Conservation Biology

Contributed Paper

Accounting for enforcement costs in the spatial allocation of marine zones

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Abstract: *Marine fish stocks are in many cases extracted above sustainable levels, but they may be protected through restricted-use zoning systems. The effectiveness of these systems typically depends on support from coastal fishing communities. High management costs including those of enforcement may, however, deter fishers from supporting marine management. We incorporated enforcement costs into a spatial optimization model that identified how conservation targets can be met while maximizing fishers' revenue. Our model identified the optimal allocation of the study area among different zones: no-take, territorial user rights for fisheries (TURFs), or open access. The analysis demonstrated that enforcing no-take and TURF zones incurs a cost, but results in higher species abundance by preventing poaching and overfishing. We analyzed how different enforcement scenarios affected fishers' revenue. Fisher revenue was approximately 50% higher when territorial user rights were enforced than when they were not. The model preferentially allocated area to the enforced-TURF zone over other zones, demonstrating that the financial benefits of enforcement (derived from higher species abundance) exceeded the costs. These findings were robust to increases in enforcement costs but sensitive to changes in species' market price. We also found that revenue under the existing zoning regime in the study area was 13–30% lower than under an optimal solution. Our results highlight the importance of accounting for both the benefits and costs of enforcement in marine conservation, particularly when incurred by fishers.*

Keywords: Chile, conservation planning, linear programming, marine stakeholders, reserve design, spatial optimization, territorial user rights

Justificación de los Costos de Aplicación en la Asignación Espacial de Zonas Marinas

Resumen: *Los stocks de peces marinos en muchos casos se extraen por encima de niveles sustentables, pero pueden protegerse por medio de sistemas de zonación de uso restringido. La efectividad de estos sistemas depende típicamente del apoyo de las comunidades pesqueras costeras. Los altos costos de manejo, incluyendo los de la vigilancia, sin embargo pueden disuadir a los pescadores de apoyar el manejo marino. Incorporamos los costos de aplicación en un modelo de optimización espacial que identificó cómo los objetivos de conservación pueden alcanzarse a la vez que se maximizan los ingresos de los pescadores. Nuestro modelo identificó la asignación óptima del área de estudio de entre zonas diferentes: sin captura, derechos del usuario territorial para las pesqueras (TURFs, en inglés) o de acceso abierto. El análisis demostró que aplicar las zonas sin captura y TURF provocan un costo pero resulta en una abundancia más alta de especies al prevenir la pesca furtiva y la sobrepesca. Analizamos qué tanto afectaron el ingreso de los pescadores afectados los diferentes escenarios de aplicación. El ingreso de los pescadores fue aproximadamente 50% más alto cuando los derechos territoriales se aplicaron que cuando no se aplicaron. El modelo asignó preferencialmente un área a la zona con aplicación de TURF sobre las otras zonas, demostrando que los beneficios financieros de*

Paper submitted February 10, 2014; revised manuscript accepted May 9, 2014.

DOI: 10.1111/cobi.12358

la aplicación (derivados de la abundancia más alta de especies) excedieron los costos. Estos hallazgos fueron robustos para los incrementos en los costos de aplicación pero sensibles a cambios en el precio de mercado de las especies. También encontramos que el ingreso bajo el régimen actual de zonificación en el área de estudio era 13-30% más bajo que bajo una solución óptima. Nuestros resultados resaltan la importancia de justificar los beneficios y los costos de la vigilancia en la conservación marina, particularmente cuando son pagados por los pescadores.

Palabras Clave: accionistas marinos, Chile, derechos del usuario territorial, diseño de reservas, optimización espacial, planeación de la conservación, programación lineal

Introduction

Growing industrial and consumer demands are negatively affecting fish stocks, which are extracted above sustainable levels in many fisheries (FAO 2012). Restricted-use management zones such as marine reserves (Alcala & Russ 1990) or territorial user rights for fisheries (TURFs) (Castilla 2010; Wilen et al. 2012) can promote sustainable extraction of marine resources and provide economic benefits through higher species' abundance in managed zones (Gelcich et al. 2012). This zoning also involves economic costs including establishment costs; management costs, particularly for enforcement (White et al. 2000; Balmford et al. 2004); and opportunity costs, such as forgone fishing or tourism revenue (Sanchirico & Wilen 2007; Smith et al. 2010). Marine zoning may also generate nonfinancial benefits and costs such as nonmarket ecosystem values (e.g., Pendleton et al. 2007).

Community support is usually necessary for successful marine management (Lundquist & Granek 2005; Klein et al. 2008), but the management and opportunity costs of marine zoning are often incurred primarily by local communities (Cinner 2007), potentially compromising support. Accounting for the costs of marine zoning, and its potential benefits, may be important for maintaining community support (Granek & Brown 2005).

Spatial optimization models, for example, Marxan with Zones (Watts et al. 2009), may be used to design marine zoning while accounting for management and opportunity costs. Previous researchers have used spatial optimization models to minimize the cost of meeting species' representation targets (Kirkpatrick 1983) and to maximize abundance—subject to area or budget constraints (Ando et al. 1998; Polasky et al. 2001). The majority of spatial optimization models used in conservation employ mixed integer programming with binary decision variables and heuristics (Pressey et al. 1996) or optimization techniques (Önal & Briers 2003) to solve the decision problems. Spatial optimization models are used in terrestrial applications, where reserves compete with forestry and other land uses (Polasky et al. 2005), and in marine applications, in the design of marine reserves and fisheries management (Klein et al. 2008).

A number of researches highlight the benefits of including economic variables in optimal reserve design (e.g., Polasky et al. 2001). This has resulted in more cost-effective solutions (better outcomes achieved given fixed resources) and in more efficient conservation planning (fewer resources required to achieve given objectives) (Ando et al. 1998; Moore et al. 2004; Naidoo et al. 2006). Marine conservation planning in which economic costs have been incorporated in optimal reserve design have focused on minimizing opportunity costs to fishers due to catch restrictions (Stewart & Possingham 2005) or fishery closures (Klein et al. 2009). There have been analyses of the optimal placement of reserves to maximize fishery yields or profit (Rassweiler et al. 2012; Yamazaki et al. 2012) and to determine optimal fishing effort (Hoff et al. 2013).

We devised a spatial optimization model which incorporates management and opportunity costs incurred through marine zoning. In particular, we focused on the management cost of enforcement and the opportunity cost of catch restrictions. Conservation goals were met by setting fixed species' abundance targets; the model minimized the opportunity costs to fisher communities of meeting these goals by maximizing fisher revenue. Our objective was to determine how including enforcement and opportunity costs into an equilibrium bioeconomic model affects optimal marine zoning allocation. We compared optimal zoning with existing zoning.

Methods

Study Site

Our study area was the central marine region of Chile, between 33°20' and 33°29'S. In this area, 3 fisher associations operated from the *caletas* (i.e., fishing coves) of Algarrobo, El Quisco, and Las Cruces. The study area had 8 locations where TURFs have been assigned (Fig. 1). The TURFs are part of the Chilean Fisheries and Aquaculture Law and allow fisher associations to apply for exclusive access rights for marine resource extraction in defined areas (Castilla 2010). These rights are granted on the condition that fisher associations comply with total allowable catch limits and other reporting requirements (Gelcich et al. 2005). The costs of monitoring and enforcement to prevent poaching are largely borne by fisher associations; consequently, we considered these costs from the

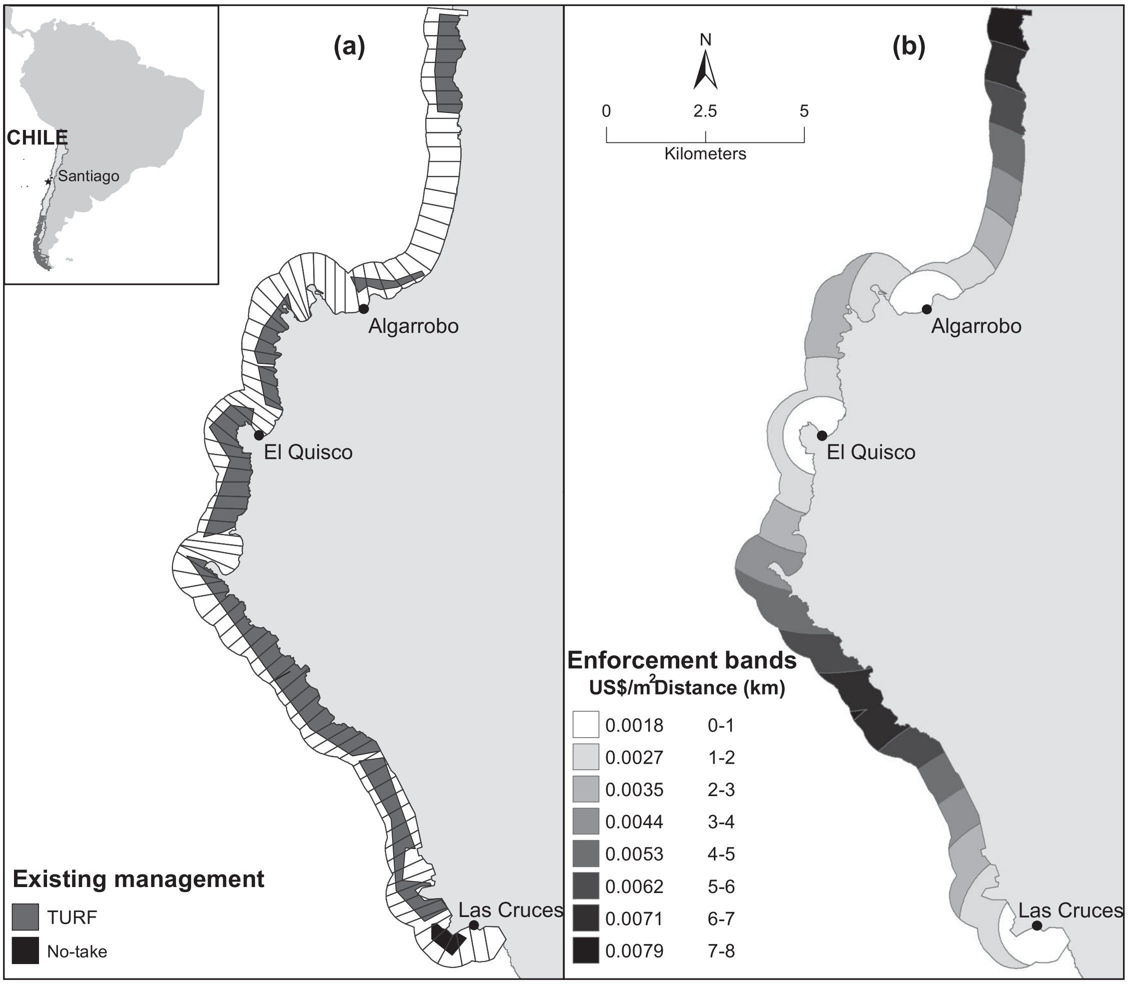

Figure 1. Study area in the central marine region of Chile (inset). (a) Locations of 3 caletas (i.e., fishing coves), existing areas of territorial user rights for fisheries (TURFs) and no-take areas, and model decision cells (i) (outlined in black). (b) Enforcement bands within the study area (costs based on data from the El Quisco fisher association [J. Moraga, personal communication]).

perspective of fisher associations and did not include enforcement costs incurred by the Chilean fisheries service or navy. The average TURF size in the study area was 136 ha. There was also 1–15-ha no-take area, which has been managed and enforced by the Pontificia Universidad Católica de Chile since 1982 (Navarrete et al. 2010).

The study area extends from the shoreline to 800m off the coast. This area was divided into 96 cells (i, $i = 1, \ldots, 96$). Each cell is approximately 30 ha in size (302,584 m^2), which ensures they are large enough to function as viable, independent reserves as evidenced from 30 years of biological monitoring of the existing no-take area (Navarrete et al. 2010).

Zones

Human activities were spatially restricted by allocating area to zones (z) representing different management activities or usage. There were 5 zones in the study area: open access (O), TURF (T), enforced-TURF (ET), no-take (N), and enforced no-take (EN). Henceforth, the term *managed area* describes all zones except open access. Marine species' abundance varied among zones. The decision problem was to allocate each of the 96 marine cells *(i)* to one or more of these zones, controlled by the decision variables $C_{O,i}$, $C_{T,i}$, $C_{\mathrm{ET},i}$, $C_{N,i}$, and $C_{\mathrm{EN},i}$. The decision variables ($C_{z,i}$) were bounded by zero and one, and fractional values were allowed. By avoiding the use of binary decision variables, the model was made more tractable and solution speed was increased (Camm et al. 1996). Each cell was fully allocated to one or more zones

$$\Sigma_z C_{z,i} = 1, \tag{1}$$

where $i = 1, \ldots, 96$.

Abundance

We analyzed 5 commercially fished species (s): 2 marine invertebrates (key-hole limpet [*Fissurella crassa*] and gastropod loco [*Concholepas concholepas*]) and 3 reef fish (biligay [*Cheilodactylus variegatus*], vieja [*Graus nigra*], and rollizo [*Pinguipes chilensis*]). The market price is the average price that one individual of a given species sells for in a first transaction at the local caleta market (Supporting Information).

Abundance levels for the 5 species were based on Gelcich et al. (2012). These authors examined abundance in 4 zones which had been established in the study area for at least 7 years: open access, TURF, enforced TURF, and enforced no-take. They found that abundance levels were a function of management; abundance (density) differed significantly between management zones ($R = 0.44$, $p < 0.01$). Abundance (A) in our model was thus an equilibrium abundance level, measured as the number of individuals per square meter of benthic habitat as observed in each zone (Supporting Information). Our data allowed the model to realistically account for species' persistence in all zones, including open access. This is in contrast to other reserve selection approaches, which commonly assume species do not survive outside managed areas (Polasky et al. 2005). We used the observed difference in abundance between the TURF and enforced-TURF zones to estimate abundance in the no-take zone from the observed abundance in the enforced no-take zone. We did not consider processes of spillover and recruitment between zones and assumed equilibrium abundance in each zone was unaffected by zone size. We assumed no net movement of species between zones because the species in question are benthic invertebrates with limited spillover potential or reef fish species with restricted home ranges (Godoy et al. 2010). We also assumed that the entire study area was available as habitat for the marine resources of interest.

The model included an abundance constraint (Eq. 2) for each species which specified a minimum level of species' abundance. This constraint functions as a conservation target and was expressed as a proportion (Aprop) of the maximum abundance (Amax) of each species (s). We calculated maximum abundance by multiplying the highest observed abundance for each species (Supporting Information) by the size of the entire study area (29 million m^2). The abundance constraint was summed across all cells and specified that the abundance of each species in the total study area must be greater than or equal to a proportion of their maximum abundance

$$\text{Aprop} \times \text{Amax}_s \leq \Sigma_i \, \Sigma_z \left(A_{s,z,i} \times C_{z,i} \right). \quad (2)$$

Stock Multiplier and Catch Levels

A stock multiplier determines what proportion of a species' total population is commercially exploitable—the exploitable stock level. We used a value of 0.30 for all species (BITECMA 2003). The catch level described the proportion of exploitable stock that could be caught in each zone: 100% in open access, 20% in TURF and enforced-TURF zones, and 0% in no-take zones (BITECMA 2003). Catch levels in TURF and enforced-TURF zones were based on the current total allowable catch limit for TURF areas. We assumed fishers catch the maximum allowable level. Chilean legislation prohibits the harvest of loco in the open access zone: a catch level of 0% was applied for this species.

Enforcement Costs

In the TURF system, fishers monitor enforced areas to counter illegal poaching (Gelcich et al. 2012), which incurs a cost. Compared with nonenforced cells, enforced cells had both higher management (monitoring and enforcement) costs and greater benefits (higher species' abundance). Henceforth, the term *enforcement costs* describes the costs of both monitoring and enforcement. Enforcement costs depend on the traveling distance from the caleta to the relevant TURF and the opportunity cost of time. The greatest distance between a location in the study area and a caleta was 8 km; therefore, we divided the study area into 8 enforcement bands (Fig. 1). Similar to Ban et al. (2009), we assumed that enforcement costs increase with distance from population centers; in our model this increase was linear. Enforcement cost in the first band was \$200/month for an average sized TURF (136 ha); this cost increased by \$100 in each subsequent, more distant band. Monetary units are in 2012 U.S. dollars, when \$1 was equivalent to CLP\$500. This data matched observed enforcement costs for the El Quisco A TURF (\$400/month) and El Quisco B TURF (\$800/month) (J. Moraga, personal communication) (Fig. 1). Only enforced zones incurred enforcement costs. We assumed enforcement costs were the same for all enforced zones.

Model Scenarios and Objectives

We analyzed 4 scenarios in which the aim was to meet abundance targets while maximizing fishers' revenue through spatial allocation of zones. The scenarios varied in their treatment of enforcement: A, no enforcement; B, enforcement, but no enforcement costs; C, enforcement with enforcement costs; and D, as for C but constrained to allocate cells to their existing zone if they were part of an existing TURF, enforced TURF, or enforced no-take.

Fishers' revenue was equal to the product of catch and market price across all species, where catch was equal to the number of individuals across all species that could be caught within the study area. Additional information regarding scenario D and model formulae are available in Supporting Information. For scenario A, we assumed no resources are spent on enforcement of user rights; the decision variables were thus $C_{O,i}$ (open access), $C_{T,i}$ (TURF), or $C_{N,i}$ (no-take).

Scenarios B and C solved for the optimal spatial allocation of zones that maximizes revenue and allowed enforced management zones to be selected ($C_{\mathrm{ET},i}$ and $C_{\mathrm{EN},i}$). We evaluated the impact of enforcement and enforcement costs by comparing scenarios B and A (with and without enforcement) and scenarios C and B (with and without enforcement cost). Scenario D accounted for the existing management in the study area—any changes to this existing allocation would likely incur costs. It was therefore worthwhile to determine the difference between the unconstrained scenario C and constrained scenario D. In scenarios C and D, the costs of enforcement were subtracted from fishers' revenue in the objective function. The model aggregated all enforcement costs incurred by fisher associations (TURF zones) and possible managers of no-take zones.

We analyzed all model scenarios at a range of abundance targets to determine how the optimum solution changed with different conservation targets. Because abundance is linked to area, proportional abundance targets were interchangeable with proportional area targets. Model scenarios A, B, and C were compared for abundance targets 0.04, 0.08, and 0.12. These targets were lower than targets generally analyzed in the literature (e.g., 20%–30% of a given habitat; Stewart & Possingham 2005; Klein et al. 2008) because the highest conservation target (Aprop) that could be achieved under scenario A was 0.12 (12% of maximum abundance) due to low abundance levels observed in nonenforced zones. Scenario A was used principally for comparison with scenarios B and C. In scenarios B, C, and D, potentially higher abundance could be achieved due to enforcement, which deters poaching. These scenarios were therefore compared at abundance targets 0.10, 0.20, 0.30, 0.40, and 0.50 to explore how solutions changed with varying conservation targets. We used the results for scenarios A, B, and C to estimate the benefit-cost ratio (BCR) of enforcement. The BCR was calculated as the ratio of benefits from enforcement relative to the costs of enforcement ([B–A]/[B–C]).

Sensitivity Analyses

Scenario C was used to test the sensitivity of the model to changes in parameter values. We determined the robustness of the optimal solution to changes in 2 key parameters: market price and enforcement cost. The relative contribution of each species to revenue was determined by its abundance and market price. Because loco is one of the major commercial species for the benthic artisanal fishing industry in Chile, it was chosen to test the sensitivity of the model. Market price varied from \$0.75/loco to \$4.50/loco, while holding all other parameters constant. We increased the enforcement cost incurred for the enforced zones (ET and EN) by factors of 2, 3, ..., 11. For each analysis, we assumed that abundance levels were unchanged; this sensitivity analysis therefore reflected uncertainty about the true costs of enforcement for given equilibrium abundance levels.

Results

Revenue under scenario A was lower than under scenarios B and C (Table 1). In scenario A, higher abundance targets resulted in the model selecting larger areas of TURF which (even with no enforcement) have somewhat higher abundance than open access areas but lower revenue due to catch restrictions.

Under scenario B, we allowed the selection of enforced zones but assumed enforcement in these zones had no cost. In this scenario, the optimal strategy was 100% enforced TURF. This strategy had higher revenues than any of the solutions for scenario A because species' abundance was greater in enforced zones. When the cost of enforcement was recognized (scenario C), the optimal strategy remained 100% enforced TURF, but fisher revenue fell by the cost of enforcement (which is undertaken by the fisher associations). This showed that the economic benefits of enforcement outweighed the costs for the 0.04–0.12 abundance targets. With no abundance target (Aprop 0.00) the BCR of enforcement was 4.6. At the highest abundance target considered under scenarios A, B, and C (Aprop 0.12), the BCR was 8.9; in all cases the BCR was much >1.

In scenarios A and B, there were no enforcement costs that varied with distance from caletas. Therefore, allocation of zones had no spatial component. Zoning solutions for scenario C did have a spatial component because enforcement costs increased with distance from the caletas. At abundance targets of 0.00–0.40, cells in lower cost-of-enforcement bands were allocated to the enforced-TURF zone (ET) (Fig. 2); these bands were closer to caletas. Cells in higher cost-of-enforcement bands were allocated to the no-take zone (*N*). This zone did not incur an enforcement cost, but has somewhat higher abundance levels than the TURF or open access zones (*T*, *O*).

The comparison of scenarios C and D showed the difference between an optimal allocation of zones and constraining allocation to account for a zones' existing status. When the model was not constrained by a zones' existing status (scenario C), more area was allocated to the enforced-TURF zone and less was allocated to the enforced no-take zone (Fig. 2). Revenues under scenario C were from 13% to 30% higher than under scenario D for all abundance targets. This showed that scenario D's

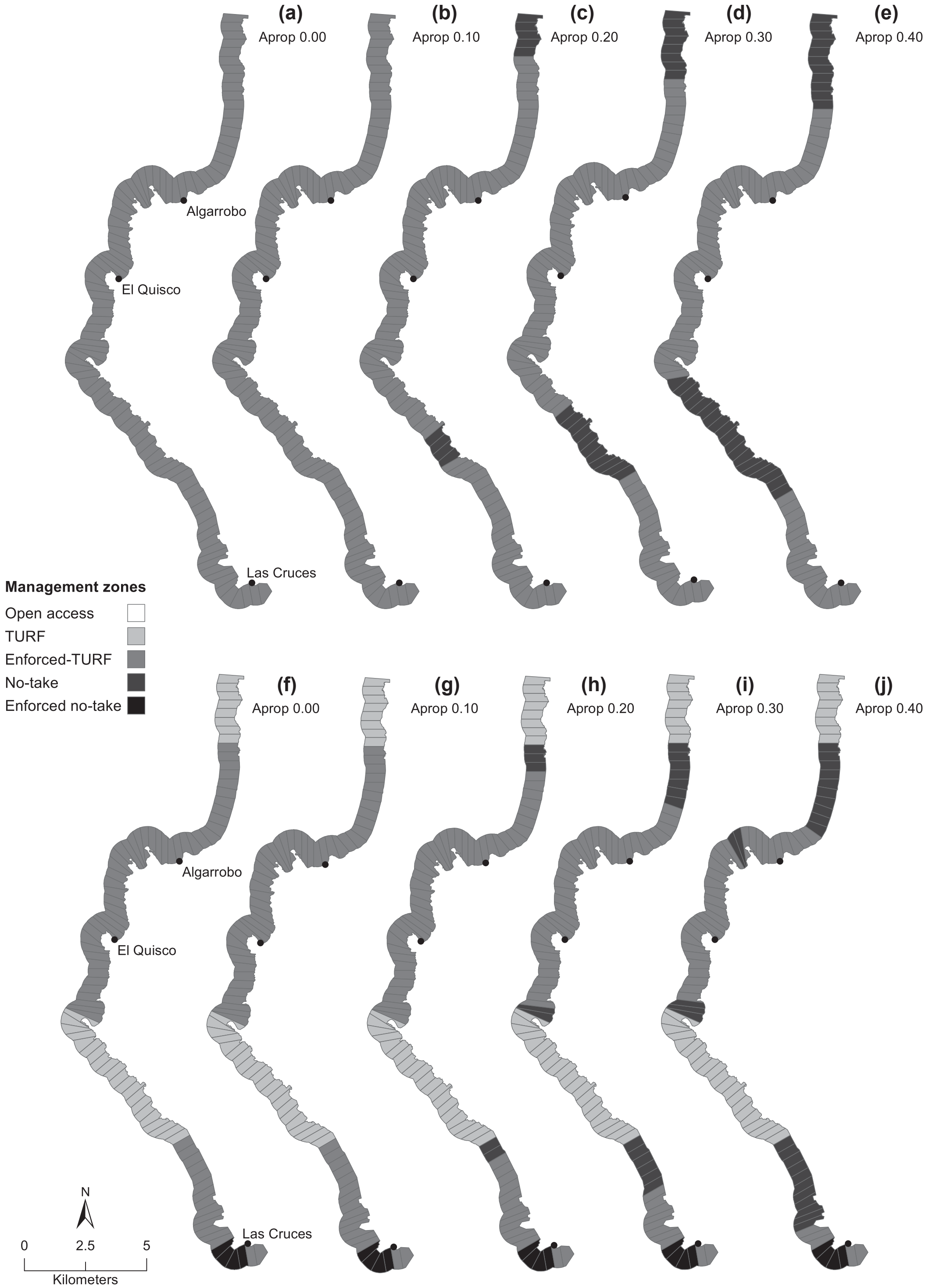

Figure 2. Zoning solutions for scenario C (upper row, spatially optimal allocation of area to different management zones when enforcement costs are incurred) and D (bottom row, spatially optimal allocation of area to different management zones when enforcement costs are incurred and the model is constrained to allocate cells to their existing zone if they were part of an existing TURF, enforced TURF, or enforced no-take): (a)–(e) different abundance targets (targets based on proportion [Aprop] of maximum abundance which was different for each species) for scenario C; (f)–(j) different abundance targets for scenario D (targets based on proportion [Aprop] of maximum abundance which was different for each species; place names, fisher association locations; TURF, territorial user rights for fisheries). Cells can be allocated to more than one zone in which case, for the purposes of illustration, we assigned them to the zone in which >50% of their area was allocated.

Table 1. Proportion of study area allocated to different management zones and fisher revenue under 3 management scenarios.

Scenario[a]	*Abundance target (Aprop)*	*Fisher revenue* [b]*(US$100,000)*	*(Net) benefits of enforcment* [c]*(US$100,000)*	*Open access (%)*	*TURF*[d] *(%)*	*Enforced TURF (%)*	*No-take (%)*	*Enforced no-take (%)*
A	0.00	2538		100	0		0	
	0.04	2465		88	12		0	
	0.08	2249		52	48		0	
	0.12	2025		17	83		0	
B	0.00	3092	554	0	0	100	0	0
	0.04	3092	627	0	0	100	0	0
	0.08	3092	843	0	0	100	0	0
	0.12	3092	1,067	0	0	100	0	0
C	0.00	2972	435	0	0	100	0	0
	0.04	2972	507	0	0	100	0	0
	0.08	2972	723	0	0	100	0	0
	0.12	2972	948	0	0	100	0	0

[a]*Model scenarios: A, no enforcement; B, with enforcement but no enforcement cost; C, with enforcement and with enforcement cost.*
[b]*Fisher revenue for scenarios A and B is equal to the product of catch and market price. Fisher revenue for scenario C is equal to the product of catch and market price minus enforcement costs, where these costs are both public (government) and private (fisher associations).*
[c]*Benefits of enforcement for scenario C represent net benefits (fisher revenue minus enforcement costs). Scenarios A and B do not include enforcement costs; benefits of enforcement are therefore equal to fisher revenue.*
[d]*Territorial user rights for fisheries.*

Table 2. Proportion of study area allocated to each management zone, and fisher revenue, under scenarios C and D.

Scenario[a]	*Abundance target (Aprop)*	*Open access (%)*	*TURF* [b]*(%)*	*EnforcedTURF (%)*	*No-take (%)*	*Enforced no-take (%)*	*Fisher revenue*[c] *(US$100,000)*
C	0.00	0	0	100	0	0	2972
	0.10	0	0	100	0	0	2972
	0.20	0	0	91	9	0	2717
	0.30	0	0	80	20	0	2386
	0.40	0	0	68	32	0	2052
	0.50	0	0	57	43	0	1716
D	0.00	0	30	66	0	4	2549
	0.10	0	30	66	0	4	2549
	0.20	0	30	61	5	4	2410
	0.30	0	30	50	16	4	2073
	0.40	0	30	38	28	4	1731
	0.50	0	30	25	20	25	1323

[a]*Model scenarios: C, with enforcement and with enforcement cost; D, as for C but constrained to allocate cells to their existing zone if they were part of an existing TURF, enforced TURF, or enforced no-take.*
[b]*Territorial user rights for fisheries.*
[c]*Fisher revenue for scenarios C and D is equal to the product of catch and market price minus enforcement costs, where these costs are both public (government) and private (fisher associations).*

existing status was suboptimal from a revenue perspective (Table 2).

A sensitivity analysis of the market price parameter showed that the allocation of zones at different abundance targets was the same under market prices \$1.50 (base case), \$3.00, and \$4.50 (Fig. 3 & Supporting Information). When the market price of loco was halved (\$0.75), area allocated to the enforced-TURF zone decreased and area allocated to the open-access and no-take zones increased. This result suggests that when the profitability of loco decreased, it was no longer as economically attractive to manage the study area as enforced TURF. At a low market price for loco, the dependence on enforced-TURF zones to meet the abundance target was no longer optimal; the same abundance was achieved with greater reliance on open-access and some no-take zones.

Changes to the enforcement cost parameter demonstrated that without an abundance target (Aprop = 0.00), the BCR of enforcement was >1 when the value of the enforcement cost multiplier was <11 (Fig. 4). It was not until the enforcement cost multiplier equaled 11 that no area was allocated to the enforced TURF zone. These results demonstrate that enforcement costs must increase substantially before enforcement is no longer beneficial.

A sensitivity analysis of changes in catch levels showed that less area was allocated to the enforced TURF and no-take zones when the catch level in TURF zones was decreased and more area was allocated to the open access

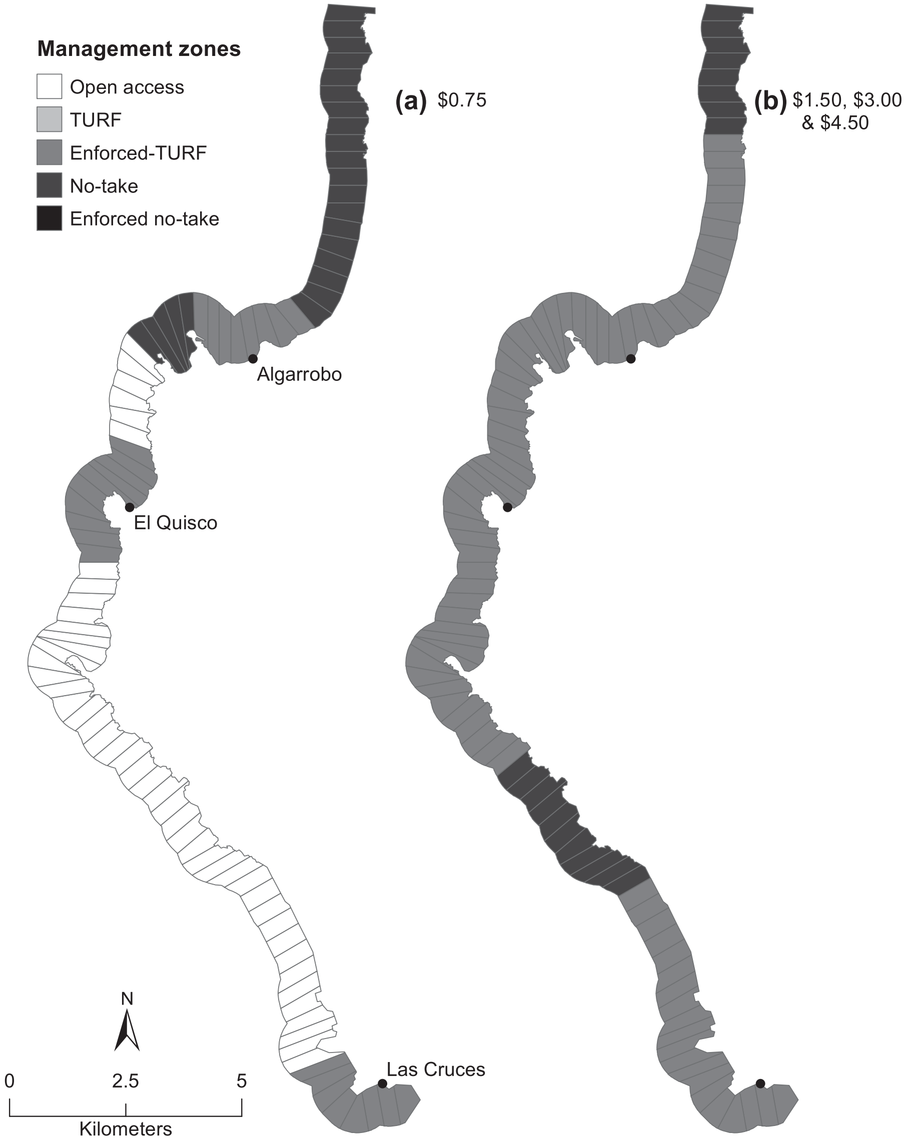

Figure 3. Sensitivity of spatial optimization model of the central marine region of Chile (scenario C [defined in Fig. 2's legend]) to different market prices of loco at an abundance target (proportion of maximum species' abundance) of 0.30: (a) loco $US0.75/individual and (b) loco $US1.50–4.50 (TURF, territorial user rights for fisheries; place names, fisher association locations).

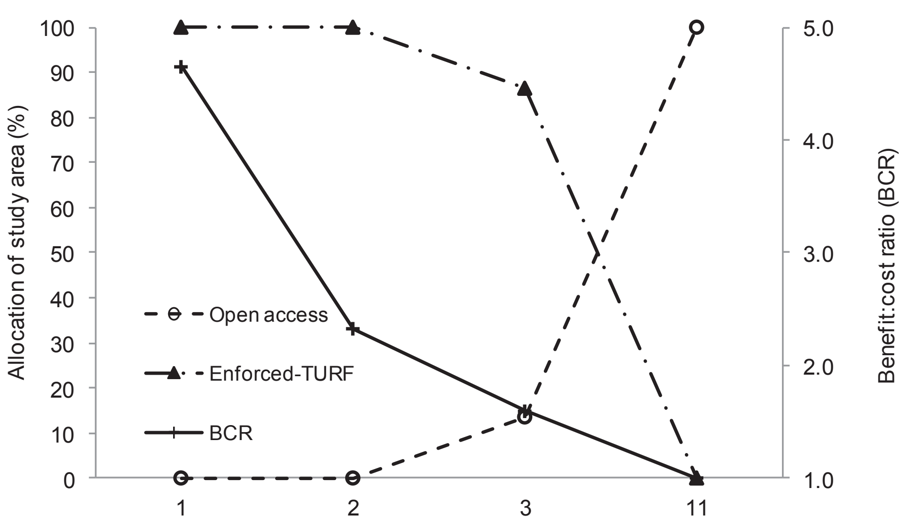

Figure 4. Spatially optimal allocation of area in the central marine region of Chile to open access and territorial user rights for fisheries (TURF) management zones and benefit:cost ratio of enforcement (proportion of maximum species abundance 0.00) for sensitivity analysis of the enforcement cost multiplier with all other variables held constant.

and enforced no-take zones. Details of the results of this analysis are in the Supporting Information.

Discussion

We investigated the impact of enforcement and opportunity costs on optimal zoning of the Chilean central marine region. Enforcement of marine protected area status is necessary to achieve the ecological benefits of protection (Guidetti et al. 2008). Where no-take areas have been enforced, higher species abundance, biomass, and richness have been observed (e.g., Jennings et al. 1996; Samoilys et al. 2007; Pierpaolo et al. 2013). It is less clear whether there are net economic benefits from enforcement: Maliao et al. (2004:352) proposed that investment in enforcement was an "efficient and necessary use of funds." Alder (1996) found that enforcement and education can significantly reduce the number of infringements in a marine park but did not comment on enforcement costs and benefits. We extended these works by estimating the net economic benefits of enforcement from the perspective of the fishing community. We found that enforcement of TURF and no-take zones resulted in substantially increased revenues for fishers and that economic benefits were much greater than enforcement costs (BCR $>$ 5). This increase in revenue can be attributed to the increase in abundance when poaching is prevented.

Fisheries theory (e.g., Cunningham 1981) demonstrates the potential for well-managed fisheries to be sustainable and maximize income for fishers without need for no-take areas. We found that the dominant zoning strategy to maximize fisher revenue with no conservation target was enforced TURF (scenarios B and C, Aprop 0.00). When conservation targets were introduced into the analyses (Aprop 0.10–0.50), area was also allocated to the no-take zone, indicating that the optimal strategy can include a mixture of multiple-use and conservation zones. This result is in line with other studies that showed that networks of no-take and managed fisheries are likely to be optimal for marine biodiversity and fisher livelihoods (Maliao et al. 2004; Claudet & Guidetti 2010).

Based on our results, one would expect fishers to enforce all TURF areas because this zone is an optimal zoning strategy even at high enforcement costs (Fig. 4). However, this was not observed in practice. Fisher associations in the study area did not enforce catch restrictions in those areas of their TURF system that were more costly to monitor. Possible explanations for this are that fishers may be underestimating the benefits of enforcement or may lack the capacity, authority, or structures needed to enforce all TURF areas. The majority of small-scale artisanal fishers cite enforcement as a major management cost that restricts their active management of areas far from a caleta (Gelcich et al. 2009, 2012). To encourage enforcement, temporary subsidies, or perhaps a training program, may be beneficial.

When areas are not enforced, it is likely that catch limits are exceeded through poaching by locals. It is possible that poaching has some social benefits which have not yet been explored. The benefits of poaching may influence community views toward poaching and limit the effectiveness of enforcement due to inability to identify and sanction poachers. However, poaching can also generate increased conflict in a community by weakening social bonds (Basurto et al. 2013). Further research into the social benefits and costs of poaching in nonenforced marine zones is needed.

Several studies suggest that, to a point, management costs per unit area will decrease as the size of managed areas increase (Balmford et al. 2004; Ban et al. 2011). Our study area was small (typical TURF size was 136 ha) relative to the areas analyzed in previous studies (e.g., the Coral Sea in Australia; Ban et al. 2011), and it is reasonable to assume that increasing marginal enforcement costs for more distant areas outweighs decreases in costs caused by efficiencies of scale when the size of a protected area increases. This is because local fishing communities have limited capacity to exploit efficiencies of scale. Here we assumed that enforcement costs increase linearly with distance, but further work is underway to investigate the potential for a nonlinear relationship.

A previous study on the design of marine reserves in the Philippines (Ban et al. 2009) included a proxy for enforcement costs of no-take areas. The authors explored how the spatial optimization model Marxan could assist in meeting conservation goals while minimizing costs. By contrast, we accounted for the impact of enforcement on species abundance. This allowed us to identify zoning solutions which meet conservation targets and maximize fisher revenue based on potential productivity in the study area under different enforcement regimes. Furthermore, we considered the enforcement costs of TURF as well as no-take areas; this allowed the optimization model to minimize the management costs of both zones.

Our results indicated that less area may be allocated to enforced TURFs when that zone is less profitable through decreases in the catch level (Supporting Information) or market price (Fig. 3). Because Chile is one of the top 10 exporters of fish and fishery products in the world (FAO 2012), the Chilean market price for commercial species is affected by fluctuations in global and domestic demand. Consequently, it may be important to understand fishers' risk management strategies in anticipation of such fluctuations. It is likely that community support for marine conservation will vary with species' market prices. We found that when the abundance target was zero, which meant the only reason for having TURFs was for fisher revenues, no area was allocated to enforced TURFs at low loco prices (Supporting Information). When conservation targets were introduced, reliance on enforced TURFs increased but remained low under low loco prices. Regulators will thus need to account for the possibility of price fluctuations and should preferably identify a spatial allocation of zones that is robust under a range of market prices.

Our model did not include ecological processes of recruitment and spillover effects between zones. Because these processes are likely to increase the benefits of enforced-TURF and no-take zones (Walmsley & White 2003; Russ & Alcala 2010), our estimates of benefits from marine management are likely to be conservative. The dominance of enforced TURF over TURF and open-access zones may consequently be understated. Incorporating recruitment and spillover processes in the model may also increase the selection of no-take zones. A second extension of the model could include habitat heterogeneity. Introducing habitat heterogeneity may alter the spatial allocation of zones, although the magnitude of such a change would depend on the magnitude of the heterogeneity in habitat condition. The study area is mainly composed of kelp-forest-dominated ecosystems (Gelcich et al. 2012). Thus, although heterogeneity no doubt exists, we anticipate that it would not change the main conclusions. Further research is being undertaken in the study area to investigate habitat heterogeneity, which may also influence management decisions. Finally, we did not consider fishing costs. Inclusion of these costs would allow the relationship between resource abundance and fishing effort (Arreguín-Sánchez 1996) to be considered more explicitly. We would expect fishing costs to decrease with higher abundance and be lowest closer to caletas. This would favor selection of enforced TURF areas that are close to caletas, which would be consistent with our results.

We found that existing management (scenario D) in the study area was less efficient than an optimal solution (scenario C): higher revenue was possible while meeting given abundance targets (Aprop 0.10–0.40; Table 2). Other researchers have also reached this conclusion (Stewart & Possingham 2005; Tognelli et al. 2009). If the costs incurred to change zoning were incorporated in the model, it would be possible to assess whether there were net benefits from that change.

Several authors have identified the need to include and understand the economic drivers of management costs when designing marine management (McClanahan 1999; Ban et al. 2011). Our model focused on the management cost of enforcement. By incorporating the impacts of distance on enforcement costs, the model minimized the costs of managed areas to marine stakeholders. It makes sense to situate managed areas in low-cost enforcement bands given that enforcement costs are spatially determined and are considered significant by fishers and that enforcement will result in significant biological benefits and associated higher revenue. If community managed marine zones are envisaged though the assignment of TURF user rights, it is important to incorporate economic impacts on local communities into the analysis to improve model predictions of fisher behavior. Nonfinancial costs and benefits of marine zoning, such as option, bequest, and existence values (Pendleton et al. 2007), are also likely to influence stakeholder support.

We found that the net benefits from enforcement of marine zones were substantial. This conclusion was sensitive to market price fluctuations, but relatively robust to increasing enforcement costs. A key implication for marine managers is that demonstrating the benefits of enforcement to fishers and supporting enforcement activities are high priorities for the future.

Acknowledgments

This research was conducted with support from the Australian Research Council Centre of Excellence for Environmental Decisions. We thank G. Doole for his input to the optimization model and N. Ban and an anonymous reviewer whose comments substantially improved the manuscript. S.G. acknowledges the financial support of Fondecyt grant 1120103, CONICYT FB 0002, The Pew fellowship, and ICM P10-033F.

Supporting Information

Market price and species' abundance data (Appendix S1), objective function formulae (Appendix S2), details of scenario D (Appendix S3), and results of sensitivity analyses of the market price (Appendix S4) and catch level (Appendix S5) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of material) should be directed to the corresponding author.

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