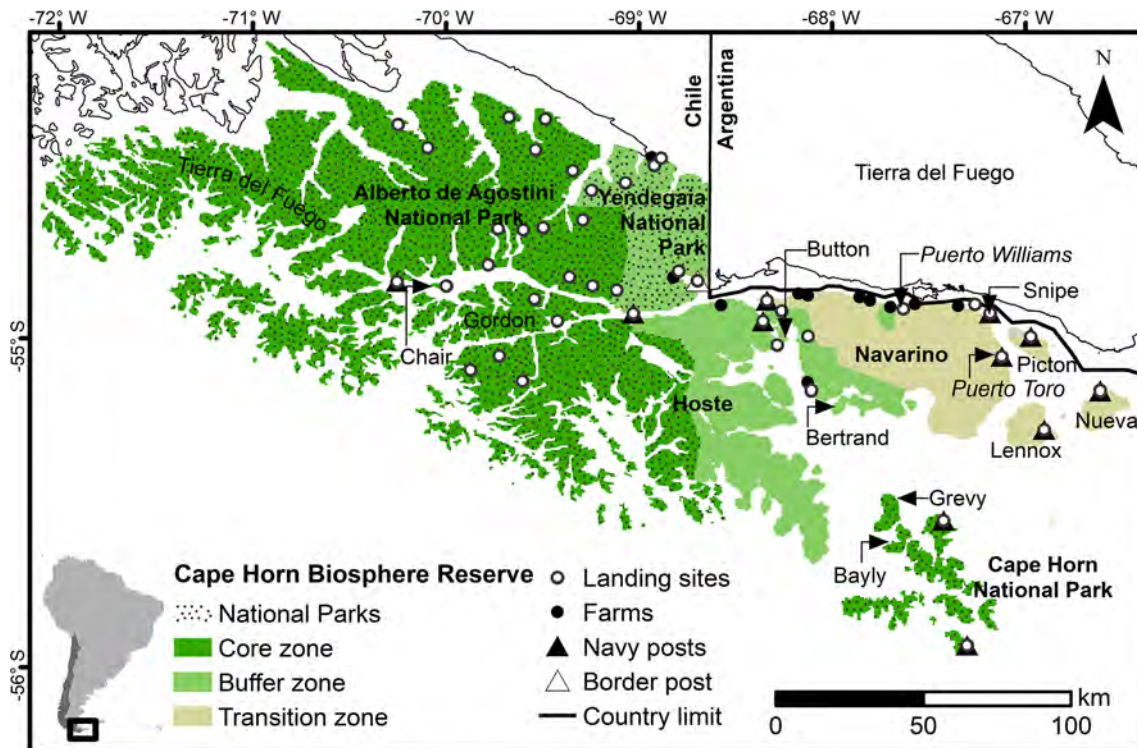
In this article, we provide an updated assessment of the current distribution and potential expansion of wild invasive and free-roaming domestic mammals in the southernmost islands of the Americas. Our specific goals are to: (1) assess the expansion of invasive and domestic mammals in the CHBR during the last decade (2006–2017), (2) assess whether human-mediated movement of invasive species is a plausible dispersal mechanism, (3) identify areas that are more suitable for invasive mammals currently present in the CHBR and therefore likely to be colonized by them, and (4) suggest priority areas for conservation that need to implement invasive species' monitoring and control programs.

## Materials and methods

### Study area

The study focuses on the CHBR (19,172 km<sup>2</sup> terrestrial surface), Chile (55°S) (Fig. 1), which is located at the southern end of the Magellanic sub-Antarctic ecoregion (Rozzi et al. 2012). The dominant habitats within this ecoregion are forests of southern beech (*Nothofagus* spp.) and Winter's bark (*Drimys winteri*), Magellanic tundra (*Sphagnum* spp.), high-Andean habitats, glaciers, and scrublands (Pisano 1977; Rozzi et al. 2012). The climate is hyperhumid, with cool temperatures and a strong precipitation gradient from 2500 mm in the west areas to 500 mm in the east (Tuhkanen et al. 1990). During winter, streams and lakes are ice-bound.

The total human population of the CHBR is 2200 inhabitants, concentrated in Puerto Williams on Navarino Island, and to a lesser extent in Puerto Toro on the eastern coast of the island. On Navarino, there are only eight permanent family-size farms throughout the island's rural zone, and infrastructure is limited to only one dirt road along the northern coast. The Chilean navy maintains 11 navy posts in strategically important sites within the CHBR, operated by



**Fig. 1** Study area in southern Chile

a new navy family each year. The other permanently inhabited sites include Kanasaka (farm on Hoste Island), Dos de Mayo (border post), María Cove (farm), and Ferrari Cove (abandoned farm, uninhabited since 2013). The latter three are located in or close to Yendegaia National Park (Fig. 1).

The main economic activities in the CHBR include artisanal fishing, small-scale livestock farming, and tourism. Currently, the area is facing economic stimuli by the Chilean government, such as the construction of docks for cruise liners, enlargement of the airport runway, land parceling and settling, and new roads (total ~190 km, Sernatur 2014), as well as salmon farming (Kol 2018). All of these activities will bring in new opportunities for species to disperse into the area via quick increase in air, ground, and sea traffic.

## Monitoring

Here, we report a decade of fieldwork (2006–2017) that includes previously published data, unreported data from previously used datasets, and novel data. As our research required access of isolated locations that are climatically and geographically hostile to scientific research procedures, we opted to use multiple sources of information that were suitable in this context. These included opportunistic (i.e., non-systematic) and systematic camera trapping, carnivore diet, questionnaires, small mammal trapping, and walks/transsects. In total, we monitored 44 sites (landing points) on 13 islands

within the CHBR (Fig. 1). Remote islands and the Chilean portion of Tierra del Fuego were visited during six 4–14-day boating expeditions and during five 3–10-day trips with the Chilean navy to each of their 11 posts. We employed more intensive fieldwork on Navarino Island due to the logistical facilities as the island hosts the only major settlement in the CHBR and the Research Center of the Sub-Antarctic Biocultural Conservation Program (detailed monitoring overview of published and new data in Online Resource 1).

## Camera trapping

We used opportunistic and published data from grid-based camera trapping to detect invasive carnivores (cats *Felis catus*, dogs *Canis familiaris*, American mink *Neovison vison*) using canned fish as bait or commercial lures. Opportunistic sampling was employed in the expeditions where the installation of camera traps was adapted to the duration of the stay and accessibility of the terrain and could not follow a random or systematic sampling. We collected data from a grid of 30 cameras on Navarino Island, during a 16-day period in summer 2013 (480 trap days). We set 22 cameras in the surrounding area of 11 navy outposts (two cameras per site) in the cold season 2015 and during the warm season 2016 (except for Horn Island) for an exposure period of two months each (cumulative effort: 2586 trap days). In Yendegaia National Park, 38 cameras, each separated by ~400 m,

were operated for up to six days in 2017 (cumulative effort: 187 trap days). The effort per site ranged between 15 and 522 trap nights (Online Resource 1). We acknowledge that—at site level—these efforts were lower than recommended for inventory purposes (Rovero et al. 2013). We added the following published data to this data base: photos from 29 cameras at three glaciers on Tierra del Fuego (87 trap days, Crego et al. 2015) and photos from systematic grid sampling on Navarino Island (3642 trap nights, Crego et al. 2018a, Online Resource 1).

### Carnivore diet

We used published accounts of mink feces ( $n = 605$ ) as indirect signs for the presence of mink on Navarino Island (Schüttler et al. 2008; Crego et al. 2016) and dog diet ( $n = 53$  feces from different islands, Schüttler et al. 2018) as a source of information on the potential ingestion of invasive prey or carcasses, such as American beavers (*Castor canadensis*), cattle (*Bos taurus*), muskrat (*Ondatra zibethicus*), European rabbits (*Oryctolagus cuniculus*), and rats (*Rattus norvegicus*, see Online Resource 1).

### Questionnaires

To record sightings of native and invasive mammal species we administered questionnaires to navy families at their outposts ( $n = 22$ ) during 2015/2016 and with artisanal fishermen targeting southern king crabs (*Lithodes santolla*) ( $n = 19$ ) in Puerto Williams during 2016. Navy families were visited during the regular provisioning journeys of the Chilean navy; fishermen were contacted following a snowballing procedure (e.g., Marshall et al. 2007). Informed consent was obtained from all participants before completing the questionnaires. Those contained an invitation to list sightings (number of individuals, month of sighting, island, habitat) next to the photos of invasive mammals (beaver, gray fox *Lycalopex griseus*, mink, muskrat, rabbit), free-roaming domestic mammals (cat, cow *Bos taurus*, dog, horse *Equus caballus*, pig *Sus scrofa*), and native mammals (culpeo fox *Lycalopex culpaeus*, guanaco *Lama guanicoe*). For the fishermen, we added questions related to the possible unintentional transport of mink/other animals and the transport of pets among fishing sites. We also asked them whether they had ever left an animal behind on an island. The questionnaires were administered in a face-to-face interview approach at the family's or fisherman's home or boat and took approximately 10–15 min. Each navy family and fisherman that was approached opted to participate in the survey. To complement the relatively low number of data for cats and dogs for modeling the areas of invasion risk, we used

published georeferenced data on unaccompanied dog and cat sightings on Navarino Island ( $n = 227$ , Schüttler et al. 2018).

### Small mammal trapping

We opportunistically captured small rodents using live-capture Sherman traps during an expedition to Tierra del Fuego in 2015, where we deployed two grids of 22–28 traps at three locations for a period of three days, baited with rolled oats. To complement this data, we included published accounts of rodent trapping on Navarino Island (total of 7396 trap nights in Crego (2017) and Crego et al. (2018b), see Online Resource 1).

### Walks and transects

During opportunistic walks at each stop of the boating expeditions, maritime replenishment of navy outposts and during field work on Navarino Island, we took notes on the evidence (sightings, tracks, feces, habitat modification) of larger invasive mammals (beaver, mink, muskrat, and domestic animals). Note that these records were collected as presence-only data without following a systematic design. The total distance walked was ~310 km (Online Resource 1). Finally, we obtained data on mink, beaver, and muskrat presence from three glaciers on Tierra del Fuego (Crego et al. 2015) and from 75 200-m-transects on Navarino Island (Crego et al. 2016).

### Areas of invasion risk

We assessed potential areas for future invasion within the CHBR that possess similar environmental conditions to those of current locations of invasive species. To do this, we ran species distribution models for six established invasive species across the CHBR (American beaver, American mink, cow, dog, horse, and muskrat), for which we were able to collect an adequate amount of georeferenced locations within our study area ( $n > 15$ , van Proosdij et al. 2016). We implemented MaxEnt models (Phillips et al. 2006) in the R Environment (R Development Core Team 2016). MaxEnt is a machine-learning method that reduces the relative entropy of the probability density of the environmental covariates at locations where the species is present relative to the probability density at random sampled background points (Elith et al. 2011). MaxEnt is widely recognized as the most reliable methodology when only presence data is available (Phillips et al. 2006; Elith et al. 2010).

We used a set of predictor variables that we thought a priori were important in modeling habitat suitability for the six species: We obtained 19 climatic variables from the WorldClim database, at 30 arc-seconds (~1 km) resolution (Hijmans et al. 2005). Given the importance of the altitudinal

**Table 1** Selected predictor variables for species distribution modeling in the Cape Horn Biosphere Reserve using MaxEnt with a spatial resolution of 1 km (mapped covariates in Online Resource 2)

Category	Variable	Variable description
Climate	ISO	Isothermality (mean diurnal range/temperature annual range *100)
	PRE	Annual precipitation (mm)
Topography	DC	Euclidean distance to marine coast (km)
	ELE	Elevation (m)
	SLO	Slope (degrees)
Vegetation	NDVI	Normalized Difference Vegetation Index
Other	BMO	Beaver presence/absence MaxEnt output (0/1, only for muskrat)

gradient in the distribution of local habitats and diversity (Rozzi et al. 2012), we included elevation and slope, which we derived from a 90-m digital elevation model (DEM) (Farr et al. 2007). We resampled the DEM raster to match the ~1 km spatial resolution of the climatic data. To consider the vegetation productivity of the system in the models we used a MODIS/Terra NDVI image (MOD13.A2; 1 km resolution). Using Google Earth Engine (Gorelick et al. 2017), we calculated the mean NDVI value for the period between 1 Jan 2008 and 1 Jan 2017. Given that the region is archipelagic, species might extend their distributions from the coastlines; therefore, we included Euclidean distance to the coast as a covariate. We also calculated the Euclidean distance to human settlements (farms and towns). For the muskrat, we added the habitat suitability model of the beaver as a predictor variable, given the facilitation effect that beavers have on muskrats in the study area (Crego et al. 2016). To avoid collinearity, we finally selected seven predictor variables (Table 1, visual overview in Online Resource 2) that we considered more biologically relevant for the distribution of mammals in the CHBR and that presented Spearman correlations  $< 0.7$ . We selected mean annual precipitation over Euclidean distance to human settlements ( $\rho = 0.91$ ) to identify potential areas with similar environmental conditions to those where the presence of invasive mammals is known. We performed all geospatial work using the *raster* package in the R programming language (R Development Core Team 2016).

For modeling in MaxEnt, we subsampled the location points for each species to ensure that each location was separated by  $> 4$  km, to maintain, what we considered, spatial independence. Due to the lack of infrastructure throughout the CHBR, more intensive monitoring took place around human settlements. We addressed this bias by creating background points with a similar sampling bias (Phillips et al. 2009). We randomly selected 5429 background points from 10-km buffer areas around species records (including Chair and Snipe Islands, where no invasive species were

detected), assuming that these areas would characterize areas of potential dispersion and occupancy (Elith et al. 2011). To validate the models, we split our data set into an 80% training data set and 20% validation data set for each species (Fielding and Bell 1997). To assess prediction accuracy of each model, we used the area under the curve (AUC) of the receiver-operating characteristic (ROC) plot (Fielding and Bell 1997). Models with AUC values close to 1.0 were interpreted as giving perfect predictions, while models with AUC values  $\leq 0.5$  should have no predictive ability (Araújo et al. 2005). For each species, we ran 20 bootstrap replicates. We then predicted a Habitat Suitability Index (HSI; logistic output of MaxEnt ranging from 0 to 1, with 0 referring to unsuitable habitat and 1 to highly suitable areas, Elith et al. 2011) of 1 km spatial resolution for each model. We estimated the mean value of the outputs of the HSI and AUC scores of all single runs (Marmion et al. 2009). For each species, we explored different regularization coefficients (1, 2, 3) and found that in all cases, the default value of 1 performed better (i.e., higher AUC values, Merow et al. 2013). We transformed the non-binary HSI results into a binary presence/absence format for each species using the HSI threshold that maximized the sum of sensitivity (correct predictions of the occurrence) and specificity (correct predictions of the absence, Liu et al. 2013). We calculated the threshold as the mean value of all 20 threshold values obtained for the 20 individual models for each species.

Finally, to illustrate areas of invasion risk in the CHBR we calculated an invasiveness index as the proportion of the six species that could be established in each pixel (1 km<sup>2</sup> spatial resolution) of the reserve based on the presence/absence output models calculated from the current invasive species distribution. The index ranges from 0 (no potential invasiveness) to 1 (highest suitable habitat for all invasive species, i.e., highest invasiveness potential). The final map identifies areas of high risk of invasion by currently established invasive mammals and thus, serves to guide management priorities within the CHBR.

## Results

### Invasive mammal assessment

We confirmed the presence of 11 invasive mammal species in the CHBR (Table 2, and excel list by species and location in Online Resource 3). Our results show new detections, particularly on the southeastern islands of the CHBR: Picton (4 spp.), Nueva (2 spp.), Lennox (2 spp.), and Wollaston (4 spp., Table 2). There were also new detections on Hoste (1 spp.), Button (3 spp.), and Bertrand (2 spp.). The mink and beaver were the two species with the highest frequency of new detections (3 islands), followed by cows, pigs, rabbits,

**Table 2** Invasive mammals and their known distributions within the major islands of the Cape Horn Biosphere Reserve. The islands are ordered from northwest to southeast. Letters refer to the data type; a = author observation (direct or indirect), c = camera-trapping, d = dog diet, q = questionnaires, r = rodent trapping. Colors refer to the current state of detection: light gray = not detected or absent, dark gray = earlier literature (Rozzi and Sherriffs 2003; Anderson et al.

2006; Valenzuela et al. 2014), black = this study. We did not include domestic dogs on Horn Island listed by Valenzuela et al. (2014), because the authors referred to those as pet dogs of the family at the navy post. Sightings of invasive mammals on islets can be found in Online Resource 3. *TDF* Tierra del Fuego, *GOR* Gordon, *HOS* Hoste, *NAV* Navarino, *BUT* Button, *BER* Bertrand, *PIC* Picton, *NUE* Nueva, *LEN* Lennox, *WOL* Wollaston, *HOR* Horn

Mammals order	Scientific name	Common name	TDF	GOR	HOS	NAV	BUT	BER	PIC	NUE	LEN	WOL	HOR
Carnivora	<i>Canis familiaris</i>	Dog	a			c, q		q	c, q				
	<i>Felis catus</i>	Cat				a, c, q			c, q			q	
	<i>Lycalopex griseus</i>	Grey fox											
	<i>Neovison vison</i>	American mink	a, c, q	a	c, q	a, c, q		a, q	c, q	q	c, q	q	
Cetartiodactyla	<i>Bos taurus</i>	Cow	a, q		q	a, c, q			q				
	<i>Sus scrofa</i>	Pig			q	a, c, q	q		q				
Lagomorpha	<i>Oryctolagus cuniculus</i>	European rabbit	a, c, q			-		q			c, q		
Perissodactyla	<i>Equus caballus</i>	Horse	a, c, q			a, c, q				a, c, q	q		
Rodentia	<i>Castor canadensis</i>	American beaver	a, c, q	a	a, q	a, d, q	a	q	q	a, c, q	a, d, q	q	
	<i>Mus musculus</i>	House mouse				r							
	<i>Ondatra zibethicus</i>	Muskrat	a		q	a, q	a		q	c, q		q	
	<i>Rattus norvegicus</i>	Brown rat											

horses, and muskrats (2 islands each) (Table 2). We decided to discard reports of gray foxes by questionnaires due to the possibility to be confounded with native culpeo foxes. The last record of a rabbit on Navarino was in 2004 (observation by Anderson et al. 2006). Since 2006, we have neither directly observed nor detected this species through camera traps. At the following sites, we did not detect invasive species during our expeditions: Chair Island, Horn, and Snipe (apart from an owned dog).

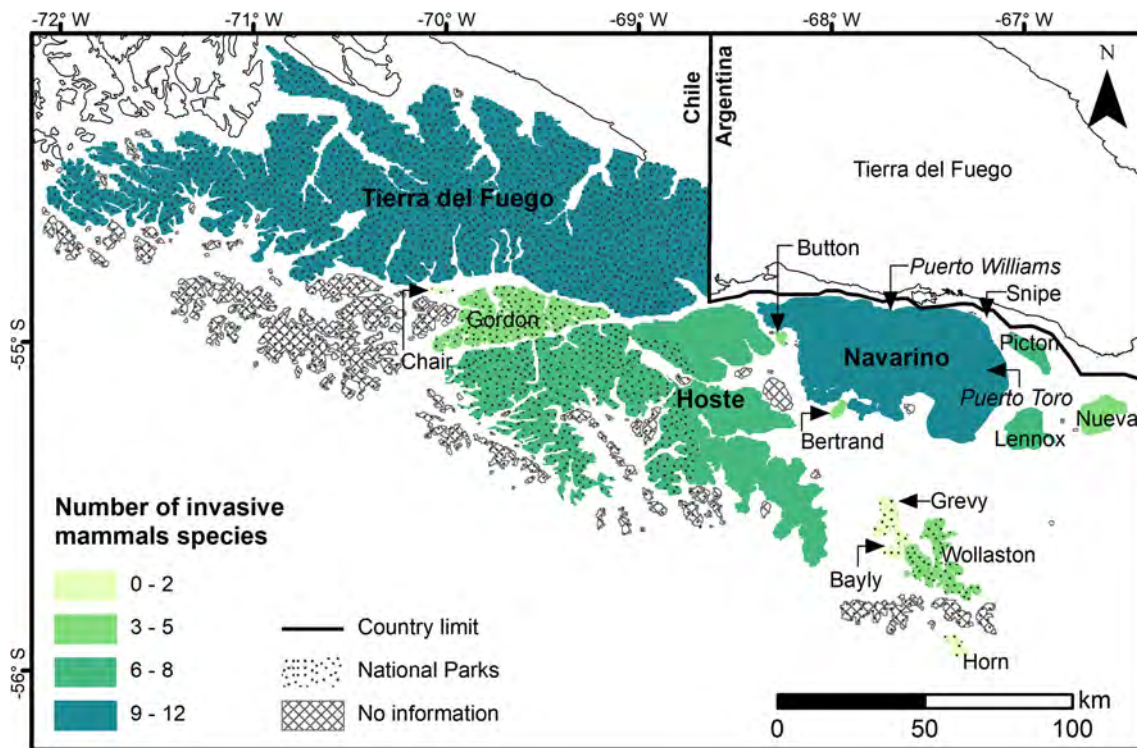
Considering the data up to this point, the two human-inhabited islands with major settlements, Tierra del Fuego and Navarino, harbor the highest number of invasive mammal species (see also Valenzuela et al. 2014), followed by Hoste, Picton, and Lennox (Fig. 2). Comparing the current data with earlier publications, almost all studied islands apart from Tierra del Fuego and Navarino Island, which had already harbored a maximum of mammal species, had an increase in the number of mammal species recorded, particularly the south-eastern islands; Picton (4 new species), Nueva (2), Lennox (2) and Wollaston (4), and also the smaller islands along the western coast of Navarino: Button (3 new species) and Bertrand (2) (Fig. 3). In contrast, Horn Island had no increase and Navarino had one species less (rabbit) than in earlier publications.

Based on records from camera traps, we revealed the presence of free-ranging cats and dogs on Picton, feral horses in Yendegaia National Park, and mink on Lennox (Fig. 4). In the case of cats, there was evidence of in situ reproduction (Fig. 4a). The cats and the dog were unknown to the family at the navy post, as well as to the personnel of the temporary military camp, the only settlement on the island, suggesting that these animals did not have owners.

Additionally, owned cats and dogs in navy posts (see Schütler et al. 2018) were recorded on camera traps on several occasions. Free-ranging dogs were detected in Lennox, Nueva, Puerto Toro, Snipe, and Wollaston, and a free-ranging cat in Wollaston. Other sites where owned free-ranging dogs were sighted was the border post Dos de Mayo and María Cove in Yendegaia National Park, and cats and dogs on Douglas farm (Navarino).

### Transport of invasive mammals

The questionnaires administered to 19 fishermen with a mean working experience of 23 years (SD 14 years, range 1.5 months to 50 years) revealed that they were possible vectors for transporting mink, dogs, and small rodents among islands within the CHBR. Fourteen fishermen (74%) have seen mink entering their boats during the last five years. Of these, ten stated that this had happened more than twice, and even regularly or many times (5 participants), probably attracted by the bait used for king crabs ( $n = 3$  comments). Five fishermen reported they might have transported mink several times among fishing sites between 1–10 km (3 participants) and 10–50 km (2 participants). Two fishermen reported to frequently transporting mice (native or invasive) and rats up to 10 km and even > 100 km. Pets (cats or dogs) accompanied 37% of the fishermen ( $n = 7$ ) occasionally ( $n = 5$ ) or frequently ( $n = 2$ ) on their fishing excursions, mostly for companionship. Finally, four fishermen reported they had lost or left dogs on different islands within the CHBR 5–20 years ago.



**Fig. 2** Number of invasive mammal species within the Cape Horn Biosphere Reserve in 2018 (species listed in Rozzi and Sherriffs 2003; Anderson et al. 2006; Valenzuela et al. 2014; Huertas-Herrera et al. 2017, and our own data)

## Areas of invasion risk

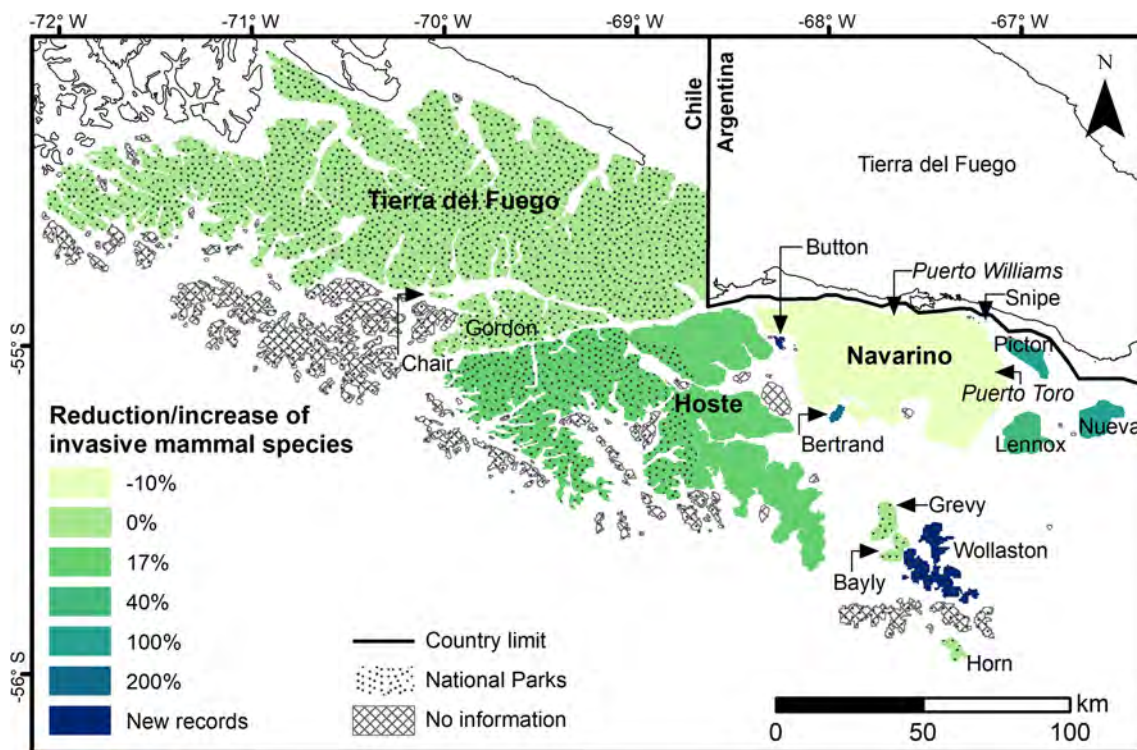
All models performed well ( $AUC \geq 0.8$ ) with the exception of beavers, for which the model performance was fair, with an  $AUC$  of 0.71 (Table 3). Habitat suitability for beavers was best explained by annual precipitation, elevation, distance to the coast, and vegetation (NDVI) (Online Resource 4). For mink and dogs, annual precipitation and vegetation played an important role. Habitat suitability for cows was best explained by precipitation, distance to the coast, and vegetation. For horses, the same variables as for cows as well as elevation were important predictors. The HSI for the muskrat was mainly affected by vegetation, beaver presence, and precipitation (Online Resource 4). The detailed results for each species (habitat suitability and presence/absence maps) can be found in Online Resource 5.

The areas of potential further invasion show a southeast pattern with sites to the southeast presenting a higher probability than sites to the west within the CHBR. Yendegaia National Park, the northeastern coast of Hoste, Navarino Island, Picton, Nueva, Lennox, and the Cape Horn National Park are areas with the highest risk for the establishment of invasive species (Fig. 5). A lower risk of the expansion of invasive species exists for the Alberto de Agostini National Park.

## Discussion

Facing the rapid development and economic interests set in the CHBR, here we raise awareness about the expansion of invasive species in one of the globe's last wilderness areas and propose priority areas for prevention. During the last decade, we did not detect new invasive mammals. However, we recorded some of those previously detected (Anderson et al. 2006; Valenzuela et al. 2014) on new islands, particularly in the southeast of the reserve (Picton, Nueva, Lennox, Wollaston). Logistics in the area are complex; therefore, we are not sure whether the new records are related to a lack of previous detection or true expansion.

Beavers and mink were detected on three new islands. These semi-aquatic species may have a higher probability of colonizing an archipelagic environment by their own means. Beavers have a linear expansion rate of 2.3–6.3 km/year in the region and water barriers of several kilometers do not represent true barriers, such as the Strait of Magellan (Skewes et al. 2006; Anderson et al. 2009). Beavers were reported by fishermen to have arrived to Wollaston Island, an island that forms part of the Cape Horn National Park. Interestingly, in the same national park (Grevy and Bayly islands) the presence of beavers was recently suggested based on the analysis of satellite imagery (Huertas-Herrera et al. 2017). This archipelago had been earlier hypothesized to represent



**Fig. 3** Possible expansion of invasive mammals within the Cape Horn Biosphere Reserve. The number of invasive mammal species detected per island in this study was contrasted with former publications (spe-

cies listed in Rozzi and Sherriffs 2003; Anderson et al. 2006; Valenzuela et al. 2014; Huertas-Herrera et al. 2017) and is displayed as a reduction or increase in percentage

an intrinsic ecological barrier to colonization due to different physical and biological characteristics compared to the rest of the CHBR (Anderson et al. 2006). In continental Patagonia, mink have linear expansion rates of 5.5–9 km/year (Fasola et al. 2011). However, mink have difficulties thermoregulating during prolonged (5 min) aquatic activity (Williams 1986), and dives usually do not last more than one minute (Harrington et al. 2012). This casts doubt on their ability to cross cold-water channels of a few kilometers and suggests that other dispersal mechanisms could operate for mink and other invasive species.

Human-mediated dispersal might play a major role in the dispersal of mink, mice, rats, dogs, and cats in the CHBR. Fishermen reported to have seen mink on their boats (74%) and to have transported them (26%) up to 50 km. Aided dispersal of mink by artisanal fishery in South America was anecdotally reported by Valenzuela et al. (2014, between Navarino and Lennox islands) and is one of the hypotheses that could explain the recent arrival of this mustelid on Chiloé Island (41°S) (Vergara and Valenzuela 2015). Although, up to this point, house mice and brown rats have only been detected on the human-inhabited Tierra del Fuego and Navarino islands (Valenzuela et al. 2014, this study), the results of our questionnaires strongly suggest that these or other rodents were transported to uninhabited islands by

fishing boats or by other vessels (e.g., sailing boats) that frequently stop at islands within the reserve. In 2015, house mice were also trapped for the first time in rural areas on Navarino Island at distances of 30 km from Puerto Williams (Crego 2017). Introducing rats and mice on islands with breeding sites of seabirds can have serious consequences (e.g., Wanless et al. 2007; Jones et al. 2008). Our interviews also revealed that some fishermen were accompanied by their pets. Suazo et al. (2013) described this practice for the Chilean Chonos archipelago, where fishermen used dogs to hunt birds for food or bait, and apparently left them on several islets. Indeed, human-mediated dog movement appears to be important for the population dynamics of dogs in Chile (Villatoro et al. 2016) and other areas of the world (Morters et al. 2014). However, not only fishermen, but any other people visiting the islands of the CHBR might introduce pets. Families at navy posts are also allowed to bring their pets with them, which are kept free-ranging and might get lost or abandoned (Schüttler et al. 2018).

Most of the new detections in our study (75%,  $n = 20$ ) came from questionnaire data and all new records for Bertrand, Hoste, and Wollaston were based on fishermen or navy family observations. While community-based monitoring is increasingly being considered a powerful instrument for conservation science (McKinley et al. 2017), particularly





**Fig. 4** Evidence of reproduction of an unowned cat (*Felis catus*) on Picton Island (a), feral horses (*Equus caballus*) in Yendegaia National Park (b), unowned dog (*Canis familiaris*) on Picton Island (c), and

American mink (*Neovison vison*) on Lennox Island (d), Cape Horn Biosphere Reserve, all photographed by camera traps

**Table 3** Mean area under the curve (AUC) and standard deviation (SD) for MaxEnt models for predicting habitat suitability of American beaver (*Castor canadensis*), American mink (*Neovison vison*),

cow (*Bos taurus*), dog (*Canis familiaris*), horse (*Equus caballus*), and muskrat (*Ondatra zibethicus*) in the Cape Horn Biosphere Reserve, based on known occurrences

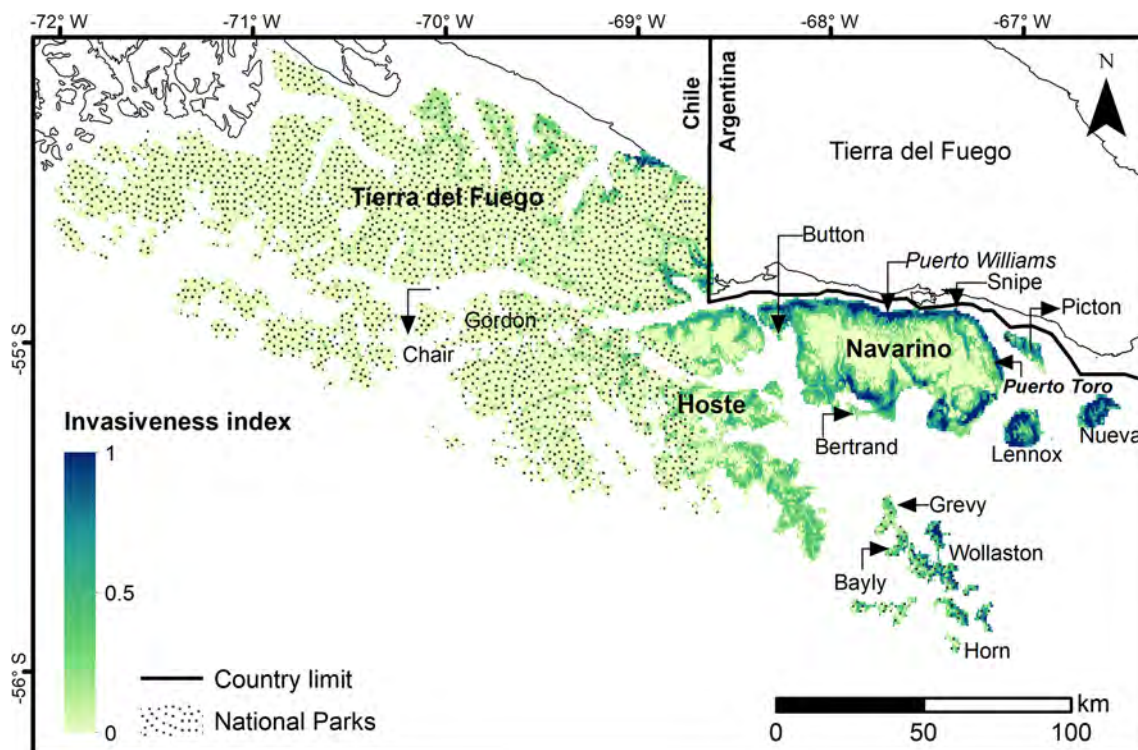
Species	Total <i>n</i>	Sub-sample	Training AUC (SD)	Test AUC (SD)
American beaver	78	47	0.80 (0.03)	0.71 (0.09)
American mink	572	59	0.88 (0.02)	0.86 (0.04)
Cow	59	23	0.97 (0.01)	0.93 (0.04)
Dog	234	45	0.94 (0.01)	0.91 (0.04)
Horse	98	20	0.95 (0.02)	0.89 (0.10)
Muskrat	43	21	0.95 (0.02)	0.89 (0.05)

Total *n* refers to the total of georeferenced locations, the sub-sample contains the number of locations used for modeling (only those locations with a distance > 4 km per species). AUC was calculated on an 80% training data set (Training AUC) and 20% validation data set (Test AUC)

in developing countries (Danielsen et al. 2009), it must be recognized that data collected by local people do not always reflect the accuracy of data collected by professionals (e.g., due to poor identification skills or results shaped by observer perceptions, Danielsen et al. 2005). Neither the fishermen nor the navy families have been trained in our study, yet we doubt that most major sized mammals are prone to misidentification. The only important exception is the gray fox (see above), and—to a lesser extent—the mink that

might be confounded with native otters (*Lontra felina*, *L. provocax*) by non-experts. In any case, further monitoring is needed to confirm each of the self-reported observations, with a special focus on Wollaston Island within the Cape Horn National Park (four new species recorded by questionnaires: beavers, cats, mink, and muskrat).

Based on the invasiveness index, the Cape Horn National Park is one of the areas with the highest risk of establishment of the six studied species (Fig. 5), and beavers may



**Fig. 5** Areas of invasion risk displayed by an invasiveness index (0–1) calculated as the proportion of species that could be established in each pixel (1 km spatial resolution) of the Cape Horn Biosphere Reserve based on the presence/absence output models calculated from the current invasive species distributions. The index ranges from

0 (no potential invasiveness) to 1 (highest suitable habitat for all invasive species). The final map shows from bright to dark - least to highest - those areas of the reserve that are vulnerable to the expansion of invasive species from established areas

already have arrived there (Huertas-Herrera et al. 2017). Although we should await further confirmation by biological data, it seems that this National Park has lost, or may soon lose, its status of being almost “free of exotics” (only three non-native plant species, Rozzi et al. 2004). In the recently created Yendegaia National Park, at least seven of the 12 invasive mammals are already present (Online Resource 3), and this area shows environmental conditions that make it prone to further invasion. This is relevant considering the increase in terrestrial connectivity and tourism (Sernatur 2014) that this park will experience in the near future. Finally, for the third national park, Alberto de Agostini, invasion risk was low, with only some fiords presenting higher risk of invasions, where beaver and mink are already present (e.g., Crego et al. 2015). This might be due to the higher precipitation and lower vegetation productivity along the southwestern gradient of the CHBR (Tuhkanen et al. 1990, Online Resource 2). However, given that many areas of the CHBR remain largely unexplored and data deficient (such as the western area), a sampling bias is possibly adding uncertainty to the models. Moreover, the high precipitation in the western study area creates a problem of no-analog conditions for model prediction (Fitzpatrick and Hargrove 2009). Therefore, we caution that model results are only a

first approximation for science-informed decision-making with regard to prevention measures.

While we consider preventing the spread of invasive species as a key element in the management of the CHBR, the control of some species of major socio-ecological concern such as the invasive predators should be urgently debated (principles for ethical wildlife control in Dubois et al. 2017). As of now, the management of invasive species in the CHBR by the Chilean Agriculture and Livestock Service has been discontinuous, distant from a societal discourse, and due to logistical and financial restraints it has been limited to certain areas of Navarino Island. Control efforts via trapping have been concentrated on beavers and mink, with hundreds of removed animals (Soto and Cabello 2007; Gallardo 2017). However, evaluating the magnitude of the effect of these efforts on the reduction of their populations (if any), as well as the consequences for biodiversity conservation, remains a challenge. With respect to free-ranging cats and dogs, management actions have not been undertaken because the hunting law in Chile (Ministry of Agriculture, Law No. 19.473) only encompasses wild and feral animals, not free-ranging pets or abandoned animals. Unfortunately, distinguishing feral dogs (i.e., self-sustaining animals, in the sense of Vanak and Gompper 2009) from other categories

of stray dogs is difficult (Green and Gipson 1994) and may not be practical for management purposes (Villatoro et al. 2019). The same applies for cats. For example, in the case of evidence of reproduction in cats on Picton Island (Fig. 4), it is unknown whether the mother was born in the wild and whether the kittens will survive. Thus, in a strict sense, we cannot conclude from our data that there is a truly feral cat population on Picton, or elsewhere in the CHBR. However, both, free-ranging dogs and cats are clearly present in the CHBR. Despite the fact that the presence of feral individuals cannot be confirmed or dismissed, it is clear that owned animals are an important part of free-roaming dogs and cats in the study area (Schüttler et al. 2018). Therefore, from a conservation perspective, the key point is the lack of responsibility towards free-ranging pets and especially their impacts on wild animals (Villatoro et al. 2019), independent of their position in the feral-owned continuum. A new law on responsible pet ownership (Ministry of Health 2017, Law No. 21.020) should trigger legal norms at a regional scale that might be helpful for increasing the restriction of pet movement, while awaiting a legal, societal, and ethical solution for the control of unowned cats and dogs.

Based on our findings, we derived the following recommendations for preventing the arrival of new species and dispersion of established species within the CHBR: (1) The three national parks require implementation (i.e., funding, park rangers based on-site, regulated access of people and pets, etc.) to pass from the status of paper parks to actively managed protected areas; (2) awareness-raising among vessel owners and rodent/mink traps in boats are possible prevention measures to reduce their human-mediated spread; and (3) the transition and buffer zones of the CHBR require more conservation actions (e.g., environmental education about invasive species, responsible pet ownership, etc.) to fulfill their role of minimizing negative effects of human-induced activities on the core area (see the Madrid Action Plan, UNESCO 2008). Finally, future scientific needs include the establishment of a systematic georeferenced long-term monitoring of invasive species in the CHBR with regular rodent trapping and camera-trap grids installed with a special focus on the Cape Horn National Park. We also recommend harnessing the significant potential of community-based monitoring to help track changes in biodiversity (gap in South America, Chandler et al. 2017), especially in remote regions with challenging research and monitoring conditions.

Ideally, these efforts should be placed in a broader framework of conservation management in the terrestrial sub-Antarctic ecosystems, including the development of common best-practice guidelines and an international conservation forum as suggested by de Villiers et al. (2006) for the Southern Ocean Islands.

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## Compliance with ethical standards

**Conflict of interest** We have no conflict of interest to declare.

**Ethical approval** Permission to work in protected areas was provided by the National Forest Corporation (CONAF, Resolutions 711/2014 and 158/2017). The Agriculture and Livestock Service (SAG, Resolutions 6518/2013, 8547/2014, and 1728/2015) issued the permits for rodent trapping. The Scientific Ethical Committee of the University of Magallanes, Chile, certified ethical approval of the questionnaires with navy families (Certificate 25/05/2015), whereas the local Chilean navy authority provided a formal permission (Resolution 119/2015).

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