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# *Endangered Species*

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# Endangered Species UPDATE

Science, Policy & Emerging Issues

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# Protecting Climate Refugia Areas: The case of the Gaviota coast in southern California



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## **Abstract**

The designation of “climate refuge areas” should be an important part of a more integrated, ecosystem-based approach to protect endangered species. Identifying “climate refugia” areas should be a priority as resource managers begin to develop adaptive policies. This article describes policy innovation in California that includes important strategic elements and goals that can support the identification and protection of climate refugia for special status species, and offers a case study of the Gaviota coast in southern California. The essay concludes with general recommendations for planning and policy development to support better protection of endangered species, and emphasizes the importance of better local land-use planning.

## **The Importance of Climate Refugia**

This article describes recent policy innovation in California to develop an ecosystem-based approach to endangered species protection, with particular emphasis on the importance of identifying and protecting climate refugia in areas that are known as hot spots for threatened biodiversity. Climate refugia is a term that has emerged in the conservation biology literature that refers to an area that is inhabited by plants and animals during a period of continental climatic change (as a glaciation), and remains as an important area from which a new dispersion and speciation may take place after climatic disturbance (Klausmeyer and Shaw 2009). Scientists have begun to describe the cumulative impacts of the multiple-use of resources, and show that these impacts will likely exacerbate an ecosystem’s ability to adapt to climate disturbance (Worm et al. 2006; Halpern et al. 2009). Large-scale climate disturbance will interact with and accelerate the existing anthropogenic pressures to endangered species. Indeed, scientists show that there are synergies among extinction drivers under global climate change that reflect the cumulative impacts of the multiple-use of resources and climate disturbance (Brook et al. 2008). Policy innovation is needed to begin to foster large-scale, ecosystem-based adaptive conservation strategies to better protect endangered species in an era of climate change.

Scholarly literature has expressed concern over the lack of region- or ecosystem-specific adaptation policy that can enable ecological resilience of threatened biodiversity with respect to climate disturbance. Current strategies include prescriptions at the state and federal government levels that support principles of ecosystem-based planning, and the establishment of

habitat reserves that support connectivity or migration corridors, habitat buffer zones, ecological core areas, the control of non-native invasive species, and collaboration across administrative, economic and political jurisdictions (Yaffee et al. 1996). Local land use planning and policy can play a fundamental role in the protection of sensitive habitat areas and ecosystems (Brody 2004).

This article describes recent policy innovation in California that focuses on the need to develop adaptive policy at the regional and local levels to protect climate refugia throughout the state. Resource managers increasingly recognize that adaptive policy must occur at regional levels with local land use decisions and management actions that can protect endangered species dependent on climate refugia. Protecting endangered species and their habitats on private and public lands will become increasingly difficult as plants, animals and insects adjust their ranges in response to climate change. More importantly, conservation effort is needed to protect endangered species in parts of their habitat range that are relatively stable “climate refugia” – areas that function as important source areas from which species can expand given climate disturbance. This article offers a number of preliminary strategies that should be developed at regional levels to begin to protect endangered species and climate refugia areas. Among the most recent recommendations in the scientific literature is the need to identify and protect climate refugia across a rapidly changing landscape and seascape (Barnosky 2007). Evidence is accumulating that emphasizes the importance of managing climate refugia that have historically supported ecological resilience during periods of dramatic climate disturbance, such as long term changes in environmental conditions.

### **Case Study: California’s Mediterranean-type Ecosystem**

Mediterranean-type ecosystems (MTEs) have a rich natural history that includes long periods of ecosystem and climate-related disturbance events, such as changes in oceanographic and climate regimes (Klausmeyer and Shaw 2009). MTEs are far from homeostatic or stable systems (Blondel and Aronson 1999). The MTEs of the world are unique biomes that share a common natural history – species of these areas have had to adapt to major climate events such as flooding, earthquakes, fire, and changes in the available of water and food. Rundel et al. (1998) note that MTEs are not steady-state ecosystems. For example, the Los Angeles River in southern California can increase its flow 3,000 fold in a 24-hour period (Davis 1998). California has experienced significant long-term droughts or extreme hydrological shifts: 892-1112 (220 years) and 1209-1350 (141 years). The longest drought of the 20th Century lasted 6 years during 1987-1992. Species adapt to these changing environmental conditions often by relying on climate refugia areas.

Human impacts on California’s MTE have led to the degradation of a range of habitats that serve the needs of endangered species: 55% of the State listed animals and 25% of the threatened plants depend on wetlands; 43% of the Federally listed species rely directly on wetlands for survival; estuarine wetlands have decline by 75-90%; riparian communities have declined by 90-95%; and vernal pools have declined by 90% (Noss et al. 1995; McGinnis 2009). The multiple impacts of human activities will likely exacerbate the ability of endangered species to adapt to climate change in California (California Resources Agency 2009; Halpern et al. 2009; Halpern et al. 2008). An example of the multiple use of MTEs is coastal urbanization, the extraction of energy resources, such as oil



and gas, agricultural activity, fishing, and the development of irrigation systems across southern California. These uses, including the associated multiple impacts have synergistic and cumulative impacts on ecosystems, and can exacerbate the ability of native species to adapt to climate disturbance (Halpern et al. 2008). The native plants unique to California are very vulnerable to global climate change such that two-thirds of these “endemics” could suffer more than an 80 percent reduction in geographic range by the end of the century (Loarie et al. 2008). Loarie et al. (2008) point to the need for identification and better protection of existing climate refugia given the evidence of ecosystem disturbance across California’s MTE.

### **Adaptive Biodiversity Policy Innovation in California**

California may be on the verge of establishing a new era of conservation policymaking that may influence how we can protect endangered species in the context of climate disturbance. In light of recent evidence of climate disturbance, policy innovation, local initiative and leadership will be needed if endangered species are to adapt to the dramatic changes in habitats. California policy requires that the public and private sectors participate in reducing California’s greenhouse gas (GHG) emissions. In addition, the existing California policy framework includes Assembly Bill (AB) 32, Senate Bill (SB) 375, SB 97, as well as a host of additional topic-specific bills, that require counties and cities to reduce GHG. State law requires that counties and cities develop Climate Action Strategies. In December 2008, the California Air Resources Board released the state’s Climate Change Scoping Plan, which describes a range of strategies that are necessary for the state to reduce its GHG emissions to 1990 levels by 2020. A move beyond a focus or emphasis in reducing GHG is required if

endangered species are to be protected.

In response to the California Governor’s Executive Order S-13-2008, the 2009 California Climate Adaptation Strategy Discussion Draft [hereafter, Discussion Draft] (2009) outlines a wide range of strategic elements that include goals and objectives for protecting biodiversity and special status species in light of increasing pressures from climate change. While California is encouraging local governments to develop plans that support these adaptive strategies, there has been little if any formal policy developed by local governments that supports the protection of climate refugia for endangered species beyond the regulatory requirements set forth by the state and federal governments, such as the Endangered Species Act, that require critical habitat designation. However, one county in California may represent the first step toward protecting climate refugia that will likely be needed for endangered species.

### **Protecting Climate Refugia: the case of the Gaviota coast in south California**

One consequence of climate disturbance in California will be a shift of biodiversity to the north (Loarie et al. 2008). Scientists from the US Geological Survey developed the Coastal Vulnerability Index (CVI) to assess the physical vulnerability of the California coast. They found that from San Luis Obispo to the Mexico border, communities along this coastline have “high” or “very high” vulnerability to climate change. One area identified as climate refugia is the Gaviota coastal (GC), which is part of one of the most threatened “hot spots” for biodiversity in the world (McGinnis et al. 2009; National Park Service 2004; Stein et al. 2000). The GC extends from Coal Oil Point to Point Sal and includes the coastal watersheds and terrestrial foothill and mountain ecosystems as-

### Map 1:

Conservation Priorities  
in the Graviota Coast.  
Source: Conception  
Coast Project 2004.

sociated with the transverse Santa Ynez Range. Map 1 depicts the GC and areas of high conservation value (Conception Coast Project 2004).

In 2004, the National Park Service (NPS) completed a feasibility study that included an evaluation of the GC as California's second national seashore (the only national seashore is the Point Reyes National Seashore north of San Francisco). According to the NPS (2004), the GC includes two of the most biologically diverse ecoregions in the world, and some of the highest concentrations globally of important, rare species in the nation. Of the approximately 1,400 plant and animal species estimated to exist within GC, there are 24 federally- or state-listed threatened or endangered plant and animal species and another 60 considered rare or of special concern. The NPS concluded that the quality and scope of GC's natural and cultural resources qualify it for inclusion in the National Park system. But because of the preponderance of private land in the area, the NPS found that it would not be feasible to add Gaviota to the National Park system at that time.

In March 2009, Santa Barbara County initiated the long-term development of a Climate Action Strategy (CAS) that would include or consider biodiversity concerns. In addition, the County Board of Supervisors voted to support a Gaviota Coast Rural Regional Planning (RRP) process, which may lead to the development of new policies in the County's General Plan and Local Coastal Plan (LCP) that can protect endangered species. The County's CAS and RRP processes are unusual insofar as they include biodiversity protection measures, and the local effort may represent the first local effort in California with regards to climate change and endangered species protection.

While climate-related policy development will necessarily include broad

spatial scales, decision makers increasingly recognize that implementation must occur at the local level with local land use decisions. With respect to the planning process for identification and protection of climate refugia, Figure 1 depicts the ideal planning process to begin to address endangered species issues at the county level.

The following planning stages are recommended to begin to identify and protect climate refugia at the local level.

**Stage 1 - Identification of Pressures.** Climate change will have direct and indirect pressures and impacts on areas designated as critical habitat and environmental sensitive habitat areas (ESHAs). Existing protected areas, such as ecological reserves, wildlife areas, undesignated lands, mitigation sites, and easements will likely be impacted by climate change. A more comprehensive, ecosystem-based and cumulative assessment, that includes the identification of multiple pressures or stressors on endangered species and their habitat, should be included in local land use planning.

**Stage 2 - A comprehensive vulnerability analysis should be conducted to establish the type and extent of potential climate changes (such as sea level rise, storm surges, and changing ocean conditions) and how these changes will impact natural habitats and endangered species. Smaller communities are particularly vulnerable as they lack many important resources for effective adaptation (California Resources Agency 2009). A vulnerability analysis must include detailed mapping that contains "measures of physical risk," identification of threatened habitats, among other factors. Coupled with an inventory, this analysis can determine the most successful places that exist for ensuring migration of sensitive habitat, such as coastal wetlands, and species.**

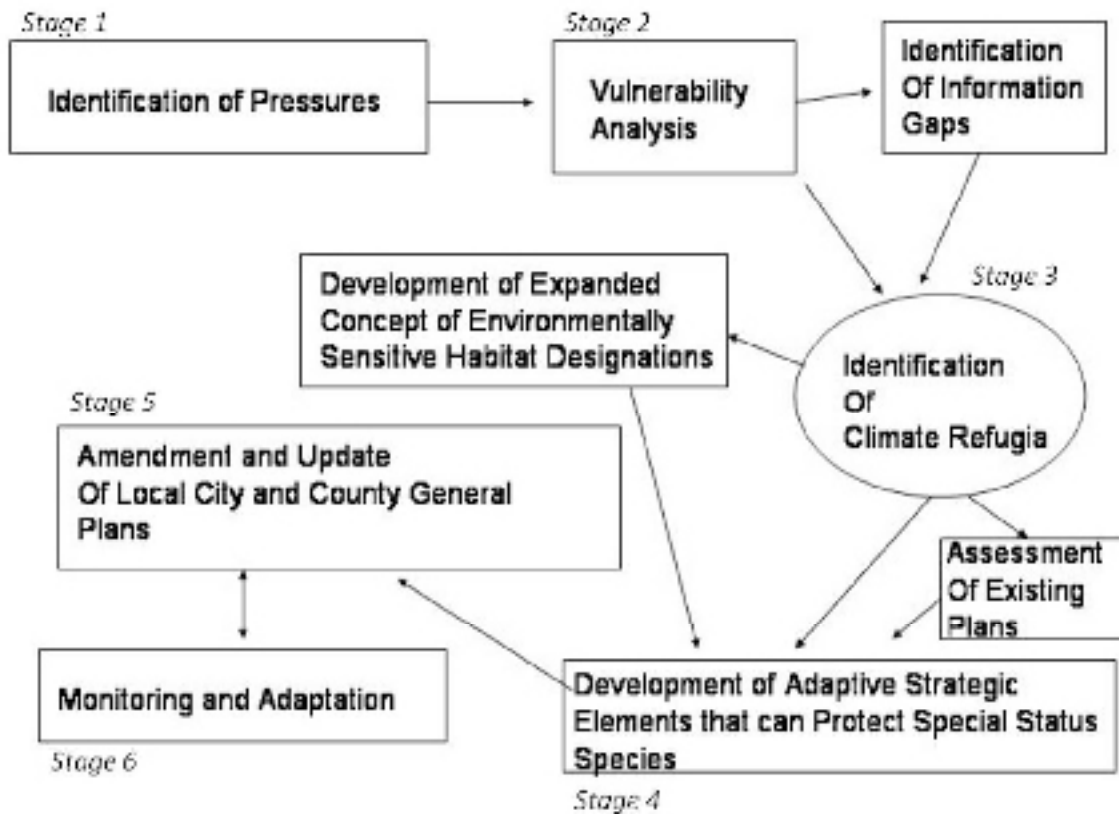
The vulnerability analysis should be used as one foundation to develop adaptation strategies (both overarching and specific) to protect endangered species. As much as possible, each adaptive strategy should be accompanied by case studies that elucidate that strategy and guidance on how it should be implemented. For example, changes in creek, wetland, and coastal bluff buffer areas and other adaptive strategies that are needed to protect sensitive habitat areas should be incorporated in local plans. In addition, the vulnerability analysis should carefully depict and describe information gaps.

To assist in the regional vulnerability analysis, regional interagency and working groups representing government and non-governmental organizations, e.g. state parks, and the private sector may be needed to discuss and recommend adoption of policies to pro-

tect biodiversity. Such a working group could also include the use of a Scientific Advisory Panel to assist in the development of guiding principles to protect biodiversity.

Stage 3 - Identification of Climate Refugia. A number of guiding principles should be emphasized to identify climate refugia including the following goals:

- o Maintain healthy, connected, genetically diverse populations
- o Improve resiliency of existing habitats in order to maintain existing or new assemblages of species
- o Reduce non-climate stressors on ecosystems (i.e. invasive species)
- o Protect coastal wetlands and accommodate sea level rise
- o Consider climate change models as well as historical data when making projections
- o Employ monitoring and adap-



**Figure 1:**  
The Planning Process  
for Establishing Climate  
Refugia.

tive management

- o Adopt adaptation approaches that reduce risks to species and habitats and provide time for species evolution and development.

With respect to the identification of climate refugia, the precautionary principle must be employed to buffer against scientific uncertainty. There are synergistic effects and positive feedback loops of human-induced climate change, other anthropogenic impacts, and natural disturbances, all of which make decisions about policy solutions difficult.

The identification of climate refugia should include priority management goals to preserve core habitat and migration corridors. Since climate disturbance will continue to cause plant communities and species' ranges to shift, adaptive corridors of continuous habitat must be preserved to enable future shifts in ranges and resiliency in ecosystems. Routes containing viable native habitats for plant pollination vectors (wind and insect) and which connect existing and predicted future habitat areas can be mapped and protected. The identification of climate refugia should also include policy development that emphasizes a careful review and assessment of existing land use plans and policies, such as Local Coastal Plans, and other elements such as biological thresholds and environmental sensitive habitat areas.

Stages 4, 5 and 6 - Development of Adaptive Strategic Elements. City and county plans should be amended to include special conservation measures that can support the ecological resilience of endangered species and climate refugia. Amendment and revision of city and county general plans should support an integrated, ecosystem-based approach that includes resources that support long-term monitoring of climate refugia areas, and land use elements should be revised as new infor-

mation becomes available.

## Conclusion

The long-term impacts of climate change on special status species are likely to be dramatic. In the context of climate change, the goal of protecting special status species is exacerbated by the cumulative or synergistic impacts of the multiple-use of resources and the expected pressures from large-scale climate disturbance. This article reviewed recent policy development in California and offered a number of recommendations for cities and counties to begin to develop biodiversity conservation measures that can better protect endangered species and climate refugia. Policy innovation that moves beyond the emphasis of reducing greenhouse gases and supports the ecological resilience and adaptation of species that are essential to the maintenance of goods and services provided by a healthy, productive ecosystem is needed today.

## Acknowledgment

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# Populations and Politics of a Plover

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The Mountain Plover (*Charadrius montanus*) is a species that inhabits cold, xeric-shrub landscapes of the western United States where it breeds in low-density, scattered populations primarily in Colorado, New Mexico, and Wyoming. To the east of this landscape, the plover is found most predictably on prairie-dog (*Cynomys* spp.) towns within western prairies from northern Montana into Nuevo Leon and San Luis Potosi. These landscapes also historically supported large herds of bison (*Bison bison*). With near eradication of bison and decreased prairie-dog presence on the landscape, the eastern breeding range of this plover became fragmented and generally of poorer quality. Thus, in contrast to westerly xeric landscapes, the current population of plovers in prairie landscapes is now restricted to fragments within the Oklahoma panhandle, north through the southwestern corner of Kansas, most of eastern Colorado, the southwestern corner of Nebraska, and eastern Wyoming and Montana (Knopf and Wunder 2006).

First collected by John Kirk Townsend along the Sweetwater River of Wyoming in 1834 and subsequently named the Rocky Mountain Plover by John James Audubon, this species of relatively nondescript plumage received little conservation attention for 150 years. It was never described as historically abundant, and only scattered references to the species appeared in the literature—much like comments are lacking about any non-charismatic species on the western frontier of America in the 1800s. Despite occasional collections of a few birds or clutches of eggs, one specific comment about plover occurring in high densities was that of an early bison hunter from the early 1870s who had killed about 200 in an hour near Dodge City, Kansas (Sandoz 1954). Those likely were from flocks of migrating birds that then flew directly south to winter in South Texas. Today, we believe that most migrants move more to the south along the Front Range of Colorado then swing west across southern New Mexico and Arizona to California and then north into the Central Valley of California. Historical records of migrating plovers are almost nonexistent within the Great Basin (Knopf and Wunder 2006).

Forty years ago, Graul and Webster (1976) estimated a continental population of 214,200–319,220 breeding Mountain Plovers, with 20,820 in the “stronghold” of Weld Co., CO. Conservation concern for the species was first expressed when the U.S. Fish and Wildlife Service (FWS) raised questions about population declines from historic levels (Leachman and Osmundson 1990). Unpublished guesses as to the contemporary population of plovers at that time oscillated around 6,000–10,000 birds, much reduced from the his-



torical estimation of Graul and Webster. This difference reflected a severe decline in the population of Weld County, Colorado, used to extrapolate the earlier continental projection. In 1999 the FWS officially proposed listing the species as “Threatened,” with evidence of decreasing population size being statistically supported by >3% annual decline across 30 years of Breeding Bird Survey data (USFWS 1999).

Beginning in the late 1990s, many studies inventoried plover populations in major breeding areas across the species range. Wunder and others (2003) estimated a population of 2,300 birds in the previously undescribed high-elevation population of South Park, Colorado. Summarizing recent inventories across eastern Colorado, FWS concluded that there were an additional 7,000

birds in eastern Colorado. Personal communications from researchers and FWS personnel projected an additional 2,000-5,000 plovers in Wyoming and 1,500 in Montana. Thus, plover breeding populations in these three critical states totaled ~12,800-15,800 birds, with additional small populations known to occur in Kansas, Nebraska, New Mexico, Oklahoma, and Utah. Since all populations seemed to be stable, FWS subsequently decided to withdraw the proposed listing of the plover as a threatened species under The Endangered Species Act (ESA) in September of 2003 (USFWS 2003).

Additional studies following the decision to withdraw the plover listing proposal confirmed, and slightly expanded, the continental population estimate. Plumb and others (2005)

Mountain plover with full clutch. Photo credit: Fritz L. Knopf



Mountain plover on the shrubsteppe in Carbon County, Wyoming. Photo credit: Fritz L. Knopf.



conducted a statewide survey within historical plover locales in Wyoming to estimate a minimum of 3,300 breeding birds. This number was within the range of estimates used in the FWS decision. However, the authors emphasized the “minimum” nature of the estimate and it still may be well short of a true statewide population due to the inadequate representation of private lands in both the historical database and contemporary sampling protocol.

Tipton and others (2009) conducted a systematic statewide survey of eastern Colorado. That study defined the Colorado population on the eastern plains to be 8,577 birds, slightly larger than the previous FWS projection of 7,000. Childers and Dinsmore (2008) subsequently estimated 1,028 birds in Northeastern Montana, supportive of the earlier FWS figure of 1500 statewide. The Wyoming and Colorado studies together, lead to a revised conti-

nenal estimate of breeding plovers totaling a minimum of 15,700 birds. The true continental population is certainly larger by an unknown quantity given (1) documented small populations in contiguous states (Ellison-Manning and White 2001, Bly et al., 2008, McConnell et al., 2009), (2) a potentially significant population in New Mexico and (3) an unknown population in Mexico (Knopf and Wunder 2006).

On 16 November 2006, Forest Guardians and the Biological Conservation Alliance challenged the withdrawal of the proposal to list the plover as Threatened (Forest Guardians, et al. v Ken Salazar et al., Case No. 3:06-cv-02560-MMA-BLM). The plaintiffs and the Federal defendants filed a settlement agreement on August 8, 2009, agreeing to reconsider the FWS 9 September 2003, decision to withdraw the proposed listing of the mountain plover (68 Federal Register 53083) and to sub-



mit to the Federal Register a notice reopening the proposal to list the Mountain Plover and providing for public comment by July 31, 2010. Thus, the decision to withdraw the proposed listing of the species in 2003 was 'vacated', and the plover is once again proposed as a threatened species under the ESA. The agreement calls for a final listing decision by 1 May 2011.

The listing process for the Mountain Plover was rather unique for the FWS. Whereas most species come to be listed following an initial petition to FWS followed by an FWS review, the impetus for increased conservation concern for this plover came from research within government research. The initial identification of plover declines came from basic science (to 1999) within (vs. external to) the Department of Interior. FWS biologists within Ecological Services pursued the review and ultimately proposed the species for listing. The process is a rather unique example of how government science and operations were intended to work within the Department of Interior. Also, whereas the scientific record for most species listed under ESA is often limited by a lack of historical and contemporary data, the science available to the proposal decision for Mountain Plover in 2003 was some of the best available to date for any species.

If FWS decides to list this plover as threatened, any plan to promote its recovery will be politically challenging. The plover is neither a montane species (as named) nor a species of shores and wetlands like other members of the Charadriidae. Rather, again, it is an upland associate of xeric landscapes to the west of the Colorado Front Range that also occurs where disturbances alter prairie landscapes to the east of the Front Range. Prairie-dogs created both historical and contemporary habitats for plovers (Dreitz et al., 2005). Contemporary sites that also attract breeding

plovers in prairie landscapes include surfaces impacted by activities such as military maneuvers, pipeline construction, petroleum development, and agricultural conversion of prairies. The most attractive of these sites are those agricultural fields that have either been recently tilled or are crop-idle at the time plovers arrive on the breeding grounds. Plovers nest on those fields, and subsequent tillage has been suspected to destroy nests and eggs. Recent research, however, shows that nest/egg destruction by tillage practices appears to be a compensatory rather than an additive constraint on reproduction; the proportion of nests lost to tillage on relatively predator-free croplands is comparable to that proportion lost to predators in contiguous, native landscapes (Dreitz and Knopf 2007).

In winter months plovers were historically found in the coastal uplands and interior valleys of California. Those habitats have been almost universally converted to urban/suburban and agricultural landscapes, respectively. Whereas the preferred habitats on those xeric plains near the ocean and the intensely grazed sites of the San Joaquin and Sacramento valleys only occur in isolated patches on the modern landscape, plovers are now found in large numbers mostly on agricultural fields (Knopf and Rupert 1995). Favored fields include those that have been recently tilled, or recently harvested and followed by either burning or grazing by domestic sheep to clean the field before replanting (Wunder and Knopf 2003).

Critics of listing the plover (including agricultural and rural development organizations among others) note that the behavioral flexibility documented for the species argues strongly against Mountain Plovers being limited by habitat. The high rate of nesting success documented range-wide (Knopf and Wunder 2006, Dinsmore et al. 2002)

and high survival rate of breeding (Dinsmore et al. 2003) and wintering (Knopf and Rupert 1995) birds further argue that the contemporary population (albeit historically depressed) is viable. Alternatively, proponents argue that the current plover population (1) is drastically reduced from the Graul and Webster 1976 projection, (2) has experienced widespread loss of native habitats, and (3) is dependent upon another species of conservation concern (prairie-dogs).

Regardless of the direction of the decision, the forthcoming process will certainly stimulate much political dialogue. If listed, however, FWS will have to develop a plan for working with agricultural producers to manage cropping practices, which goes well beyond contemporary ESA vs. private-lands conflicts. The timing of agricultural practices is weather-driven, and interjecting ESA considerations into daily management decisions at the level of local farms would have a major economic impact on the agricultural community. Ironically, that political and administrative theater will focus on a semi-desert species that historically and currently occurs secondarily in altered shortgrass prairie landscapes.

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# Spatial analysis of incidental mortality as a threat for Franciscana dolphins (*Pontoporia blainvillei*)



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Fisheries bycatch is considered one of the main threats for most large marine vertebrates, such as seabirds, marine mammals, turtles and sharks (review, among others, by Shaughnessy *et al.* 2003, Read *et al.* 2006, Zydelis *et al.* 2009). Incidental entanglement is a direct human threat with a straightforward effect on species mortality. Three problems arise when results from bycatch research are translated into wildlife management actions. First, predictions of population trends are normally weak because large-scale estimates of demographic parameters are difficult to generate (*e.g.*, Moore and Read 2008). Second, bycatch impact is rarely compared to other threats, such as depletion of prey base, ecosystem changes, habitat degradation or disease. The effects of these stressors are not always as conspicuous as bycatch, and may be more difficult to evaluate (Taylor *et al.* 2007). Third, fisheries can offer short-term benefits to some marine vertebrates that overshadow the long-term costs of this threat.

An emerging method since the development of GIS and spatial statistics is the comparison between distributions of species and threats that are formalized in habitat suitability models or ecological-niche models. These models relate presence-absence or abundance observations based on random or stratified field sampling with stressors and other environmental variables (Guisan and Thuilier 2005, Sims *et al.* 2008).

Franciscana dolphins (*Pontoporia blainvillei*) are small cetaceans restricted to shallow waters of the South Atlantic Ocean, from southeastern Brazil (18°25'S) to northern Patagonia (42°10'S) (Crespo *et al.* 1998). It has been classified as 'vulnerable' by the IUCN (Reeves *et al.* 2008). There is no current abundance estimate for the species as a whole.

Incidental entanglement or bycatch of Franciscana dolphins was reported repeatedly: Di Benedetto *et al.* (2001), Kinas (2002), Dans *et al.* (2003), Freitas Netto and Barbosa (2003), Secchi (2003), Bordino and Albareda (2005), Capozzo *et al.* (in press). IUCN defines incidental mortality in gillnet fisheries as the main threats to this species (Reeves *et al.* 2008). Secchi *et al.* (2001) first estimated bycatch in the overall Franciscana population via population abundances based on aerial surveys. They calculated that removal by gillnets ranged between 1.1% and 3.5% per year. The Scientific Committee of the International Whaling Commission has noted that incidental mortality of 1% is sufficient for concern about the conservation future of coastal dolphins. However the various assumptions and parameters of these estimations are difficult to test. For example, Secchi *et al.* (2001) counted an average of 4.25

individuals per flight and extrapolated an overall abundance of 42,000 Franciscana dolphins (Reeves *et al.* 2008). It is expected that slight changes in parameters assumptions could produce significant changes to these results.

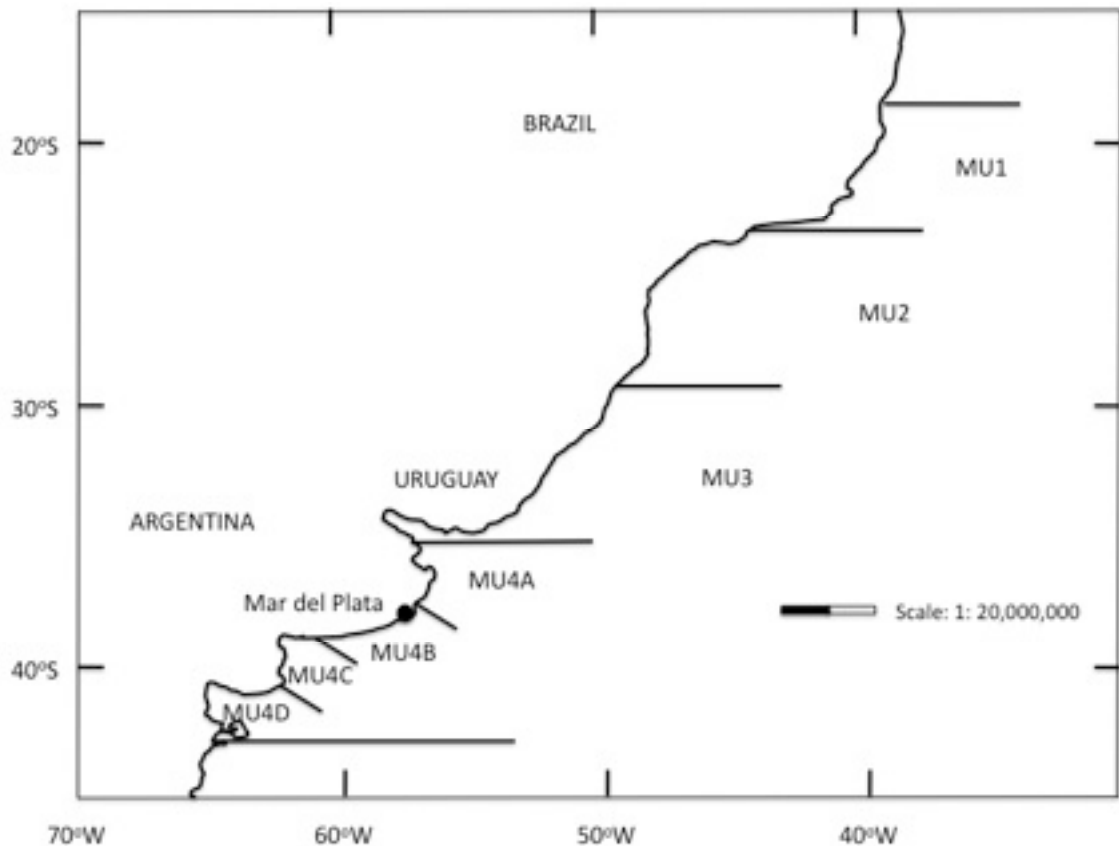
I used a spatial approach to analyze the role of incidental mortality as a threat to Franciscana dolphins. I compared information from different areas of their distribution with different levels of fishery activity. I followed the main assumption of most habitat models, i.e., that distribution of population abundance is positively related to habitat qualities (Guisan and Thullier 2005). If incidental entanglement is a main mortality factor, then areas with higher incidental mortality should have lower population densities.

Based on the genetic structure of the species, Secchi *et al.* (2003) proposed four management units (MU) (Fig. 1). MU1= coastal waters of Espírito Santo

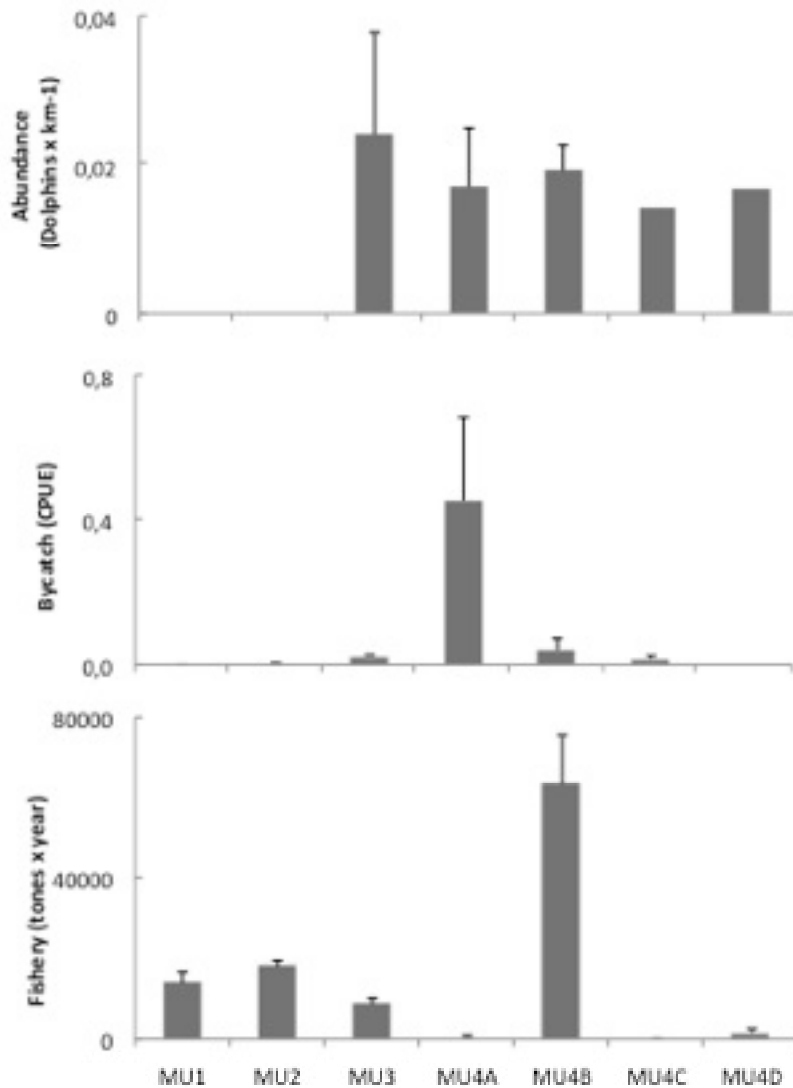
and Rio de Janeiro states, Brazil; MU2= waters off Sao Pablo, Paraná and Santa Catarina states, Brazil; MU3= waters off Rio Grande Sul State (Brazil) and Uruguay; MU4= waters off Argentina. Two studies have estimated abundance via aerial surveys for MU3 (Secchi *et al.* 2001) and MU4 (Crespo *et al.* 2009). In MU3, 8 flights of equal length (185.4 km except one of 129.8 km) with a single-engine aircraft were conducted at the end of summer 1996 and distributed equally between morning and afternoon. In MU4, 17 flights were conducted of variable lengths, ranging from 185.2 km to 564.9 km, with a double-engine aircraft in summer 2003 and autumn 2004. On the 13th and 18th of February, two flights were conducted on the same day, thus afternoon flights were not considered in the analyses to minimize pseudo-replicate effects and dolphin behavioral response to aircraft noise of the first daily flight. Relative abundance

**Figure 1:**

Geographic range of Franciscana dolphins (*Pontoporia blainvillei*) with four management units (MU) and MU4 divided into four sectors.







**Figure 2:** Population abundances, bycatches and artisan fishery activity in the sixth areas of the distribution range of Franciscana dolphins. Captures per unit of effort (CPUE) = 100 x deaths x km of net<sup>-1</sup> x d<sup>-1</sup>. Fishing was averaged with data from 2001 to 2007. MU1 and MU2 without data on abundance.

of *Pontoporia* in each MU is expressed as number of dolphins per km surveyed. I avoided the use of absolute estimations of abundance to minimize errors in parameter estimations of equations. Because there are several reports from MU4, I divided it in four sectors: MU4A, MU4B, MU4C (corresponding to northern, central and southern coasts of Buenos Aires Province, respectively), and MU4D, corresponding to northern coast of Patagonia (Fig. 1).

Di Benedetto (2003) provided systematic data on annual incidental mortality rate from MU2, Secchi *et al.* (2001) from MU3, and Bordino and Albareda

(2004) and Cappozzo *et al.* (2007) from MU4. Bycatch is expressed in 'captures per unit of effort' (CPUE units), which consisted of the number of dolphins killed in nets, divided by the km of fishery nets of the total float, and the number of fishing days per year. At MU1, there were no comparable measurements of bycatch effect, but Netto and Barbosa (2003) counted 13 records of Franciscana dolphins stranded along the Espirito Santo coast presenting wounds that may suggest interactions with fisheries between 1994 and 2001. MU1 and MU2 have similar characteristics with regard to fishery activity (Di

Beneditto and Netto 2008), therefore bycatch incidence at MU1 was estimated using the same fishing effort calculated by Di Benedetto (2003). This value of CPUE was included only for illustrative purpose, although it is well established that bycatch is comparatively very low at MU1 (review by Di Benedetto and Netto 2008). Information on fishing activity was obtained from official websites of governmental agencies of Brazil and Argentina (date: 09/09/09) who are responsible for fishery activity regulation: Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis and Secretaría de Agricultura, Ganadería, Pesca y Alimentación de la Argentina.

Data on abundance, bycatch incidence and fishing activity are summarized in Figure 2. There were no statistical differences in the relative abundance between MU3, MU4A, MU4B and MU4C (Kruskal-Wallis ANOVA by ranks,  $H_2 = 1.94$ ,  $P = 0.38$ ,  $n = 22$  with data on MU4C and D together in a category). However, there were significant differences in bycatch incidence, mainly due to a substantially high value observed in MU4A ( $H = 14.0$ ,  $P = 0.007$ ,  $n = 17$ ) and in fishery activity, due to a substantially high value observed in MU4B ( $H = 40.7$ ,  $P < 0.0001$ ,  $n = 7$  yrs  $\times$  7 areas). Cappozzo *et al.* (2007) provided information both on Franciscana mortality and fishing effort between 2002 and 2003 from 16 localities of MU4A, B and C: San Clemente del Tuyu, Las Toninas, Santa Teresita, Mar del Tuyú, La Lucila del Mar, San Bernardo, Mar de Ajó, Villa Gesell, Mar del Plata, Quequén - Necochea, Claremeccó, Monte Hermoso, Villa del Mar, Puerto Rosales - Ingeniero White. The correlation of mortality and fishing effort among these localities was not significant (Spearman rank correlation,  $r_s = 0.23$ ,  $P = 0.41$ ,  $n = 16$ ).

In summary, there was no clear positive relationship between fishery activity and Franciscana mortality, sug-

gesting that catchability depends on several co-varying factors, such as meteorological and economic conditions, individual behavior of fishermen and social and foraging behavior of the species (Crespo *et al.* 2009). For example, if dolphins tend to be more aggregated and at the same time overlap with fishing areas, a clumped distribution pattern could be associated with predation for sciaenid fish in nursery areas (Crespo *et al.* 2009).

Geographical differences on bycatch mortality did not appear to influence population abundance, given that the number of dolphins/km surveyed between the areas did not differ. This unexpected result could be caused by various mechanisms: (1) different initial population abundances between areas, (2) meta-population dynamics that dilute geographical differences, (3) compensation of the detrimental effects by benefits of the use of nets, and/or (4) overestimation of the impact of bycatch mortality on population growth rate. I will briefly analyze each hypothesis.

1) There is no information of historical abundances of Franciscana's dolphins, so it is not possible to test directly the first mechanism. An indirect approach is to project a population growth model into the past. Kinas (2002) developed such model, with the equation:

$$N_{(t-1)} = N_t e^{r(1-h_t)} \quad (1)$$

Kinas (2002) used, for MU3, a growth rate of  $r = -0.053$ , and a fraction of the population dying in year  $t$  as a result of entanglement  $h_t = 0.047$ . Secchi *et al.* (2001) estimated total population size in MU3 as  $N_t = 42,078$  dolphins in approximately 64,000 km<sup>2</sup>. At MU4A, estimations are:  $N_t = 8,279$  dolphins in approximately 22,000 km<sup>2</sup> (Crespo *et al.* 2009) and a maximal entanglement rate of  $h_t = 0.097$  (Bordino and Albareda, cited by Crespo *et al.* 2009). In accordance

to these estimations, present abundance proportion of MU3: MU4A is almost 5:1. Assuming the same value of  $r$ , constant impact of entanglements and lack of density-dependency, it is possible to apply equation (1) to predict population densities at MU3 and MU4A in the past. Applying this equation suggests that abundances in MU3 and MU4A could have been equalized 30 years before. At that time, population density in MU4A should have been almost triple that in MU3. There are no obvious reasons to expect such spatial differences in population densities between Brazil and Argentina when stranding effect is eliminated or substantially reduced from calculations.

2) Another possibility is that those populations exposed to a high bycatch incidence retain high levels of population density due to immigration processes, as in a typical source-sink system (Pulliam 1988). However, recent studies using tracking devices on Franciscana dolphins showed that animals did not tend to disperse significantly from the target location, which suggests a clear resident pattern for the studied animals (Bordino and Wells 2005, Bordino *et al.* 2008). A new series of genetic studies that identified several genetic stocks along the relatively small geographic range of this species confirms low levels of interaction between populations (Secchi *et al.* 2003, Mendez *et al.* 2008). These results do not support the source-sink model.

3) Another possible mechanism is that Franciscanas receive some kind of benefit from artisan fishing that compensates the cost of entanglement. The main potential benefit is facilitation of food patch detection. Foraging theory has investigated the effect of food distribution on foraging behavior and its population consequences (Sutherland 1996). Fish schools are typically characterized as rich food patches with an unpredictable distribution. Marine verte-

brates develop foraging strategies that are designed to maximize food-searching strategies. Fishery gillnets facilitate finding prey location and bycatch can occur when predators are attracted to these overabundant and fixed patches of food. This phenomenon is evident for seabirds for which gillnets do not normally represent a risk of passive entanglement, however, they may drown when they actively attempt to snatch baited hooks deployed by longline fishing vessels (Gales 1997).

Díaz López (2006) described how gillnet fishing appeared to benefit the Mediterranean bottlenose dolphin (*Tursiops truncatus*) in Sardinia, Italy. These dolphins were frequently observed feeding near the location of nets, suggesting dolphins try to exploit a concentrated food source at gillnets. Interviews of fishermen in Argentina revealed that they frequently observe Franciscana dolphins near the boats when they install gillnets (Negri, unpubl. data). It appears that, like *Tursiops sp.*, Franciscana dolphins also approach gillnet areas, although there is no evidence that they increase foraging efficiency by doing so. If the benefit in terms of survival and breeding success provided by this alternative source of food equals the cost due to incidental mortality, no apparent effect will be measured in the short term until the population collapses.

4) The final consideration is that the effect of incidental mortality rates have been overestimated due to biases in estimations of abundance. Equations for calculating population abundance and overall impact of entanglement require the use of several parameters. Potential error terms of these parameters are difficult to calculate. Field data are restricted to a small fraction of the coastline representing only approximately 0.7% of the possible range in the MU3 subpopulation (Reeve *et al.* 2008). Thus, final numbers frequently contain un-

controlled and large error terms that reduce reliability of conclusions. Reeve *et al.* (2008) stated that estimates of Franciscana abundance and, consequently, of density or entanglement, need to be treated cautiously.

This paper provides the first application of a spatial approach to the analysis of anthropogenic threats to Franciscana dolphin survival. Bycatch does not produce the expected spatial pattern in population abundance if incidental entanglement is a main threat to this species. Franciscana dolphins are threatened by several human impacts, other than bycatch, mainly: (1) water pollution, (2) human depletion of dolphin food, and (3) marine debris swallowed by the dolphins. Conservation agencies do not have resources to solve all these problems at once, thus criteria to define priorities is essential. It is urgently required that research on Franciscana dolphins provides clues to evaluate the precise impact of incidental bycatch on their conservation.

#### Acknowledgements

M. F. Negri kindly provided information on Franciscana dolphins. M. H. C is researcher of the the Argentinean Research Council (CONICET). GEMA group is supported by the Department of Basic Sciences from the University of Lujan and CONICET.

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# Observations on the reproductive seasonality of *Atlantoraja platana* (Günther, 1880), an intensively fished skate endemic to the Southwestern Atlantic Ocean

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## Abstract

Specimens of the La Plata skate (*Atlantoraja platana*) were collected monthly from commercial fishing landings at Guarujá, São Paulo State, Brazil, from March of 2005 to April of 2006. One hundred males ranging from 13.1 to 70.0 cm and 88 females ranging from 12.5 to 76.0 cm of total length were collected and their gonads analysed to determine maturity stages. Gonadosomatic and hepatosomatic indexes did not significantly vary among seasons between the sexes. Ovulation and egg-laying were continuous throughout the year. These observations suggest an annual cycle with eventual -though not well delimited- peaks in the reproductive activity. This pattern has been reported for skates of the same genus and for other species. *Atlantoraja platana* is intensely exploited, though as a non-target species, and retained for exportation over the South and Southeast Brazilian coast. For these reasons the species is already considered 'vulnerable' by the IUCN Red list of Threatened Species. Present data may be the base for future studies in order to protect the populations of *A. platana* from local disappearance.

## Introduction

Genus *Atlantoraja* Menni, 1972, is endemic to the Southwestern Atlantic Ocean (McEachran and Aschliman 2004). The La Plata skate's (*Atlantoraja platana*) (Günther 1880) distribution ranges from São Paulo's littoral area to Argentina, and is common in Rio Grande do Sul State in Southern Brazil (Figueiredo 1977). In Southern Brazil, it is found at depths of 40-100 m (Vooren 1997), though Marçal (2003) recorded its occurrence at up to 231 m deep. In the Southeastern Brazilian continental shelf, *A. platana* is commonly caught in the range of 20-120 m deep (Oddone and Amorim 2007).

The assessment of chondrichthyan populations requires a quantitative approach to the study of reproduction (Walker 2005). Oddone and Amorim (2007) reported the size at maturity of male and female *A. platana* in Southeastern Brazil. Data on the reproduction of *A. platana* were also provided by Marçal (2003), Oddone et al. (2008) and Oddone and Vooren (2008). However, so far, the trend in the seasonality of the reproduction of this species is unknown. But Vooren and Klippel (2005) demonstrated that intensive fisher-

ies in the South-western Atlantic have led to overexploitation of several species of demersal elasmobranchs, such as the congeneric *Atlantoraja castelnaui* and *A. cyclophora*, already 'endangered' and 'vulnerable' species according to the IUCN Red list of Threatened Species. (Hozbor et al. 2004, Massa et al. 2006).

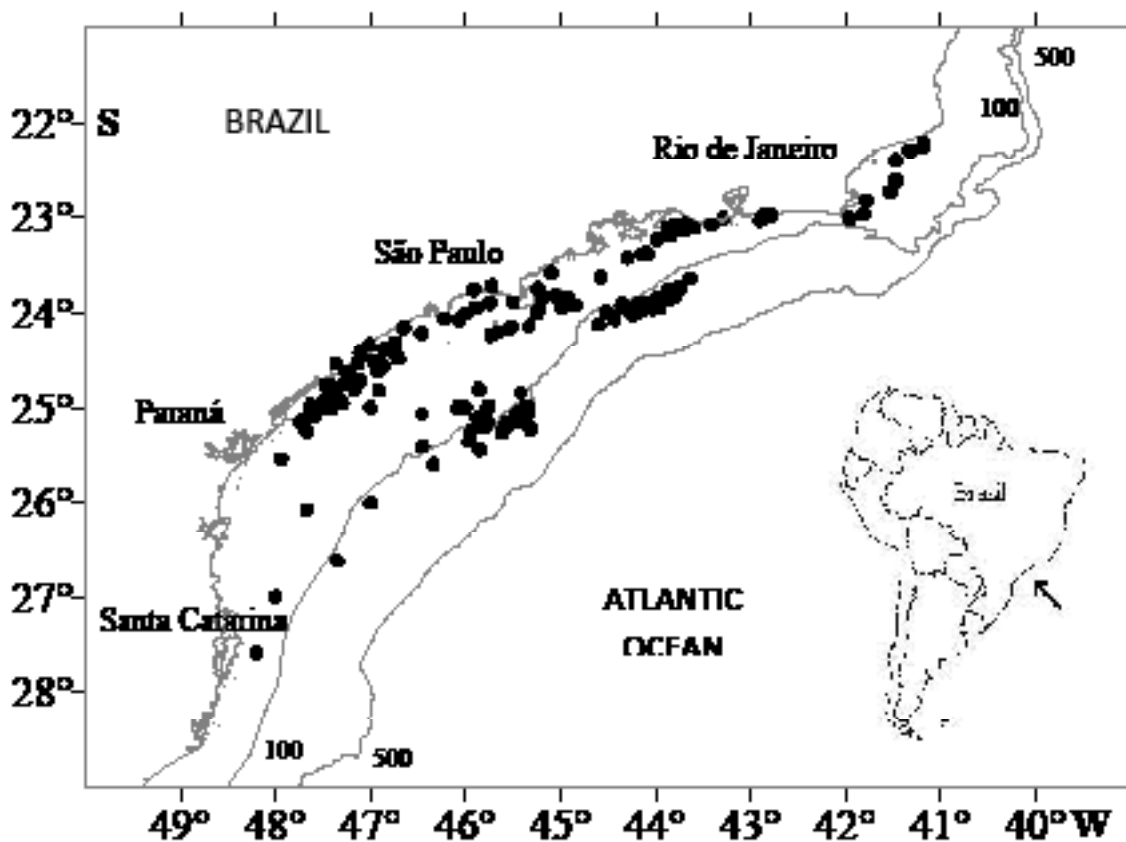
Over the past thirty years, catches of rajoids (skates) have increased in the

known in order to make decisions on stock management of chondrichthyan fishes.

The conservation status of *A. platana* is considered 'vulnerable' by the IUCN Red list of Threatened Species (San Martin et al. 2007). This is a matter of concern for a commercially exploited species because when life history characteristics are coupled with the selective

**Figure 1:**

Map of the study area, southeast Brazil, south-western Atlantic Ocean. Symbols represent the total number of fishing hauls (when registered by fishermen) in the area where samples of *Atlantoraja platana* were collected.



western Atlantic, mainly as a by-catch of bony-fish target fisheries, yet, sustainable catch rates are completely unknown (Frisk et al. 2002). This situation is also valid for the Brazilian continental shelf, where bottom trawling fisheries affect populations of *A. Platana*, which is incidentally captured. Thus, the knowledge of all the events compounding the reproduction of a species needs to be

removal of large individuals (as is the case for this species) of a given population subjected to intense fishery pressure, such a population may become highly susceptible to overexploitation and even disappearance, as has been the case for several rajoids (Brander 1981; Hoenig and Gruber 1990). Specimens of *Atlantoraja* spp. are commonly landed and sold in Santos and Guarujá

(São Paulo state, Brazil), especially the largest individuals (Oddone, unpublished data). In the present paper, we aim to analyze the trend of the reproductive variables for *A. platana* on an annual basis for this species in southeastern Brazil, which is critical information for developing a successful conservation program.

### Materials and Methods

Specimens of *Atlantoraja platana* were collected monthly from commercial fishing landings at Guarujá, São Paulo State, Brazil, from March of 2005 to April of 2006 by eight fishing vessels regularly bringing (once or twice a month) samples of this species. The study area was situated between latitudes 23°37'S and 27°40'S, at depths between 10 and 120 m (Fig. 1).

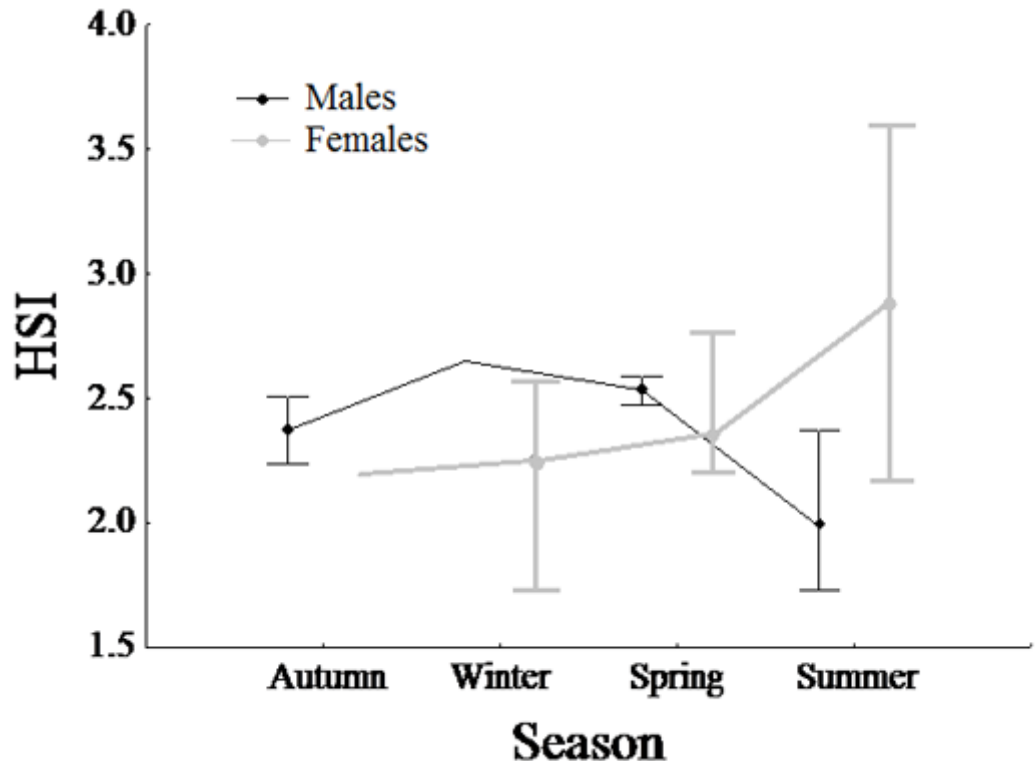
Specimens were measured to the nearest millimetre below in total length (TL) and weighted as gutted ( $M_c$ ) mass (g). Gonad and liver weight (g) were recorded for both sexes. Electric scales used had 1 and 5 g precision. For weighting material of less than

1.0 g, a precision scale was used. Classification of the specimens by maturity stage was done according to criteria defined by Oddone et al. (2007) for *Rioraja agassizi* (Müller and Henle, 1841). The presence of sperm in the males' seminal vesicle was noted.

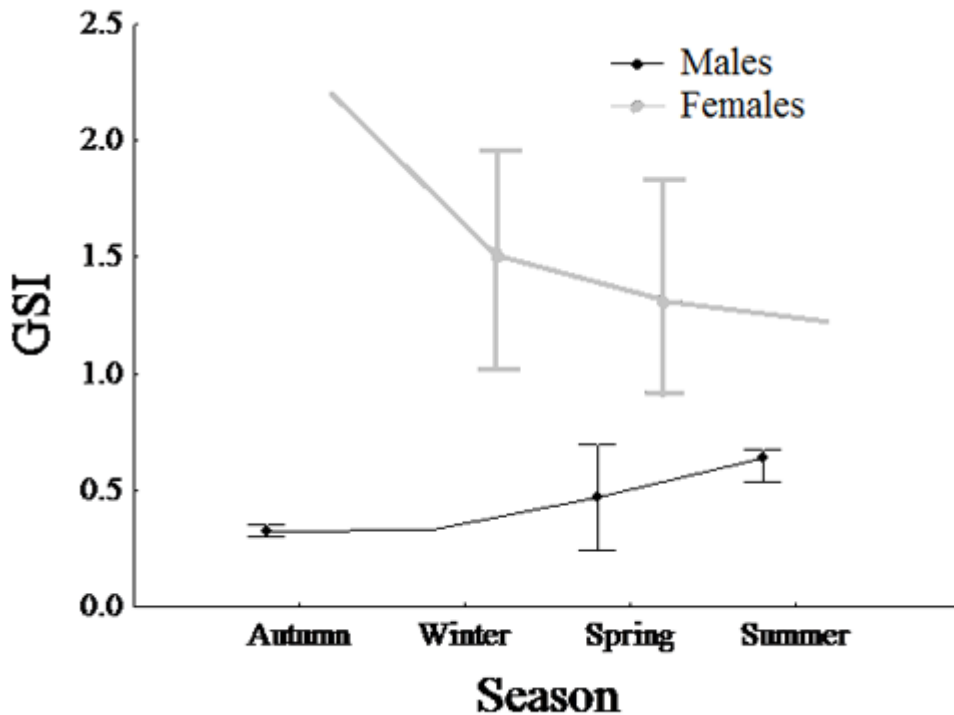
To compare the reproductive activity in males and females among seasons, the gonadosomatic and hepatosomatic indexes were calculated as:  $GSI = (\text{gonad weight}/M_c) * 100$  and  $HSI = (\text{liver weight}/M_c) * 100$ , respectively. To use parametric/non-parametric tests, normality and homoscedasticity of the variables were tested by Lilliefors' and Levene's tests, respectively. Parametric comparisons were performed using a Student t-test. Comparisons among monthly HSI and GSI were performed using Kruskal-Wallis' H-test (Sokal and Rohlf 1995). Variables range was expressed along with the mean value and the standard deviation, as 'range (mean  $\pm$  SD)'. Significance level considered in all cases was 0.05.

**Figure 2:**

Seasonal variation of hepatosomatic (HSI) for males and females of *Atlantoraja platana*, from March 2005 to March 2006. Whiskers represent the non-outlier minimum and maximum (whiskers' middle point represents the median).







**Figure 3:** Seasonal variation of gonadosomatic index (GSI) for males and females of *Atlantoraja platana*, from March 2005 to March 2006. Whiskers represent the non-outlier minimum and maximum (whiskers' middle point represents the median).

## Results

In all, 100 males ranging from 13.1 to 70.0 cm of total length (TL) were collected for analysis. Regarding the females, a total of 88 specimens ranging from 12.5 to 76.0 cm TL were recorded.

The HSI peaked in summer for the females, while it was lowest for the males. However, it did not significantly vary among seasons in males ( $H_{(3,10)}=6.67$ ,  $p=0.0830$ ) nor females ( $H_{(3,10)}=1.34$ ,  $p=0.7184$ ) (Fig. 7a). Just the opposite pattern was observed for the GSI, which showed a trend to be lowest in summer for the females, when was highest for the males. This variation of the GSI was neither significant for males ( $H_{(3,10)}=3.37$ ,  $p=0.3515$ ) nor for females ( $H_{(3,9)}=2.84$ ;  $p=0.4162$ , Fig. 7b).

Egg-bearing females were observed in spring ( $n=2$ ), summer ( $n=2$ ) and winter ( $n=1$ ). Males with sperm in the seminal vesicle were caught in spring ( $n=2$ ) and summer ( $n=3$ ).

## Discussion

We are aware that much more data should be obtained in order to infer the reproductive cycle of this species. However, this species occurs less frequently than other rajoids in the SE Brazil trawl fisheries' catches, as shown by Oddone (2007). Nevertheless, according to the observations presented in this paper and information about this species in other regions, and even other related species in the same region, we inferred that female *A. platana* undergo an annual cycle, with continuous reproduction throughout the year, and eventual seasonal variation (though not statistically significant). Marçal (2003) analysed the reproductive trends of *A. platana* for summer and winter in Southern Brazil, recording the GSI and HSI of egg-bearing females in both seasons and found no seasonal variation.

Our hypothesis of lack of seasonality in the reproduction of *A. platana* is,

A 128.0 cm total length and 410 g total weight immature female of *Atlantoraja platana*, captured in November 2005 off southeast Brazil. Black bar represents 3.0 cm. Photo credit: Maria Cristina Oddone



as stated above, reinforced by observations done in other species inhabiting the Southwestern Atlantic, including those belonging to the genus *Atlantoraja* Menni, 1972, i.e., *A. castelnaui* (Ribeiro, 1907) and *A. cyclophora* (Regan, 1903). Oddone and Vooren (2005) proposed that *A. cyclophora* in Southern Brazil have an annual cycle with continued reproductive activity, but with no peaks; or an annual cycle with at least one peak in reproductive activity in spring and/or autumn. Oddone et al. (2008), with a large data set for the same species elsewhere, demonstrated that the second hypothesis was correct. A similar pattern, with a peak in the GSI and HSI detected in autumn was reported for female *A. castelnaui* for the area in question (Oddone et al., 2008).

For other sympatric genera, as *Rioraja* Whitley, 1939 (also endemic to the southwestern Atlantic), it was observed that for the southeastern Brazilian area,

*R. agassizi* has an annual cycle, also with year long ovulation that peaks twice a year (during September and December, austral Spring) (Oddone et al., 2007). In Argentinean and Uruguayan waters, Colonello et al. (2007) also recorded continuous reproduction for *R. agassizi*, with a peak in the reproductive activity of males during late spring and summer and evidence of a partially defined annual cycle with two peaks for the females, one from

November (Spring) to February (Summer) and another in July (Winter). We infer that this presumably occurs in *A. platana* as well. Braccini and Chiaramonte (2002) reported a continuous reproductive cycle during the year, with a maximum number of females carrying egg-cases in summer, for *Psammobatis extenta* Garman, 1913.

Examples of these patterns observed in species of other areas can be seen in *Raja 'pulchra'* Liu, 1932 in the Yellow Sea, which undergo egg-laying throughout the year except in August and September, peaking from April to June and during November and December (Yeon et al. 1997). Annual reproductive cycles have been also described for *Raja clavata* Linnaeus, 1758 in British waters and *Dipturus chilensis* (Guichenot, 1848) in the Chilean coast (Holden 1975; Fuentelba and Leible 1990).

The observations carried out for all



A 70.0 cm total length and 2490 g total weight mature male of *Atlantoraja platana*, captured in November 2005 off southeast Brazil. Black bar represents 2.0 cm. Photo credit: Maria Cristina Oddone.

these rajoid species agree with one of the three reproductive cycles proposed by Wourms, continuous reproduction with eventual peaks in reproductive activity, (1977) for elasmobranch fishes and would also apply to *A. platana*. So far, this is surely the case of the females. For the males, reproductive activity probably takes place continuously during the year with no apparent peaks. This affirmation should be confirmed through histological examinations of the testicular tissues throughout the

year.

The same data set as the present data was reported for sexually resting females of *A. platana* by Oddone and Amorim (2008). A sexual resting period for the females to the ovary level was noted in several rajoid species (Holden et al. 1971; Capapé 1974, 1976; Capapé and Quignard 1974; Oddone and Voooren 2005; Ebert 2005, Oddone et al. 2007). However, as in other species, the temporal length of that period is very difficult to assess, especially because it

is not synchronized to the population level, but occurring at the species level, as it was observed in *A. cyclophora* (Oddone and Vooren 2005). For determining more details on this phenomenon, more observations should be done on a larger number of specimens and over a longer time period.

Even when taken as by-catch, skates are often subjected to high fishing mortality and as a consequence, some species have been extirpated from large regions (Stevens et al., 2000) and at least nine skate species have already disappeared from their distribution range (Brander 1981; Dulvy and Reynolds 2002). Our deeper concern is that data presented here may be the base for future studies on this species that may contribute to its conservation status and also support rational fisheries management in order to protect the populations of *Atlantoraja* spp. from disappearance. Population information such as reproductive season and area, altogether with other crucial reproductive parameters as size-at-maturity (provided for this species by Oddone and Amorim, 2008) and reproductive potential are essential for developing management and conservation plans.

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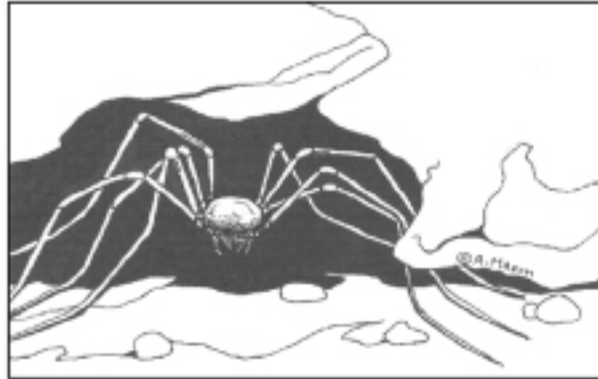


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