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Risk assessment and relative impact of the Uruguayan pelagic longliners in seabirds

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Risk assessment and relative impact of the Uruguayan pelagic longliners in 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 then 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 for most of the populations or species of albatross and petrel with high association with this fishery. Of 15 species addressed, were finally evaluated 11 for which we obtained a ranking of risk. The concept of PBR was applied to the eight most at risk species. The impact of fishing on populations could not be straightforwardly presumed from their by-catch rates. The 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 would 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. Assess the overall impact of pelagic longline fleets operating in this region should be considered a high priority in some populations.

Introduction

Fisheries are major anthropogenic activities carried out in all oceans over 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 & 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) results often in bycatch and mortality of seabird in some fisheries (e.g. longline, trawl and gillnet; Brothers et al. 1999, Tasker et al. 2000). Undoubtedly, the bycatch of albatrosses and petrels in longline has historically drawn most of the attention at a global level (Alexander et al. 1997, Robertson & Gales 1998, Brothers et al. 1999), as it has been associated as one of the main causes of the decline of several populations (Gales 1998).

At least 15 species of seabirds have been caught by the Uruguayan pelagic longline fleet, being the continental slope of the South-Western Atlantic and the period from May to November the area and time, respectively, with greater interaction for many of these species (Jiménez et al. 2009a, 2009b, 2010). Estimates indicate that catches during 2004-2007 were in the order of several hundreds of birds caught annually, with *Thalassarche melanophrys*, *Procellaria aequinoctialis* and *Thalassarche chlororhynchos* the main species caught. In turn, other species are captured in very small proportions (Jiménez et al. 2008, 2010). However, the significance of these captures on their populations still remains poorly understood. Differences in population sizes and recovery capabilities (in terms of productivity) of each

population could determine that the impact caused by this fishery in their populations is not a reflection of their relative bycatch.

Procellariiformes shared extreme demographic characteristics, exhibiting a high adult survival rate, high longevity and low fertility (highly delayed sexual maturity, a single egg). Within this order, the great albatrosses (*Diomedea* spp) have the most extreme characteristics (Wales 1998, Tasker et al. 2000, Furness 2003, Brooke 2004), being the less productive species. Furthermore, empirical evidence shows that species that are more likely to access to baited hooks during setting in the pelagic longline fishery, are those with greater ability to dive or with larger beaks that can feed on large items such as baits. However, body size is a determining factor in the outcome of inter-specific interactions for baits (Jiménez et al. ACAP BVWF-4a, ACAP BVWF-4b), which states that great albatrosses have a greater advantage to access them. Given its low productivity and potentially high susceptibility to bycatch, great albatrosses would be located in a high risk scenario.

This paper aims to obtain a view of the relative impact that the Uruguayan pelagic longline fleet has on the different population of Procellariiformes. Specifically, it seeks to answer if bycatch by the Uruguayan pelagic longline fleet represents a relatively greater risk for great albatross species than to other smaller Procellariiformes. In order to address this hypothesis, we make a Productivity and Susceptibility Analysis (PSA; Hobday et al. 2011) to determine those species that are at greater risk. Also, we apply the concept of PBR (Wade 1998) to assess the relative impact.

Materials and Methods

The fishery and study area

The Uruguayan pelagic longline fishery fleet directs its fishing effort towards swordfish (*Xiphias gladius*), tuna (*Thunnus albacares*, *Thunnus obesus* and *Thunnus alalunga*), and pelagic sharks (principally *Prionace glauca*) since 1981 in a large area of the southwestern Atlantic Ocean (Pons & 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 et al. 2010), using as its main fishing gear an American style longline (monofilament mother 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). Based on the data on incidental capture employed, the study covers the period 2004-2007. However, we used data obtained in 2005-2008 (i.e. bird counts) and 2005-2010 (i.e. observations of attacks on baits) (see below). The principal oceanographic characteristic defining the study area is the occurrence of the Brazil-Malvinas Confluence (BMC), formed by the meeting of the currents from Brazil and the current of Malvinas (Olson et al. 1988, Acha et al. 2004, Ortega & Martínez 2007).

Productivity and Susceptibility Analysis (PSA)

The PSA, in the framework of an Ecological Risk Analysis (ERA) for species/populations exploited by a fishery, captured incidentally and threatened or protected, helps identify which of a group of species that interact with a given fishery are the most vulnerable and which are

exposed to the greatest risk. (Cortés et al. 2010, Patrick et al. 2010, Arrizabalaga et al. 2011, Hobday et al. 2011). In the context of an ERA, this analysis is considered primarily a level 2 or semi-quantitative analysis, due to the nature of the information required (Hobday et al. 2007, 2011). The risk (r) is estimated as the Euclidian distance from the point (productivity = 0, susceptibility = 1) of the graph of productivity against susceptibility (Hobday et al. 2011). 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* on both adjacent archipelagos situated in the South central Atlantic (Brooke 2004, Onley & 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, Nueva Zelanda; ACAP 2009b), *Thalassarche steadi* (>99% of the population on Auckland Islands, Nueva Zelanda; 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 principal population that visits the study area. For the case of *T. melanophrys* we assumed that the majority of individuals that arrive in Uruguay and adjacent waters come from the Falkland Islands/Islas Malvinas, while a substantially smaller proportion arrives from South Georgia (Tickell 1967, Phillips et al. 2005, Jiménez et al. 2010). Of the recaptures of birds ringed as chicks on the Falkland Islands/Islas Malvinas, 88% ($n = 66$) took place on the Atlantic coast of South America, while only 4% ($n=278$) of the recaptures of chicks ringed on South Georgia occurred in this region (Tickell 1967, Prince et al. 1998). If it is considered that the Falkland Islands/Islas Malvinas population is more than five times greater in terms of number of mated 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. In *P. aequinoctialis*, we assume that the birds come from South Georgia, neglecting the very small population of the Falklands Islands/Islas Malvinas (ver Phillips et al. 2006, Jiménez et al. 2009a, 2010). For *Phoebetria 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 they occupy an important proportion of the observed birds in Uruguay and adjacent waters. 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, M. Abreu pers. comm.). 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 Argentinian 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. Consequently, we consider the *M. halli* population to be from South Georgia and the *M.*

giganteus population to be from the coast of Argentina. *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, in the majority, come from other South Georgia islands where a ringing program has not been undertaken extensively. Finally, the 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 & Lebreton (2005) in the context of conservation biology and is highly applicable to Procellariiforms (see Niel & Lebreton 2005, Dillingham & Fletcher 2008, Babraud et al. 2009, Kirby et al. 2009, Waugh et al. 2009), where there are frequently few data for threatened or rare species. The productivity can be estimated using the rate of maximum growth (λ_{max}), which is the rate of annual growth of a population of a species without limiting factors and a low population density (Niel & Lebreton 2005). A key assumption of the MID is that the λ_{max} per generation does not vary and is a nondimensional number independent of the body size. This method allows the calculation of λ_{max} in long-living bird species, knowing only the age of first reproduction (α) and the survival of adults (s). It yields results similar to matrix models with complete demographic models according to the following equation (Niel & Lebreton 2005):

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

This formulation assumes constant fecundity and constant survival of adults after first reproduction. The maximum rate of population recruitment (R_{max}) and λ_{max} are related by $R_{max} = \lambda_{max} - 1$ (Dillingham & 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 & Lebreton 2005). In the present study we estimated R_{max} for all the species with a high level of association with the fishery (when available information permitted) (according to Jiménez et al. ACAP BCWG-4). The parameters for each of the species were obtained primarily from the evaluations of species in the Agreement on Conservation of Albatross and Petrels, ACAP (<http://www.acap.aq/acap-species?id=12>), and also from the available bibliography. Whenever possible we considered estimated values for populations known or considered to occur in the SW Atlantic (see above). In those populations where it was not possible to find these values, we used the values obtained for other colonies of the same species as an approximation.

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 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 waste, 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 attributes 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. proportion of the population that overlaps in space and time with the fishery) we 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 an 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 0.333 (low), 0.667 (medium) and 1.000 (high). In order to relativize the observed %FO with their 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 %FOs 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. (ACAP BCWG-4) and corresponds to the period 2005-2008. For the calculations of %FO the counts realized during the setting and hauling of the longline were used, discounting counts in which no fishing activity was carried out (i.e. during navigation). Two pairs of species were grouped during the counts (see Jiménez et al. ACAP BCWG-4), *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: 1) based on data for incidental capture obtained in the study area we know that the majority of *D. exulans* s.l. belong 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 to this species a %FO of 5-10% in the area considered. 2) For the case of *Macronectes* spp we discriminated between species in 97 of 113 counts where this genera 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 during all the period 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 sets during 2005-2010 (Jiménez et al. ACAP BCWG-4). These observations were used to estimate the probability of access to bait. The methodology is detailed in Jiménez et al. (ACAP BCWG-4). For each species we used the total number of contacts with the bait obtained during exclusively primary attacks (i.e. primary attacks that did not lead to multiple attacks) added to the total number of contacts with the bait as the last individual during multiple attacks. In order to obtain a probability value for the access to bait, the result of this sum was divided by the number of individuals present during the observations (Table 2). Primary attacks from the surface in which no contact was observed between the bird and the bait were not considered in the analysis.

SELECTIVITY.

As an attribute of selectivity we used the ratio between the length of the beak of each species and the hook (Table 2). To determine the probability of capture (i.e. selectivity) we used as a proxy the ratio of the length of the beak (culmen) and the length of the hook 9/0 for swordfish used by the fleet. 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 0.333 (low), 0.667 (medium) and 1.000 (high) we considered a species to have a low probability of being captured when its beak length was less than the front length of the hook (Fig. 1). The probability of capture is medium when the length of the beak is greater or equal to the front length of the hook and less or equal to the total length of the hook (Fig. 1). Finally, it was considered high when it was greater than the total length of the hook (Table 2). Note that results other than capture 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 species, the point assigning system was considered appropriate. A total of 6 types of hooks "J" 9/0 used by the fleet were measured (Fig. 1), with a median total length of 80.0 (SE ±0.8) mm and a median front length of 43.1 (SE ±0.9) mm. The length of the beaks of each species was obtained from incidentally captured individuals from the pelagic longlines of Uruguay and Japan which operate in Uruguayan and adjacent waters in the southwestern Atlantic (n = 541 birds; DINARA unpublished manuscript), or from the literature.

Potential Biological Removal Level

The PBR (Wade 1998) of a population is determined by:

$$PBR = N \min \frac{1}{2} R \max Fr$$

where N_{min} is an estimation of the minimum population size, $1/2R_{max}$ is half the rate of maximum population recruitment and Fr is a factor of recuperation between 0.1 and 1. A limiting value of 0.5 for Fr is suggested for most populations (Wade 1998, ver también Niel & Lebreton 2005 y Barbraud et al. 2009) as it is derived from protection against biases caused by population size estimates, R_{max} and estimations of mortality. This value can also be selected based on the status of the populations of a species (according to the IUCN), 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 & Fletcher 2008). In the current study we used this approximation and as all the species analyzed are threatened (Table 1) we used a value of $Fr = 0.1$. To calculate the PBR, a conservative estimate of the population size (N_{min}) in number of individuals is required. In this study we estimated the total number of individuals of each population based on the available estimations of the number of annually reproducing pairs (Table 1). However, it must be considered that part of the population does not return to the colonies in a given year, which complicates the estimation of the population size. In the species considered that have biennial reproduction (i.e. *Diomedea* spp) the only estimation of number of individuals reported (including all ages) was for *D. dabbenena*. For this species we estimated a population size of 1514 pairs during 2008 on Gough Island and circa 11300 individuals considering all age classes (Wanless et al. 2009). Assuming a similar relation between the number of annual pairs and the population size, we estimated a population of circa 10600 individuals for *D. exulans* in South Georgia (1420 annual pairs), circa 58200 individuals for *D. epomophora* (7800 annual pairs) and circa 43300 individuals for *D. sanfordi* (5800 annual pairs). In the two populations of petrels considered (*Procellaria* spp.) there is no existing estimation of the number of individuals of all ages calculated directly from information about their population. The scarcity of studies estimating the proportion of mature and immature individuals in populations of Procellariiforms is mentioned in Martin et al. (2009). Based on information estimated for *Pterodroma sandwichensis*, which indicates that 48% of the population would be of reproductive age, these authors estimated that *P. aequinoctialis* in South Georgia have a population size of circa 3.7 million birds of all ages (889122 annual pairs). The same relation applied to its sister species *P. conspicillata* (14400 annual pairs) suggests a population size of circa 60000 individuals. The same approximation was carried out for *T. melanophrys* and *T. chlororhynchos*, as we were unable to find estimations in the literature of the number of immature birds (or the ratio of immatures to mature individuals) for species with an annual reproductive cycle like the petrels mentioned. For *T. melanophrys* in the Falkland Islands/Islands Malvinas (399416 annual pairs) the population size was estimated at circa 1650000 individuals. For the population of this species in South Georgia (74296 annual pairs) we estimated a population size of circa 310000 individuals. For *T. chlororhynchos* in Gough Island (5300 annual pairs) we estimated a population size of circa 22100 individuals. For *T. chlororhynchos* in the Tristan da Cunha group (including Tristan, Gough, Nightingale and Inaccessible) the estimations of the number of pairs is less precise (i.e. 21,700-35,800, Table 1). Consequently the population was estimated as circa 90400-149000 individuals. Finally we estimated N_{min} as the 20th percentile assuming a coefficient of variation $CV = 0.5$ for each population, following Dillingham & Fletcher (2008). For *T. melanophrys* and *T. chlororhynchos* we estimated PBR using the sum of the two estimated population sizes (Falkland Islands/ Islas Malvinas -South Georgia and Tristan da Cunha and Gough, respectively).

The values of PBR were related to the available estimations of incidental capture of the species with greatest risk according to the PSA. For the Uruguyan pelagic longline fleet estimations exist of the number of birds captured annually during the period 2004-2007 for the three most captured species (Jiménez et al. 2010). During the mentioned 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). During the period mentioned the National Program of Observers aboard the Uruguyan Tuna Fleet (PNOFA) of DINARA covered approximately 35% of the total effort. At the same time,

within the framework of a campaign of ring recovery (<http://www.cicmar.org.uy/archives/246>) by the Albatross and Petrel Project –Uruguay, since 2004-2005 an effort has been underway along with fishermen to collect rings of birds captured incidentally on voyages not monitored by PNOFA, allowing us to obtain several records of capture of *D. exulans* from South Georgia (DINARA, unpublished data). During the period 2004-2007 both sources of information combined showed greater values of capture in 2005 and 2006, with at least 9 and 11 *D. exulans* respectively captured by the fleet. For the remaining species of large albatross (i.e. *D. epomophora*, *D. sanfordi* and *D. dabbenena*), and for *P. conspicillata*, whose annual captures are also very low, additional information is unavailable due to the rings being only rarely recovered. With rates of capture less than *D. exulans* (DINARA unpublished manuscript), it is reasonable to suppose that during the same period the annual capture is unlikely to exceed 10 individuals, *D. dabbenena* being the least captured species.

Results

Productivity/Susceptibility Analysis (PSA)

The results of the estimation of productivity and susceptibility for the various populations/species are given in Tables 1 and 3, respectively. Despite 15 species being considered, the PSA was ultimately carried out for 11 populations of albatross and petrels making strong use of waste and which are captured in the southwestern Atlantic by pelagic longliners. The PSA indicated that the population of South Georgia *D. exulans* presents the highest risk on the Uruguayan continental slope and adjacent waters (Table 4, Fig. 2). The second population on the scale of risk was *T. melanophrys* on the Falkland Island/ Islas Malvinas, followed in third place by *D. dabbenena* on Gough Island. In fourth and fifth place, and close together (Fig. 2), were both species of royal albatross, *D. sanfordi* on the Chatham Islands and *D. epomophora* from the Auckland Island group, respectively. In sixth and seventh place were *P. aequinoctialis* and *T. chlororhynchos* (using the values of Rmax from Tristan da Cunha), two of the main species captured in the fishery. Finally, positions 8 through 11 were occupied by *P. conspicillata*, *M. giganteus*, *D. capense* and *F. glacialoides*, respectively (Table 4). Four species were left out of the PSA (Table 4) due to lack of basic information needed to determine Rmax (i.e. *T. steadi*, *M. halli* and *P. gravis*) or because we did not obtain information about their catchability in order to estimate susceptibility (*P. fusca*).

“Potential Biological Removal Level”

The application of the concept of PBR was carried out for the populations of the eight species most affected by the fishery according to the PSA (Table 5). The values of PBR estimated for these eight species suggest that the population most affected by this fishery corresponds to *D. exulans* in South Georgia, with records of at least 11 birds captured during 2006 (see Materials and Methods), which is 56% 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 (110 birds) between 31% and 46% of PBR. Although uncertainty exists with respect to the number of birds of *D. dabbenena* captured annually, the capture of few individuals per year (about 5 birds) would locate this species in third place due to its low PBR (22 individuals). In the most conservative scenario (10 individuals) this species would be located in second place with *T. chlororhynchos*. In fourth place is *T. melanophrys*. The mean capture is estimated at 575

birds during 2006, which would correspond to about 16% of the sum of PBR for the two colonies considered (Falkland Islands/Islas Malvinas and South Georgia). The low annual captures (i.e. ≤ 10 individuals) of *D. sanfordi*, *D. epomophora*, and *P. conspicillata* indicate that these species follow in the order of affected populations. Finally, *P. aequinoctialis* is the least affected by the fishery of all the species analyzed with an estimated capture (i.e. 113 birds in 2006) near 1% of its PBR.

Discussion

This study considered for the first time the relative impact of a fishery in the southwestern Atlantic on various affected populations. The approximation was carried out in two steps, first the application of an PSA and then an evaluation using the concept of PBR, permitting us to understand the relative impact of the Uruguayan pelagic longline fleet for the majority of populations/species of albatross and petrel that are highly associated with it. The PSA, using a semi-quantitative analysis, offered a scenario illustrating which species are exposed to the greatest risk from pelagic longliners operating on the continental slope of Uruguay and adjacent waters. Consequently, the order of risk estimated with the PSA should be interpreted with caution (see below). A second step in the analysis was to apply the concept of PBR for the populations of the eight species with greatest risk, according to the PSA. Note that, unlike for the PSA, the estimations of capture used to evaluate the relative impact on the various species includes data from areas outside the continental slope of Uruguay and adjacent waters and involves the entire area of operation of the fleet (i.e. 20°-40°S and 20°-55°W). Together, both approximations permit us to affirm that the impact of the fishery on the various populations is not proportional to the rate of capture of each species. A commonality of both approximations is that the population most affected by the Uruguayan fleet is the population of *D. exulans* in South Georgia. The main species in terms of number of annually captured individuals are *T. melanophrys*, *T. chlororhynchos* y *P. aequinoctialis* (Jiménez et al. 2010). Nevertheless, other annually captured populations which in terms of numbers are very small (*Diomedea* spp; Jiménez et al. 2008, 2010) are found among the potentially most affected. Though the present study supports the hypothesis that the species at greatest risk are the large albatross, *T. chlororhynchos* would also be among the most affected species.

It is important to highlight some points that generate uncertainty for both approximations. For example, we used qualitative information to estimate some parameters for susceptibility (i.e. availability and selectivity). Further investigation that obtained these data would help to estimate the susceptibility of these species with greater precision. Dillingham & Fletcher (2008) discuss the sensitivity of the survival of adults (s) and the age of first reproduction (α) in the estimation of λ_{max} and thus PBR (Dillingham & Fletcher 2008). In general the use of an s from populations in optimal conditions is desirable; however, in practice these values are not available. In some cases in this study it was necessary to use values of α and/or s other than those predominant in the southwestern Atlantic. However, the MDI was developed to apply in populations with limited and incomplete demographic information (Niel & Lebreton 2005, Dillingham & Fletcher 2008). This is the principal argument in favor of using R_{max} as a measure of productivity for each population in the PSA and also for estimating PBR (see the same approximation in Waugh et al. 2009, Filippi et al. 2010).

Implications for conservation

Not all of the populations of the eight species identified here as facing a higher risk of impact are decreasing. Doubtless the most dramatic decline is 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 (circa 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). The situation of *T. chlororhynchos* is not clear due to the uncertainty that exists with respect to its population size. However, population modelling of the better known reproductive sites predicts an annual rate of reduction of 1.5-2.8% on Gough Island, and 5.5% (though not significant) on Tristan da Cunha (Cuthbert et al. 2003). The worldwide population of *P. conspicillata* is increasing at a rate of 7% annually (Ryan & Ronconi et al. 2011), while the first demographic study of the population in the Falkland Islands/ Islas Malvinas 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 Methods). 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 its tendency. However, it may be stable (ACAP 2009a). Finally, for 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 in order 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/ Islas Malvinas), although not to be supposed unaffected by pelagic longlines, could be considered to have lower priority.

Leaving aside threats on land (e.g. predation by invasive species), the fisheries are without doubt the principal source of mortality at sea for the populations of albatross and petrel while they are in the southwest Atlantic, yielding at least a combined 10000 avian mortalities per year (Favero et al. 2003, 2011, González Zevallos & Yorio 2006, Bugoni et al. 2008, Jiménez et al. 2010). Some species, such as *T. melanophrys* and *P. aequinoctialis* are affected by various 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 & Favero 2007, Seco Pon et al. 2007, González Zevallos & Yorio 2006, Bugoni et al. 2008, Jiménez et al. 2010) and currently it is not possible to attribute greater importance to any of these as the principal 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 CBM, overlap more with the pelagic longline fisheries (e.g. *D. exulans*, *D. dabbenena*, *P. conspicillata*, *T. chlororhynchos*), compared to species that are mainly abundant over the 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 hubbsi*), and to a lesser extent demersal longliners, are also a source of mortality (Favero et al. 2003, 2011, González Zevallos & Yorio 2006). Considering the population statuses mentioned above, three populations (i.e. *D. exulans* on South Georgia, *D. dabbenena* and *T. chlororhynchos* on Tristan da Cunha and Gough) could be being 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 by PBR. At the same time, though derived from a semi-quantitative analysis, the PSA 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 southwestern Atlantic, mainly over part of the BMC. Considering the fishing effort that the pelagic longline fleets expend over the region of the BMC, clearly the area of greatest concentration of effort over the last decade in the southwestern Atlantic (Tuck et al. 2003, Huang 2011), there is no doubt that some populations are seriously affected. In this study we showed that the capture of a few individuals the population of *D. exulans* in South Georgia is the most affected species. Nevertheless, 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. The species present different catchabilities with respect to each of these fleets, due to various factors, such as differences in the operation of fishing, the sinking rates, and the size of the hooks. The catchability of the PSA realized 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 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 be responsible for many of the observed declines in the previously mentioned populations of *D. exulans*, *D. dabbenena* y *T. chlororhynchos*. In order to evaluate this hypothesis it would be necessary to carry out studies modelling in a more detailed manner the impact of pelagic longline fishing in the southwest Atlantic on the aforementioned species. 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 & Fletcher 2008). Considering the total effort of the fleets in the region, the values of PBR could be easily exceeded. The approximation that we used in this study (i.e. using $Fr = 0.1$ and N_{min}) leads to a conservative estimation of PBR. A nonconservative approximation e.g. $Fr = 0.5$, thought not recommended for threatened species (Dillingham & Fletcher 2008), would result in a PBR five times higher than what we estimate here. However, considering this latter scenario the situation would still be complicated for these populations.

The remaining populations also require attention, mainly those whose statuses are not clear and whose population sizes are 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/Islas Malvinas may be stable, and *P. aequinoctialis* is potentially decreasing, the first species requires more attention due to being very small and endemic to a single island. The populations less affected by the pelagic longlines in the studied area would be *M. giganteus*, *D. capense* and *F. glacialoides*. It was not possible to obtain an understanding of the impact of this fishery on *P. gravis*, *M. halli*, *T. steadi* and *P. fusca*. However it is highly probable that the two first species are little affected. The remaining two require further investigation, especially *P. fusca* whose population on Tristan da Cunha and Gough is small (Table 1).

A high priority should be to undertake a more detailed analysis (level 3 in the terminology of Risk Analysis, Hobday et al. 2011) where the impact of the pelagic longline fleets that operate in the southwestern Atlantic on populations of *D. exulans* (South Georgia), *T. chlororhynchos* (Tristan da Cunha and Gough) and *D. dabbenena* (Gough) are modelled. 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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Tabla 1. Principales poblaciones de albatros y petreles que interactúan con palangreros pelágicos en Uruguay y aguas adyacentes (Atlántico Sudoccidental) y su productividad expresada como la tasa máxima de reclutamiento poblacional (Rmax). Rmax fue estimada mediante el método invariante demográfico usando la tasa de sobrevivencia de adultos (S) y edad de primera reproducción (α). Los valores expresados en *itálicas* fueron obtenidos de otras colonias de la especie.

Especies	Código	IUCN Estatus*	Población (Islas/ Archipiélagos)	Frec. reprod.	Parejas reprod. anuales**	S	α	Fuente	Rmax
<i>Diomedea exulans</i>	DEX	VU	South Georgia	Bienal	1,420	0.926	10.6	1	0.0565
<i>Diomedea dabbenena</i>	DDA	CR	Gough	Bienal	1,514	0.910	11	2	0.0584
<i>Diomedea epomophora</i>	DEP	VU	Campbell	Bienal	7,800	0.935	9 (6-12)□	3	0.0609
<i>Diomedea sanfordi</i>	DSA	EN	Chatham	Bienal	5,800	0.946	8.5	4	0.0594
<i>Thalassarche steadi</i>	TST	NT	Auckland Group	Bienal	97,089	-	-	-	-
<i>Thalassarche melanophrys</i>	TME _{FM}	EN	Islas Malvinas/Falklands Islands	Anual	399,416	0.942	10	5, 6	0.0544
<i>Thalassarche melanophrys</i>	TME _{SG}	EN	South Georgia	Anual	74,296	0.910	10	7, 6	0.0627
<i>Thalassarche chlororhynchos</i>	TCH _{T.C.}	EN	Tristan da Cunha	Anual	21,700-35,800	0.920	9.7	8	0.0618
<i>Thalassarche chlororhynchos</i>	TCH _{G.}	EN	Gough	Anual	5,300	0.840	9.7	8	0.0754
<i>Phoebastria fusca</i>	PFU	EN	Tristan da Cunha, Gough	Bienal	7,625-8,750	0.898	11.8	9	0.0574
<i>Macronectes halli</i>	MHA	LC	South Georgia, otras islas sub-antárticas	Anual	4,310	0.910	-	10	-
<i>Macronectes giganteus</i>	MGI	LC	Islas en costa Argentina	Anual	2,831	0.917	7.5	11, 12	0.0752
<i>Procellaria aequinoctialis</i>	PAE	VU	South Georgia	Anual	889,122	0.895	6.1	13	0.0945
<i>Procellaria conspicillata</i>	PCO	VU	Tristan da Cunha	Anual	14,400	0.970	5	14	0.0664
<i>Daption capensis</i>	DCA	LC	Antartida, varias islas sub-antárticas	Anual	-	0.942	5	15, 16	0.0872
<i>Fulmarus glacialisoides</i>	FGL	LC	Antartida, varias islas sub-antárticas	Anual	-	0.923	7	17	0.0769
<i>Puffinus gravis</i>	PGR	LC	Tristan da Cunha, Gough	Anual	6,000,000	-	-	-	-

* <http://www.birdlife.org/datazone/species/search>. **Para las especies listadas en ACAP la frecuencia de reproducción y el tamaño poblacional fueron obtenidos de los Species Assessments de ACAP (<http://www.acap.aq/acap-species>). □ Se estimó la mediana del rango reportado para calcular Rmax. Tamaño poblacional obtenido de: ●Martin et al. 2009, ●● Ryan & Ronconi 2011, ●●● Cuthbert et al. 2005. *** Las fuentes de S y α se dan en el apéndice 1 (dos números indican que S y α fueron tomados de referencias distintas). TME_{FM}, TME_{SG}, TCH_G y TCH_{TC} refieren a las colonias de Falkland Islands/ Islas Malvinas, South Georgia, Gough e islas de Tristan da Cunha, respectivamente.

Tabla 2. Límites o “cut-offs” utilizados para estimar los distintos atributos de la susceptibilidad para las poblaciones/especies de albatros y petreles asociadas a palangreros en el talud uruguayo y aguas adyacentes. El atributo marcado con asterisco indica que fue estimado cuantitativamente (ver Materiales y Métodos). %F.O. = frecuencia relativa de ocurrencia, LFA = longitud frontal del anzuelo, LTA = longitud total del anzuelo, N/A = no aplicable.

Atributo	Baja (0.333)	Media (0.667)	Alta (1.000)
Disponibilidad o solapamiento con la pesquería			
Si la población es:			
> 100 000 parejas	%F.O. <25%	%F.O. 25–50%	%F.O. >50%
10 000 - 100 000 parejas	%F.O. <10%	%F.O. 10–25%	%F.O. >25%
< 10 000 parejas	%F.O. <5%	%F.O. 5–10%	%F.O. >10%
Acceso a las carnadas *	Nº accesos a las carnadas/ Nº aves presentes		
Selectividad	Culmen < LFA	LFA < Culmen > LTA	Culmen > LTA
Mortalidad post captura	N/A	N/A	Nº aves muertas/Nº aves capturadas ~ 1

Tabla 3. Susceptibilidad para las principales poblaciones/especies de albatros y petreles (códigos en Tabla 1) asociadas a palangreros en el talud uruguayo y aguas adyacentes. Fue estimada como el producto de las probabilidades condicionales de: disponibilidad, capturabilidad (incluye acceso a las carnadas y selectividad) y mortalidad post captura (ver Materiales y Métodos).

Código	Disponibilidad				Capturabilidad							Mortalidad Post captura	Susceptibilidad	
	%FO	FO (n=349 conteos)	Result.	Acceso	ES	n lances	Selectividad	Longitud media del pico (mm)	ES	n	Fuente	Result.		
DEX	40.7	142	1.000	0.660	0.385	13	1.000	166.7	1.2	19	1	0.660	1.000	0.660
DDA	*		0.667	0.660	0.385	13	1.000	147.9	1.9	7	1	0.660	1.000	0.440
DEP	13.2	46	1.000	0.300	0.249	10	1.000	172.5	1.0	35	1	0.300	1.000	0.300
DSA	25.8	90	1.000	0.300	0.249	10	1.000	161.3	0.8	46	1	0.300	1.000	0.300
TST	18.9	66	0.667	0.495	0.168	13	1.000	134.8	0.7	24	1	0.495	1.000	0.330
TME	72.5	253	1.000	0.497	0.140	33	1.000	118.1	0.2	319	1	0.497	1.000	0.497
TCH	59.9	209	1.000	0.245	0.245	30	1.000	115.8	0.5	71	1	0.245	1.000	0.245
PFU	2.0	7	0.333	-	-	1	1.000	107.2	-	13	2	-	1.000	-
MHA	47.6	173*	1.000	0.028	0.019	20	1.000	89.3	-	71	3	0.028	1.000	0.028
MGI	21.8		1.000	0.028	0.019	20	1.000	80.9	-	27	4	0.028	1.000	0.028
PAE	68.8	240	1.000	0.413	0.139	36	0.667	52.1	0.4	20	1	0.275	1.000	0.275
PCO	71.3	249	1.000	0.231	0.074	25	0.667	50.6	-	20	5	0.154	1.000	0.154
DCA	52.4	183	1.000	0.008	0.008	20	0.330	32.0	-	53	6	0.003	1.000	0.003
FGL	21.5	75	0.667	0.000	0.000	6	0.667	45.2	0.5	15	7	0.000	1.000	0.000
PGR	67.0	234	1.000	0.406	0.140	30	0.667	46.9	-	30	8	0.271	1.000	0.271

* Ver Materiales y Métodos por detalles sobre estimación de %FO. Referencias: 1 Aves capturadas en palangre pelágico el Atlántico Sudoccidental, DINARA datos sin publicar, 2 Brooke 2004 (hembras), 3 González-Solís 2004 (hembra), 4 Copello et al. 2006 (volantón hembra), 5 Hall 1987, 6 Ryan 1998, 7 Pinder 1966, 8 Spear & Ainley 1998, 9 Cuthbert 2005.

Tabla 4. Resultados del análisis PSA para las poblaciones/especies de albatros y petreles (códigos en Tabla 1) que interactúan con la flota uruguaya de palangre pelágico en el talud de Uruguay y aguas adyacentes. Rmax la y susceptibilidad provienen de la Tabla 1 y 2 respectivamente.

Código	Rmax	Susceptibilidad	Distancia Euclideana (Rmax = 0, Suscep=1)	Ranking de Riesgo
DEX	0.057	0.660	0.3447	1
TME	0.054	0.497	0.5059	2
DDA	0.058	0.440	0.5628	3
DSA	0.059	0.300	0.7025	4
DEP	0.061	0.300	0.7026	5
PAE	0.094	0.275	0.7307	6
TCH	0.062	0.245	0.7575	7
PCO	0.066	0.154	0.8485	8
MGI	0.075	0.028	0.9749	9
DCA	0.087	0.003	1.0012	10
FGL	0.077	0.000	1.0030	11
TST	-	0.330	-	-
PFU	0.057	-	-	-
MHA	-	0.028	-	-
PGR	-	0.271	-	-

Tabla 5. Estimación en número de individuos del PBR y tamaño poblacional mínimo (Nmin) para las poblaciones (códigos en Tabla 1) con mayor riesgo según en PSA.

Código	N _{min}	PBR
DEX	6,964	20
DDA	7,425	22
DEP	38,251	116
DSA	28,443	83
TME _{FM}	1,093,479	2975
TME _{SG}	203,400 59,397 -	637
TCH _{Tc}	98,009	183-303
TCH _G	14,510	55
PAE	2,434,145	11496
PCO	39,423	131

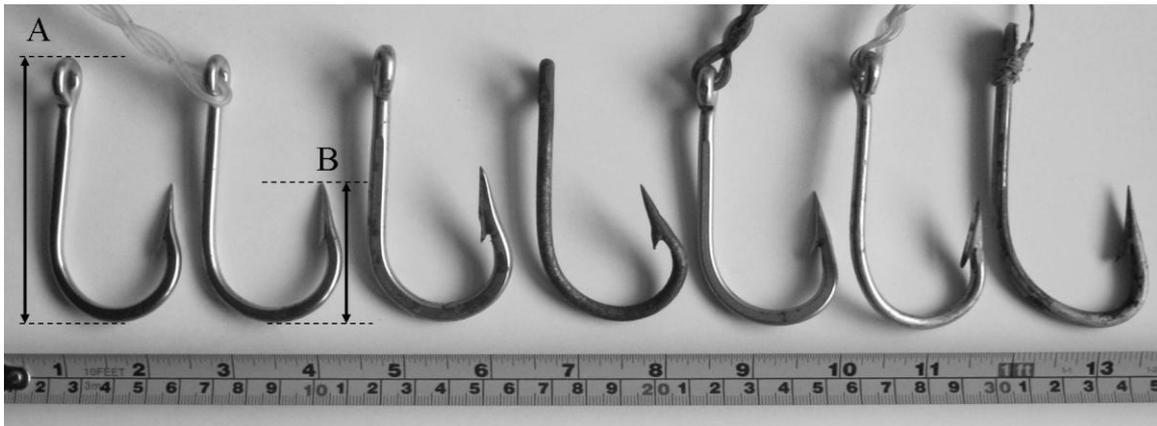


Figura 1. Anzuelos utilizados por palangreros pelágicos uruguayos. Se indican las medidas consideradas por evaluar la susceptibilidad a la captura de las distintas especies. A. longitud total, B, longitud frontal. De izquierda a derecha, los primeros 6 anzuelos son usados en el palangre tipo americanos (y usados en este trabajo) y el último en palangre español (no abordado).

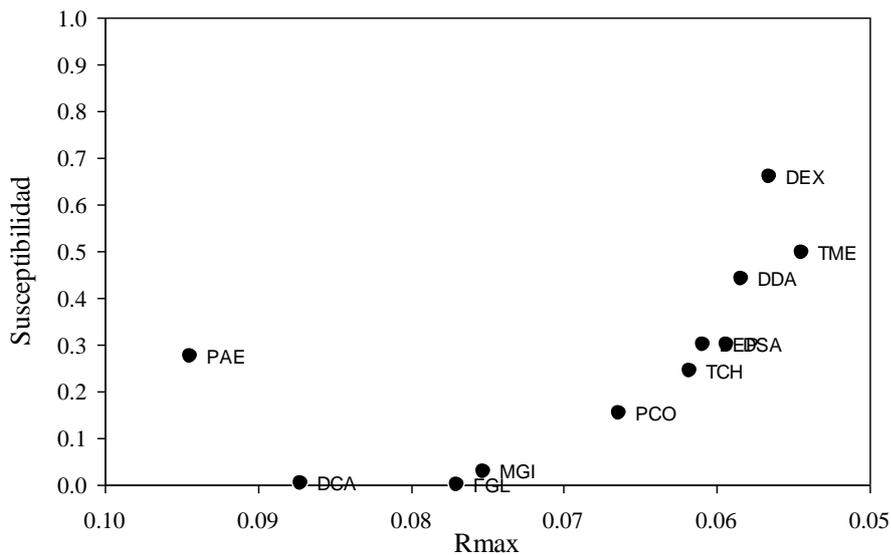


Figura 2. PSA análisis para la principales especies (códigos en Tabla 1) de aves marinas asociadas a la flota uruguaya de palangre pelágico en el talud uruguayo y aguas adyacentes. La productividad es medida como la tasa máxima de reclutamiento R_{max} . La susceptibilidad es el producto de las probabilidades disponibilidad, capturabilidad (acceso a las carnadas y selectividad) y mortalidad post captura (ver Materiales y Métodos).