



## Risk assessment and relative impact of Uruguayan pelagic longliners on seabirds

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**Abstract** – Bycatch in longline fisheries is considered one of the main threats for the conservation of albatrosses and petrels worldwide. However, the relative impact of fisheries on all the affected populations or species still remains poorly understood. This paper applied a Productivity and Susceptibility Analysis (PSA) and the concept of “Potential Biological Removal Level” (PBR) to assess the relative impact caused by the Uruguayan pelagic longline fishery on several populations. This two-step approach allowed us to obtain an objective view of the relative impact of the Uruguayan pelagic longline fleet on most of the populations or species of albatrosses and petrels with high association with this fishery. Of fifteen species considered, fourteen were finally assessed and a ranking of risk derived. The concept of PBR was applied to the nine most at-risk species. The impact of fishing on populations could not be straightforwardly inferred from their bycatch rates. Results indicate that large albatrosses (*Diomedea* spp.) and *Thalassarche chlororhynchos* are more affected than some of the main species caught by the fishery (i.e. *Thalassarche melanophrys* and *Procellaria aequinoctialis*). *Diomedea exulans* from South Georgia is likely to be the population most affected by the Uruguayan fleet. This work should be seen as a case study of the fisheries operating in the southwestern Atlantic. The Uruguayan fleet within its operation area was responsible for only the 4.3% to 12.5% of the total annual effort deployed by the different fleets during 2004–2008. The combined impact of these fleets could be sufficiently high to account for many of the observed declines in the populations of *D. exulans*, *D. dabbenena* and *T. chlororhynchos*. However, the seabird bycatch numbers for most of the pelagic longline fleets that operate in the southwest Atlantic remain unknown. Applying mitigation measures to reduce the impact of pelagic longline fleets operating in this region should be considered a high priority.

**Keywords:** Bycatch / Ecological Risk Assessment / Demographic invariant method / Fishery Management / Albatross / Petrel / South West Atlantic

### 1 Introduction

Fisheries are major anthropogenic activities carried out in all oceans of the planet, with a great impact on seabirds (Tasker et al. 2000; Montevecchi 2002; Furness 2003). Albatrosses and petrels (Procellariiformes) live most of their lives at sea, and are capable of traveling vast distances to feed (e.g. Prince et al. 1998; Weimerskich and Wilson 2000; Nicholls et al. 2002; Croxall et al. 2005). The use of a supplementary source of food offered by fishing vessels (as discards or bait) often results in seabird bycatch and consequent mortality in some fisheries (e.g. longline, trawl and gillnet; Brothers et al. 1999; Tasker et al. 2000). Undoubtedly, bycatch of albatrosses and petrels in longlines has historically drawn

most of the attention at a global level (Alexander et al. 1997; Robertson and Gales 1998; Brothers et al. 1999), as it has been associated as one of the main causes of decline of several populations (Gales 1998).

At least fifteen species of seabirds have been caught by the Uruguayan pelagic longline fleet, with the continental slope of the southwest Atlantic and the period from May to November the area and time, respectively, with greatest interaction for many of these species (Jiménez et al. 2009a, 2009b, 2010). Estimates indicate that during 2004–2007 several hundred birds were caught annually, with *Thalassarche melanophrys*, *Procellaria aequinoctialis* and *Thalassarche chlororhynchos* the main species caught (Jiménez et al. 2010). In contrast, other species are captured in very small proportions (Jiménez et al. 2008, 2010). However, the significance of these captures on their populations remains poorly understood. Differences in

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population sizes and recovery capabilities (in terms of productivity) among populations could mean that the impact caused by this fishery on their populations is not a direct reflection of their relative bycatch.

Procellariiformes share extreme demographic characteristics, exhibiting high adult survival rate, high longevity and low fecundity (highly delayed sexual maturity, a single egg). Within this order, several species are biennial breeders (including the great albatrosses, *Diomedea* spp.), making them even less productive (Gales 1998; Tasker et al. 2000; Furness 2003; Brooke 2004). Furthermore, empirical evidence shows that species that are more likely to gain access to baited hooks during setting of pelagic longline gear, are those with greater ability to dive or with larger beaks that allow them to feed on large items such as baits. However, the existing dominance hierarchy related to body size (Jiménez et al. 2011) establishes that larger species have greater access to baits (Jiménez et al. 2012). Given their low productivity and potentially high susceptibility to bycatch, great albatrosses would be in a high-risk situation.

This paper aims to measure the relative impact that the Uruguayan pelagic longline fleet has on different populations of Procellariiformes. Specifically, we seek to answer whether bycatch by the Uruguayan pelagic longline fleet represents a relatively greater risk for great albatross species than for other smaller Procellariiformes. To address this hypothesis, we conduct a Productivity and Susceptibility Analysis (PSA; Hobday et al. 2011) to determine those species that are at greater risk. Then, we apply the concept of Potential Biological Removal Level (PBR; Wade 1998) to assess the quantitative impact.

## 2 Materials and methods

### 2.1 The fishery and study area

The Uruguayan pelagic longline fleet has been targeting swordfish (*Xiphias gladius*), tuna (*Thunnus albacares*, *Thunnus obesus* and *Thunnus alalunga*), and pelagic sharks (mostly *Prionace glauca*) since 1981 in a large area of the south-west Atlantic Ocean (Pons and Domingo 2009a, 2009b, 2010). During the last decade the fleet has operated mainly between 20°–40°S and 20°–55°W (Forselledo et al. 2008; Jiménez et al. 2009a, 2010; Pons & Domingo 2010), using as its main fishing gear an American style longline (monofilament main line), the description of which is given in Domingo et al. (2005) and Jiménez et al. (2009a). This study focuses on the region of the continental slope of Uruguay and the north of Argentina (within the Argentina-Uruguay Common Fishing Zone, ZCPAU), and adjacent waters (including the external border of the continental shelf and the deep waters of Uruguay and nearby international waters 34°–38°S 48°–54°W), where historically the major seabird bycatch events have been recorded in the South-western Atlantic (Jiménez et al. 2009a). The present work covers the period of 2004–2007, where exists estimates of seabird bycatch for the main capture species (Jiménez et al. 2010). During such period no mitigation measures were used except for the night setting. Night setting was practiced mainly as a fishing strategy; however, a previous research in the study fleet demonstrated that

it has a mitigating effect on seabird bycatch (Jiménez et al. 2009a). During this period, sets starting during daylight hours (before sunset) were observed through the year, increasing its proportion between late spring and early fall (Jiménez et al. 2010). The principal oceanographic characteristic defining the study area is the occurrence of the Brazil-Malvinas Confluence (BMC), formed by the meeting of the Brazil and Malvinas currents (Olson et al. 1988; Acha et al. 2004; Ortega and Martínez 2007).

### 2.2 Productivity and Susceptibility Analysis (PSA)

The PSA, in the framework of an Ecological Risk Analysis (ERA) for exploited species or populations, helps to identify which species of those that interact with a given fishery are the most vulnerable and thus exposed to the greatest risk (Cortés et al. 2010; Patrick et al. 2010; Arribabalaga et al. 2011; Hobday et al. 2011; Ormset & Spencer 2011; Waugh et al. 2012; Tuck et al. 2011). In the context of an ERA, the present analysis is a level 2 or semi-quantitative analysis due to the nature of the information available (Hobday et al. 2007, 2011). Risk is estimated as the Euclidian distance from the point (productivity = 1, susceptibility = 0) of the graph of productivity against susceptibility. In this study the risk is understood as a measure of the degree to which the impact caused by the fishery to a population/species exceeds its biological capacity to reproduce (Cortés et al. 2010).

#### Productivity

**Assignment of reproductive islands/archipelagos.** For some species the origin of the individuals that interact in the fishery is clear due to their being endemic to one or a few islands or archipelagos nearby. This is the case for *Procellaria conspicillata*, which reproduces solely on the island Tristan da Cunha, for *Diomedea dabbenena*, which reproduces on Gough Island and *T. chlororhynchos*, which reproduces on both adjacent archipelagos in the south central Atlantic (Brooke 2004; Onley and Scofield 2007). Other species, though not endemics, have more than 90% of their population in one archipelago, such as *Diomedea epomophora* (99% of the population on Campbell Island, New Zealand; ACAP 2009a), *Diomedea sanfordi* (>99% of the population on Chatham Islands, New Zealand; ACAP 2009b), *Thalassarche steadi* (>99% of the population on Auckland Islands, New Zealand; ACAP 2011) and *Puffinus gravis* (>99% of the population on Tristan da Cunha and Gough, Cuthbert 2005).

For the other species, when possible, we considered primarily the main population that visits the study area. In the case of *T. melanophrys* we assumed that the majority of individuals arriving to Uruguayan and adjacent waters come from the Falkland Islands/Islands Malvinas, while a substantially smaller proportion arrives from South Georgia (Tickell 1967; Phillips et al. 2005). Of the recaptures of birds ringed as chicks on the Falkland Islands/Islands Malvinas, 88% ( $n = 66$ ) took place on the Atlantic coast of South America, while only 4% ( $n = 278$ ) of recaptures of chicks ringed on South Georgia occurred in this region (Tickell 1967; Prince et al. 1998). If we consider that the Falkland Islands/Islands Malvinas population is more than five times greater in terms of number

of breeding pairs (see Table 1), a minimal proportion of the birds observed in Uruguay would be from South Georgia. Consequently, the first population is given priority throughout the study. However, to assess the quantitative impact through the PBR both populations were considered (see Results). In the case of *P. aequinoctialis*, we assume that the birds come from South Georgia, neglecting the very small population of the Falklands Islands/Islands Malvinas (see Phillips et al. 2006; Jiménez et al. 2009a, 2010). For *Phoebastria fusca*, we assume that the birds come from Tristan de Cunha and Gough. Although other colonies exist in the sub-Antarctic Islands outside the Atlantic, it is unlikely that an important proportion of the observed birds in Uruguay and adjacent waters originate from them. Within this zone, the species is observed in association with other species from Tristan de Cunha and Gough (e.g. *P. conspicillata*, *Pterodroma incerta*; SJ and MA pers. obs.). In the case of the giant petrels (*Macronectes* spp), there are records in Uruguay and/or adjacent waters of individuals from sub-Antarctic and Antarctic colonies (Escalante 1980; Olmos 2002). However, it is possible that a large portion of the individuals of *Macronectes halli* belong to the population on South Georgia (see González-Solís et al. 2000). On many occasions, young birds (the age mainly observed) of *Macronectes giganteus* were observed with plastic rings fitted in Argentinean colonies (pers. obs.; see Copello et al. 2009), suggesting that a large proportion of the birds associated with the longliners in the area belong to the aforementioned colonies. In turn, evidence from some individuals tracked using geolocators indicates that a smaller proportion arrives from South Georgia (González-Solís et al. 2008). Consequently, we consider the *M. halli* population to be from South Georgia and the *M. giganteus* population to be from the coast of Argentina (including Isla Gran Robredo, Isla Observatorio, Isla Arce and Isla de los Estados) and South Georgia.

*Diomedea exulans* also reproduces on various sub-Antarctic islands; however, in Uruguayan and adjacent waters the captured birds present a very high proportion of rings from Bird Island, South Georgia (Jiménez et al. 2008, unpublished data). We assume that the remaining birds (see Table 1), in their majority, come from other South Georgia islands where a ringing program has not been undertaken extensively. Finally, identification of the origin of the other species that reproduce on various sub-Antarctic islands or on the Antarctic continent is more difficult (e.g. *D. capense*, *Fulmarus glacialisoides*), which is why the analysis is carried out at the species level.

**Estimation of productivity.** In order to estimate the productivity of the various species of seabird we used the approximation developed by Waugh et al. (2009) to produce a PSA for birds in the fisheries of New Zealand, and by Filippi et al. (2010) for the longline fleets in the central and western Pacific. This approximation is based on the “Demographic Invariant Method” (DIM) developed by Niel and Lebreton (2005) in the context of conservation biology, and it is highly applicable to Procellariiformes (see Niel and Lebreton 2005; Dillingham and Fletcher 2008, 2011; Babraud et al. 2009; Kirby et al. 2009; Waugh et al. 2009; Tuck 2011), where there are frequently few data for threatened or rare species. Productivity is estimated as the rate of maximum population growth ( $\lambda_{\max}$ ), which is the rate of annual growth of a

population of a species without limiting factors and at low density (Niel and Lebreton 2005). A key assumption of the DIM is that the  $\lambda_{\max}$  per generation does not vary and is a nondimensional number independent of body size. This method allows the calculation of  $\lambda_{\max}$  in long-living bird species, knowing only the age of first reproduction ( $\alpha$ ) and the survival of adults ( $s$ ). It yields results similar to those of matrix models according to the following equation (Niel and Lebreton 2005):

$$\lambda_{\max} = \frac{(s\alpha - s + \alpha + 1) + \sqrt{(s + s\alpha - \alpha - 1)^2 - 4s\alpha^2}}{2\alpha}.$$

This formulation assumes constant fecundity and constant survival of adults after first reproduction. The maximum rate of population recruitment ( $R_{\max}$ ) and  $\lambda_{\max}$  are related by  $R_{\max} = \lambda_{\max} - 1$  (Dillingham and Fletcher 2008). In optimal demographic conditions, the population cannot increase by a proportion greater than  $\lambda_{\max} - 1$  (that is, the sustainable incidental mortality cannot exceed  $\lambda_{\max} - 1$ ) (Niel and Lebreton 2005). In the present study we estimated  $R_{\max}$  for all the species with a high level of association with the fishery (according to Jiménez et al. 2011). Because  $s$  and  $\alpha$  should represent optimal values rather than current value (Dillingham and Fletcher 2008, 2011), we used the categories proposed by Dillingham and Fletcher (2011) as an approximation of these values. These four combinations of survival and age at first breeding, based in the body size (and breeding frequency), are as follows: 1)  $s = 0.93$  and  $\alpha = 6$  for shearwater *Puffinus* spp.) and small petrels (*Daption* and *Fulmarus* in our case), 2)  $s = 0.94$  and  $\alpha = 7$  for medium sized petrels (*Procellaria*), 3)  $s = 0.95$  and  $\alpha = 8$  for large petrel *Macronectes*) and annually breeding albatrosses (*Thalassarche*), 4)  $s = 0.96$  and  $\alpha = 10$  for biennially breeding albatrosses (*Diomedea* and *Phoebastria*).

## Susceptibility

Susceptibility, understood as a measure of potential impact, in this case of the pelagic longline fishery, is expressed as the product of four conditional probabilities: availability (i.e. overlap of fishery effort with the distribution of the population/species), encounterability (i.e. the probability that a population/species could encounter fishing gear released within the geographic range of the species, in this case with the hooks baited), selectivity (probability that the fishing gear captures individuals of a population/species that encounters the gear; the hooks in this case) and post-capture mortality (Cortés et al. 2010; Hobday et al. 2011). In the current study, for species that consume discards, offal, bait and/or are captured by the fishery, we estimated the susceptibility to incidental capture considering the availability, encounterability or access to bait and probability of remaining captured or selectivity, and the probability of post-capture mortality, considered to be 1. This is because birds captured in the hooks or tangled in the line during setting are found during the hauling with a probability of mortality of 1 (or very close to 1). The probabilities considered were multiplied as suggested by Hobday et al. (2011).

**Availability or overlap with the fishery.** In the course of estimating the availability or overlap (i.e. the proportion of the population that overlaps in space and time with the fishery) we

**Table 1.** Main populations of albatrosses and petrels that interact with pelagic longline fishing in Uruguay and adjacent waters (southwestern Atlantic) and their productivity expressed as the maximum rate of population recruitment ( $R_{\max}$ ).  $R_{\max}$  was estimated by the demographic invariant method using the adult survival rate ( $s$ ) and age at first breeding ( $\alpha$ ). Values of  $s$  and  $\alpha$  were assumed following Dillingham and Fletcher (2011). Breeding frequency: annual (A), biennial (B).

Species	Code	IUCN Status*	Population (Islands or Archipelagos)	Breeding Freq.	Annual breeding pairs**	$s$	$\alpha$ (year)	$R_{\max}$
<b>Albatrosses</b>								
<i>Diomedea exulans</i>	DEX	VU	South Georgia	B	1 420	0.96	10	0.048
<i>Diomedea dabbenena</i>	DDA	CR	Gough	B	1 514	0.96	10	0.048
<i>Diomedea epomophora</i>	DEP	VU	Campbell	B	7 800	0.96	10	0.048
<i>Diomedea sanfordi</i>	DSA	EN	Chatham	B	5 800	0.96	10	0.048
<i>Thalassarche steadi</i>	TST	NT	Auckland Group	A	97 089	0.95	8	0.060
<i>Thalassarche melanophrys</i>	TME <sub>F/M</sub>	EN	Islas Malvinas/ Falkland Islands	A	399 416	0.95	8	0.060
<i>Thalassarche melanophrys</i>	TME <sub>SG</sub>	EN	South Georgia	A	74 296	0.95	8	0.060
<i>Thalassarche chlororhynchos</i>	TCH <sub>T.C.</sub>	EN	Tristan da Cunha	A	21 700–35 800	0.95	8	0.060
<i>Thalassarche chlororhynchos</i>	TCH <sub>G.</sub>	EN	Gough	A	5 300	0.95	8	0.060
<i>Phoebastria fusca</i>	PFU	EN	Tristan da Cunha, Gough	B	7 625–8 750	0.96	10	0.048
<b>Petrels</b>								
<i>Macronectes halli</i>	MHA	LC	South Georgia, other Sub- Antarctic islands	A	4 310	0.95	8	0.060
<i>Macronectes giganteus</i>	MGI <sub>A</sub>	LC	Argentina (Gran Robredo, Observatorio, Arce, de los Estados)	A	2 831	0.95	8	0.060
<i>Macronectes giganteus</i>	MGI <sub>SG</sub>	LC	South Georgia	A	5 500	0.95	8	0.060
<i>Procellaria aequinoctialis</i>	PAE	VU	South Georgia	A	889 122 •	0.94	7	0.070
<i>Procellaria conspicillata</i>	PCO	VU	Tristan da Cunha	A	14 400 ••	0.94	7	0.070
<i>Daption capensis</i>	DCA	LC	Antarctica, various Sub- Antarctic islands	A	2 000 000 •••	6	5	0.083
<i>Fulmarus glacialisoides</i>	FGL	LC	Antarctica, various Sub- Antarctic islands	A	4 000 000 •••	0.93	6	0.083
<i>Puffinus gravis</i>	PGR	LC	Tristan da Cunha, Gough	A	6 000 000 ••••	0.93	6	0.083

\* <http://www.birdlife.org/datazone/species/search>. \*\*Breeding frequency and population size of ACAP listed species were obtained from ACAP Species Assessments (<http://www.acap.aq/acap-species>). Population size obtained from: • Martin et al. 2009, •• Ryan and Ronconi 2011, ••• Booke 2004 (Presented values are approximations), •••• Cuthbert et al. 2005 (Presented value is an approximation). TME<sub>F/M</sub>, TME<sub>SG</sub>, TCH<sub>G</sub>, TCH<sub>T.C.</sub>, MGI<sub>A</sub> and MGI<sub>SG</sub> refer to the colonies of Falkland Islands/ Islas Malvinas, South Georgia, Gough, Tristan da Cunha, Argentina and South Georgia, respectively. IUCN: the International Union for Conservation of Nature.

encountered several gaps in the information related to the distribution of the different populations. As the main objective of this study was to estimate the relative impact of the Uruguayan pelagic longline fleet on different populations of seabirds affected, we used the following approximation: the relative frequency with which a given species occurred in the vicinity of Uruguayan longliners in the area of study (FO- number of counts in which a species occurred as a percent of the total number of counts of seabirds associated with the boats) was used as a proxy for spatiotemporal overlap of the species with the fishery. In order to give a probability value to the availability we used a system of assigning probabilities of 0.33 (low), 0.67 (medium) and 1.00 (high). To make comparable

the observed %FO between species with different population sizes, we took different arbitrary cut-offs (Table 2). For species with population sizes greater than 100 000 reproductive pairs, we considered as high FO greater than 50%, medium between 25 and 50% and low as less than 25%. For populations with 10 000 to 100 000 pairs, we considered an overlap high when the FO was greater than 25%, medium between 10 and 25% and low as less than 10%. For species with population sizes less than 10 000 pairs, the limits for high, medium and low overlap were greater than 10%, between 5 and 10% and less than 5%, respectively.

The information used to estimate FO was obtained from Jiménez et al. (2011) and comprises 20 fishing trips conducted

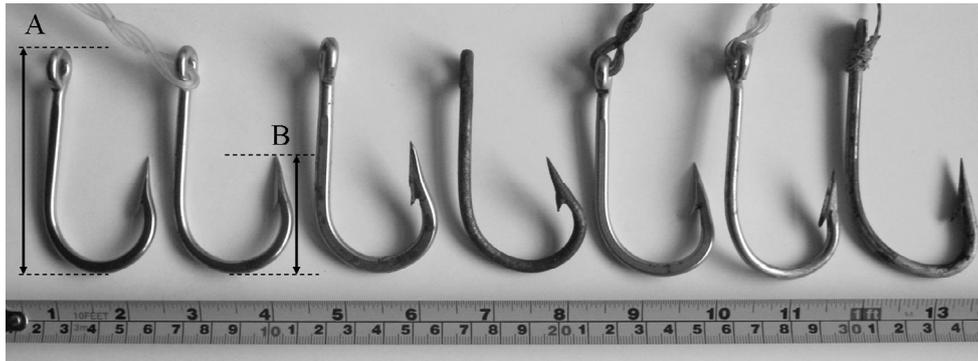
**Table 2.** Cut-off used to estimate the different attributes of the susceptibility for the populations/species of albatrosses and petrels associated with longline vessels in the Uruguayan slope and adjacent waters. For each species, the probability ( $p$ ) for each attribute was scored in low (0.33), medium (0.67) and high (1.00) according these cut-off. The attribute marked with an asterisk indicates it was estimated quantitatively and the resulted values were then translated into a semi-qualitative score using the cut-off (see Sect. 2). FO: relative frequency of occurrence (%), FL: front length of the hook, TL: total length of the hook, N/A: not applicable.

Attribute	Low (0.33)	Medium (0.67)	High (1.00)
<b>Availability or overlap with the fishery</b>			
If population size is:			
>100 000 breeding pairs	FO < 25%	FO 25–50%	FO > 50%
10 000–100 000 breeding pairs	FO < 10%	FO 10–25%	FO > 25%
<10 000 breeding pairs	FO < 5%	FO 5–10%	FO > 10%
<b>Access to bait</b> (N of bait access/ N of bird present during the observations)	<33th percentile	33th percentile <Access to bait > 66th percentile	> 67th percentile
<b>Hook Selectivity</b>	Culmen < FL	FL < Culmen > TL	Culmen > TL
<b>Post-capture mortality</b>	N/A	N/A	N of bird dead/N of bird captured ~1

during 2005–2008 in the Uruguayan pelagic longline fleet. The methodology is described in Jiménez et al. (2009b, 2011). Basically, it included 3 counts  $d^{-1}$  (morning, around noon, and afternoon–evening) from aft of the boat, where an area encompassing about  $200 \times 400$  m (~200 m from the stern and ~200 m to port and to starboard) was observed during approximately 30 min, during which the number of individuals of each species was counted (Jiménez et al. 2009b, 2011). Morning counts (initiated between 06:30 and 09:30 h) coincided with the first hour of longline hauling. Midday counts took place (started between 10:30 and 13:30 h) were performed mainly during hauling or at the end of hauling (navigating). Afternoon counts (initiated between 16:30 and 19:30 h) were during daylight hours, before or the early stages of gear setting. In some cases it was impossible to complete the 3 daily counts due to bad weather (Jiménez et al. 2009b, 2011). For the purpose of the present study, counts realized during the setting and hauling of the longline ( $n = 349$  counts) were used to calculate FO, discounting counts in which no fishing activity was carried out (i.e. during navigation). The data set included seabird counts performed in all season during the four years mentioned above. Two pairs of species were grouped during the counts (see Jiménez et al. 2011), *D. exulans* and *D. dabbenena* and the two species of giant petrels *M. halli* and *M. giganteus*. To obtain a %FO for each species we made the following corrections. First, based on data for incidental capture obtained in the study area we know that the majority of *D. exulans s.l.* belongs to *D. exulans* and particularly to South Georgia (Jiménez et al. 2008; unpublished data). *Diomedea dabbenena* has been captured principally to the northeast and also to the east of the study area. A conservative approximation attributes a FO of 5–10% to this species in the area considered.

Second, in the case of *Macronectes* spp. we discriminated between species in 97 of 113 counts where this genus was present during 2006–2008 ( $n = 279$ ). Of these counts *M. halli* was present in 93 counts (96% of the counts), while *M. giganteus* was present in only 43 counts (44% of the counts). Assuming that these proportions were constant throughout 2005–2008, the FO of 49.6% obtained for *Macronectes* spp. can be divided into 47.6% and 21.6% for *M. halli* and *M. giganteus*, respectively.

**Access to bait (encounterability).** Access to the bait, both primary (by diving or on the surface) or via other species (secondary attacks), was quantified during observations of attacks on baits in this fishery during 48 diurnal sets during 2005–2010 (Jiménez et al. 2012). These observations were used to estimate the probability of access to bait. The methodology is detailed in Jiménez et al. (2012). Briefly, each time that a baited hook was set to the water this was observed, using as a reference the snap that connected it to the mother line of the longline, at a maximum distance of approximately 150 m from the stern of the vessel. For each primary attack that was observed, the species, the distance from the stern of the boat, whether contact with the bait was observed or not, and the mode of attacking the bait (i.e. if the bird descended and attempted to take it at the surface (i.e. surface attack) or if the bird dived (i.e. body fully submerged) in order to try to take it (i.e. diving attack) were recorded. In multiple attacks, the secondary species were also noted, taking account of the order in which they occurred. Before beginning observations the birds were counted, recording the number of individuals present per species (Jiménez et al. 2012). In the present study, for each species we used the total number of contacts with the bait obtained during exclusively primary attacks (i.e. primary



**Fig. 1.** Hooks used by Uruguayan pelagic longline vessels. The measures considered to assess the selectivity are indicated. A. total length, B. front length. From left to right, the first 6 hooks were used in the American-type longline gear (and used in this work) and the last in the Spanish-type longline gear (not addressed).

attacks that did not lead to multiple attacks) plus the total number of contacts with the bait as the last species during multiple attacks. This was because all previous species in multiple attacks are considered unsuccessful in remaining with the bait. Because in the present work the objective is to quantify the bait access, primary attacks from the surface in which no contact was observed between the bird and the bait were not considered in the analysis. Since it is not possible register the contact in diving attacks, except when the birds return with the bait, we assumed 100% contact for these attacks. This may overestimate the access of diving petrels (i.e. *Procellaria* spp. and *Puffinus gravis*), but this bias may be offset to some extent if we consider that some species of albatrosses (mainly *Thalassarche* spp.) also dive pursuing baits. Primarily, to obtain a probability value for the access to bait, the resulted total number of contacts was divided by the number of individuals present during the observations for each species (Table 2). However, when we use these values as a measure of encounterability, it was observed that this component dominated the outcome of the analysis. In order to correct its dominance in the results we took a qualitative approach assigning probabilities ( $p$ ) of 0.33 (low), 0.67 (medium) and 1.00 (high). The quantitative access (i.e. mean number of access/number of bird) was normal distributed (Shapiro-Wilk  $p > 0.05$ ,  $n = 14$  taxa). We used the percentiles 33th and 66th as the better approach to setting cut-offs. It was considered a high ( $p = 1.00$ ) assess to the bait when the quantitative access was greater than 66th percentile (i.e. 0.41), medium ( $p = 0.67$ ) between 33th (i.e. 0.23) and 66th percentiles and low ( $p = 0.33$ ) as less than 33th percentile (Table 2).

**Selectivity.** To determine the probability of capture (i.e. selectivity), we used as a proxy the ratio of the length of the bill (culmen, i.e. the dorsal ridge of the upper mandible) and the length of the 9/0 hook for swordfish used by the fleet (Table 2). The size of the bait is also useful in this case; however, it can be very variable and/or be unavailable for many fisheries in relation to the size of the hook. Using a system of assigning probabilities of 0.33 (low), 0.67 (medium) and 1.00 (high) we considered a species to have a low probability of being captured (hooked) when its bill length was less than the front length of the hook (Fig. 1). The probability of capture was medium when the length of the bill was greater or equal to the front length of the hook and less or equal to the total length of the

hook (Fig. 1). It is assumed that these species have a greater chance of getting caught, but the chance to completely swallow the hook is still limited. Finally, it was considered high when it was greater than the total length of the hook (Table 2), since there is a great chance of being completely swallowed. Note that outcomes other than capture can occur when a bird obtains the bait (i.e. robbing the bait or failing to rob the bait), so the probability of capture is never 1. However, as the objective is to obtain a relative value of the susceptibility of the species, the point-assigning system was considered appropriate. A total of 6 types of “J” 9/0 hooks used by the fleet were measured (Fig. 1), with a median total length of 80.0 (SE  $\pm$  0.8) mm and a median front length of 43.1 (SE  $\pm$  0.9) mm. Bill length of each species was obtained from individuals incidentally captured in pelagic longlines of Uruguay and Japan that operate in Uruguayan and adjacent waters in the southwest Atlantic ( $n = 541$  birds; DINARA data), or from the literature.

### 2.3 Potential Biological Removal Level

The “Potential Biological Removal Level” PBR (Wade 1998) is a threshold of additional annual mortality, which could be sustained by the population. This concept has been widely applied in Procellariiformes and other seabirds (Niel and Lebreton 2005; Dillingham and Fletcher 2008, 2011; Barbraud et al. 2009; Kirby et al. 2009; Waugh et al. 2009, Žydeli et al. 2009; Tuck 2011) as it is calculated with minimal demographic information. If there is information on the level of bycatch, the application of this concept offers the opportunity to assess the relative impact of the fishery in the different populations/species. In order to estimate the PBR of each population we used the recent specific method developed for Procellariiformes (Dillingham and Fletcher 2011).

The PBR of a population of Procellariiformes could be determined by:

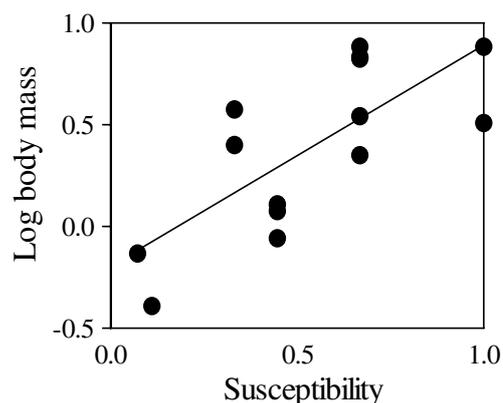
$$PBR = \tau f \hat{B}$$

where  $\tau$  is a coefficient that incorporates a species’ maximum growth rate (i.e.  $R_{max}$ ) and a species appropriate population multiplier ( $M$ ), and also included uncertainty in the estimate of the number of breeding pairs,  $\hat{B}$  is the estimated number of breeding pairs (Dillingham and Fletcher 2011) and  $f$  is a recovery factor between 0.1 and 1. A limiting value of 0.5

for  $f$  is suggested for most populations (Wade 1998, see also Niel and Lebreton 2005; Barbraud et al. 2009) as it is derived from protection against biases caused by estimates of population size,  $R_{\max}$ , and mortality. This value can also be selected based on the status of the populations of a species (according to IUCN criteria), with values of 0.1 for threatened species (i.e. Vulnerable, Endangered, Critically Endangered), 0.3 for species almost threatened and 0.5 for species not of conservation concern (Dillingham and Fletcher 2008, 2011). In the present study we used this approximation. The numbers of annual breeding pairs used to estimate the PBR are presented in Table 1. For *T. chlororhynchos* in the Tristan da Cunha group (including Tristan, Gough, Nightingale and Inaccessible) the estimation of the number of pairs is less precise (i.e. 21 700–35 800, Table 1). Consequently, for estimate the PBR we used the midpoint of this range. For *T. melanophrys* and *T. chlororhynchos* we estimated the PBR using the sum of the two estimated population sizes (Falkland Islands/Islands Malvinas – South Georgia and Tristan da Cunha and Gough, respectively). Finally, the coefficient  $\tau$  was estimated by the following equation  $\tau = 0.5M_{\min}R_{\max}$  (Dillingham and Fletcher 2011) using the calculation of  $R_{\max}$  explained above.  $M_{\min}$  is a species appropriate population multiplier that allow to estimate the 20th percentile of the population size (in individuals) from the estimated annual breeding pairs. This conservative estimate of the population is required for the estimation of the PBR (Dillingham and Fletcher 2008; Dillingham and Fletcher 2011). Therefore, we used the  $M_{\min}$  calculated in Dillingham and Fletcher (2011) assuming a coefficient of variation in the estimates of breeding pairs ( $CV_B$ ) of 0.5. The used values are as follows: 1)  $M_{\min} = 3.0$  for medium sized petrels (*Procellaria*), 2)  $M_{\min} = 3.2$  for large petrel (*Macronectes*) and annually breeding albatrosses (*Thalassarche*), 3)  $M_{\min} = 6.0$  for biennially breeding albatrosses (*Diomedea* and *Phoebetria*).

The values of PBR were further related to available estimates of incidental capture of the species with greatest risk according to the PSA. For the Uruguayan pelagic longline fleet estimate exist of the number of birds captured annually during 2004–2007 for the three-most frequently caught species (Jiménez et al. 2010). The average percentage of the annual observed effort by the national observer program (Programa Nacional de Observadores a bordo de la Flota Atunera Uruguaya, PNOFA) with respect to the total realized effort by the fleet during this period was 37% (range 26%–49%). This coverage was used to predict the total number of birds caught (Jiménez et al. 2010). During that period the mean number of captured individuals for *T. melanophrys*, *T. chlororhynchos* and *P. aequinoctialis* was 308–575 individuals, 26–110 individuals and 18–113 individuals, respectively. The other species are captured annually in low numbers, which complicates estimating the annual number of individuals captured with precision (Jiménez et al. 2010).

At the same time, within the framework of a campaign of ring recovery (<http://www.cicmar.org.uy/archives/246>) by the Proyecto Albatros y Petreles –Uruguay (PAP), effort has been underway with cooperating fishermen to collect rings of birds captured incidentally on trips not monitored by PNOFA since 2004–2005, allowing us to obtain several records of capture of *D. exulans* from South Georgia (Jiménez et al. 2008;



**Fig. 2.** Relationship between susceptibility and body mass (0.5 to 8 kg) for the seabird populations associated with the Uruguayan pelagic longline fleet in the Uruguayan slope and adjacent waters. Susceptibility is the product of the probabilities availability, access to the baits, selectivity and post-capture mortality (see Sect. 2; estimated values are presented in Table 3). The regression of susceptibility on body mass (Log transformed data) is significant ( $r^2 = 0.55$ ,  $p < 0.01$ ,  $n = 14$ ). Sources of body mass are: Pinder (1966); Tickell (2000); Brooke (2004); Cuthbert (2005) and Copello et al. (2006). When body mass values were available for both sexes, we used the average of the females.

PAP & DINARA unpublished data). During 2004–2007 both sources of information (i.e. PNOFA and ring reports) combined showed greater values of capture in 2005 and 2006, with at least 9 and 11 *D. exulans*, respectively, captured by the fleet. However, this number should be considered as a minimum since some ringed albatrosses could have been captured and not reported by fishermen. Moreover, it is unknown if there were captures of not ringed albatrosses in trips without observers. For the remaining species of large albatross (i.e. *D. epomophora*, *D. sanfordi* and *D. dabbenena*), and for *T. steeadi* and *P. conspicillata*, whose annual catches are also very low, additional information is unavailable due to the rings being only rarely recovered. With rates of capture less than those for *D. exulans* (Jiménez et al. 2008; PAP & DINARA unpublished data), it is reasonable to assume that during the same period the annual catch is unlikely to have exceeded 10 individuals, with *D. dabbenena* being probably the species least frequently caught.

### 3 Results

#### 3.1 Productivity/Susceptibility Analysis (PSA)

The information available allowed us to estimate productivity (Table 1) and susceptibility (Table 3) for 15 and 14 species, respectively. Although we initially considered 15 species, the PSA was ultimately carried out for 14 populations of albatrosses and petrels that make strong use of discards, offal and which are captured in the southwest Atlantic by pelagic longliners. The estimated susceptibility showed a significant linear relationship with the body mass (Fig. 2). The PSA indicated that the population of South Georgia *D. exulans* has the highest risk on the Uruguayan continental slope and

**Table 3.** Susceptibility for the main populations/species of albatrosses and petrels (codes in Table 1) associated with longline vessels in the Uruguayan slope and adjacent waters. Susceptibility was estimated as the product of the conditional probabilities: availability ( $p1$ ), access to bait ( $p2$ ), selectivity ( $p3$ ) and post-capture mortality ( $p4$ ) (see Sect. 2). Availability was measured using the observed frequency of occurrence (FO), around fishing vessels considering the different population sizes. Access to bait was estimated quantitatively from 48 longline sets and then translated into a semi-qualitative score; presented values are as follows: mean  $\pm$  standard error (SE) and the number of set with occurrence of the species from where estimates were performed. Hook selectivity was qualitatively estimated from the ratio of the length of the culmen and the length of the hook; presented values are as follows: mean  $\pm$  standard error (SE) of the culmen and the number of birds measured. Post-capture mortality assumed the following relationship: number of bird dead/number of bird captured  $\sim 1$ . The cut-off used to estimate the different attributes are presented in Table 2.

Species	Availability			Access to bait				Hook selectivity				Post-capture mortality	Susceptibility	
	%FO	FO	$p1$	Access	SE	$n$ of sets	$p2$	Mean culmen length (mm)	SE	$n$ of birds	Sources (*)	$p3$		$p4$
DEX	41	142	1.00	0.660	0.385	13	1.00	167	1	19	1	1.00	1.00	1.00
DDA	*		0.67	0.660	0.385	13	1.00	148	2	7	1	1.00	1.00	0.67
DEP	13	46	1.00	0.300	0.249	10	0.67	173	1	35	1	1.00	1.00	0.67
DSA	26	90	1.00	0.300	0.249	10	0.66	161	1	46	1	1.00	1.00	0.67
TST	19	66	0.67	0.495	0.168	13	1.00	135	1	24	1	1.00	1.00	0.67
TME	73	253	1.00	0.497	0.140	33	1.00	118	1	319	1	1.00	1.00	1.00
TCH	60	209	1.00	0.245	0.245	30	0.67	116	1	71	1	1.00	1.00	0.67
PFU	2	7	0.33	–	–	1	–	107	–	13	2	1.00	1.00	–
MHA	48		1.00	0.028	0.019	20	0.33	89	–	71	3	1.00	1.00	0.33
MGI	22	173*	1.00	0.028	0.019	20	0.33	81	–	27	4	1.00	1.00	0.33
PAE	69	240	1.00	0.413	0.139	36	0.67	52	–	20	1	0.67	1.00	0.45
PCO	71	249	1.00	0.231	0.074	25	0.67	51	–	20	5	0.67	1.00	0.45
DCA	52	183	1.00	0.008	0.008	20	0.33	32	–	53	6	0.33	1.00	0.11
FGL	22	75	0.33	0.000	0.000	6	0.33	45	1	15	7	0.67	1.00	0.07
PGR	67	234	1.00	0.406	0.140	30	0.67	47	–	30	8	0.67	1.00	0.45

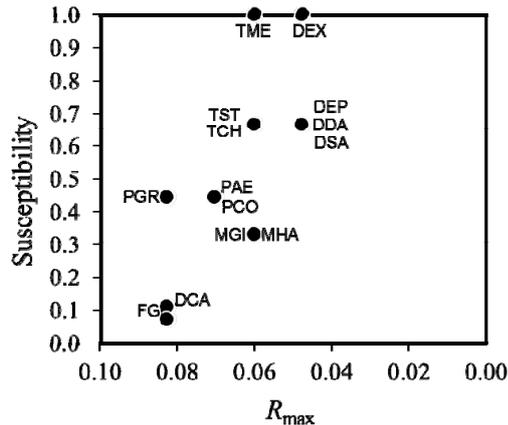
(\*) Sources: 1 Birds captured in pelagic longliners in the Southwest Atlantic, DINARA unpublished data, 2 Brooke 2004 (females), 3 González-Solís 2004 (females), 4 Copello et al. 2006 (fledgling females), 5 Hall 1987, 6 Ryan 1998, 7 Pinder 1966, 8 Spear and Ainley 1998, 9 Cuthbert 2005.

adjacent waters (Table 4, Fig. 3). The second population on the risk scale was *T. melanophrys* on the Falkland Island/ Islas Malvinas, followed in third place by *D. dabbenena* on Gough Island, *D. epomophora* on Campbell Island and *D. sanfordi* on the Chatham Islands. In fourth place, were *T. chlororhynchos* and *T. steadi* and in the fifth place were *P. aequinoctialis* and *P. conspicillata* (Table 4, Fig. 3). Sixth place was occupied by *P. gravis* and in the seventh place were both species of giant petrels (i.e. *M. halli* and *M. giganteus*). Finally, positions 8 and 9 were occupied by *D. capense* and *F. glacialoides*, respectively (Table 4). *Phoebetria fusca* was left out of the PSA (Table 4) because we did not obtain information about their catchability to estimate susceptibility.

In order to examine the robustness of the susceptibility results a simple exercise was conducted to evaluate their sensibility to the used arbitrary cut-offs. Considering that for a management strategy it is important not to under-estimate the susceptibility of the different species, potential forms of enhance susceptibility are reduce the cut off set to assign the probabilities of availability, access to the baits and selectivity. Therefore, we calculated the effects on the susceptibility of a proportional decrease in the cut off established to estimate qualitatively these attributes. Because two cut off were used by attribute in order to obtain qualitative probability values (i.e. low, medium and high), attributes were modified one at a time, simultaneously reducing both cut off at 5%, 10%, 20%,

**Table 4.** Results of the Productivity and Susceptibility Analysis (PSA) analysis for the populations/species of albatrosses and petrels (codes in Table 1) that interact with the Uruguayan pelagic longline fleet in the slope of Uruguay and adjacent waters.  $R_{max}$  and susceptibility values are from Tables 1 and 2 respectively.

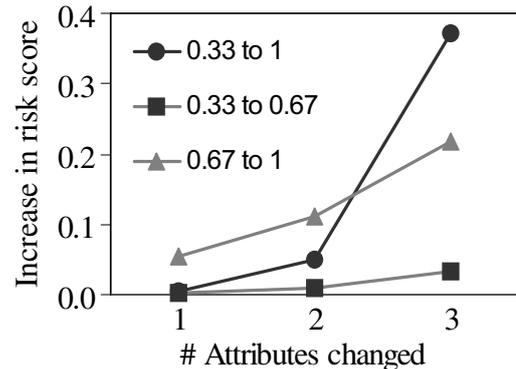
Species	$R_{max}$	Susceptibility	Euclidian distance ( $R_{max} = 1$ , Susceptibility = 0)	Rank of Risk
DEX	0.048	1.00	1.381	1
TME	0.060	1.00	1.372	2
DDA	0.048	0.67	1.163	3
DEP	0.048	0.67	1.163	3
DSA	0.048	0.67	1.163	3
TST	0.060	0.67	1.153	4
TCH	0.060	0.67	1.153	4
PAE	0.070	0.45	1.031	5
PCO	0.070	0.45	1.031	5
PGR	0.083	0.45	1.020	6
MHA	0.060	0.33	0.997	7
MGI	0.060	0.33	0.997	7
DCA	0.083	0.11	0.924	8
FGL	0.083	0.07	0.920	9
PFU	0.048	–	–	–



**Fig. 3.** PSA analysis for the mean species (codes in Table 1) of seabirds associated with the Uruguayan pelagic longline fleet in the Uruguayan slope and adjacent waters. Productivity was measured as the maximum rate of recruitment  $R_{max}$ . Susceptibility is the product of the probabilities availability, access to the baits, selectivity and post-capture mortality (see Sect. 2).

25%, 30% and 40% with respect to the lower cut off value. As expected, this analysis affected at first instance those species whose estimated values for some of its attributes were closer to the established cut off. In our case the species that increased their susceptibility faster were *P. aequinoctialis* and *P. gravis* due to a minimal decrease (i.e. 5%) in the cut off access to the baits. Greater changes in the cut off (i.e. reduction of 20% and 30% in the cut off) of the other two attributes were necessary to increase the susceptibility of other species (*F. glacialoides* modifying availability and *D. capense* modifying selectivity, respectively). The absolute change in species with greater initial susceptibility (*P. aequinoctialis* and *P. gravis*) generated a greater impact on the risk score with respect to species with lower initial susceptibility values (*F. glacialoides* and *D. capense*).

With the purpose of visualize this, a second analysis was conducted following Ormset and Spencer (2011) to test the sensitivity of the PSA to changes in the susceptibility attributes scores. Different PSAs focused on only one hypothetical population with an  $R_{max}$  of 0.048 were conducted. Risk scores were calculated for successive increases in the number of changed attribute scores. Because only three attributes have an effect in our estimated susceptibility (post capture mortality is supposed to be 1; see Sect. 2), this was the number of attributes used in the analysis. In our case three types of changes were considered: (1) all attributes starting at 0.33 were increased to 0.67; (2) all attributes starting at 0.67 were increased to 1; (3) all attributes starting at 0.33 were increased to 1. The increase in risk was greatest for the PSAs where susceptibility attributes scores were changed from 0.33 to 1 (Fig. 4). PSAs where attributes scores increased from 0.67 to 1 had greater increases in risk than those where attributes scores were increased by the same extent. This means that PSAs with higher initial susceptibility attributes scores produced a greater absolute change in risk than those with



**Fig. 4.** Effects of increasing susceptibility attributes scores on the overall risk score of a Productivity–Susceptibility Analysis for a hypothetical population with an  $R_{max} = 0.05$ . Three types of changes were considered: all attributes starting at 0.33 and increasing to 1; all attributes starting at 0.33 and increasing to 0.67; all attributes starting at 0.67 and increasing to 1. In our score system, 0.33, 0.67 and 1 represents low, medium and high, respectively (see Sect. 2).

lower susceptibility attributes scores, as observed by Ormset and Spencer (2011). Therefore, underestimate any of the attributes used to estimate the susceptibility in species with an intermediate or high susceptibility value can generate a greater absolute underestimation of risk, with respect to an underestimate of the same amount in species with low initial susceptibility values. As an example, an underestimation in the access to the baits in species with high availability and selectivity scores (most albatross species) has a greater impact on the estimated risk, than in those species of small petrels with low access to the bait as *F. glacialoides* and *D. capense*, in which underestimate the overlap or selectivity, respectively, produces an underestimate in the risk considerably lower. This justifies a second step to determine the most affected species by the studied fishery, which considers only the species most at risk according to the PSA (see next section).

In order to select a final PSA, we performed three alternative PSAs with those species that increased their susceptibility due to the above performed decreases in the cut off: (1) with an increased susceptibility of *P. aequinoctialis* and *P. gravis* due to an increased access to the bait (from 0.67 to 1.00); (2) with an increased susceptibility of *F. glacialoides* by an increase in the availability score (from 0.33 to 0.67); (3) with an increased susceptibility of *D. capense* caused by an increase in the selectivity score (from 0.33 to 0.67). In the first case *P. gravis*, exchange its risk place with *P. conspicillata* and in the other two scenarios *F. glacialoides* and *D. capense* exchanged the positions 8 and 9, depending on which species increased their susceptibility. These PSAs did not generate great changes in the ranking of risk; although there were slight differences in the order, these were among species with similar characteristics and no changes in the order between taxa with notable differences (i.e. body mass, diving ability, breeding frequency) were observed. The medium-sized petrel (*Procellaria* and *Puffinus*) did not reach the levels of risk of annual breeding albatrosses, while small petrels (*F. glacialoides* and *D. capense*) did not reach the levels of risk of the giant petrels (*Macronectes* spp.). Importantly, the differential productivity between taxa affects

the ranking of risk, even though different taxa obtained similar values of susceptibility. Therefore, the PSA presented in Figure 3 was considered appropriate.

### 3.2 Potential Biological Removal Level

The application of the concept of PBR was carried out for the populations of the nine species (Table 5) most affected by the fishery according to the PSA (PSA ranks 1 to 5; Table 4). The values of PBR estimated for these nine species suggest that the population most affected by this fishery is that of *D. exulans* in South Georgia, with records of at least 11 birds captured during 2006 (see Sect. 2), which is 54% of the PBR estimated (i.e. PBR = 20 individuals). In second place are the combined populations of *T. chlororhynchos* (Tristan da Cunha and Gough) with an estimated capture during 2006 of 110 birds, which is 34% of PBR. Although there is uncertainty with respect to the number of birds of *D. dabbenena* captured annually, the capture of few individuals per year (about 5 birds) would place this species in third place due to its low PBR (23 individuals). In the most conservative scenario (10 individuals) this species would be located in second place. In fourth place is *T. melanophrys*, whose mean capture is estimated at 575 birds during 2006, which is about 13% of the sum of PBR for the two colonies considered (Falkland Islands/Islands Malvinas and South Georgia). The low annual captures (i.e.  $\leq 10$  individuals) of *D. sanfordi*, *D. epomophora*, and *P. conspicillata* indicate that these species follow in the order of affected populations. Caution should be taken with these results, particularly with royal albatrosses, since they have very small populations and are biennial breeders, making them even less productive species. In the most conservative scenario (15 individuals caught per year) *D. sanfordi* and *D. epomophora* would be located in the fourth and fifth place, respectively, moving *T. melanophrys* to the sixth place. In eighth place is *P. aequinoctialis* with an estimated capture (i.e. 113 birds in 2006) near 1% of its PBR. Finally, there is uncertainty with respect to the number of birds of *T. steadi* captured annually (Jiménez et al. 2009b) in the Uruguayan fleet. The capture of 10 individuals per year would place this species in last place due to its high PBR (2913 individuals). Even in the most conservative scenario (20 individuals caught per year) this species would be located in the last place, with captures representing less than 1% of its PBR. It is important to mention that because this species is considered Near Threatened (Table 1), PBR was estimated using a  $f = 0.3$  (see Sect. 2). This resulted in a PBR three times bigger than a PBR estimated using a  $f = 0.1$  (here used for threatened species). In turn, this species is considered an annual breeder; however, there is recent evidence that suggests it is a biennial breeder species (Thompson and Sagar 2008; Agreement of the conservation of albatrosses and petrels-ACAP 2011). Therefore, caution should be taken if the global status of this species change and/or the biennial breeding frequency is confirmed.

## 4 Discussion

This study considered for the first time the relative impact of a fishery in the southwest Atlantic on various affected

**Table 5.** Estimation in the number of individuals of the Potential Biological Removal Level (PBR) and annual bycatch for the populations of the nine species most affected by the fishery according to the PSA (codes in Table 1). Species were listed in descending order of impact according to the percentage of PBR that comprised the estimates of bycatch. Sources of the bycatch estimation are presented in Sect. 2. Estimates correspond to the year 2006. \*: the catch values could be underestimated, \*\*: catch values were assumed due to the absence of information.

Species	Annual breeding pairs	PBR	Bycatch Estimate	% of the PBR
DEX	1 420	20	11*	54
TCH <sub>T.C.Is.</sub>	28 750	277		
TCH <sub>G.Is.</sub>	5 300	51		
TCH sum	34 050	328	110	34
DDA	1 514	22	5**	23
TME <sub>F/M</sub>	399 416	3 844		
TME <sub>SG</sub>	74 296	715		
TME sum	473 712	4 559	575	13
DSA	5 800	83	10**	12
DEP	7 800	112	10**	9
PCO	14 400	152	10**	7
PAE	889 122	9 385	113	1
TST	97 089	2 803	–	<1

seabird populations. The approximation was carried out in two steps, first the application of a PSA and then an evaluation using the concept of PBR, allowing us to understand the relative impact of the Uruguayan pelagic longline fleet on the majority of populations/species of albatross and petrel that are highly associated with it. The PSA, using a semi-quantitative analysis, illustrated which species are exposed to the greatest risk from pelagic longliners operating on the continental slope of Uruguay and adjacent waters. However, the ranking of risk estimated with the PSA should be interpreted with caution because of the numerous assumptions (see below). A second step in the analysis was to apply the concept of PBR to the populations of the eight species with greatest risk found in the PSA. Unlike for the PSA, the estimates of capture used to evaluate the relative impact on the various species include data from areas outside the continental slope of Uruguay and adjacent waters and involve the entire area of operation of the fleet (i.e. 20°–40°S and 20°–55°W). Together, both approaches allow us to conclude that the impact of the fishery on the different populations is not proportional to the rate of capture of each species. A commonality of both approaches is that the population most affected by the Uruguayan fleet is that of *D. exulans* in South Georgia. The main species in terms of number of annually captured individuals are *T. melanophrys*, *T. chlororhynchos* and *P. aequinoctialis* (Jiménez et al. 2010). However, other populations with low numbers of annual captures (*Diomedea* spp.; Jiménez et al. 2008, 2010) are among those most potentially affected. Though the present study supports the hypothesis that the species at greatest risk are the large albatrosses, *T. chlororhynchos* would also be among the most affected species.

It is important to highlight some sources of uncertainty in both approaches. For example, we used qualitative information

to estimate some parameters for susceptibility (i.e. availability and selectivity). Further work aimed at obtaining more quantitative data would help to estimate the susceptibility of these species with greater precision. DIM was developed to apply in populations with limited and incomplete demographic information (Niel and Lebreton 2005; Dillingham and Fletcher 2008). This is the main argument in favor of using  $R_{\max}$  obtained from the DIM as a measure of productivity for each population in the PSA and also for estimating PBR since this method requires fewer biological attributes compared to other approaches, notably it does not require estimates of maximum age or fecundity (see the same approach in Waugh et al. 2009; Filippi et al. 2010). Consistent with this point, Skalski et al. (2008) highlights how large the sample size needs to be to estimate maximum age for species with high survival rates, and that it may be better to ignore senescence. Dillingham and Fletcher (2008) discussed the influence of adult survival ( $s$ ) and age of first reproduction ( $\alpha$ ) in the estimation of  $\lambda_{\max}$  and thus PBR (Dillingham and Fletcher 2008). In general the use of  $s$  from populations in optimal conditions is desirable, but in practice these values are not available. Since current estimates of adult survival come from populations subject to different levels of depletion, we used assumed values based in the body mass and breeding frequency (following Dillingham and Fletcher 2011) in order to make valid interspecific comparisons. A comparison of the assumed  $s$  optimal values against the  $s$  current values for the addressed albatross populations (Table 6) denotes the level of stress in these populations. Long-lived seabird populations, as Procellariiformes, are highly sensitive to changes in adult survival (Croxall and Rothery 1991; Weimerskirch et al. 1997; Cuthbert et al. 2003), and there is a significant relationship between adult survival and observed population growth rates (Cuthbert et al. 2003). Between *Diomedea* spp., the greatest level of stress appears to occur in *D. dabbenena* followed by *D. exulans* from South Georgia (Table 6). Although both populations are affected by fisheries, the first population it is also known to be highly affected by introduced mice at Gough Island (Wanless et al. 2009); however, it is important to mention that there is not published data on the  $s$  of the South Georgia population of *D. exulans* for the last decade. During this last period the situation of this population has aggravated (Poncet et al. 2006; see below). Leaving aside the population of South Georgia of *T. melanophrys*, which is probable the rarest in the study area (see Sect. 2), both populations of *T. chlororhynchos* appear to be more stressed than the population of *T. melanophrys* of Falkland Islands/Islands Malvinas (Table 6).

#### 4.1 Implications for conservation

Not all the populations of the nine species identified here as facing a higher risk of impact are decreasing. The most dramatic decline is undoubtedly occurring in the population of *D. exulans* in South Georgia, with an annual rate of reduction of 4% (Poncet et al. 2006). Also important is the reduction (ca. 1% annually) of the world population of *D. dabbenena*, attributed to the combined effects of predation by invasive mammals and incidental capture in longlines (Wanless et al. 2009).

**Table 6.** Comparison between values of adult survival ( $s$ ) assumed for optimal conditions (following Dillingham and Fletcher 2011) and current values (with their respective sources) for the population of albatrosses (codes in Table 1) associated with longline vessels in the Uruguayan slope and adjacent waters.

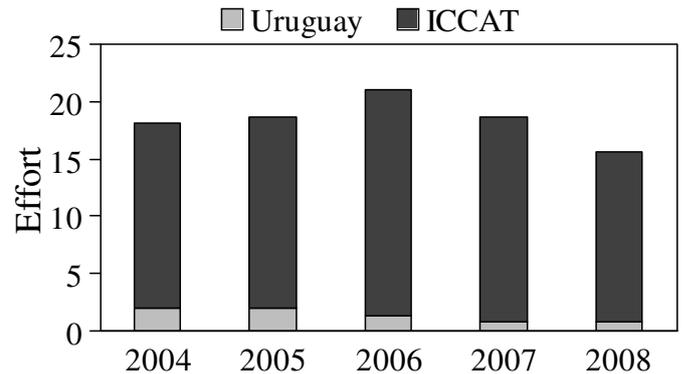
Species	$s$		Sources
	Assumed	Current	
DEX	0.96	0.93	Croxall et al. 1998
DDA	0.96	0.91	Wanless et al. 2009
DEP	0.96	0.94	ACAP 2009a
DSA	0.96	0.95	ACAP 2009b
TME <sub>F/M</sub>	0.95	0.94	Catry et al. 2011
TME <sub>SG</sub>	0.95	0.91	Arnold et al. 2006
TCH <sub>T.C.Is.</sub>	0.95	0.92	Cuthbert et al. 2003
TCH <sub>G.Is.</sub>	0.95	0.84	Cuthbert et al. 2003

The situation of *T. chlororhynchos* is not clear due to the uncertainty that exists with respect to its population size. However, population modeling of the better known reproductive sites predicts an annual rate of reduction of 1.5–2.8% on Gough Island, and 5.5% on Tristan da Cunha (Cuthbert et al. 2003). The worldwide population of *P. conspicillata* is increasing at a rate of 7% annually (Ryan and Ronconi et al. 2011), while the first demographic study of the Falkland Islands/Islands Malvinas population of *T. melanophrys* suggests that there is no evidence of decline (Catry et al. 2011). On South Georgia the latter species is declining (Poncet et al. 2006), but this appears to be atypical of the area considered here (see Sect. 2). For *P. aequinoctialis* on South Georgia it is not possible to know if there is a negative tendency in the population, though this population would be declining (Martin et al. 2009). For *D. epomophora*, the absence of regular and comparable counts on Campbell prevents a clear understanding of population trends, but it is thought they may be stable (ACAP 2009a). Finally, in the case of *D. sanfordi*, the population tendency on Chatham Island is unknown (99% of the global population). Consequently, some populations are experiencing declines or their situation is unknown, requiring greater attention to evaluate the impact of any cause of mortality caused by humans, while others (i.e. *P. conspicillata* and *T. melanophrys* on the Falkland Islands/Islands Malvinas), although not assumed to be unaffected by pelagic longlines, could be considered to have lower priority.

Leaving aside threats on land (e.g. predation by invasive species), fisheries are without doubt the main source of mortality at sea for the populations of albatross and petrel while in the southwest Atlantic, yielding at least a combined 10 000 avian mortalities per year (Favero et al. 2003, 2011; González Zevallos and Yorio 2006; Bugoni et al. 2008; Jiménez et al. 2010). Some species, such as *T. melanophrys* and *P. aequinoctialis*, are affected by several fisheries (e.g. demersal and pelagic longline, trawling; Vaske 1991; Favero et al. 2003, 2011; Gandini and Frere 2006; Gómez Laich et al. 2006; Gómez Laich and Favero 2007; Seco Pon et al. 2007; González Zevallos and Yorio 2006; Bugoni et al. 2008; Jiménez et al. 2010; Tuck et al. 2011) and currently it is not possible to attribute greater importance to any particular fishery as the leading source of mortality. However, some species with more oceanic distribution ranges (i.e. from the continental slope to the oceanic waters) and/or which range

over warm waters to the north of the Brazil-Malvinas Confluence (BMC), overlap more with the pelagic longline fisheries (e.g. *D. exulans*, *D. dabbenena*, *P. conspicillata*, *T. chlororhynchus*), compared to species that are mainly abundant over the shelf and continental slope in cold waters associated with the Malvinas current (i.e. *T. melanophrys*, *D. epomophora*, *D. sanfordi* y *P. aequinoctialis*), where the trawling fishery directed at hake (*Merluccius hubssi*), and to a lesser extent demersal longliners, are also a source of mortality (Favero et al. 2003, 2011; González Zevallos and Yorío 2006). Considering the population status mentioned above, three populations (i.e. *D. exulans* on South Georgia, *D. dabbenena* and *T. chlororhynchus* on Tristan da Cunha and Gough) could be affected by bycatch in pelagic longliners more than by any other form of mortality, while in the southwest Atlantic. In this study these were the three-most affected species when assessing the impact of the Uruguayan pelagic longline fleet through the PBR approach. Although derived from a semi-quantitative analysis, the PSA also ranked the first two as among the three species with greatest risk on the Uruguayan continental slope and adjacent waters.

We present here a case study of one of the fisheries that operates in the southwest Atlantic, mainly over part of the BMC. Considering the fishing effort that the pelagic longline fleets expend over the region of the BMC, undoubtedly the area of greatest concentration of effort over the last decades in the southwest Atlantic (Tuck et al. 2003; Lewison et al. 2005; Huang 2011), there is no doubt that some populations are seriously affected. Fleets flying the flags of Taiwan, Japan, Spain, and Brazil, among others (see ICCAT, Task II Catch & Effort (T2CE) <http://iccat.int/Data/t2ce.rar>), operate in this region throughout the year. In fact, the Uruguayan fleet within its operation area was responsible for only the 4.3% to 12.5% of the total effort deployed by the different fleets (Fig. 5). The species have different catchabilities with respect to each of these fleets, due to various factors, such as differences in fishing operations, sinking rates of the baited hooks, and hook sizes. The catchability used in the PSA for the Uruguayan fleet was estimated based on observations of attacks on bait under a regimen of sinking rates and hook sizes which does not necessarily correspond to those of other fleets. We also considered a period when mitigation methods in Uruguay were not in effect, the application of which is also uncertain in many fleets operating in the region. However, the combined impact of these fleets could be sufficiently high to account for many of the observed declines in the populations of *D. exulans*, *D. dabbenena* and *T. chlororhynchus*. Particularly, in this study we showed that despite the capture of only a few individuals, the population of *D. exulans* from South Georgia is the most affected. Moreover, the bycatch of this species in the study region is known to have very strong bias towards adult females (Croxall and Prince 1990; Jiménez et al. 2008), which strongly exacerbates the population impact. This is because sex-biased mortality decreases the number of potential breeding pairs and reduces the fecundity of the population (Nel et al. 2002) and adult survival is the most important demographic parameters influencing on population trends (Croxall and Rothery 1991; Tasker et al. 2000; Véran et al. 2007).



**Fig. 5.** ICCAT fishing effort (millions of hooks) reported for the region between 20°–45°S and 20°–55°W and the period 2004–2008. The Uruguayan fishing effort is represented in Grey. Sources: ICCAT fishing effort (Task II Catch & Effort (T2CE) <http://iccat.int/Data/t2ce.rar>); Uruguayan fishing effort (DINARA and Jiménez et al. 2010).

It should be noted that this study used the PBR as a tool for understanding the relative impact of a specific fleet in a particular fishery, but this concept has a broader scope. The concept of PBR suggests that any source of mortality that comes close to the PBR value could result in a population decline (Wade 1998; Dillingham and Fletcher 2008, 2011). However, this threshold may be compared with the total number of human-caused mortalities worldwide from all possible sources. Even leaving aside other threats, if we considering the total effort of the pelagic longline fleets operating in the region (c.a. 15.5 to 21 million of hooks per year; Fig. 5), the values of PBR could be easily exceeded. The approximation that we used in this study (i.e. using  $f = 0.1$  for threatened species) leads to a conservative estimation of PBR. A non-conservative approximation, e.g.  $f = 0.5$ , though not recommended for threatened species (Dillingham and Fletcher 2008), would result in a PBR five times higher than what we estimated for the threatened albatross and petrels species. Even with such higher PBR values the populations of *D. exulans* (South Georgia), *D. dabbenena* and *T. chlororhynchus* could still face significant threats.

The remaining populations also require attention, mainly those whose status is not clear and whose size is very small, such as both species of royal albatross (*D. sanfordi* and *D. epomophora*). Though the population of *P. conspicillata* has increased, *T. melanophrys* in the Falkland Islands/Islands Malvinas may be stable, and *P. aequinoctialis* is potentially decreasing, the first species requires more attention because population size is very small and it is endemic to a single island. The populations less affected by pelagic longlines in the studied area would be *T. steadi*, *P. gravis*, *M. halli*, *M. giganteus*, *F. glacialoides* and *D. capense*. It was not possible to gain an understanding of the impact of this fishery on *P. fusca*. This species require further investigation, because its population on Tristan da Cunha and Gough is small (Table 1). However, the bycatch of this species in the study fleet it is considered very rare.

Applying mitigation measures to reduce the impact of pelagic longline fleets operating in southwest Atlantic should be considered a high priority for the populations of

*D. exulans* (South Georgia), *T. chlororhynchos* (Tristan da Cunha and Gough) and *D. dabbenena* (Gough). Other priority populations that require more attention are, in order of importance, *D. sanfordi*, *D. epomophora*, *P. conspicillata*, *T. melanophrys* on the Falkland Islands/ Islas Malvinas, *T. melanophrys* on South Georgia and *P. aequinoctialis* (South Georgia).

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