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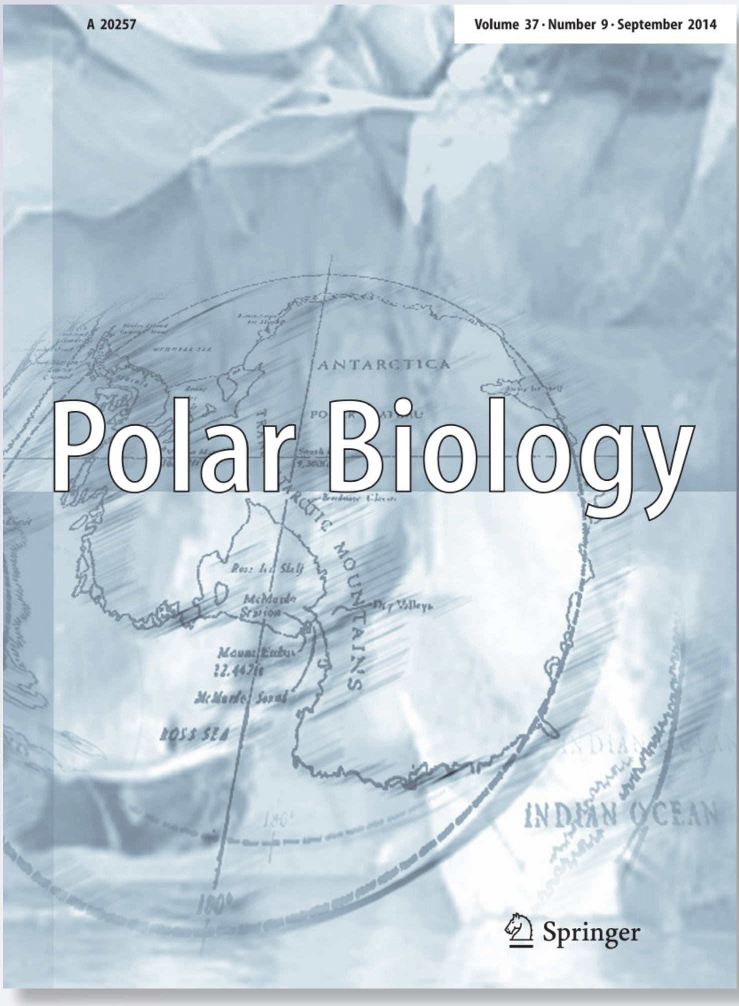

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ORIGINAL PAPER

Species-specific population trends detected for penguins, gulls and cormorants over 20 years in sub-Antarctic Fuegian Archipelago

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Abstract Understanding the dynamics and causes of population trends are essential for seabird conservation. Long-term studies of seabirds at high-latitude (Antarctic, sub-Antarctic and Arctic) regions have shown contrasting species-specific trends in population size in response to climate change and anthropogenic pressures. We have studied for the last 20 years (1992–2012) the population trends of seven seabird species that breed in the Beagle Channel, south-eastern Tierra del Fuego and at Staten Island, a sub-Antarctic region in southern Argentina. The numbers of Magellanic and Gentoo Penguins increased significantly since 1992 (by >15 % $year^{-1}$). In comparison, the populations of Imperial Cormorants, Dolphin Gulls and Kelp Gulls increased at slower rates (<5 % $year^{-1}$), while the Rock Cormorant population even decreased by 1.3 % $year^{-1}$. At Staten Island, the numbers of Rockhopper Penguins decreased by 24 % between the censuses of 1998 and 2010, whereas the population of Magellanic Penguins increased by 227 % during the same period. Over the study period, air and sea-surface temperatures remained stable in our study area, suggesting that the detected population changes are not driven by the climate. This finding contrasts with the detected links between increasing temperature trends and seabird population changes reported from Antarctic and Arctic regions. The level of tourism and size of the permanent human population has increased in the Beagle Channel area during the last 20 years and could be responsible for the increase of gull populations. The seabird species that received the highest number of visitors (Imperial Cormorants and penguin species) seem to be adapted or at least indifferent to pressures exerted by tourism, as their populations increased during the study period. In addition, increasing numbers of seabirds in the area may generally be leading to higher abundances of scavenging species (e.g. gulls).

Keywords Dolphin Gull · Kelp Gull · Imperial Cormorant · Rock Cormorant · Magellanic Penguin · Gentoo Penguin · Southern Rockhopper Penguin · Population trends · Sub-Antarctic · Climate · Tourism

Introduction

Information on population trends and understanding the causes regulating population dynamics is essential for seabird conservation (Lewison et al. 2012). Population changes can be due to several factors that affect seabirds either at their breeding colonies and/or at sea (Croxall et al. 2012). Invasive species, some native species, human disturbance and climate change can have an effect on seabird populations (Micol and Jouventin 2001; Croxall et al. 2002; Wanless et al. 2007; Trathan et al. 2008). Among the at-sea factors that influence birds at their foraging, premoult trips and migration areas, bycatch is the most important (Furness 2003; Delord et al. 2008), followed by sea-surface temperature (SST) changes (Croxall et al. 2002; Jenouvrier et al. 2009), pollution (Velando et al. 2005) and overfishing (Tasker et al. 2000; Croxall et al. 2012). How a seabird population reacts to a certain suite of

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factors appears to differ among species and locations (Weimerskirch et al. 2003; Lescroël and Bost 2006; Ainley and Hyrenbach 2010). The threats described are particularly relevant to pelagic seabirds, and this has resulted in a disproportionate number of these species being listed as threatened (Croxall et al. 2012).

Seabirds are an important component of the marine ecosystem and can be valuable economic resources, particularly through tourism (Yorio et al. 2001). Seventeen seabird species breed along the Argentinean coast. Although the distribution and abundance of these species have been recorded since the 1970s (Yorio et al. 2005a), information on fluctuations in population sizes is scarce, as it is based on only a few long-term studies on a small number of species (e.g. Malacalza and Bertellotti 2001; Yorio et al. 2005b; Schiavini et al. 2005; Frere et al. 2005; Quintana et al. 2006; Lisnizer et al. 2011). These studies have shown that seabird species in Argentina face different threats and this is reflected in different population trajectories. The Olrog Gull (*Larus atlanticus*) populations are decreasing due to habitat destruction (GarciaBorboroglu and Yorio 2007), while populations of the Kelp Gull (*L. dominicanus*) are increasing. Due to this species' predatory nature, this trend is recognised as a threat to other biodiversity (Bertellotti and Perez Martinez 2008; Lisnizer et al. 2011). Among other species, Magellanic Penguins (*Spheniscus magellanicus*) and Imperial Cormorants (*Phalocrocorax atriceps*) are confronted with various pressures that affect their survival, such as bycatch mortality (Yorio and Caille 1999; González-Zevallos and Yorio 2006) and pollution (Boersma 1987a, b; Gandini et al. 1994; García Borboroglu et al. 2006). Seabird species differ in their sensitivity to human disturbances due to tourism (Yorio et al. 2001). Some penguin species habituate to human presence, and no differences can be detected in their survival and breeding success among sites of differing tourist numbers (Cevasco et al. 2001; Trathan et al. 2008). Other seabird species, such as the Rock Cormorant (*Phalacrocorax magellanicus*), generally abandon temporarily their nests when people approach the colony at distances <100 m with inflatable boats (Schiavini and Yorio 1995). In general, tourism has demonstrated to be compatible with the well-being of most seabird species in Argentina (Yorio et al. 2001).

The Fuegian Archipelago (Isla Grande de Tierra del Fuego and Staten Island) is one of the world's 24 remaining wilderness areas (Mittermeier et al. 2003) and one of the endemic bird biodiversity conservation zones (Brooks et al. 2006). Yet, parts of this area are experiencing increasing human activities, including tourism, oil extraction, fishing, and commercial and fishing vessel traffic (Rosciano et al. 2013). In the Beagle Channel, tourist vessels regularly visit cormorants and penguin colonies, sometimes approaching to less than 10 m from the islands where the birds are nesting (Schiavini and Yorio 1995). The number of tourist vessels has increased from six (with a capacity of about 200 tourists) in 1997 to 25 (with a capacity of more than 2,000 tourists) in 2007 (Secretaría de Turismo Municipalidad de Ushuaia 2012).

The first studies on seabirds in the area provided information on species composition (Reynolds 1932, 1934; Olrog 1948, 1950; Venegas 1991). Eight of the 17 breeding seabird species in Argentina breed along the Beagle Channel, including two species each from the families Laridae, Phalacrocoracidae and Spheniscidae, and one species each from the families Stercorariidae and Sternidae (Schiavini and Yorio 1995). In addition, there are two penguin species on Staten Island (Schiavini et al. 1998; Schiavini 2000): Southern Rockhopper Penguin (*Eudyptes chrysocome*) and Magellanic Penguin (*S. magellanicus*).

Given the increasing human activities and potentially also changes to oceanic conditions, it is critical to determine the health of seabird species within the Fuegian Archipelago. The study also provides useful information on trends of near-shore sub-Antarctic seabirds (e.g. gulls and cormorants) which are generally less monitored than pelagic offshore species such as albatrosses and penguins. Various authors analysed distribution and abundance of colonial seabirds in the Argentinean sector of the Beagle Channel during the 1990s (Schiavini and Yorio 1995; Schiavini and Raya Rey 2001) and at sea (Raya Rey and Schiavini 2000). Since 2002, there has been considerable effort to monitor the seabird species at sites where previous counts exist. In this study, we present the current population trends for seven seabird species breeding in the Argentinean territory of the Fuegian Archipelago that have been monitored at various frequencies during the last 20 years. We evaluated population trends from 1992 to 2011, using the numbers calculated during this study (2002–11) and counts presented in published studies (Schiavini and Yorio 1995; Schiavini 2000; Schiavini and Raya Rey 2001; Schiavini et al. 2005; Ghys et al. 2008). We consider three possible factors influencing population numbers of the seven species.

Materials and methods

Study area, seabird distribution and colony size

The study area extended from the western limit of the Argentinean sector of the Beagle Channel near Roca Peron to the waters around Becasses Island at the eastern limit and Franklin Bay in Staten Island (Figs. 1, 2, 3). The Beagle Channel runs in an east–west direction along the southern coast of the Isla Grande de Tierra del Fuego at

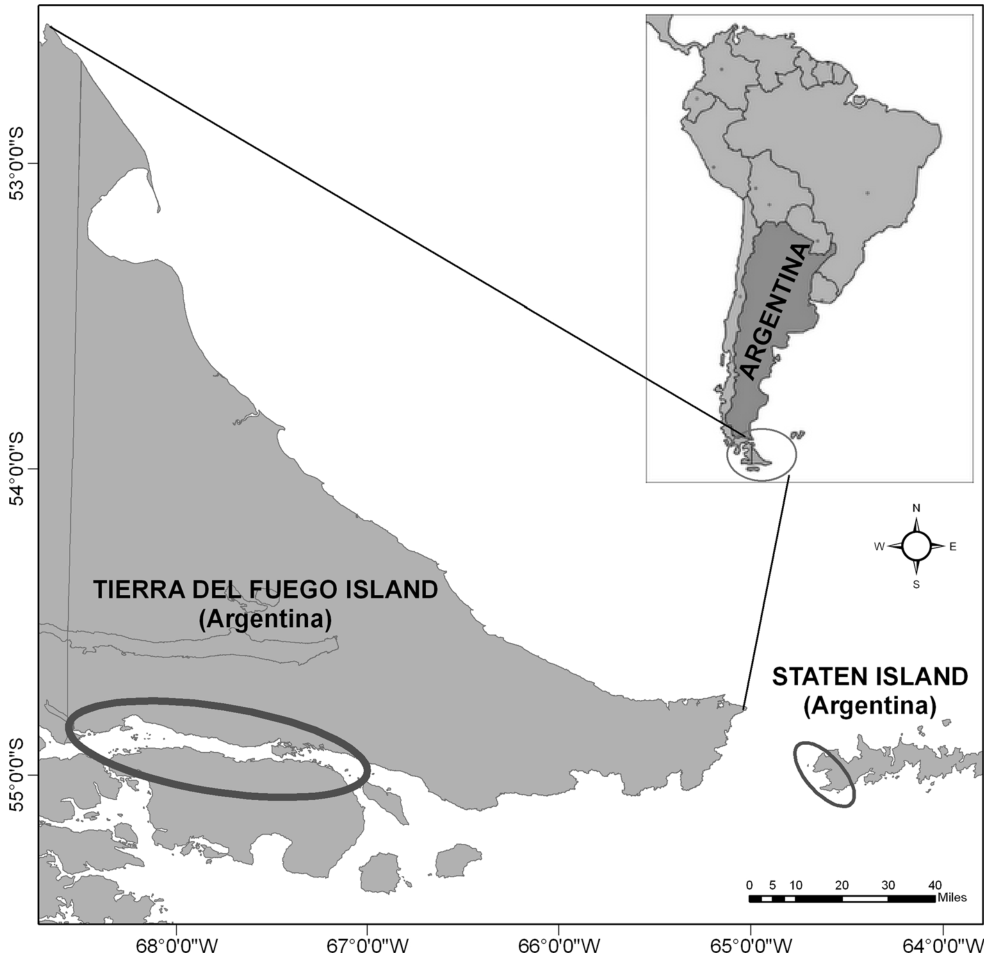

Fig. 1 Map showing the two study areas: the Beagle Channel and Franklin Bay (Staten Island)

about 55°S. There are 32 islands and islets in the Argentinean sector of the Beagle Channel, where a total of eight seabird species nests.

Colonies' locations along the Beagle Channel and Staten Island were determined from previous studies (Schiavini and Yorio 1995; Schiavini et al. 1998; Schiavini 2000). Then, we assessed colony abundance of seven of these seabird species that nest along the coasts of the Beagle Channel and/or in Franklin Bay, Staten Island: Rock Cormorant, Kelp Gull, Dolphin Gull (*Larus scoresbii*) and Gentoo Penguin (Beagle Channel only), Imperial Cormorant, as well as Magellanic Penguin (Beagle Channel and Staten Island) and Southern Rockhopper Penguin (Staten Island only). While performing the survey, we visited by boat all islands along the Argentine sector of the Beagle Channel and walked along Franklin Bay in Staten Island looking for changes in colony distribution and/or new settlements in particular for gulls and cormorants.

Surveys were carried out in mid-November for gulls and cormorants (from 2002 to 2011) and between the end of October and early November for penguins, when most breeding pairs of the species monitored are settled and incubating (Schiavini and Yorio 1995; Raya Rey et al. 2007a). Not all species were surveyed every year.

Different monitoring methods were used for each species and/or site. The Imperial Cormorant breeding colonies in the Beagle Channel were monitored by counting observable nests on aerial digital images taken from a plane flying at a height of 200 m at a speed of 222 km/h. The single count of the Imperial Cormorant colonies of Franklin Bay (2012–2013) was surveyed by direct counts of nests using binoculars from an elevated position. Rock Cormorant active nests were counted directly from an inflatable boat using binoculars. Kelp Gull, Dolphin Gull and Gentoo Penguin active nests were counted directly by walking throughout the nests carefully trying to avoid nests abandonment.

In 1992 and 2000 on Martillo Island, and in 1998 and 2010 at Franklin Bay, Staten Island, Magellanic Penguins were monitored by walking through the colony area and counting all burrows (Schiavini and Yorio 1995; Schiavini and Raya Rey 2001). From 2004 onwards on Martillo Island, the species was monitored using the point-transect methodology (Bibby et al. 2000; Buckland et al. 2001).

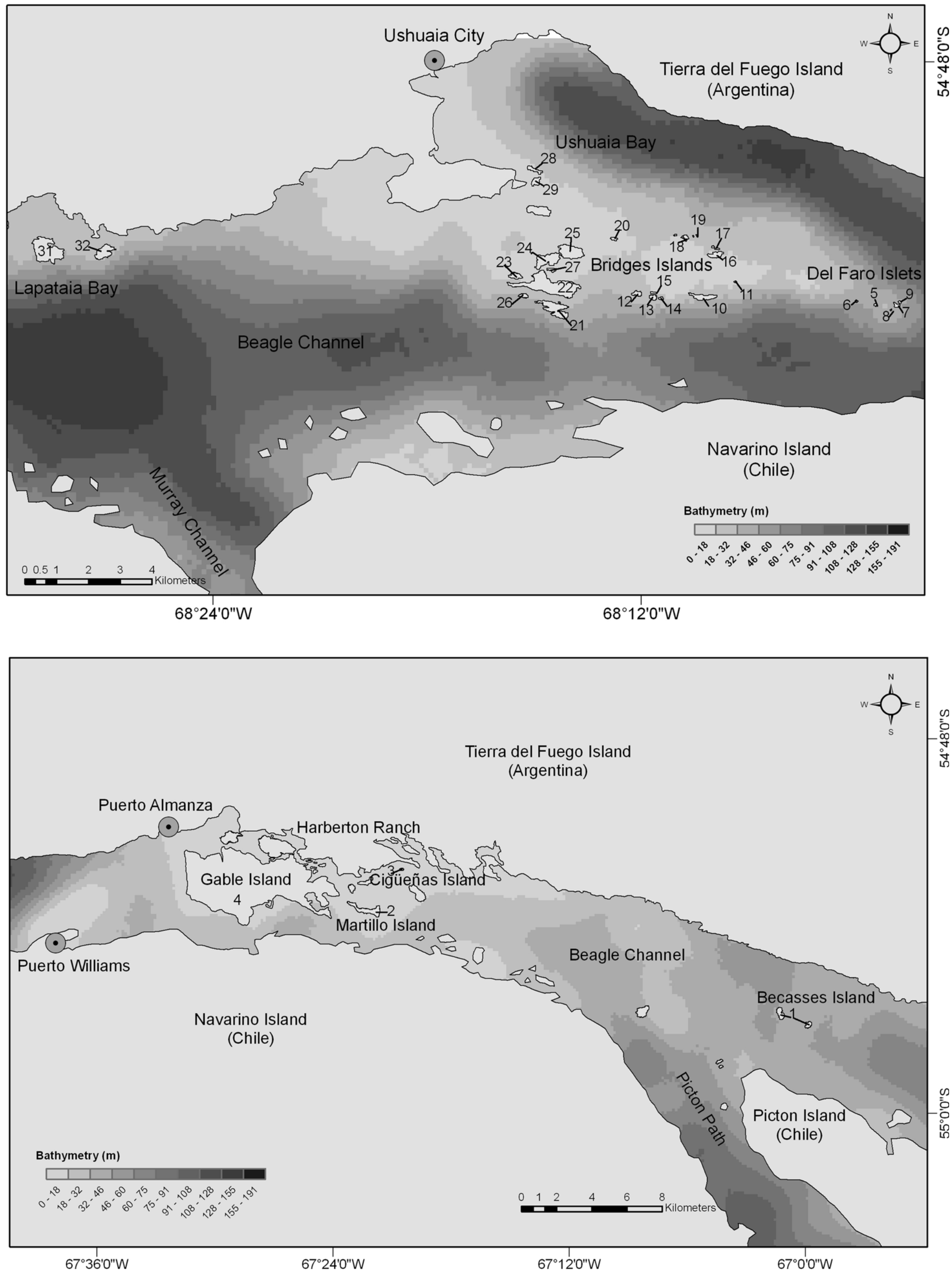

Fig. 2 Detail of the Beagle Channel area showing the islands where the colonies are located. *Numbers* correspond to the name of the islands in Table 1

The colony perimeter was surveyed with GPS, and a permanent grid of points evenly spaced every 50 m was established. The distance from each point to all the active nests was recorded to the nearest metre. Density and abundance were estimated using DISTANCE 6.0 software (Thomas et al. 2010). The confidence interval for the estimates was assessed empirically assuming a Poisson distribution of observations.

Due to the large size and geographic spread of the Southern Rockhopper Penguin colonies at Franklin Bay and the difficulty in physically accessing all the colonies, the number of active nests was estimated by assessing nest density at Franklin Bay and the minimum area occupied by nests. Nest density at Franklin Bay was estimated in October 2010 by counting the number of active nests in 168 circular plots of 100 m^2 distributed randomly in 57 subcolonies within Franklin Bay. A subcolony was defined as a tightly knit group of nests whose limit is easily identifiable on the ground and from aerial pictures by differences in soil and vegetation modified by birds. The subcolonies surveyed extended over two-thirds of the total area of the colony (Fig. 3). We assumed that nest density

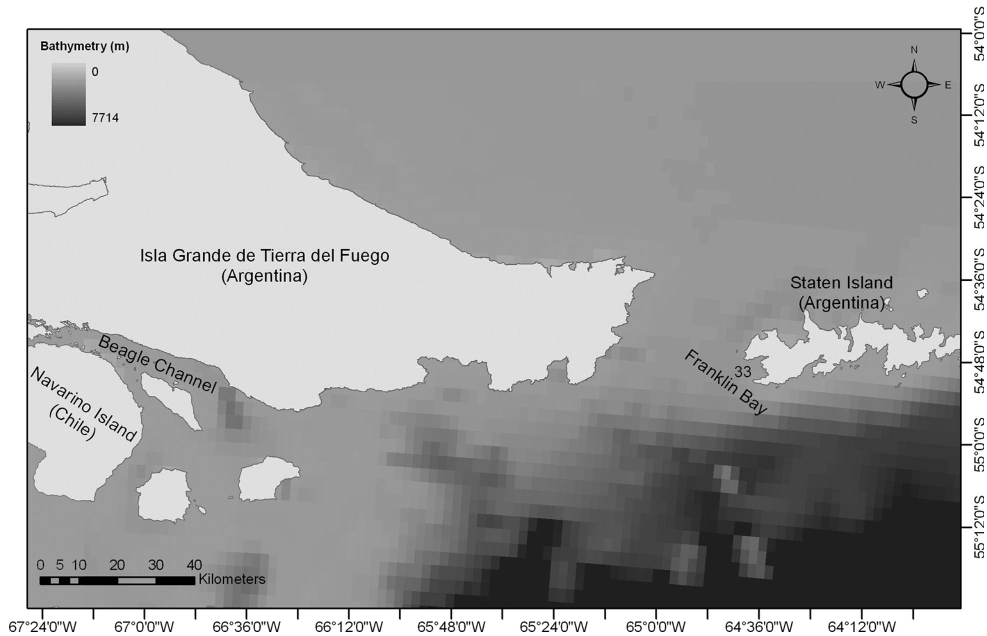

Fig. 3 Map showing the location of Franklin Bay on Staten Island

was similar in the unsurveyed one-third of the colony. The area of the colony was estimated from aerial pictures obtained with a high-wing twin engine (Cessna 337 Skymaster) flying at an altitude of 500 m. Images were taken as perpendicularly to the ground as possible. The locations of southern Rockhopper Penguin subcolonies were identified where ornithogenic soil could be clearly recognised on the images. The aerial images were geo-referenced in ArcGIS 9.3 (Esri), and the limits of the subcolonies were digitalized and the area calculated using spatial analysis tools. Estimates of Southern Rockhopper nest numbers are presented together with 95 % confidence intervals, based on the confidence interval of the mean density of nests.

Trends in seabird populations

For the purpose of the study, we estimated population trends for the whole population of these species on the Beagle Channel (Imperial Cormorants for Staten Island are taking into account separately as it is far away and could not move from Staten to the colonies in the Beagle Channel). As the movements in between the Argentinean Islands of the Beagle Channel are common, we estimated trend for the entire population and not for each site.

To analyse the overall population trend (1992–2011) of each species, we combined the time series with missing counts from all the colonies and fitted a log-linear regression model with Poisson error terms using the program TRIM version 3.53 (Trends and Indices for Monitoring Data; Pannekoek and van Strien 2005). We started the analysis with a model with change-points at each time-point and used the stepwise selection procedure to identify change-points with significant changes in slope, based on Wald tests with a significance-level threshold value of 0.01 (Pannekoek and van Strien 2005). We took into account over-dispersion and serial correlation for all species, since they can have important effects on standard errors, although they have usually only a small effect on the estimates of parameters (Pannekoek and van Strien 2005). For each species, annual population rates (r) were calculated using the relationship:

$$r = \ln \lambda = \ln(N_{t+1}/N_t)$$

where N_t and N_{t+1} are the numbers of breeding pairs in year t and $t + 1$, respectively, and λ the population is the growth rate (Caughley 1977). Both values N_{t+1}, N_t were given by TRIM as imputed counts that represent both the observed counts (data from the surveys) and estimated counts if the count for a given year is missing (population numbers calculated by the program for the years when there were no survey). All population size estimates are presented ± 1 SE. It was assumed that all the incubating birds within the monitoring site were detected on each monitoring occasion.

Natural and anthropogenic factors

We used factors that have been proved to affect seabird populations at other locations worldwide that we considered may be relevant to the Fuegian Archipelago (Micol and Jouventin 2001; Croxall et al. 2002; Wanless et al. 2007; Trathan et al. 2008). These were annual and monthly trends of SST, air temperature and numbers of tourists that visit the area. We downloaded monthly measurements of

Table 1 Location of breeding colonies along the Beagle Channel and in Franklin Bay, Staten Island

	Location name	Location position	PA	PM	PP	SM	LD	LS	EC
1	Becassess Island	54°58′S, 67°01′W	6,209	38			62	161	
2	Martillo Island	54°54′S, 67°23′W		163	31	5,677			
3	Cigüeñas Island	54°53′S, 67°21′W		27					
4	Gable Island	54°53′S, 67°33′W							
5	Faro Centro Islet	54 52′S, 68°05′W							
6	Faro Oeste Islet	54 52′S, 68°06′W							
7	Faro Sur Islet	54°52′S, 68°06′W		22					
8	Roca al sur Faro Sur Islet	54°52′S, 68°06′W		3					
9	Faro Norte Islet	54°52′S, 68°05′W							
10	Despard Island	54°53′S, 68°11′W	2,222	11					
11	NE Despard Islet	54 52′S, 68°10′W							
12	Major Islet of Lucas	54°52′S, 68°13′W					8		
13	East Islet of Lucas	54°52′S, 68°12′W		30					
14	Lucas no name	54°52′S, 68°11′W							
15	NE Islet Lucas	54°52′S, 68°12′W							
16	Major Islet of Willie	54°52′S, 68°10′W		22			15		
17	Islet NO of Willie	54°52′S, 68°11′W		7					
18	Major Islet of Bertha	54°52′S, 68°11′W		8			16		
19	East Islet of Bertha	54°51′S, 68°11′W		8					
20	Alicia Island	54°51′S, 68°13′W	284						
21	H Island	54°53′S, 68°15′W		57			3		
22	Bridges Island	54°53′S, 68°15′W					1		
23	Reynolds Island	54°52′S, 68°16′W		4			31		
24	Leelom Island	54°52′S, 68°15′W		3			48		
25	Mary Ann Island	54°52′S, 68°15′W		26			256		
26	Thomas Island	54°53′S, 68°15′W					74	294	
27	Mary Island	54°53′S, 68°15′W		27			3		
28	Casco Island	54°50′S, 68°16′W					56		
29	Chata Island	54°51′S, 68°16′W					49		
30	Conejo Island	54°51′S, 68°16′W					336		
31	Redonda Island	54°52′S, 68°30′W		21					
32	Estorbo Island	54°52′S, 68°28′W		57					
33	Franklin Bay (Staten Island)	54°53′S, 64°39′W	4,592			1,633	nc	nc	127,325
	Total		13,307	534	31	7,310	958	455	127,325

Numbers of seabird breeding pairs for the last breeding season survey (2011–2012), except for Bahia Franklin which correspond to the season 2010–2011 for penguins and 2012–2013 for cormorants

PA: *Phalacrocorax atriceps*, PM: *Phalacrocorax magellanicus*, PP: *Pygoscelis papua*, SM: *Spheniscus magellanicus*, LD: *Larus dominicanus*, LS: *Larus scoresbii*, EC: *Eudyptes chrysocome*

SST from the NOAA website (http://ingrid.Ideo.columbia.edu/SOURCES/.NOAA/.NCEP/.EMC/.CMB/.GLOBAL/ReynSmithOlv2/monthly/.sst) for the Beagle Channel region (54.8–54.9°S; 68.2–68.3°W) and for two marine sectors intensively used by Southern Rockhopper Penguins during winter dispersion (Pütz et al. 2006; Raya Rey et al. 2007b): an area around Staten Island (52–54°S; 67–59°W) and the Drake Passage (55–61°S; 42–72°W). Air temperature for Ushuaia city was obtained from the Ushuaia Navy station (for the 1992–93 period) and from the Servicio de Información Ambiental y Geográfico (CADIC-CONICET) (for the 1994–2011 period). The number of visitors to Ushuaia (from this 80 % took the at sea excursion to visit the islands) and number of ships sailing in the area during the study period were provided by the Ushuaia City Hall (Secretaria de Turismo Municipalidad de Ushuaia 2012). We used linear regression and ANOVA to evaluate the significance of the regression curve for each data set.

Table 2 Number of breeding pairs and colony location of Imperial Cormorants for each breeding season survey

	Location name	Location position	Imperial Cormorant						
			1992–1993	2000–2001	2007–2008	2008–2009	2009–2010	2010–2011	2011–2012
1	Becassess Island	54°58′S, 67°01′W	4,109	4,005	3,106	4,164	3,618	*	6,209
2	Martillo Island	54°54′S, 67°23′W	–	–	–	–	–	–	–
3	Cigüeñas Island	54°53′S, 67°21′W	–	–	–	–	–	–	–
4	Gable Island	54°53′S, 67°33′W	–	–	–	–	–	–	–
5	Faro Centro Islet	54 52′S, 68°05′W	322	217	–	40	201	–	–
6	Faro Oeste Islet	54 52′S, 68°06′W	–	85	239	263	–	91	–
7	Faro Sur Islet	54°52′S, 68°06′W	–	–	–	–	–	–	–
8	Roca al sur Faro Sur Islet	54°52′S, 68°06′W	–	–	–	–	–	–	–
9	Faro Norte Islet	54°52′S, 68°05′W	–	–	–	–	–	–	–
10	Despard Island	54°53′S, 68°11′W	–	1,014	1,482	3,040	2,330	2,177	2,222
11	NE Despard Islet	54 52′S, 68°10′W	1,444	–	–	–	–	–	–
12	Major Islet of Lucas	54°52′S, 68°13′W	–	–	–	–	–	–	–
13	East Islet of Lucas	54°52′S, 68°12′W	–	–	–	–	–	–	–
14	Lucas no name	54°52′S, 68°11′W	–	–	–	–	–	–	–
15	NE Islet Lucas	54°52′S, 68°12′W	–	–	–	–	–	–	–
16	Major Islet of Willie	54°52′S, 68°10′W	–	–	–	–	–	–	–
17	Islet NO of Willie	54°52′S, 68°11′W	–	–	–	–	–	–	–
18	Major Islet of Bertha	54°52′S, 68°11′W	–	–	–	–	–	–	–
19	East Islet of Bertha	54°51′S, 68°11′W	–	–	–	–	–	–	–
20	Alicia Island	54°51′S, 68°13′W	–	1,390	779	196	–	1,440	284
21	H Island	54°53′S, 68°15′W	–	–	–	–	–	–	–
22	Bridges Island	54°53′S, 68°15′W	–	–	–	–	–	–	–
23	Reynolds Island	54°52′S, 68°16′W	–	–	–	–	–	–	–
24	Leelom Island	54°52′S, 68°15′W	–	–	–	–	–	–	–
25	Mary Ann Island	54°52′S, 68°15′W	–	–	–	–	–	–	–
26	Thomas Island	54°53′S, 68°15′W	–	–	–	–	–	–	–

Table 2 continued

	Location name	Location position	Imperial Cormorant						
			1992–1993	2000–2001	2007–2008	2008–2009	2009–2010	2010–2011	2011–2012
27	Mary Island	54°53′S, 68°15′W	–	–	–	–	–	–	–
28	Casco Island	54°50′S, 68°16′W	–	–	–	–	–	–	–
29	Chata Island	54°51′S, 68°16′W	–	–	–	–	–	–	–
30	Conejo Island	54°51′S, 68°16′W	–	–	–	–	–	–	–
31	Redonda Island	54°52′S, 68°30′W	–	–	–	–	–	–	–
32	Estorbo Island	54°52′S, 68°28′W	–	–	–	–	–	–	–
	Total		5,875	6,711	5,606	7,703	6,149	3,708	8,715

* No survey was conducted at Becassess Island in the 2010–2011 breeding season

Results

Cormorants and gulls breed on a number of islands along the Argentinean side of the Beagle Channel and the locations of the colonies where the nest varied between years, in particular for cormorants (Tables 1, 2, 3).

Beagle Channel Islands (except Martillo) 1992–2011

Imperial Cormorant

The overall breeding population increased 0.8 % during the study period ($\lambda = 1.008 \pm 0.001$, CI 95 % 1.006–1.009) (Fig. 4). The stepwise procedure for the selection of change-points indicated that there were significant changes for all years (1992, 2000, 2007, 2008 and 2009; Wald test $P < 0.01$), showing a first period of growth between 1992 and 2000 ($\lambda = 1.017 \pm 0.002$, CI 95 % 1.012–1.021), followed by a decline of the population between 2000 and 2007 ($\lambda = 0.975 \pm 0.003$, CI 95 % 0.970–0.980) and then significant fluctuations (Fig. 4) with a maximum reached in 2011 (8715 $\pm$ 93; Table 1).

Rock Cormorant

Several colonies (18 in the last census) with a small number of individuals were found in the Beagle Channel. The overall breeding population decreased by 1.27 % ($\lambda = 0.987 \pm 0.002$, CI 95 % 0.983–0.992) (Fig. 4). Results suggested two significant change-points ($P < 0.05$ for Wald test), with a significant decrease between 1992 and 2000 ($\lambda = 0.970 \pm 0.007$, CI 95 % 0.956–0.983) and an increase in 2007–2008 ($\lambda = 1.172 \pm 0.071$, CI 95 % 1.033–1.310). However, no stepwise procedure for automatic deletion of change-points was applied given a warning message of no convergence after 200 iterations.

Changes for Imperial and Rock Cormorants were noted not only in the Beagle Channel numbers for all colonies together, but also in the number of breeding pairs on the different islands between years (Tables 2, 3).

Kelp Gull

Almost 1,000 pairs breed in the area distributed along 14 islands of the Beagle Channel, most of them concentrated on two islands and a few pairs or even solitary pairs in small islands or islets. The overall breeding population increased 2.05 % ($\lambda = 1.021 \pm 0.003$, CI 95 % 1.015–1.026) (Fig. 4). The stepwise procedure for the selection of change-points indicated that there were significant changes for all years (1992, 2000 and 2007; $P < 0.01$ for Wald test), with a significant decline between 1992 and 2000 ($\lambda = 0.896 \pm 0.008$, CI 95 % 0.880–0.911), followed by an increase in the next period (2000–2007: $\lambda = 1.098 \pm 0.012$, CI 95 % 1.075–1.120; 2007–2011: $\lambda = 1.150 \pm 0.015$, CI 95 % 1.120–1.180), reaching the peak of abundance in 2011 (958 $\pm$ 31; Fig. 4; Table 1).

Dolphin Gull

Two colonies of dolphin gull are known within the Beagle Channel. The overall breeding population increased 5.13 % ($\lambda = 1.051 \pm 0.005$, CI 95 % 1.041–1.061) (Fig. 4). The stepwise procedure for the selection of change-points indicated that there were significant changes for all years

Table 3 Number of breeding pairs and colony location of Rock Shags for each breeding season survey

	Location name	Location position	Rock Shag						
			1992–1993	2000–2001	2007–2008	2008–2009	2009–2010	2010–2011	2011–2012
1	Becassess Island	54°58′S, 67°01′W	38	50	64	62	–	*	38
2	Martillo Island	54°54′S, 67°23′W	185	169	179	155	191	183	163
3	Cigüeñas Island	54°53′S, 67°21′W	168	68	29	21	28	28	27
4	Gable Island	54°53′S, 67°33′W	–	–	–	–	–	–	–
5	Faro Centro Islet	54 52′S, 68°05′W	–	–	–	–	–	–	–
6	Faro Oeste Islet	54 52′S, 68°06′W	–	–	–	2	–	–	–
7	Faro Sur Islet	54°52′S, 68°06′W	30	10	15	16	18	22	22
8	Roca al sur Faro Sur Islet	54°52′S, 68°06′W	5	4	2	2	2	3	3
9	Faro Norte Islet	54°52′S, 68°05′W		–	–	–	–	–	–
10	Despard Island	54°53′S, 68°11′W	4	–	8	11	12	13	11
11	NE Despard Islet	54 52′S, 68°10′W	–	–	–	–	–	–	–
12	Major Islet of Lucas	54°52′S, 68°13′W	–	–	–	–	–	–	–
13	East Islet of Lucas	54°52′S, 68°12′W	8	17	19	22	25	26	30
14	Lucas no name	54°52′S, 68°11′W	–	–	–	–	–	–	–
15	NE Islet Lucas	54°52′S, 68°12′W	–	–	–	–	–	–	–
16	Major Islet of Willie	54°52′S, 68°10′W	13	14	17	22	25	15	22
17	Islet NO of Willie	54°52′S, 68°11′W	7	5	4	8	8	7	7
18	Major Islet of Bertha	54°52′S, 68°11′W	1	5	4	7	8	7	8
19	East Islet of Bertha	54°51′S, 68°11′W	12	3	8	7	8	9	8
20	Alicia Island	54°51′S, 68°13′W	–	–	–	–	–	–	–
21	H Island	54°53′S, 68°15′W	27	41	52	40	67	70	57
22	Bridges Island	54°53′S, 68°15′W	–	–	26	2	–	1	–
23	Reynolds Island	54°52′S, 68°16′W	40	30	10	9	9	3	4
24	Leelom Island	54°52′S, 68°15′W	–	–	–	3	3	2	3
25	Mary Ann Island	54°52′S, 68°15′W	21	16	15	40	18	25	26
26	Thomas Island	54°53′S, 68°15′W	–	–	–	35	–	–	–

Table 3 continued

	Location name	Location position	Rock Shag						
			1992–1993	2000–2001	2007–2008	2008–2009	2009–2010	2010–2011	2011–2012
27	Mary Island	54°53′S, 68°15′W	–	15	–	28	28	31	27
28	Casco Island	54°50′S, 68°16′W	–	–	–	–	–	–	–
29	Chata Island	54°51′S, 68°16′W	–	–	–	–	–	–	–
30	Conejo Island	54°51′S, 68°16′W	–	–	–	–	–	–	–
31	Redonda Island	54°52′S, 68°30′W	7	26	29	35	33	31	21
32	Estorbo Island	54°52′S, 68°28′W	98	67	26	67	59	49	57
	Total		664	540	507	594	542	525	534

* No survey was conducted at Becassess Island in the 2010–2011 breeding season

(1992, 2000 and 2007; $P < 0.01$ for Wald test), showing a decline in the first period between 1992 and 2000 ($\lambda = 0.962 \pm 0.014$, CI 95 % 0.934–0.989) followed by an increase in the next period (2000–2007: $\lambda = 1.102 \pm 0.017$, CI 95 % 1.068–1.135; 2007–2011: $\lambda = 1.155 \pm 0.023$, CI 95 % 1.110–1.199) reaching the peak of abundance in 2011 (455 ± 21; Fig. 4; Table 1).

Martillo Island 1992–2011

Magellanic Penguin

The overall breeding population increased 14.8 % ($\lambda = 1.148 \pm 0.002$, CI 95 % 1.145–1.152) (Fig. 4). The stepwise procedure for the selection of change-points indicated that there were significant changes for all years except for 2004 (1992, 2000, 2005, 2006, 2008 and 2009; $P < 0.01$ for Wald test), with all the years showing an increase (1992–2000: $\lambda = 1.100 \pm 0.005$, CI 95 % 1.091–1.109; 2000–2004: $\lambda = 1.200 \pm 0.007$, CI 95 % 1.186–1.214; 2005–2006: $\lambda = 1.565 \pm 0.028$, CI 95 % 1.509–1.620; 2008–2009: $\lambda = 1.302 \pm 0.017$, CI 95 % 1.269–1.335; 2009–2010: $\lambda = 1.013 \pm 0.006$, CI 95 % 1.001–1.024).

Gentoo Penguin

The overall breeding population increased 18.8 % ($\lambda = 1.188 \pm 0.006$, CI 95 % 1.071–1.305) (Fig. 4). The stepwise procedure for the selection of change-points indicated that there were non-significant changes in the slope between the years ($P > 0.05$ for Wald test), indicating a linear annual rate of increase.

Franklin Bay, Staten Island

At Franklin Bay, the Imperial Cormorant was found nesting within 24 subcolonies of southern Rockhopper Penguins. The species was counted for the first time comprising an important percentage of the total for the species in Patagonia (Table 1).

The total population size of southern Rockhopper Penguin population at Franklin Bay, Staten Island decreased by 24 % between 1998 and 2010 (1998: 166,762 breeding pairs, CI 150,350–183,175; 2010: 127,325 breeding pairs, CI 122,647–132,003), representing a decline of 2.22 % per annum. In contrast, over the same period, the Magellanic Penguin colony at Franklin Bay increased from 500 nests in 1998 to 1,633 nests in 2010, representing an increase of 227 % over the whole study period, corresponding to an annual growth rate of 10.4 %.

Natural and anthropogenic factors

Monthly mean values of SST and air temperatures for the Beagle Channel during the 1992–2011 period and of SST for wintering areas of Rockhopper Penguins during the 1998–2011 period are shown in Table 4. Monthly SST for the Beagle Channel area did not vary between the years during the study period (1992–2011) except for May when it decreases ($F = 6.8$, $P = 0.01$; Fig. 5) as well as January, July and August for the Le Maire Strait ($F = 6.6$, $P = 0.02$; $F = 5.8$, $P = 0.03$; $F = 6.9$, $P = 0.02$, respectively) and April for the Drake Passage ($F = 5.3$, $P = 0.03$). Air temperature for the study period (1998–2011) decreased only during April and November ($F = 5.2$, $P = 0.03$; $F = 4.3$, $P = 0.05$, respectively). Also, the only warm event in our

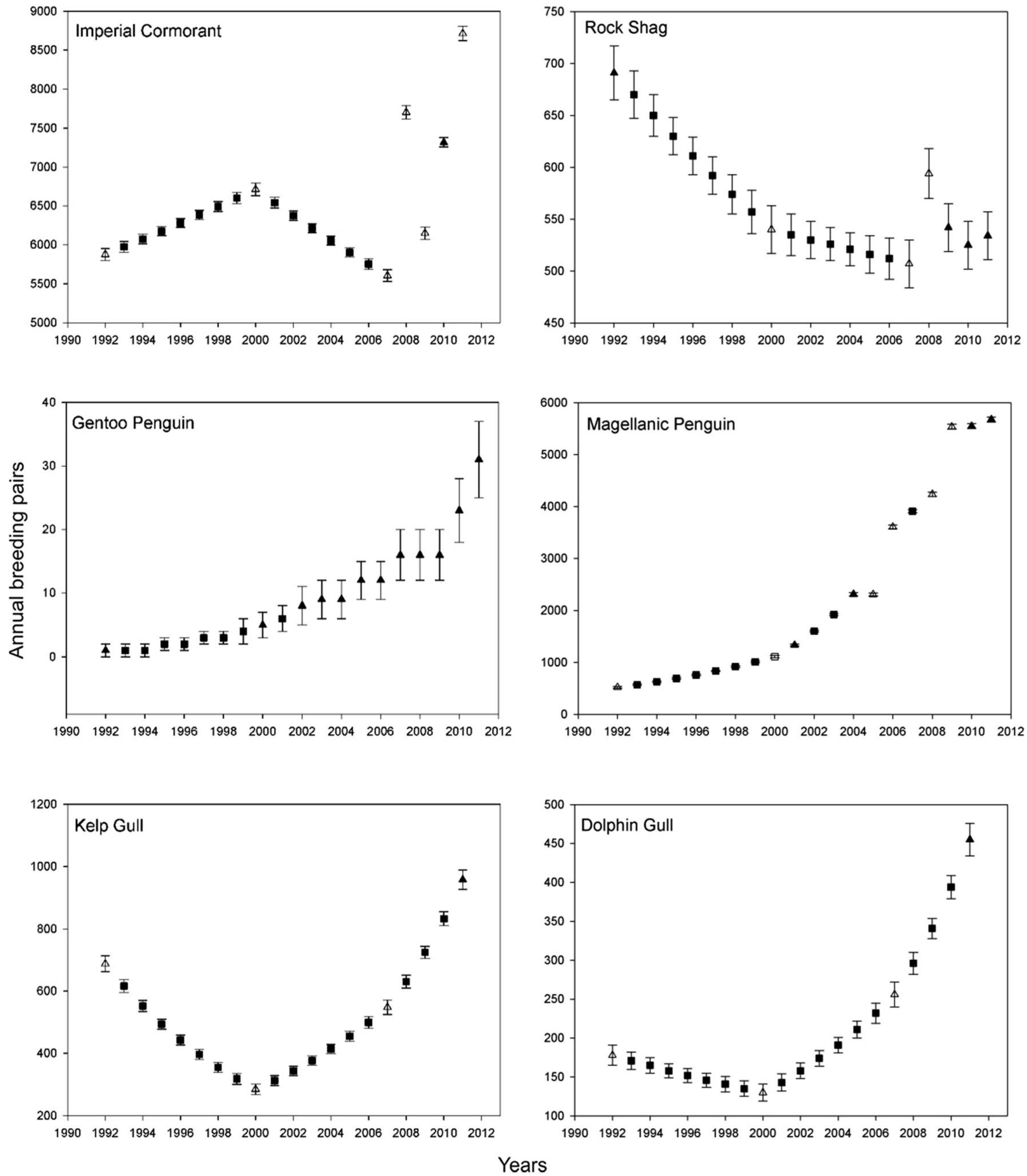

Fig. 4 Estimates of annual breeding population size of Imperial Cormorant, Rock Shag, Dolpping and Kelp Gull and Magellanic and Gentoo Penguins breeding along the Beagle Channel from 1992 to 2011. *Triangles* indicate the breeding numbers we actually counted. *Squares* indicate the imputed values of annual breeding pairs estimated using program TRIM. *Black* indicates the significant change-points years. *Errors bars* indicate ± SE

time series corresponds to an *El Niño* year (1998), when an increase in SST is noticed for all months and then we identified a cold episode in 2000 (see Fig. 5).

The number of tourist arriving at Ushuaia, on the other hand, increased by 84 % in the last 20 years ($F = 346$, $P < 0.01$; Fig. 6). Approximately 80 % of them do a maritime excursion (Secretaria de Turismo Municipalidad de Ushuaia 2012).

Discussion

Seabird population trends at the southern extreme of Argentina over the past 20 years differed depending on the species. In general, population numbers presented a slight increase for cormorants and gulls, except for Rock Cormorants whose numbers decreased slightly, while penguins showed contrasting trends, with an increase in Magellanic

Table 4 Monthly mean values of sea-surface and air temperatures in the study areas

	SST BC	Air temperature	SST SI	SST DP
January	8.7 ± 0.5	9.8 ± 0.8	8.1 ± 0.5	4.5 ± 0.6
February	9.1 ± 0.5	9.5 ± 1.2	8.3 ± 0.6	4.7 ± 0.6
March	8.9 ± 0.4	8.2 ± 0.8	8.1 ± 0.5	4.4 ± 0.5
April	8.3 ± 0.4	5.8 ± 1	7.4 ± 0.5	3.6 ± 0.4
May	7.6 ± 0.6	3.7 ± 1.3	6.1 ± 0.6	2.9 ± 0.4
June	6.8 ± 0.6	1.5 ± 1.2	6.8 ± 0.6	2.3 ± 0.4
July	6.1 ± 0.5	1.5 ± 1.3	5.7 ± 0.6	2.0 ± 0.3
August	5.7 ± 0.4	2.6 ± 0.9	5.5 ± 0.6	1.8 ± 0.3
September	5.6 ± 0.3	4.2 ± 1.2	5.3 ± 0.4	1.8 ± 0.3
October	6.0 ± 0.4	6.3 ± 0.8	5.7 ± 0.4	1.9 ± 0.3
November	6.9 ± 0.5	7.7 ± 1.2	6.6 ± 0.5	2.6 ± 0.3
December	7.9 ± 0.6	9.1 ± 1.1	7.6 ± 0.5	3.5 ± 0.4

Values are mean ± SD. Sea-surface temperature for the Beagle Channel (SSF BC), for the Le Maire Strait Staten Island (SST SI) and for the Drake Passage (SST DP), and air temperature for Ushuaia city

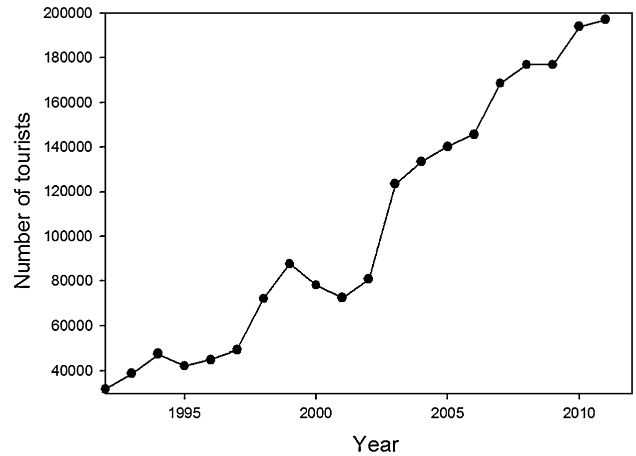

Fig. 6 Number of tourists visiting Ushuaia city during the study period (1992–2012)

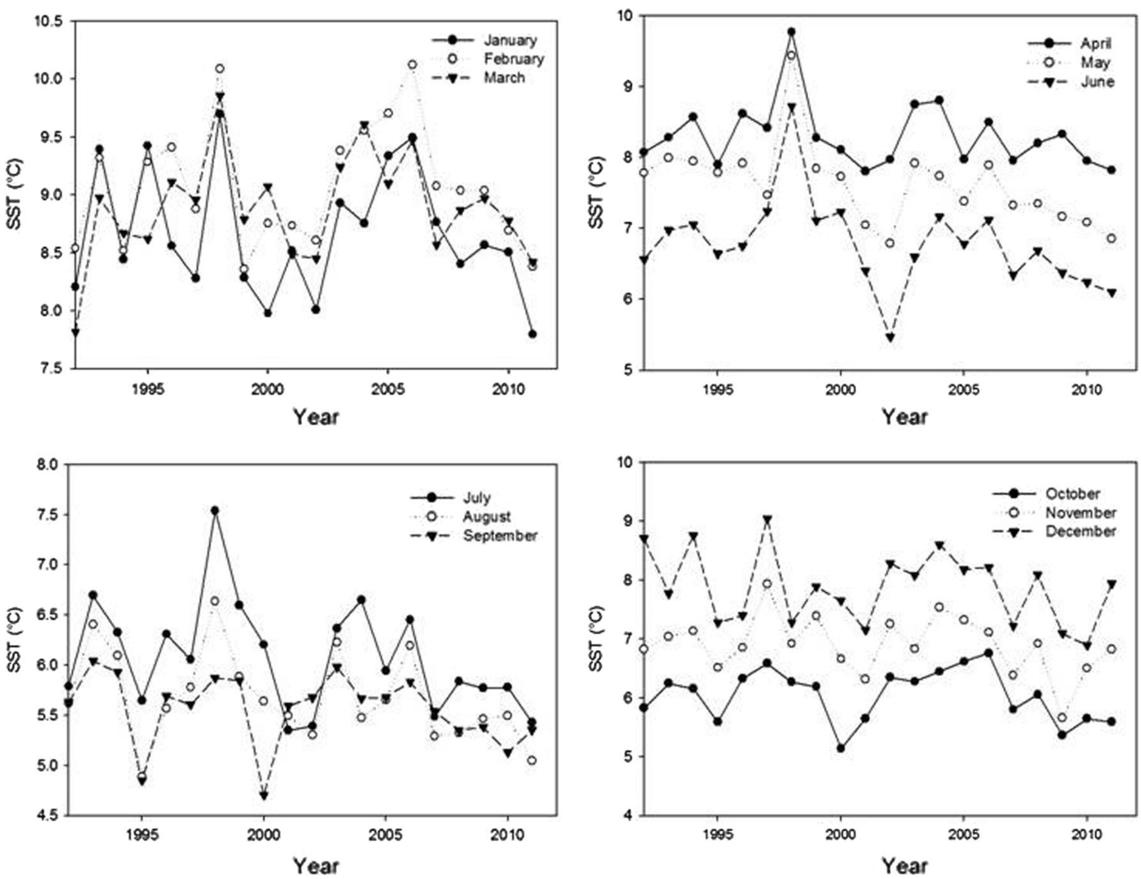

Fig. 5 Monthly sea SST for the Beagle Channel area during the study period (1992–2012)

and Gentoo colonies, and a decrease in southern Rockhopper Penguins. Air and at sea temperatures remained unchanged for most months, while tourism increased constantly during the study period.

Kelp Gull

Colony sizes are smaller than those reported in northern Patagonia (Lisnizer et al. 2011) but similar to populations from Antarctica, Africa, Brazil and Chile (Croxall et al. 1984; Higgins and Davies 1996; Simeone et al. 2003; Branco 2004; Kemper et al. 2007). Population numbers have increased at a low rate during the past 20 years compared to increasing colony sizes at sites in northern Patagonia (Yorio et al. 2005b), being estimated recently at an annual growth rate of 2.7 % (Lisnizer et al. 2011). At some places in northern Patagonia, waste food deposited at tips and discards from trawl fisheries that operate in the area have been identified as supplementary food sources and the reason for the increase in the population (Yorio and Caille 1999; Bertellotti and Yorio 2000; Yorio and Giaccardi 2002; González-Zevallos and Yorio 2006). The effect of supplementary anthropogenic food in gull colonies has also been observed in South Africa (Crawford et al. 2009). In our study site, the population increase in Ushuaia city could be responsible for the Beagle Channel gull population increase as the gulls are known to take advantage of waste food (Raya Rey and Schiavini 2000).

Dolphin Gull

The dolphin gull is endemic to southern South America and Falkland Islands/Islas Malvinas (Yorio et al. 2005b). Two colonies of dolphin gull are known within the Beagle Channel, and both colonies are located close to where other animals breed colonially (mainly cormorants and seals), where dolphin gulls feed (Raya Rey and Schiavini 2000). The exact population trends for dolphin gull colonies in Patagonia are unknown, but Yorio et al. (2005b) suspect that they have stable trajectories. In contrast, the population in the Beagle Channel, this study, is increasing which is in accordance with the imperial cormorant population, a species on which gulls rely on for eggs and waste (Raya Rey and Schiavini 2000). No population trend information is available from the Falkland/Islas Malvinas.

Imperial Cormorant

The Imperial Cormorant, one of the most abundant seabird species along the Patagonian coast (Frere et al. 2005), is the most abundant seabird species in the Beagle Channel (Schiavini and Yorio 1995; this study). The population at Franklin Bay, Staten Island, was estimated for the first time, and the population size reported (4,600 pairs) shows that it represents an important site for the species. Population trends are unknown in northern Patagonia, whereas increasing trends were found in Chubut (Yorio and Harris 1997; Yorio et al. 1998; Frere et al. 2005) and decreasing ones in Santa Cruz (Frere et al. 2005) in a north–south gradient. Meanwhile, the population along the Beagle Channel, further south, has been increasing, although at a very low rate, at least over the past 20 years. Causes of decrease in Santa Cruz were not established but seemed to be local (Frere et al. 2005).

Rock Cormorant

The distribution pattern of the species (small colonies throughout the area) was similar to the one found at other locations (Frere et al. 2005). Population numbers in northern Patagonia were stable (Yorio and Harris 1997; Yorio et al. 1998), while the trend in the Beagle Channel is decreasing.

The dynamics of Cormorant colonies in the Beagle Channel is remarkable, regarding the changes in the number of breeding pairs on the different islands between years (Tables 2, 3). At the Columbretes archipelago, Spain, Martínez-Abraín et al. (2002) documented changes in the number of nesting pairs of Eleonora's falcon *Falco eleonorae* on different islands. They found a decrease in the number of breeding pairs on islands subject to increasing presence of tourism and vice versa, with no changes in the overall number of couples, suggesting a rearrangement of breeding pairs nesting on the islands. A rearrangement of the colonies of Cormorants and Cormorants within each colony throughout the Beagle Channel may be occurring in the study area, such as described for *F. eleonorae* (Martínez-Abraín et al. 2002). However, we could not conclude whether tourism pressure or natural factors (i.e. late snow on certain islands) were the cause of the trends found in both cormorants and the movements of birds between islands as we found opposite trends in islands visited and non-visited by tourists (i.e. Becasess Island and Cigüeña are non-visited, and Faro Sur and H are visited, see Tables 2 and 3).

Magellanic Penguin

Both Magellanic Penguin colonies on Isla Martillo, Beagle Channel, and at Franklin Bay, Staten Island, showed considerable increases in the last 20 years (15 and 10 % annual growth rate for each colony, respectively). In spite of there being a change in the methodology used to estimate Martillo Islands penguin numbers (see "Materials and methods"), the analysis showed an increase in all consecutive years. Elsewhere, the species showed mixed trends. Two of the largest known colonies which are in central Patagonia (Punta Tombo and Isla Leones) have declined (Boersma 1987a, b, 2008). Colonies in northern Patagonia (the northern extent of the species) are increasing (Schiavini et al. 2005), and new colonies are being established at the northern end of the range (Boersma 2008). The population at Falkland/Malvinas Islands is in general considered to have been stable between 1989 and 2008, but with mixed trends at different colonies (Pütz et al. 2001;

Pistorius 2009). There is no population trend data for Magellanic Penguin colonies on the Pacific Ocean region of their range.

The main threats faced by Magellanic Penguins are oil pollution, fishing bycatch and entanglement, and competition for prey with fisheries (Gandini et al. 1994, 1999; Yorio and Caille 1999; Yorio et al. 2010). Other lesser threats include climate change, toxic algae blooms, disease, harvest of eggs and adults, and unregulated tourism (GarciaBorboroglu et al. 2006, 2008; Boersma 2008; GarciaBorboroglu et al. 2010). The decrease in breeding pairs at the Punta Tombo colony has been attributed to an increase in foraging range during the last years, resulting in lower breeding success (Boersma 2008; Boersma and Rebstock 2009; García Borboroglu et al. 2010). Different foraging areas were used given the changes in food availability suggested as a consequence of climate change (Boersma and Rebstock 2009).

The population at Martillo Island has grown continuously since its establishment 50 years ago, similar to what happened to the colonies in Santa Cruz province (Schiavini et al. 2005). The increase is likely due to the high breeding success reported at the colony (Scioscia 2012) but could also be due to high survival of adults.

This increase could be related to a lack of commercial fisheries in the areas where Magellanic Penguins from this colony forage (Raya Rey et al. 2010), and hence an untampered abundant food supply. At this location, penguins are known to feed on fuegian sprat *Sprattus fuegensis*, a non-commercial fish (Schiavini et al. 2005) that is abundant in the Beagle Channel (Casarsa 2005). During winter, penguins from Martillo Island migrate into northerly coastal waters (Pütz et al. 2007). Although they are potentially more exposed to human threats in coastal waters, human activity in this area is lower than in coastal waters utilised by Magellanic Penguins in more northern colonies.

Gentoo Penguin

The colony of Gentoo Penguins on Martillo Island continues to grow since its establishment in 1992 (Ghys et al. 2008). Across its circumpolar range, the Gentoo Penguin has different population trends. Populations at Falkland/Malvinas Islands, Macquarie Island and Antarctica have increased (Woehler et al. 2001; Huin 2006; Zhu et al. 2007; Pistorius et al. 2010; Lynch et al. 2010, 2012). In the Falkland/Malvinas Islands, a decrease of 42 % was found and attributed to a paralytic shellfish poisoning in 2002, with a later increase of 95 % since 2005 (Pistorius et al. 2010). However, a recent study by Baylis et al. (2012) in the Falkland showed interannual fluctuations without a clear trend. Moreover, substantial decreases have been reported for the last 10–20 years for colonies at islands in the Southern Indian Ocean (Crawford et al. 2003; Lescroël and Bost 2006) and at South Georgia (Woehler et al. 2001). Lescroël and Bost (2006) suggested that the population decrease in the Indian Ocean, where Gentoo Penguins forage in coastal waters in contrast to elsewhere, is likely to be related to food availability. On the other hand, monitoring of colonies on the Antarctic Peninsula has shown substantial increases and, in addition, there has been southward expansion of where colonies are located, due to the decline of spring sea ice, which provides the ice-free territory needed for breeding (Lynch et al. 2010, 2012).

The Gentoo population on Martillo increases in spite of tourists being present on the island and catamarans cruising along the shore near the breeding colony. Studies on the effect of tourist visitation at the Antarctic colonies have shown contrasting results. While at Port Lockroy, the reproductive success was lower in visited colonies compared to non-visited ones (Trathan et al. 2008), no effect was found at other colonies or even at the same colony in different years (Cobley and Shears 1999). However, this study was conducted only during one season. Cobley and Shears (1999) suggested that Gentoo Penguins from disturbed colonies are not at a disadvantage during particularly successful breeding seasons, but eventually during years of poor prey availability, the impact of tourism could become evident. Gentoo Penguins at Martillo Island have received regulated tourist visits since 2004 without an apparent effect on the population numbers (Ghys et al. 2008). The colony in Martillo Island is in steady increase and thus does not show the fluctuation presented in the closest sites in the Falkland/Malvinas Islands (Baylis et al. 2012). The small colony size could be the reason why fluctuations are not evident in this population, but also the probable stability of the coastal system within the Beagle Channel.

Rockhopper Penguin

Over most of their range, the populations of *Eudyptes* penguins have undergone considerable declines, with populations of the once most numerous rockhoppers declining by up to 90 % in some locations during the last century (Cunningham and Moors 1994; Bingham 1998; Pütz et al. 2001, 2003; Birdlife 2010). Recent trends for the Falkland/Malvinas Islands showed a 30 % decline in the population between the 2000–2005 censuses (Huin 2006) followed by a 51 % increase by 2010 (Baylis et al. 2013). The increasing trend for the last 15 years is thought to be given by the high survival rate of juveniles and adults around the islands linked with favourable SST values in the waters where they forage (rockhopper penguins survive best at lower SST, Dehnhard et al. 2013).

At Staten Island, only two censuses were carried out, one in 1998 and the other one in 2010. The population was suggested to be stable since the 1970s given the occupied area estimated from the aerial surveys at that time (Schiavini 2000). This study shows a decrease in the population size within Franklin Bay from the first to the second survey. Given that rockhopper populations can undergo opposing trends within a 5-year period in the Falkland/Malvinas Islands, we cannot reliably infer a population trend based on two population counts. Eight putative factors have been found as potential threats for rockhopper populations (Birdlife 2010). Among them, climate change has been identified as one of the factors most likely to be responsible for declining numbers through a resultant reduction on prey availability (Birdlife 2010). Both a decrease (Guinard et al. 1998) and an increase (Cunningham and Moors 1994; Hilton et al. 2006; Raya Rey et al. 2007b; Dehnhard et al. 2013) in SST are related to a lower survival in Rockhopper Penguins. This effect both on adult survival in particular but also on breeding success can influence the population trend in the long term.

Environmental and anthropogenic change

Seabird numbers may fluctuate from year to year between limits that are imposed by reproductive and mortality rate, as well as the biological age at which they first reproduce (Lack 1954). Populations remain relatively stable when conditions at breeding, moulting and foraging sites stay undisturbed either by human activities or by natural environmental variability, or when species have sufficient biological flexibility to adapt to varying conditions (Micol and Jouventin 2001). Seabird populations in the Antarctic region and several sub-Antarctic islands have been affected by previous years' climate fluctuations (increase in temperature and reduction of sea ice), either decreasing or increasing (i.e. Taylor et al. 1990; Fraser et al. 1992; Trathan et al. 1996; Micol and Jouventin 2001; Croxall et al. 2002; Jenouvrier et al. 2005; Forcada et al. 2006; Lynch et al. 2010). Population changes driven by variability in ocean productivity have also been observed in seabird populations of the northern hemisphere, where increases in SST and air temperature have been identified as possible drivers of population changes, through changes in the food web (Ainley and Hyrenbach 2010).

Contrary to what was found in Antarctica and other sub-Antarctic areas (i.e. South Georgia, Kerguelen), air and sea temperature in the Argentinean territories on the Fuegian Archipelago have remained stable for the past 20 years, with only some months showing a decreasing trend in temperature. This relative stability in weather conditions, and hence in the marine ecosystem for the area, may explain the increasing population trends found in the Beagle Channel given that there are no other pressure factors for most species.

Disturbance by human activities is known to affect seabirds at several levels, including population size (Gill 2007). However, little is known on the effect of vessel-based tourism on seabirds (Rodgers and Schwikert 2002; Ronconi and Clair 2002; Velando and Munilla 2011). Most of the populations we studied show increasing trends, indicating that during the study period, no signs of detrimental effects were evident. Tourists walk within the penguin colony at Martillo Island, though in small numbers (80 visitors per day), and between three to five vessels observe penguins from the beached vessels. Given the population increase and the high breeding success of the colony (Scioscia 2012), this level of tourism seems not to affect the well-being of the penguins' population.

In summary, we have shown that for the Beagle Channel, most species presented an increase in their population numbers which is in accordance with the stability in weather and oceanographic parameters (which could be related to the stability in food availability). The increase in human population and tourism seems not to affect seabird numbers, although the most sensitive seabird species studied of the area, the Rock Cormorant, presented a decreasing trend. Finally, for the southern Rockhopper Penguin, we showed a decline for the 1998–2010 period and thus we consider the proposed re-evaluation of the status (Baylis et al. 2013) to be undertaken carefully given our results and that the trends are unknown for the Chilean archipelago.

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