Priority areas for the conservation of coastal marine vertebrates in Chile

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Abstract

In the past decade, there has been growing concern about the rapid degradation of marine ecosystems due to anthropogenic causes. Consequently, identifying priority areas for the conservation of marine biodiversity has become a crucial conservation issue. Taking into account the influence of human population density, we performed complementarity analyses to identify priority areas for the conservation of all coastal marine vertebrate species in Chile (265 species), and evaluated congruence among the different target groups. The distribution ranges of all species were digitized in a geographic information system and analyses were performed on latitudinal bands of 0.5°. Our results show that 12 latitudinal bands (~16% of all latitudinal bands) are necessary to conserve at least one population of each species. Ten of these bands are irreplaceable, whereas two are flexible. Many of the irreplaceable sites lie within areas that have high human population density. In order to conserve all threatened and endemic species, six and three latitudinal bands are needed, respectively. Four latitudinal bands are needed to represent all species of fish, reptiles, and mammals, whereas nine bands are needed to protect all bird species. Taking flexible sites into account, reserve networks that meet the minimum representation goal for each taxonomic group, and for threatened and endemic species, represent subsets of the 12 latitudinal band network selected for all species. Spatial congruence among reserve networks selected for each target group was relatively low and only significantly higher than random in 9 out of 21 pairwise comparisons. However, with the exception of reptiles, conservation areas selected for different surrogate groups represented other groups relatively well, compared to randomly selected sites.

Keywords: Vertebrate diversity; Marine conservation; Chile; Priority areas

1. Introduction

Because biodiversity is distributed heterogeneously, no single area would support all the important processes and species of value for conservation. Moreover, areas containing high numbers of rare, endemic, and endangered species are not congruent in space and vary across taxonomic groups, as has been shown by studies in terrestrial taxa (Prendergast et al., 1993; Dobson et al., 1997; van Jaarsveld et al., 1998). Thus, the problem of identifying and selecting areas for conservation usually requires the application of optimization methods that maximize the preservation of species in the long-term (Williams et al., 1996; Csuti et al., 1997; Prendergast et al., 1999 for a review; Leslie et al., 2003). In this regard, reserve selection strategies, such as those based on the complementarity principle (Kirkpatrick, 1983; Vane-Wright et al., 1991) are particularly useful when resources for conservation are limited and the data available are scant.

The identification of high priority areas for conservation has usually been based on high species richness and high concentrations of endemic, rare or endangered species (Ceballos and Brown, 1995; Rodríguez and Rojas-Suárez, 1996; Ceballos et al., 1998; Reid, 1998; Dobson

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et al., 1997; Myers et al., 2000; but see Olson and Dinerstein, 1998; Kareiva and Marvier, 2003). However, the majority of these analyses concerned terrestrial habitats, and only a few of them have identified priority areas of conservation in marine habitats (Arriaga Cabrera et al., 1998; Sullivan Sealey and Bustamante, 1999; Turpie et al., 2000; Roberts et al., 2003; Leslie et al., 2003). The fact that 37% of the human population worldwide lives within 100 km of a coastline (Cohen et al., 1997) and the increasing recognition of the profound effect of human activities upon marine ecosystems (GESAMP, 1991; Norse, 1993; Lubchenco et al., 1995; Botsford et al., 1997; Coleman and Travis, 2000) have led to a strong marine conservation advocacy (Kelleher et al., 1995; Roberts et al., 2003). Although valuable, the large scale nature of these studies precludes the identification of areas that might be important within a given biogeographic zone or within particular political boundaries which, do not represent natural limits for ecological systems, but are essential to consider in order to turn conservation science into useful conservation policy.

Chile has a coastline about 4200 km long, but, so far, no national marine parks or reserve systems have been developed; only a few small marine areas have been protected to pursue long-term research projects by universities or industrial environmental monitoring (Castilla, 1996; Castilla, 1999). Recently, Fernandez et al. (2000) reviewed the current state of scientific knowledge for biodiversity conservation in Chile. They conclude that studies are needed that assess large-scale patterns in species diversity aimed at identifying key areas for biodiversity conservation. In comparison to other South American countries, Chile has set aside a relatively large proportion of land in natural reserves or protected areas (18%, e.g., Pauchard and Villarroel, 2002), although biodiversity hotspots are under-represented (e.g., Armesto et al., 1998).

Recent studies have shown a positive correlation between human population density and biodiversity (Cincotta et al., 2000; Balmford et al., 2001; Araújo, 2003; Luck et al., 2004), suggesting a spatial conflict between human settlement patterns and conservation goals. Therefore, when selecting areas for conservation, it is crucial to assess the level of overlap between densely populated areas and areas of conservation importance to minimize potential conflicts (Luck et al., 2004). Accordingly, in this study, we use information on the distribution of coastal marine vertebrate species of Chile (mammals, birds, reptiles, and fish), and on human population density to identify priority areas for conservation, and to evaluate the degree of spatial congruence among the different target groups. This is a preliminary analysis, at a broad-spatial scale, that we hope will serve as a framework for more refined analyses leading towards the establishment of a marine protected area network in Chile.

2. Methods

2.1. Data

We compiled data from the published literature on geographic distribution for a total of 265 species (25 species of mammals, 93 species of birds, 13 species of reptiles and 134 species of teleost fish) registered as resident or occasional on the coast of Chile and continental islands located less than 10 km offshore (a list of species and the sources of their distribution is available from the corresponding author upon request). We considered all species inhabiting coastal ecosystems (rocky and sandy intertidal and subtidal areas, cliffs, fjords and estuaries); pelagic species were excluded from the analysis. For birds, we compiled data for all species registered on the Chilean coast, including those that secondarily occupy coastal habitats and that might be more commonly registered in other kinds of environments (e.g. turkey vultures, egrets). For fish, we considered only marine teleost fish inhabiting coastal waters that have been captured from subsurface waters down to a maximum depth of 60 m, excluding species found only in oceanic and deep-water habitats. Additionally, we compiled data on endemism, conservation status, and habitat for each species when available. We considered endemic those species inhabiting the Chilean territory only. Data for each species' conservation status was based on Glade (1993) for birds, reptiles, and fish, on Comisión Nacional del Medio Ambiente (1996) and Glade (1993) for marine mammals, and on the IUCN red list (Baillie and Groombridge, 1996) for turtles. We included species classified as critically endangered, endangered or vulnerable to be threatened species.

We mapped the geographic distribution of each species on the Chilean coast in a grid of 76 coastal latitudinal bands of 0.5° each (approximately 50 km), between 18 and 56°S (Fig. 1). Geographic distribution was assumed continuous between range end points. Species richness was calculated as the number of species recorded in each latitudinal band.

We used the LandScan, 2002 Global Population database (Oak Ridge National Laboratory) to obtain values of human population density. Human population density was measured within the first 10 km inland from the coast. We assume that people living within this distance will have an influence on the coastal marine environment. For each latitudinal band, we calculated the percentage of its coastal length that have high (>10 people/km^2) and low ($\leqslant$10 people/km^2) human population density.

2.2. Analyses

We performed complementarity analysis to determine the "near-minimum" number of latitudinal bands

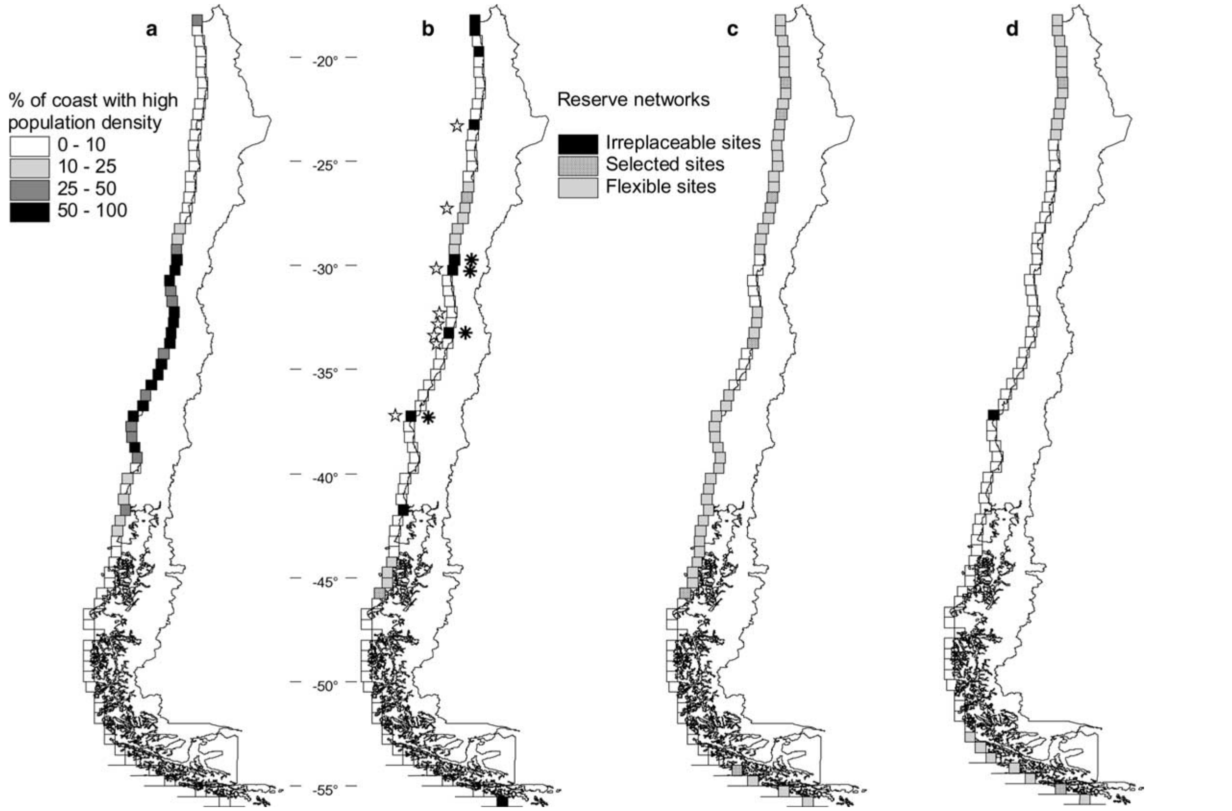

Fig. 1. Map of percentage of coastal length in each latitudinal band with high human population density (a). Near-minimum area sets representing all vertebrate species (b), endemic species (c), threatened species (d), mammals (e), birds (f), reptiles (g), and fish (h) at least once (irreplaceable in black, selected with diagonal hatch, and flexible bands in grey; asterisks denote irreplaceable bands that conflict with human population density; stars represent sites selected by Castilla (1976)). Note that latitudinal bands are represented as squares for graphical purposes only.

(minimum area sets can only be achieved using optimization procedures) that would contain at least one population of each vertebrate species (Vane-Wright et al., 1991). Complementarity is an iterative process that selects cells in a step-wise manner, such that at each step the newly selected cell includes the greatest number of species not yet represented among selected cells (Vane-Wright et al., 1991). We calculated ‘‘near-minimum’’ area sets for all vertebrate species, for endemic species, for threatened species, and for each taxonomic group independently (mammals, birds, reptiles, and fish). Complementarity analyses were performed with MARXAN software (Ball and Possingham, 2000). We used the summed rarity algorithm to obtain reserve network solutions (other algorithms gave similar results). For each target group (all species, endemic species, threatened species, and each taxonomic group) we ran the complementarity analysis 1000 times. To assess the conservation value of each latitudinal band, we used a measure of irreplaceability employed by Andelman and Willig (2003). The level of irreplaceability was then determined by the number of times a particular latitudinal band was selected. For example, a latitudinal band that was selected 1000 times was considered completely irreplaceable. Other bands that were selected in some but not all solutions were considered flexible. In all cases, when more than one latitudinal band provided the same number of unrepresented species to the solution (i.e. flexible bands), we selected those that conflict the least with human activities (i.e. those bands that had the smallest percentage of coast with high human population density), considering also that human population densities in the two immediately adjacent bands was low.

Additionally, we used two different methods to measure the level of congruence among the different target groups analyzed. The first one is the Jaccard coefficient, which is a similarity index that ranges from 0 to 100, and measures the percentage of sites in a reserve network that are shared across two networks (van Jaarsveld et al., 1998):

$$J = N_c/(N_1 + N_2 - N_c) \times 100,$$

where N_c is the number of common sites in a pair of reserve networks, and N_1 and N_2 are the number of sites in the pair of reserve networks. To test the significance of reserve network similarity, we compared the observed values with randomly generated Jaccard values (Warman et al., 2004). For each pairwise comparison, we generated Jaccard values for 1000 pairs of randomly

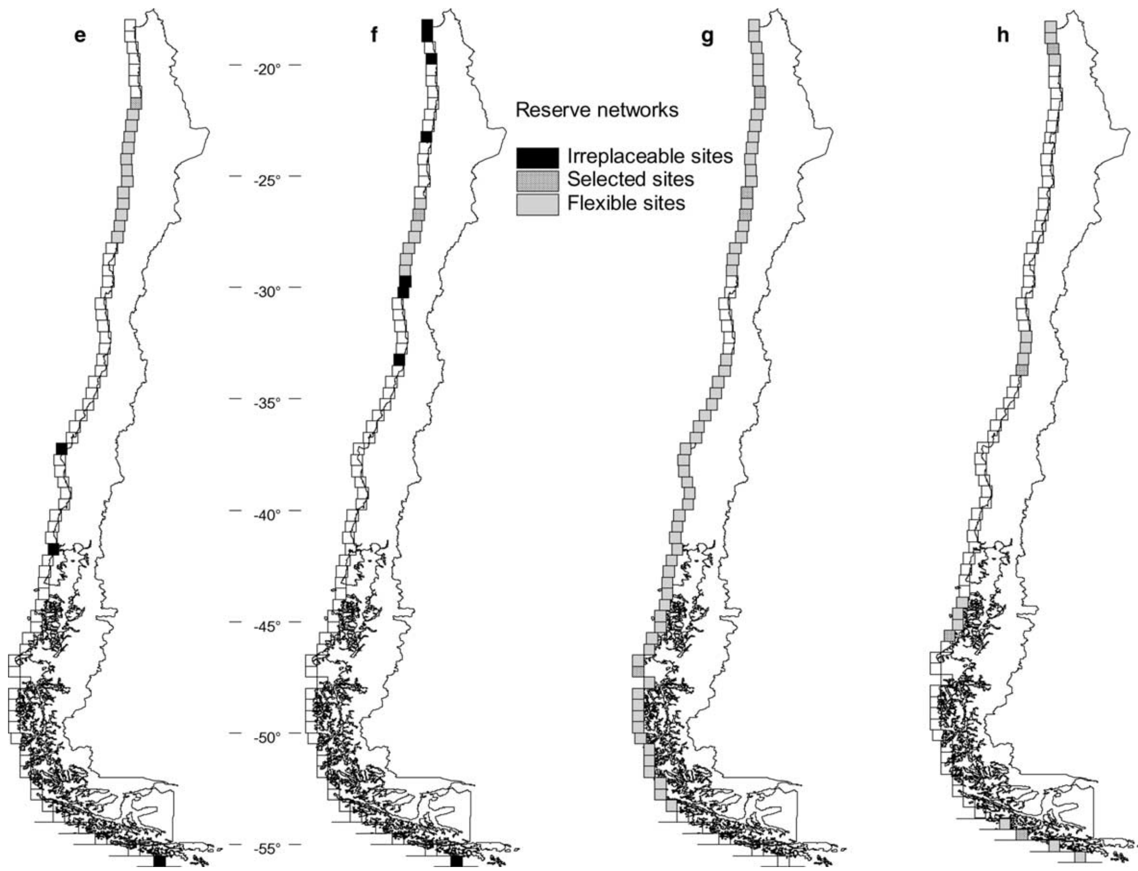

Fig. 1 (*continued*)

selected reserve networks with the same number of latitudinal bands as in the observed networks. The second measurement of congruence we term "surrogacy", and measures the proportion of a target group that is represented when a reserve network for a focal group (surrogate group) is selected. For each near-minimum set, we generated 1000 random sets with the same number of latitudinal bands as in the near-minimum set. We used the upper 99% confidence interval of random sets to evaluate the effectiveness of surrogate groups in representing other groups.

3. Results

Latitudinal bands in central Chile (between 30 and 40°S) have a large percentage of their coastal length with high population density (Fig. 1a). We found a positive significant correlation between human population density (mean population density per latitudinal band) and total species richness (Spearman-rank correlations, $r_s = 0.51$, $p < 0.001$), endemic species richness ($r_s = 0.72$, $p < 0.001$), and threatened species richness ($r_s = 0.67$, $p < 0.001$).

Based on the principle of complementarity, to conserve at least one population of all 265 species of marine vertebrates the near-minimum area set includes 12 latitudinal bands (Fig. 1b, mid-point of latitudinal bands selected with alternative selection for flexible bands shown in brackets (data in decimal degrees): −18°25′, −18°75′, −19°75′, −23°25′, −26°75′ [−26°25′, −27°25′, −27°75′, −28°25′, −28°75′, −29°25′], −29°75′, −30°25′, −33°25′, −37°25′, −41°75′, −44°25′ [−44°75′, −45°25′, −45°75′], −55°75′). Ten of these latitudinal bands are considered irreplaceable (i.e. they were selected 1000/1000 times), whereas two are flexible. There were seven bands that were flexible for one site, and four for the other site (Fig. 1b). From the two sets of flexible bands two sites were selected (shown with diagonal hatch in Fig. 1) that had a small percentage of the coast with high human population density. In Fig. 2b–h a single near-minimum set for a target group is composed of the irreplaceable bands and the selected bands. Flexible bands are the alternative for selected bands, so that replacing selected bands with any of the flexible bands would produce the same result. In all but one case, all 1000 runs for each target group were split evenly among flexible bands. Therefore, flexible bands were selected approximately the same number of times. We also performed the analyses including human

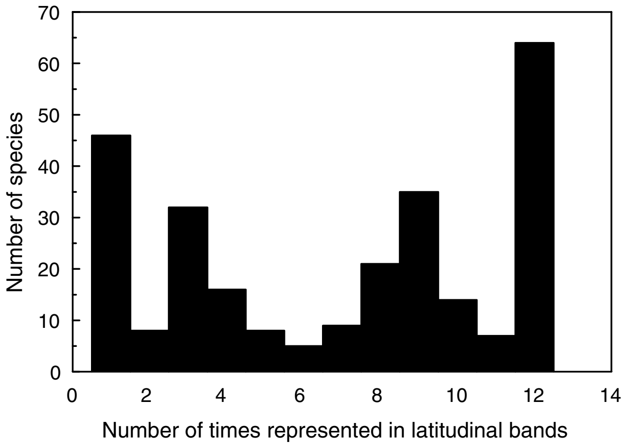

Fig. 2. Frequency distribution of the number of bands in which a species was represented in the near-minimum area set for all marine vertebrate species in coastal Chile.

population density in the cost function in the selection algorithm with the same results. The only difference was that the two bands selected from the alternative flexible bands were included in all solution sets (i.e. they became irreplaceable). Similar results were obtained for the other target groups. However, since this is a preliminary, broad-brush study, we deemed it appropriate to present the results including the alternative flexible bands. Thus, when more refined analyses are performed at regional scales, managers and decision makers will have all available information regarding the alternative sites that meet the representation goals.

Of the total number of species (265), 13 (~5%) are restricted to one latitudinal band within the coast of Chile. Although the target representation used was 1, only 12% of the species present in more than one latitudinal band within Chile were represented in only one of the selected sites, 3% were represented in two, and the remaining were represented in three or more of the twelve selected sites (Fig. 2). Species represented in all 12 latitudinal bands (Fig. 2) are distributed along the entire coast of Chile. This might be an overestimation due to the continuous distribution assumption between range end points. Taking into account human population density, four of the 10 irreplaceable latitudinal bands have more than 50% of their coastal length with high population density (asterisks in Fig. 1b).

In order to conserve all endemic species the near-minimum area set consists of six latitudinal bands, none of which were found to be irreplaceable (Fig. 1c). If we consider flexible bands, this solution set is nested within the set selected for all species. The analysis based on threatened species indicates that to conserve all endangered and vulnerable species three latitudinal bands are needed (Fig. 1d). Only one of these was found to be irreplaceable. Remarkably, this band had a high percentage (>50%) of its coastal length affected by humans. Again, considering flexible bands, these three areas are also a subset of the near-minimum area set selected to conserve all species.

Four latitudinal bands are needed to represent all species of mammals, reptiles and fish (Fig. 1e, g, and h), whereas nine are required to represent all bird species (Fig. 1f). Three and eight latitudinal bands were considered irreplaceable in the mammal and bird priority sets, respectively, whereas no irreplaceable bands were found in the reptile and fish priority sets (Fig. 1e–h). One and three irreplaceable bands, for mammals and birds

Table 1
(a) Spatial congruence (in percentages) among reserve networks selected for the different target species groups, measured by Jaccard coefficients. All values are significant except for those noted with ns (not significant). (b) Measure of surrogacy among the different target groups. Values are the proportion of species in a target group (rows) represented within the near-minimum set for the surrogate group (columns). The values in parentheses are the 99% confidence intervals for the 1000 randomly selected reserve networks. All values are significant except for those noted with ns (not significant)

	All species	Endemic	Threatened	Mammals	Birds	Reptiles	Fish
(a)							
All species		50.00	25.00	33.33	75.00	33.33	33.33
Endemic			50.00^{ns}	42.85^{ns}	50.00^{ns}	66.66	66.66
Threatened				37.50^{ns}	20.00^{ns}	40.00^{ns}	40.00^{ns}
Mammals					18.18^{ns}	33.33^{ns}	14.28^{ns}
Birds						44.44^{ns}	60.00
Reptiles							33.33^{ns}
Fish							
(b)							
All species		0.98 (0.83)	0.84 (0.73)	0.90 (0.80)	0.95 (0.79)	0.73^{ns} (0.79)	0.85 (0.74)
Endemic			0.60^{ns} (64.25)	0.72 (0.70)	0.96 (0.85)	0.84 (0.71)	0.92 (0.71)
Threatened		0.95 (0.85)		1.00 (0.82)	0.95 (0.89)	0.71^{ns} (0.81)	0.95 (0.71)
Mammals		0.84 (0.71)	0.88 (0.59)		0.88 (0.77)	0.36^{ns} (0.64)	0.80 (0.78)
Birds		0.90 (0.73)	0.84 (0.75)	0.87 (0.78)		0.84 (0.78)	0.87 (0.78)
Reptiles		1.00 (0.73)	0.61 (0.54)	0.61^{ns} (0.62)	1.00 (0.83)		0.84 (0.64)
Fish		0.98 (0.88)	0.88 (0.78)	0.94 (0.83)	0.97 (0.91)	0.73^{ns} (0.83)	

respectively, have conflicts with high concentration of people (Fig. 1e and f). Considering flexible bands, priority sets selected for the four taxa (mammals, birds, reptiles, and fishes) are nested within the near-minimum area set needed to conserve all species.

The degree of spatial overlap among reserve networks selected for each target group was low for most comparisons, with the exception of those between birds and all species, reptiles and endemic species, fish and endemic species, and fish and birds (Table 1a). Only 9 of 21 pairwise comparisons were significantly higher than random. The level of surrogacy among the different target groups was generally high (Table 1b). The worst surrogate group, in terms of representing other species, was the reptiles. Also, mammals were not very good at representing reptiles, and threatened species represented a significantly lower proportion of endemic species with respect to random sets. In general, the best surrogate groups were, in order of importance, birds, endemics, and fish.

4. Discussion

Chile has a long tradition of studies documenting the effects of human activities on marine ecosystems (see Castilla, 1999 and Fernandez et al., 2000 for reviews). However, despite this tradition and despite the social and economic importance of coastal ecosystems for Chile, there currently are no organized efforts for protecting marine biodiversity (Fernandez et al., 2000). Our paper attempts to fill this gap from the perspective of vertebrate species associated with coastal marine environments. Our results identify 12 latitudinal bands, which, if set aside as marine reserves, will collectively represent at least one population of each of the vertebrate species considered in our analysis. Similarly, six areas would do the same for endemic vertebrate species and three for threatened ones. However, some of the sites selected – mostly irreplaceable sites – overlap with densely populated areas with more than 50% (and in one case, with up to 89%) of their coastal length with high population density. Therefore, conservation efforts should concentrate in the areas of the coast within these irreplaceable bands that have low population density, and should be of highest priority before they are converted through human use. On the other hand, when there are alternative bands from which to choose (flexible bands) it is possible to minimize conflict between species representation and human population density by selecting those bands that have a low percentage of their coast impacted by humans.

More than 20 years ago, Castilla (1976; see also 1986, 1996) stressed the urgent need to establish a network of marine protected areas in Chile, and proposed specific places where this could be done. However, lack of data and ad hoc methods precluded, at that time, the performance of a more systematic priority-setting analysis. More recently, Sullivan Sealey and Bustamante (1999) identified the northern part of the Chilean coast as a high priority for conservation. Our results show a remarkable concordance with the sites highlighted by Castilla (1976) almost 30 years ago. Four of the eight sites identified by Castilla (see stars in Fig. 1b) are found to be irreplaceable latitudinal bands in this study. The remaining sites identified by Castilla (1976) lie within flexible cells, either of the reserve network for all species or of the other groups (Fig. 1b). It is worth stressing that the proposal made by Castilla (1976) was based on his knowledge of ecosystem processes, human disturbance and marine species distribution. Overall, we found that the spatial overlap among the near-minimum sets selected for the different target groups was low. In general, congruence patterns between complementary sets of priority areas for different target groups are ambiguous (Dobson et al., 1997; van Jaarsveld et al., 1998; Andelman and Fagan, 2000; Warman et al., 2004). The inconsistency of these results may be due to the use of different methods to measure surrogacy, different scales of study and sampling units, and different underlying geographic patterns (Warman et al., 2004). Additionally, the problem with some conservation assessments is that only a single set is considered for comparisons and, although the existence of flexible areas has been recognized, they rarely are considered in the analyses (Hopkinson et al., 2001). When flexible sites are taken into account significantly higher levels of overlap are often found between priority sets for different taxonomic groups (Hopkinson et al., 2001). In contrast to the low congruence of network sites selected for each target group, we found that the level of surrogacy among the different groups was relatively high. Our results are in agreement with Warman et al. (2004), in that the representation of other groups is a more suitable measure of surrogacy than the congruence of network sites selected for each target group. Indeed, two target groups may have only a few sites of their respective reserve networks in common but these may harbor a large number of species.

Marine reserves, or no-take areas may be designed to serve different purposes such as conservation of populations and habitats (e.g. Agardy, 1994; Castilla, 1996; National Research Council, 2001), fisheries management (e.g. Holland and Brazee, 1996; Lauck et al., 1998; Castilla and Fernandez, 1998; Coleman and Travis, 2000), tourism, education, scientific research, or recreation (Roberts et al., 2003; see also National Research Council, 2001 for a general assessment). The recent increase in the number of newly established marine reserves (Allison et al., 1998; Palumbi, 2000) has raised the issue of adequately complementing some of these conflicting demands (e.g., Villa et al., 2002). The Chilean case is not different, for its coastal areas are subjected to several uses, including recreational, scientific, touristic, and extractive. The latter use is manifest in the large area of coast devoted to shellfisheries through management and exploitation areas (MEA; Castilla, 1994). MEAs represent small-scale (artisanal) fisheries of invertebrates based on a co-management (joint management involving resource users and government) and have helped assure the sustainable exploitation of several economically important species (Castilla and Fernandez, 1998); they number over 500 and many more are already in the process of formation. The large number of these areas already in place and functioning and associated high human population density will likely make difficult the establishment of preserves. In a preliminary attempt to evaluate the potential conflict of MEAs and vertebrate marine conservation, we estimated the percentage of the coast in each irreplaceable band that is occupied by MEAs. Overall, half of the irreplaceable bands have >30% of their coastal length with MEAs, whereas the other half have a relatively low percentage of their coast with MEAs (Table 2). Not surprisingly, most of the irreplaceable bands with high population density also have a large percentage of their coast dedicated to MEAs. Based on this preliminary assessment, urgent action may be needed in terms of assuring compatibility between marine conservation sites and MEAs.

Although there is no clear threshold of population density above which there is significant impact on biodiversity, we assumed that densities higher than 10 people/km^2 have a significant impact to habitats, following Sanderson et al. (2002) who gave the maximum score of human influence to densities above 10 persons/km^2. Therefore, our threshold of 10 people/km^2 seems reasonable. We also calculated the percentage of latitudinal bands with high and low human population density using different criteria (2 km inland from the coast, and a threshold of 15 people/km^2), and the results were very similar. Indeed, there were no significant differences in the percentage of the coastal length of each latitudinal band with high and low human population densities when using different distances from the coast (2 vs. 10 km inland) and human population thresholds (10 vs. 15 people/km^2; Kolmogorov–Smirnov tests for both percentage of latitudinal band with high and low human population density $p = 0.97$).

Table 2
Number of marine exploitation areas (MEAs), and percentage of the coast occupied with MEAs in each irreplaceable latitudinal band

Latitudinal band	Number of MEAs	Percentage of the coast with MEAs
1	1	54.10
2	0	0.00
4	1	5.16
11	1	2.87
24	12	51.86
25	10	39.34
31	14	61.31
39	20	30.81
48	58	20.38
76	0	0.00

Latitudinal bands are ordered from north to south (see Fig. 1(b) for location of irreplaceable bands).

Some important caveats apply to the site-prioritization exercise in this study. First, the distribution data was considered as presence-absence of each species in latitudinal bands of 0.5°. This may introduce some error by considering a species present when its actual distribution range partially overlaps a latitudinal band. Also, the assumption that species are distributed continuously between range end points may represent a serious source of error, and limits the scale at which our results can be applied. Second, we did not consider ecosystem processes, species' population dynamics, differences in vulnerability to environmental disturbance or catastrophe, or habitat representation, such as nursery grounds, areas of larval retention and upwelling zones (Allison et al., 2003; Sala et al., 2002) in our analyses. Such information is not available at this time, but could be incorporated if it becomes available. Third, species have differential area requirements for population viability, thus a 0.5° band may be sufficient to maintain viable populations of a restricted-ranging species (because a greater percentage of its total range is protected), but not enough for a wide-ranging one. Finally, the planning units used in this study are relatively large (latitudinal bands of 0.5°), and we do not pretend that they should be set aside in their entirety for conservation. However, these analyses should help direct conservation planning in those areas identified as of highest priority. Further studies, at a finer spatial resolution, should be undertaken in order to complement the present analyses, which in this context represents a first step towards a fully developed conservation planning project for the Chilean coastal marine environment.

When designing national and regional priorities for conservation several factors in addition to biological data should be taken into account. The establishment of marine protected areas implicitly acknowledges the threat posed by humans through consumption and/or extraction of marine resources, and the associated habitat degradation resulting from these uses (Roberts et al., 2003). However, human population is not always explicitly considered during the conservation planning process (but see Abbitt et al., 2000; Balmford et al., 2001; Cowling et al., 2003; Rouget et al., 2003; Luck et al., 2004). Given the current national and global trends in human population growth in coastal marine areas, our study represents a coarse approach of how conflict between human population and marine biodiversity can be minimized, and can be used as a first step towards the conservation planning of the marine coast in those areas around the world where fine scale data on biodiversity and human impacts is not yet available.

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