Accounting for enforcement is essential to improve the spatial allocation of marine restricted-use zoning systems

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This paper has been published in a peer-reviewed journal as:

Davis, K., M. Kragt, S. Gelcich, S. Schilizzi, and D. Pannell. 2015. Accounting for Enforcement Costs in the Spatial Allocation of Marine Zones. *Conservation Biology*. 29, 226-237.

9 January 2014
Working Paper 1406
School of Agricultural and Resource Economics
http://www.are.uwa.edu.au

Citation: Davis, K.J., Pannell, D.J., Kragt, M., Gelcich, S., and Schilizzi, S. (2014) *Accounting for enforcement is essential to improve the spatial allocation of marine restricted-use zoning systems*, Working Paper 1406, School of Agricultural and Resource Economics, University of Western Australia, Crawley, Australia.

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Abstract

Growing industrial and consumer demands are negatively affecting fish stocks, which are increasingly extracted above sustainable levels. Successful management of marine resources through restricted use zoning systems such as reserves and territorial user rights schemes relies on support from marine stakeholders; particularly coastal fishing communities. Restricted use zoning results in both management costs and benefits to stakeholders. To increase support for management decisions these need to be taken into account when designing optimal marine management.

A linear spatial optimisation model was developed to identify zoning solutions which maximize fishers' revenue, while meeting conservation targets. Targets were based on maximum population abundance levels for two invertebrate and three reef fish species in Chile. Revenue was maximised by allocating the study area to different management zones: no-take, territorial user rights for fishing (TURFs), or open access. Costs are incurred to enforce no-take and TURF areas; but enforcement results in higher species abundance by preventing poaching and overfishing. Several scenarios were analysed to determine the impact of enforcement on revenue.

Results demonstrated net benefits from enforcement: revenue under scenarios with enforcement was approximately 50% higher than under scenarios without it; and enforced-TURF areas were preferentially selected over other zones. Enforcement costs are one of the chief reasons that fishers in the study area stop actively managing TURFS. However, our analysis demonstrates that the often hidden benefits of enforcement far exceed the visible costs. These findings highlight the importance of accounting for both the benefits and costs of management in marine spatial design; particularly as they relate to marine stakeholders.

Key words: Spatial optimization; conservation planning; linear programming; marine stakeholders; reserve design; territorial user rights

JEL classifications: Q57

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1. Introduction

Growing industrial and consumer demands are negatively affecting fish stocks, which are extracted above sustainable levels in many fisheries (FAO 2012). Regulation such as marine reserves (Alcala and Russ 1990; Grafton et al. 2005) and other restricted-use management zones (e.g. territorial user rights for fishing, TURFs; (Gelcich et al. 2010; Sanchirico and Wilen 2007; Wilen et al. 2012)) can be used to moderate negative impacts. These marine regulations can yield economic benefits, in particular from 'spill-over' effects; relocation of adults or export of juveniles from managed to non-managed areas (Gell and Roberts 2003; Roberts et al. 2001; Russ and Alcala 2010), or from higher levels of fish abundance in managed zones (Gelcich et al. 2012). However, restricting or prohibiting human activities in managed areas will also involve economic costs. These include direct costs, such as those spent by regulators or managers on creating and then managing protected areas (Balmford et al. 2004; White et al. 2000) or TURFS (Gelcich et al. 2009); and opportunity costs (Adams et al. 2010), such as lost fishing or tourism revenue (Sanchirico and Wilen 2002; Smith et al. 2010).

Community support is usually necessary for the success of marine zoning as a conservation vehicle (Helvey 2004; Jones 1999; Klein et al. 2008; Lundquist and Granek 2005; Rudd et al. 2001), but the opportunity costs of marine regulation are often concentrated on those same communities (Cinner 2007); compromising the level of popular support. Consequently, accounting for the net costs imposed on marine stakeholders, and their likely reactions to management decisions, is important to develop successful marine zoning strategies (Granek and Brown 2005).

The resources available to fund conservation activities like marine reserves and other zoning structures are typically limited (Myers and Mittermeier 2000). To inform policy makers regarding the best use of scarce funds, optimization models have been developed to determine the best spatial configurations of reserves and other zoning structures within environments of high conservation value (Williams et al. 2004). Optimisation models can be designed to minimise the cost of meeting species' representation targets (Kirkpatrick 1983), or maximising abundance subject to area or budget constraints (Ando et al. 1998; Church et al. 1996; Cocks and Baird 1989; Polasky et al. 2001). Spatial optimisation models have been used in terrestrial applications, where reserves compete with forestry and other land uses (Polasky et al. 2005); and in marine contexts, in the design of marine reserves and other fishery management structures (Klein et al. 2008).

There are few studies that include comprehensive economic analyses in optimal reserve design (Grafton et al. 2005; Polasky et al. 2001; Thorpe et al. 2011). This is despite the fact that accounting for economic variables such as differentiated land prices, budget constraints, and the costs of management, has been shown to lead to more cost-effective solutions (Moore et al. 2004; Polasky et al. 2001). It has also been demonstrated that accounting for heterogeneity in economic costs can increase the efficiency of conservation planning. For example, by including heterogeneity in land prices, Ando et al. (1998) found that the same number of species could be protected for lower costs, or more species could be protected given a fixed budget. In a review of the economic costs involved in conservation planning, Naidoo et al. (2006) found that more species could be protected when costs are considered at the outset of planning. Economic incentives which determine fisher participation and spatial effort decisions have been found to be a critical determinant of the impact of reserves as a conservation and fisheries policy tool (Smith and Wilen 2003).

There are examples of terrestrial reserve design which have attempted to include more comprehensive measures of economic costs. In particular, there has been a focus on the role of land prices (Ando et al. 1998), and feedbacks between reserve creation and its impact on these prices (Armsworth et al. 2006; Tóth et al. 2011). Polasky et al. (2008) found that economic returns from land use could be maintained while conserving biodiversity. Similarly, trade-offs between ecological and economic goals were examined by Nalle et al. (2004) who used a spatially and temporally explicit model to evaluate land-use decisions. They found that conservation objectives could be improved without reducing the present value of timber production. The central concept behind these analyses is that accounting for heterogeneity in cost can capture synergies or complementarities in resource use.

There has been less comprehensive treatment of economics in marine reserve design. Studies where comprehensive economic costs have been included have chiefly focused on minimizing opportunity costs to fishers due to catch restrictions (Stewart and Possingham 2005) or fishery closures (Klein et al. 2009). Other studies have analysed the optimal placement of reserves to maximise fishery yields or profit (Rassweiler et al. 2012; Richardson et al. 2006; White et al. 2013; Yamazaki et al. 2012), or have optimised fishing effort (Hoff et al. 2013). There have also been studies which have focused on equity considerations implicit in marine zoning; the distribution or concentration of costs and benefits as they pertain to stakeholders (Klein et al. 2009). Grantham et al. (2013) identified potential no-take zoning solutions which had both smaller and more equitable impacts across local fishing communities. Halpern et al. (2013) explored how an explicit assessment of equity could be incorporated into spatial planning methods; they identified that significant tradeoffs between more equitable outcomes and achievement of conservation goals may be encountered.

In this paper we present a spatial optimization model which more comprehensively includes relevant economic costs involved in marine zoning, including direct and opportunity costs. Direct costs represent the management or transaction costs incurred by government, or other marine stakeholders, to enforce user rights. Our objective in this study is to determine the impact that including these costs has on optimal zoning allocation of the study area; the central marine region of Chile. We then compare this optimal zoning allocation with an allocation constrained by the existing management. The model analyses how to maintain a specified level of species' abundance, while maximizing catch revenue for the artisanal fishing industry operating within the study area. The model was designed to provide decision makers with information on how best to protect marine resources through optimal allocation of restricted use marine areas.

2. Materials and Methods

2.1 Theory

There are a number of models which have been developed for conservation planning, including Marxan with Zones (Watts et al. 2009), the project prioritization protocol (Joseph et al. 2009), and Zonation (Moilanen et al. 2005). The majority of these and other reserve design models use mixed integer programming with binary decision variables, and heuristics (Pressey et al. 1996) or optimisation techniques (Önal and Briers 2003) to solve the decision problems[1].

In this study, we optimise a mathematical programming problem using linear programming (LP). LP is designed to find the best or optimal value for a given objective function, subject to resource constraints (Pannell 1997). LP was preferred over mixed integer programming as it is less computationally difficult to solve. LP avoids the use of binary decision variables, giving the model greater tractability and speed (Camm et al. 1996). This also ensures a single optimal solution, if it exists.

2.2 Study site

The study area for this investigation is in the central marine region of Chile, between 33º20´ and 33º 29´S (Figure 1). There are three fisher associations which work from specific coves (called *caletas*) operating in this area: Algarrobo, El Quisco and Las Cruces. The study area has eight areas where territorial user rights for fisheries (TURFS) have been assigned (Figure 1). 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 marine areas (Gelcich et al. 2005). These user rights are granted on the condition that fisher associations are responsible for enforcing their user rights, and comply with total allowable catch limits and other reporting requirements. The average TURF size in the study area is 136ha. Within the study area there is one no-take area (NTA) (Figure 1), which is managed and enforced by the Pontificia Universidad Católica de Chile. This area has been managed as a NTA since 1982 and is currently the only well enforced and biologically monitored area in the country (Gelcich et al. 2012).

The study area extends from the shoreline to 800m off the coast. The study area was divided into 96 cells (C_i *where* $i = 1,2,\ldots,96$) of equal size. The cells are slightly larger than 30 ha (302 584m^2), to ensure that each cell is large enough to function as a viable reserve, as evidenced from 30 years of biological monitoring of the 15 ha NTA of the Pontificia Universidad Católica de Chile (Gelcich et al. 2012; Navarrete et al. 2010).

[1] For a review of the use of optimisation procedures versus heuristics see Csuti et al. (1997) and Rodrigues et al. (2002).

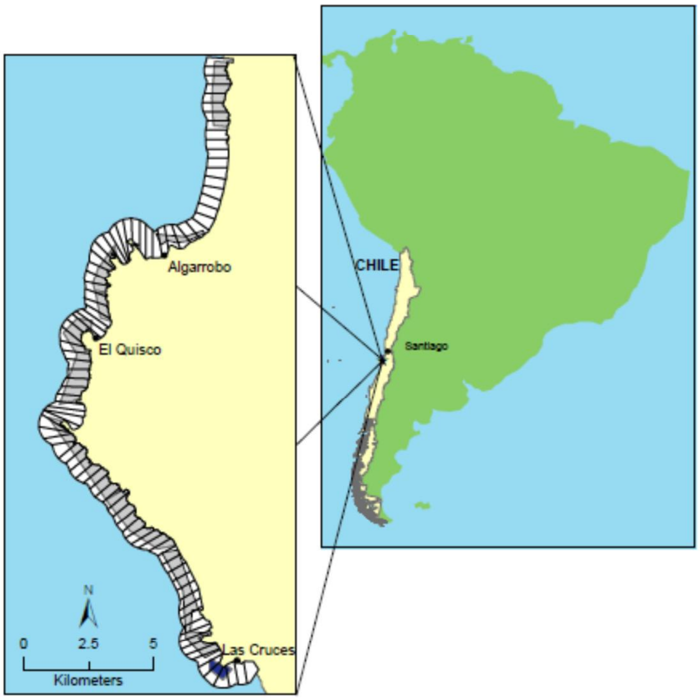

Figure 1. Chile, detail (left) shows study area with cells outlined in black, and territorial user rights for fishing (TURF) areas shaded grey. The no-take area of the Pontificia Universidad Católica de Chile is shaded in blue. Locations of caletas are indicated.

2.3 Zones

In this analysis, human activities within the study site are spatially restricted by the allocation of area to different management zones. Zones therefore represent management activities or uses of an area. There are five possible options for marine management in the study area (Table 1). The term 'managed area' is used to describe all management zones except open access (O) (i.e. TURF (T), enforced-TURF (ET), no-take (N), and enforced-no-take (EN)). Species' abundance will vary in each zone, depending on whether catch restrictions are in place and poaching is deterred (see Section 2.7). The spatial allocation of areas to different zones determines the impact on the environmental resource which is being managed: marine species' abundance.

Table 1. Management zones.

Management zone	Description
O	Open access
T	Territorial user rights for fishing
ET	Enforced territorial user rights for fishing
N	No-take area
EN	Enforced no-take area

2.4 Enforcement costs

In the Chilean TURF system, artisanal fisher associations are responsible for enforcing their own managed areas. Fishers monitor enforced zones to counter illegal poaching (Gelcich et al. 2012). This monitoring incurs a cost (*Ec*). Enforced cells therefore have higher management costs, but also result in higher abundance for each target species than non-enforced areas. Enforcement costs are primarily composed of surveillance costs, which depend on the travelling distance from the *caleta* to the relevant TURF and on the opportunity cost of time.

To spatially represent the relationship between distance and enforcement cost, enforcement bands were created around each of the three *caletas* in the study region. The innermost band area has a radius of 1km. Each subsequent band extends a further 1km using the relevant *caleta* as the reference point. The greatest distance between a marine cell and the nearest *caleta* is 8km (Figure 2). The El Quisco *caleta* provided data on enforcement costs. To enforce a TURF close to the *caleta*, this fisher association spent $400[2] per month. To enforce a TURF further from the *caleta* they spent $800 per month. This data showed that monthly enforcement costs for an average sized TURF (136ha) are approximately $200 for the innermost buffer zone, or an annual $0.0018 per m (pers. comm. Moraga 2013[2])[3]. These costs were incrementally increased by $100 with each additional kilometre from the *caleta* (Table 2). Note that only the enforced-TURF and enforced-no-take zones incur enforcement costs, and these costs are assumed to be the same for both zones.

Table 2. Enforcement costs for each enforcement band within study area.

Enforcement band	Distance to caleta (km)		Enforcement cost (average turf size)	Cost per m^2	
	Minimum	Maximum	US$ per month	US$ per month	US$ per year
1	0	1	200	0.00015	0.00176
2	1	2	300	0.00022	0.00265
3	2	3	400	0.00029	0.00353
4	3	4	500	0.00037	0.00441
5	4	5	600	0.00044	0.00529
6	5	6	700	0.00051	0.00617
7	6	7	800	0.00059	0.00705
8	7	8	900	0.00066	0.00794

[2] Jose Moraga, Presidente del Sindicato de Trabajadores Independientes Narciso Aguirre, El Quisco.

[3] All dollars are expressed in US$ (2012: one dollar is equivalent to 500 pesos).

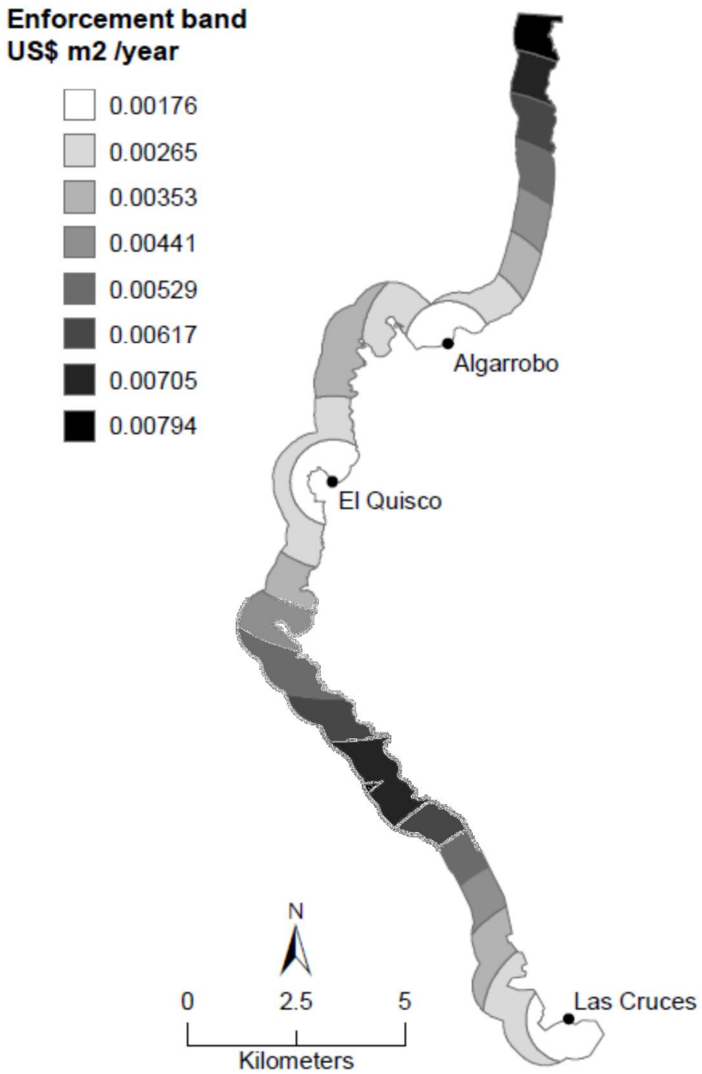

Figure 2. Enforcement bands within study area. The locations of the three caletas, or fisher associations, are indicated. Costs are based on data from the El Quisco fisher association (pers.comm. Moraga 2013).

2.5 Decision variables - zone selection

For each of the 96 marine cells C_i, there are five decision variables, representing the allocation of the cell to one of the management zone types (Table 1). Each of these variables is bounded between zero and one, and fractional values are allowed. There is a constraint for each cell requiring that each cell be allocated to one of the zone types (or to more than one zone type, summing to one):

$$C_{O_i} + C_{T_i} + C_{TE_i} + C_{N_i} + C_{NE_i} = 1 \qquad i = 1,\ldots,96 \qquad (1)$$

2.6 Species

Five different species (Sp_y) were included in the analysis: two marine invertebrates (key-hole limpet (*Fissurella crassa*) and gastropod loco (*Concholepas concholepas*)); and three reef fish (*Cheilodactylus variegatus*, *Graus nigra*, and *Pinguipes chilensis*). All five species are commercially fished. The market price (*Mp*) of each species represents the average price for which one individual of that species sells in a first transaction at the local fish cove market (Table 3).

Table 3. Market price (US$/individual) (Caleta El Quisco 2013) and species abundance predicted to occur in each management zone (number/m^2).

Name	Market price per individual (US$)	Abundance (number/m^2)				
		Open access	TURF	Enforced-TURF	No-take	Enforced-no-take
Key-hole limpet	0.50	0.024	0.176	0.296	0.447	0.497
Loco	1.50	0.014	0.074	0.537	0.365	0.405
Cheilodactylus variegates	8.00	0.014	0.041	0.038	0.260	0.288
Graus nigra	20.00	0.001	0.007	0.008	0.027	0.030
Pinguipes chilensis	8.00	0.020	0.055	0.044	0.114	0.127

2.7 Abundance

Abundance data was available from Gelcich et al. (2012) for four management zones: open access, TURF, enforced-TURF, and enforced-no-take. Having monitoring data for all management zones (including open access) makes the model more realistically account for species' persistence. This is in contrast to other reserve design models, which commonly assume that species do not persist outside of managed areas (Polasky et al. 2005). Abundance (*A*) is measured as the number of individuals per m^2 of benthic habitat (Table 3). Abundance in each zone is represented as the average equilibrium abundance specific to the management type. The total number of a species in a cell is calculated by multiplying cell size by the abundance per m^2 for that species. For example, if cell 1 were an enforced-no-take zone, this would result in 302 584 x 0.497 = 15 384 individuals of *F. crassa* in that cell. The abundance in the no-take zone was calculated based on the proportional difference in abundance in the TURF zone with and without enforcement. It was assumed that there is no net movement of species between zones. This assumption was justified on the basis that the species in question are benthic invertebrates with limited spill-over potential and reef fish species with restricted home ranges (Godoy et al. 2010). It was also assumed that all habitat was available for resources.

The model includes an abundance constraint for each species, specifying the minimum level of species' abundance (2). This constraint can be thought of as a conservation target. The constraint is expressed as a proportion (*Aprop*) of the maximum abundance (*Amax*) for each species (Sp_y). The maximum abundance is calculated by multiplying the highest observed abundance per m^2 for a species in any location of the study site, by the site's total area (29 million m^2), e.g. for keyhole limpets this calculation is *Amax* (keyhole limpet) = 0.497 (individuals per m^2, see Table 3) x 29 million (m^2) = 14 million (individuals in the study site). The constraint therefore specifies that the total abundance in the study site (summed across all cells) must be greater than or equal to a given proportion of this maximum abundance.

$$Aprop \times Amax_{Sp_y} \leq \sum_{C_i} (A_O \times C_O) + (A_T \times C_T) + (A_{ET} \times C_{ET}) + (A_N \times C_N) + (A_{EN} \times C_{EN}) \tag{2}$$

2.8 Stock multiplier and catch levels

The stock multiplier (*Sm* < 1) dictates what proportion of a species' total population is commercially exploitable. For all species a value of 0.3 was used (Bitecma 2003). This means that 30% of a given species' abundance level is of sufficient size to be harvested. From this 30%, the catch level (*Cl*) determines the number of individuals of each species that can be caught within each zone. It is expressed as a proportion of the exploitable stock level (ESL) for each species in each zone. In open access areas 100% of each species' ESL can be caught, and in the TURF and enforced-TURF zone 20% of each species' ESL can be caught (Bitecma 2003). In the Chilean legislation it is illegal to harvest Loco in the open access zone, so the catch level for this species in this zone was set at 0. The catch levels in TURF and enforced-TURF zones are based on the current total allowable catch limit negotiated for TURF areas. It was assumed that fishers catch the maximum allowable level. In the no-take zones no catch is allowed (0%). The abundance levels are equilibrium levels at the specified catch rate (Table 3).

2.9 Model scenarios and objective functions

There are four scenarios:

a is catch maximization with no enforcement;

b is catch maximization with enforcement, but no enforcement costs;

c is catch maximization with enforcement, and with enforcement costs; and

d is as *c* but constrained to take account of the existing management status of cells (*T*, *ET*, and *EN* zones) in the study region.

The objective of scenario *a* is to maximize fishers' revenue through an optimal spatial allocation of management zones across the study region. Revenue is equal to the product of catch and market price (fisher revenue) across all species, and catch is equal to the number of individuals across all species that can be caught within the study area. This scenario assumes that no resources are spent on enforcement of property rights. The decision variables (C_i) can take the value of *O* (open access), *T* (TURF) or *N* (no-take area).

Scenarios *b* and *c* solve for the optimal spatial allocation of management zones that maximises revenue, but include the option to choose management zones that are enforced (*ET* and *EN*). Scenario *d* represents the existing management in the study area (see Figure 3). Any changes to this existing management allocation are likely to incur costs; it is therefore worthwhile determining how much better the unconstrained scenario *c* is so that the benefits of this change can be evaluated.

The objective function for scenario *b* is expressed in (3). For scenarios *c* and *d*, the costs of enforcement (4) are incurred as a penalty on the objective function. This objective function is equal to the product of catch and market price across all species, minus the costs of enforcement. Poacher revenue is not included in the objective function for any scenario. The model aggregates enforcement costs although they are incurred by separate organizations; fisher associations (TURFS) and the Chilean government (NTAs).

$$Total\ catch = \sum_{C_i Sp_y} \Big((A_O \times Sm \times Cl_O \times Mp_{sp} \times C_O) + (A_T \times Sm \times Cl_T \times Mp_{sp} \times C_T) + (A_{ET} \times Sm \times Cl_{ET} \times Mp_{sp} \times C_{ET}) + (A_N \times Sm \times Cl_N \times Mp_{sp} \times C_N) + (A_{EN} \times Sm \times Cl_{EN} \times MP_{sp} \times C_{EN}) \Big) \quad \textbf{(3)}$$

$$Total\ enforcement\ cost = \sum_{C_i} (Ec_O \times C_O + Ec_T \times C_T + Ec_{ET} \times C_{ET} + Ec_N \times C_N + Ec_{EN} \times C_{EN}) \quad \textbf{(4)}$$

All model scenarios were analysed at a range of abundance target constraints to determine how the optimum solution changed when the conservation target was increased. In scenario *a*, abundance can only be as high as levels observed in non-enforced zones. Because of this limitation, the highest abundance target *(Aprop*) which could be achieved was less than or equal to 0.12 (i.e. 12 per cent of maximum potential abundance). The area allocated to each zone under model scenarios *a*, *b* and *c* was compared for abundance targets 0.04, 0.08 and 0.12. In the other three scenarios, potentially higher abundance can be achieved (due to enforcement which deters poaching). Therefore model scenarios *b*, *c*, and *d* could be compared at abundance targets 0.10, 0.20, 0.30, 0.40, and 0.50.

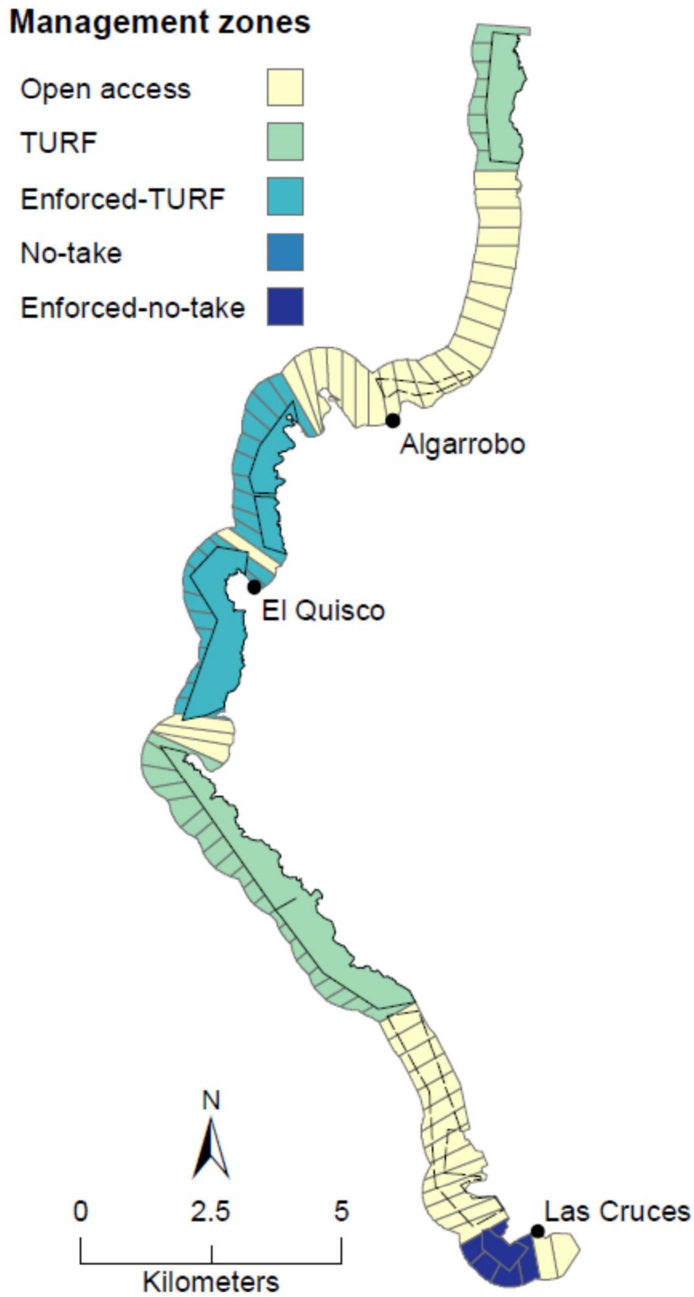

Figure 3. The base case management status for scenario d. Locations of caletas are indicated. The size of existing managed areas is over-stated due to the size (30ha) of the planning cells used in the analysis. Note that two existing territorial user rights for fishing (TURF) areas (dashed outlines) were excluded from the analysis as they were missing data (top) or the management status (bottom) was uncertain. Cells coinciding with existing managed cells (TURF, enforced-TURF, enforced-no-take, as indicated) are assigned to that management type.

3. Results

Table 4 presents the model results of scenarios *a*, *b*, and *c*. In scenario *a*, higher abundance targets result in the model selecting larger areas of TURF, but this causes lower revenue to fishers. This result reflects that TURFs with no enforcement of reduced fishing effort have somewhat higher abundance, but lower revenue due to reductions in take by complying fishers (though not by non-complying fishers).

When the option of TURFs with costless enforcement is introduced (scenario *b*), the optimal strategy for all of the shown abundance targets is 100% enforced-TURF. This strategy has higher revenues than any of the solutions for scenario *a*. When the cost of enforcement is recognised (scenario *c*), the optimal strategy remains 100% enforced-TURF, but fisher revenue falls by the cost of enforcement (which is undertaken by the fishers' associations).

Under the studied targets, revenue increases by 22% to 53% under scenario *b* compared to scenario *a*, and by 17% to 47% under scenario *c* compared to *a*. The higher revenues under scenarios *b* and *c* arise from the greater species' abundance associated with enforcement. This increase in revenue thus captures the economic benefits of enforcement.

Table 4. Proportion of study area allocated to each management zone, and revenue under model scenarios *a*, *b* and *c*.

Scenario	Abundance target (Aprop)	Revenue[a] ('000 USD)	(Net) Benefits of enforcment[b] ('000 USD)	Open access %	TURF %	Enforced-TURF %	No-take %	Enforced-no-take %
(a)	0.00	2,538		100	0		0	
	0.04	2,465		88	12		0	
	0.08	2,249		52	48		0	
	0.12	2,025		17	83		0	
(b)	0.00	3,092	554	0	0	100	0	0
	0.04	3,092	627	0	0	100	0	0
	0.08	3,092	843	0	0	100	0	0
	0.12	3,092	1,067	0	0	100	0	0
(c)	0.00	2,972	435	0	0	100	0	0
	0.04	2,972	507	0	0	100	0	0
	0.08	2,972	723	0	0	100	0	0
	0.12	2,972	948	0	0	100	0	0

[a]Revenue for scenarios *a* and *b* is equal to the product of species abundance and market price. Revenue for scenario *c* is equal to the product of species abundance and market price minus the costs of enforcement, where enforcement costs are both public (government) and private (fisher associations). [b]Benefits of enforcement for scenario *c* represent net benefits.

Enforcement provides benefits but comes at an economic cost, which is incurred in scenarios *c* and *d* (Table 5). Comparing results for scenarios *a* and *c*, the benefits of enforcement substantially outweigh the costs. The benefit:cost ratio (BCR) of enforcement is calculated as the ratio of benefits from enforcement relative to the costs of enforcement ((b-a) / (b-c)). With no abundance target

(*Aprop* 0.00) the BCR of enforcement is 4.6. At the highest abundance target considered by scenarios *a*, *b* and *c* (*Aprop* 0.12), the BCR is 8.9.

In scenario *a*, as the abundance target increases to 0.04, 0.08 and 0.12, area is increasingly converted from open access to TURF. Only at abundance target 0.12 is any area allocated to the no-take zone (4%). By contrast, the optimal zoning strategy for scenarios *b* and *c* does not change as the abundance target is increased to 0.12. In scenarios *a* and *b*, the allocation of zones has no spatial component as there are no enforcement costs that vary with distance from *caletas*.

Figure 4 shows the scenario *c* zoning solutions at abundance targets of 0.00, 0.10, 0.20, 0.30, and 0.40. At abundance targets of 0.00 to 0.40, enforced-TURF and no-take zones (*ET* and *N*) are preferentially selected over all other zones. Cells in the lower cost-of-enforcement bands are allocated to the enforced-TURF zone (Figure 4). These bands are located closer to the headquarters of fisher associations. Cells in the higher cost-of-enforcement bands (further from fisher associations) are allocated to the no-take zone.

Table 5. Proportion of study area allocated to each management zone, and revenue, under scenarios *c* and *d*.

Scenario	Abundance target (*Aprop*)	Open access %	TURF %	Enforced-TURF %	No-take %	Enforced-no-take %	Revenue[a] ('000 US$)
(*c*)	0.00	0	0	100	0	0	2,972
	0.10	0	0	100	0	0	2,972
	0.20	0	0	91	9	0	2,717
	0.30	0	0	80	20	0	2,386
	0.40	0	0	68	32	0	2,052
	0.50	0	0	57	43	0	1,716
(*d*)	0.00	0	30	66	0	4	2,549
	0.10	0	30	66	0	4	2,549
	0.20	0	30	61	5	4	2,410
	0.30	0	30	50	16	4	2,073
	0.40	0	30	38	28	4	1,731
	0.50	0	30	25	20	25	1,323

[a]Revenue for scenarios *c* and *d* is equal to the product of species abundance and market price minus the costs of enforcement, where enforcement costs are both public (government) and private (fisher associations).

The difference between scenarios *d* and *c* is that the allocation of zones in scenario *d* is constrained such that TURF and no-take zones that currently exist in the real world cannot be altered to other zone types. In both scenarios, the model allows enforced areas to be chosen, and the costs of this enforcement are incurred in the objective function. Revenue is between 13% and 30% higher for all abundance targets considered under scenario *c* compared to *d*, reflecting the cost of constraining existing zone status. The greatest difference in revenue ($424 000) between scenarios *c* and *d* is at abundance targets (*Aprop*) of 0.00 or 0.10. At these target levels, all area in scenario *c* is allocated to the enforced-TURF zone; in scenario *d*, area is constrained to be allocated to the existing management zones (T, ET and EN) (Table 5). When the abundance target (*Aprop*) increases (0.20 to 0.40), area is removed from the enforced-TURF zone and allocated to the no-take zone in both scenarios *c* and *d* (Table 5, Figures 4 & 5). At these target levels scenarios *c* and *d* are similarly constrained so there is less difference in revenue (between $307 000 and $321 000). At the highest target level analysed (0.50) there is a difference in revenue between scenarios *c* and *d* of $392 000.

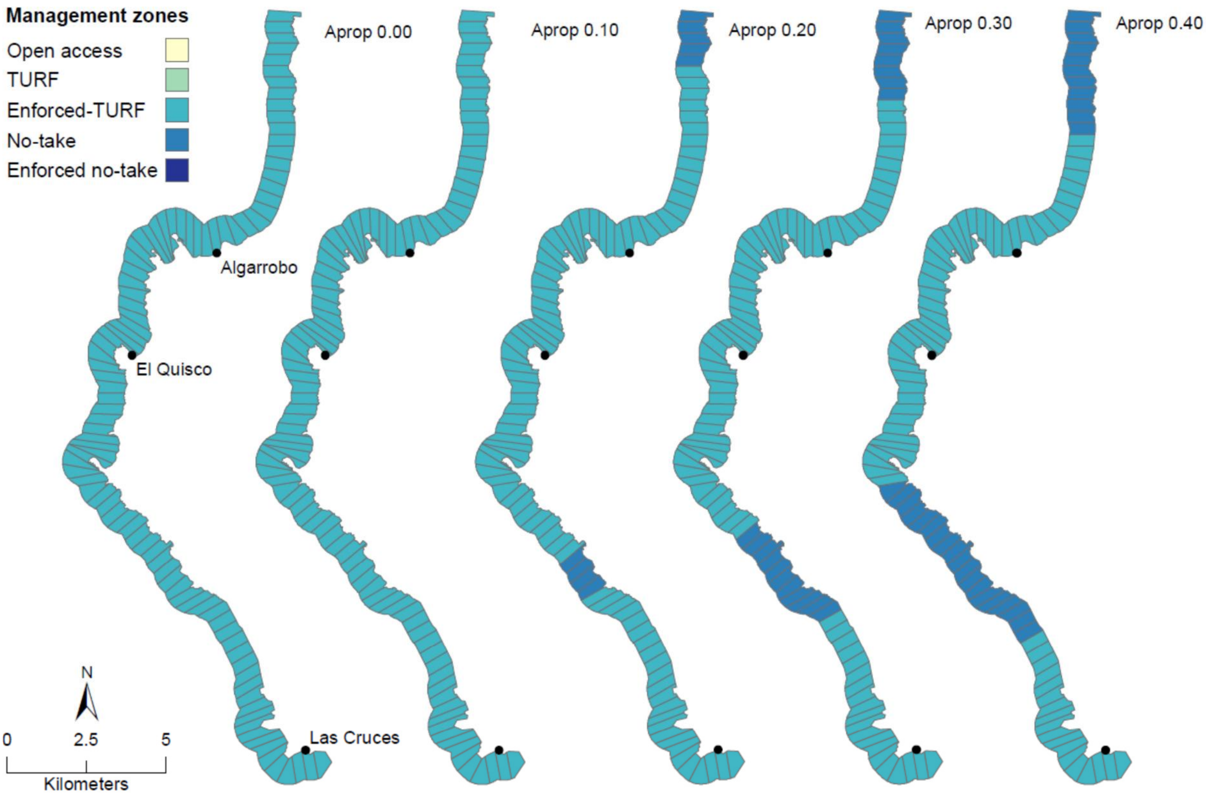

Figure 4. Zoning solutions for scenario c. Abundance targets are described top right, and are based on proportion of maximum abundance potential (different for each species). Note that for purposes of illustration, cells are colour-coded for the management zone with greater than 50% of their total area allocation.

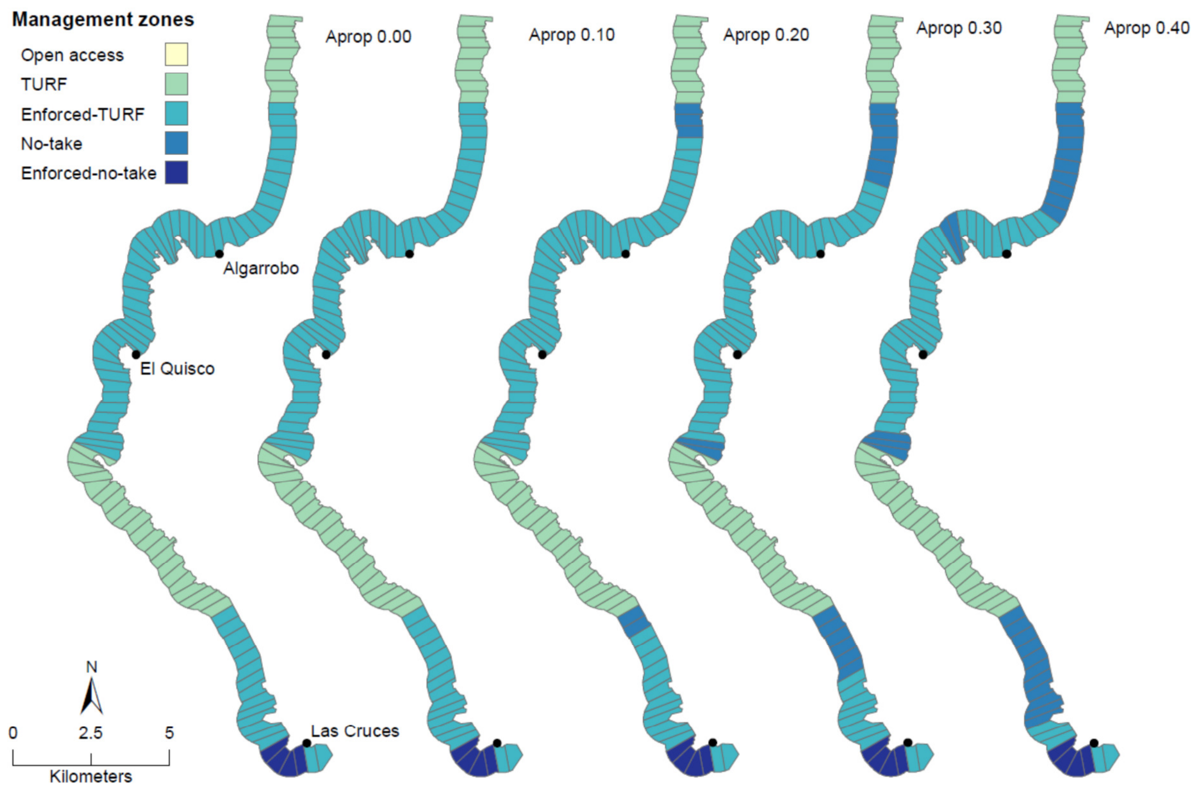

Figure 5. Solutions for scenario d. Abundance targets are described top right, and are based on proportion of maximum abundance potential (different for each species). Note that for purposes of illustration, cells are colour-coded for the management zone with greater than 50% of their total area allocation.

3.1 Sensitivity Analysis

Scenario *c* was used to test the sensitivity of the model to changes in parameter values. Three parameters were tested, market price (*mprice)*, catch level (*cprop*) and enforcement. The purpose of the sensitivity analysis is to determine the robustness of the optimal solution to changes in these key parameters, and to identify the nature of changes in response to parameter changes.

3.1.1 Market price

The relative 'value' (contribution to revenue) of each species is determined by the abundance of each species in each zone and its market price. Loco is the major commercial species for the artisanal fishing industry in Chile. Therefore Loco was chosen to test the sensitivity of the model to different market prices. The market price parameter for Loco was varied between \$0.75 and \$4.50 (US\$ per individual) while holding all other parameters constant (Table 6). Thus we are assuming that the price of Loco relative to other species could change substantially. The allocation of area to management zones was the same under market prices \$1.50 (base case), \$3.00 and \$4.50. Revenue increases as market price increases: from \$1.7-\$3.0 million (\$1.50) to \$3.3-\$5.8 million (\$4.50). When the market price of Loco was halved (\$0.75), then area allocated to the enforced-TURF zone decreased, and area allocated to the open access and no-take zones increased (Figure 6). This result suggests that when the profitability of Loco is decreased, it is no longer as economically attractive to manage the study area as enforced-TURF. This result is logical as Loco abundance is highest in the enforced-TURF zone, and no Loco is able to be extracted in the open access zone. At a low market price for Loco, the high dependence on enforced-TURF zones to meet the abundance constraint is no longer optimal; the same abundance can be achieved with greater reliance on open access and some no-take.

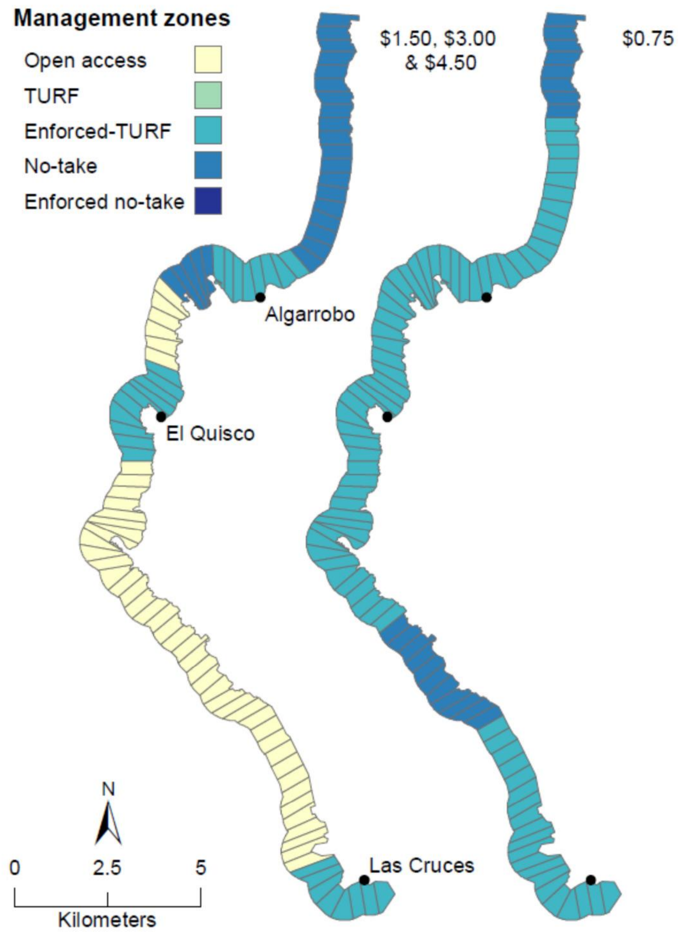

Figure 6. Sensitivity of the model to the market price of Loco, at an abundance target (Aprop) of 0.30.

Table 6. Sensitivity analysis of the market price (US$ per individual) of Loco.

Market price Loco[a] US$	Abundance target (Aprop)	Open access %	TURF %	Enforced-TURF %	No-take %	Enforced-no-take %
1.50, 3.00, & 4.50	0.00	0	0	100	0	0
	0.10	0	0	100	0	0
	0.20	0	0	91	9	0
	0.30	0	0	80	20	0
	0.40	0	0	68	32	0
	0.50	0	0	57	43	0
0.75	0.00	100	0	0	0	0
	0.10	81	0	14	5	0
	0.20	65	0	21	15	0
	0.30	48	0	27	25	0
	0.40	30	0	36	34	0
	0.50	11	0	45	44	0

		Revenue[b] US$ ('000)			
Market price (US$)	***Aprop***	**0.75**	**1.50**	**3.00**	**4.50**
	0.00	2,538	2,972	4,377	5,781
	0.10	2,389	2,972	4,377	5,781
	0.20	2,123	2,717	3,997	5,277
	0.30	1,857	2,386	3,506	4,626
	0.40	1,590	2,052	3,012	3,972
	0.50	1,321	1,716	2,516	3,315

[a]Prices of all other species were held constant throughout. [b]Revenue is equal to the product of species abundance and market price minus the costs of enforcement, where enforcement costs are both public (government) and private (fisher associations).

3.1.2 Catch limits/levels

Catch levels represent the proportion of a given species' population that can be caught. The catch level for the TURF and enforced-TURF zones was varied between 0.1 and 0.4, to determine the sensitivity of the model solution to changes in the total allowable catch for the TURF zones. The TURF zones were used for this analysis as no catch is allowed in the no-take zones, and there is no restriction on catch in the open access zone. The catch levels for the open access, no-take and enforced-no-take areas were held constant. In each case, assumed abundance levels were left unchanged. This sensitivity analysis explores the consequences of uncertainty about the level of take that would be compatible with the assumed abundance level.

The results show that increasing the catch level for the TURF zones increases the amount of area which is allocated to the enforced-TURF and no-take zones. Abundance in open access areas is lower than in both TURF zones, but there is no catch restriction. This means that when *Cprop* (T, ET) is 0.1, catch in the open access zone is higher than in the TURF or enforced-TURF zones. At this catch level, it is optimal to allocate the majority of the study area to the open access (65%) and enforced no-take zones (28%) (Figure 7). When *Cprop* (T, ET) is greater than or equal to 0.2, a greater proportion of the abundance in the enforced-TURF zone is available as catch. This changes the optimal zoning allocation to the enforced-TURF and no-take zones; area allocated to these zones increases by 73% and 20% respectively (*Aprop* 0.30). Revenue also increases as catch levels increase. Given abundance target (*Aprop*) 0.30, revenue increases from $1 728 000 *(Cprop* 0.1) to $4 851 000 at (*Cprop* 0.4).

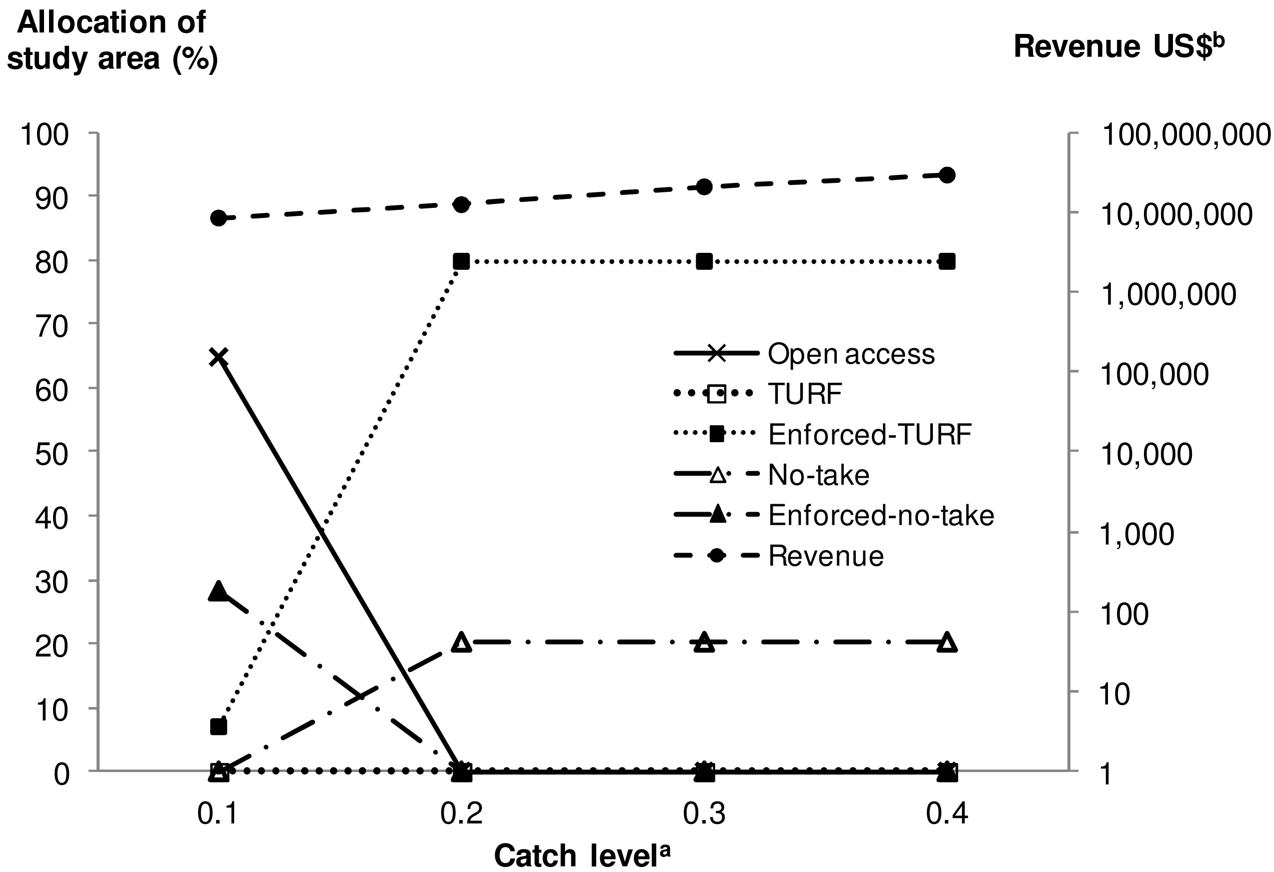

Figure 7. Area allocation and revenue for sensitivity analysis of the catch level at abundance target (*Aprop*) 0.30. [a]Catch levels for the open access, no-take, and enforced-no-take zones were held constant for all scenarios (1.0, 0.0 and 0.0 respectively). [b]Revenue is equal to the product of species abundance and market price minus the costs of enforcement, where enforcement costs are both public (government) and private (fisher associations).

3.1.3 Enforcement costs

The third sensitivity analysis examines the impact of changes to the proportion of enforcement cost incurred by the enforced-TURF and enforced-no-take zones. Enforcement costs for the enforced-TURF and enforced-no-take zones were increased by factors of 2, 3 and 11. Similar to the analysis of catch levels, in each case it is assumed that abundance levels are unchanged. This sensitivity analysis therefore reflects uncertainty about the true costs of enforcement for given equilibrium abundance levels. The BCR for enforcement (see Results) was calculated for each variation in the enforcement cost parameter. The results showed that at abundance target (*Aprop*) 0.00, the BCR will be greater than 1 when the value of the enforcement cost multiplier is less than 11 (Figure 8). At this point, all area in the study region (scenario *c*) is allocated to the open access zone. This analysis demonstrates that enforcement costs must increase substantially before enforcement is no longer beneficial.

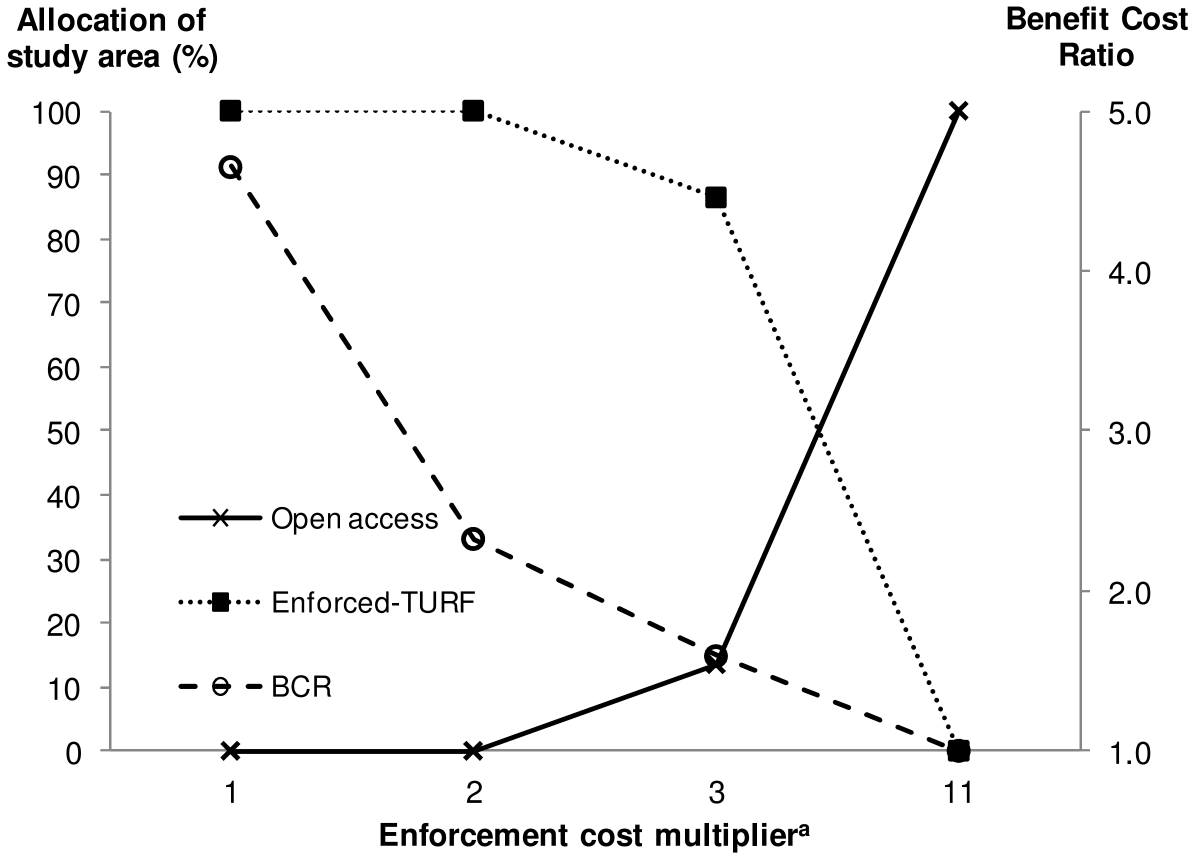

Figure 8. Area allocation and benefit:cost ratio (*Aprop* 0.00) for sensitivity analysis of enforcement cost multiplier. [a]All other variables were held constant.

4. Discussion

Our objective in this study was to determine how incorporating the costs of enforcing territorial user rights affects optimal zoning allocation. A spatial optimisation model that incorporated enforcement costs was developed for a study area of marine reserves and TURFS in Chile. Enforcement of TURF and no-take areas resulted in substantially increased revenues for fishers, at negligible cost. Revenue under scenarios where catch limits were enforced (*b* and *c*), was approximately 50% higher than under a non-enforced scenario (*a*). This increase in revenue was attributed to the increase in equilibrium abundance when poaching is prevented. The impact on fisheries from deterring poachers and ensuring fisher compliance with catch levels has previously been identified by Byers and Noonburg (2007), who found that the cost of enforcement needs to be quantitatively assessed to maximize fisheries benefits. The present analysis found that the benefits from active enforcement of a zone were greater than the costs of that enforcement (BCR > 9).

With no abundance constraint (*Aprop* 0.00), scenarios *b* and *c* allocated 100% of the study area to the enforced-TURF zone. This suggests that enforced-TURF areas are an optimal strategy for increasing fisher revenue. If an abundance target was introduced and increased (*Aprop* 0.10 to 0.40), area was also allocated to no-take zones, signalling the need for joint management and conservation networks. The results also demonstrated that enforcement expenditure can be minimized by allocating enforced zones to areas of low enforcement cost (i.e. closer to the location of *caletas*). By incorporating the impacts of distance on enforcement costs, the model minimised the costs of managed areas to marine stakeholders.

A surprising result from the sensitivity analysis was that selection of enforced-TURF areas was still an optimal strategy even if enforcement costs were increased substantially. This result is at odds with observed fisher behaviour; fishers cite costs of enforcement as one of the chief reasons for TURF operators to stop actively managing areas that are further away from a *caleta* (Gelcich et al. 2012). The vast majority of small-scale artisanal fishers (97%) perceive enforcement as a major cost involved with managing TURF areas (Gelcich et al. 2009). It may be the case that fishers do not perceive the benefits of enforcement to be as great as have been shown by this investigation. In addition, the decision not to enforce zoning restrictions may represent ecological or other economic motivations that were not included in the current analysis. For example, areas with high enforcement cost could be situated in areas with relatively lower biological productivity, or the effectiveness of enforcement might be limited due to inability to identify and sanction poachers. The opportunity cost of time spent enforcing has not been considered in this analysis and may also play a role in fishers' decision not to enforce.

The findings of this study clearly show that transaction costs and pre-existing management structures may need to be included when assessing the optimal spatial allocation of marine reserves. If community managed marine zones are envisaged (by the assignment of user rights through TURFs), it is particularly important to incorporate economic impacts on local communities into the analysis to improve model predictions of fisher behaviour. Given the knowledge that enforcement will result in significant biological benefits and associated higher revenue, that enforcement costs are spatially determined, and that these costs are considered significant by marine stakeholders; it makes sense to situate managed areas in low-cost enforcement bands.

One thing to keep in mind when considering the zoning of restricted-use areas are the geographical characteristics of reserves and their management costs. Balmford et al. (2004) found that marine protected areas cost more to run, per unit area, where they are small, close to inhabited land and where cost structures are high. However, Potts and Vincent (2008) found that management of several smaller reserves could sometimes be economically superior to a single larger one. In the present study, the impact of reserve size on management cost was not examined, however it was shown that there would be efficiencies from monitoring and enforcing management areas that are close to *caleta* locations.

In its current form, the model considers the transaction costs involved with enforcing marine protected areas. However, some transaction costs associated with the establishment or rezoning of cells were not included in the assessment. The existing management system (scenario *d*, Table 5) was shown to be less efficient than the non-constrained optimum: higher revenue can be achieved while meeting abundance targets (*Aprop* 0.10 to 0.40) when the model is not constrained to contain existing managed areas (scenario *c*). If the transaction costs involved with rezoning cells could be incorporated into the model, then it would be possible to assess whether the benefits obtained from changing the existing management outweighed the costs of doing so.

The results indicate that, at any given abundance target, less area is allocated to enforced-TURFs when species' market price falls (Table 6). As Chile is one of the top ten exporters of fish and fishery products in the world (FAO 2012), the Chilean market price for commercial species is affected by fluctuations in global as well as domestic demand. Consequently it may be important to understand how fishers behave in anticipation of such fluctuations. It is highly likely that community support for marine reserves will vary with species' market price. Indeed, if the abundance target is zero, so that the only reason for having TURFs is the benefit to fishers, the optimal area of enforced-TURFS at a low Loco price falls to zero. As general abundance becomes more of a priority, reliance on enforced-TURFs increases, but remains less than under high Loco prices. Regulators will thus need to account for the possibility of price fluctuations, and should preferably attempt to identify a spatial allocation of zones that is robust under a range of market prices.

Also of note is the impact that different catch levels have on management recommendations. If the catch level in non-enforced zones is decreased, it changes the relative allocation of area to management zones. The total allowable catch limit currently in use in the Chilean TURF system is approximately 20% of a given population's stock (Bitecma 2003). If catch limits in TURF areas decrease, then less area is optimally allocated to the enforced-TURF zone and more is allocated to the open-access and no-take zones. In open access and non-enforced-TURF zones it is likely that catch limits are exceeded through poaching by locals. In the current analysis, the possible social benefit of this poaching is not included. It is possible that while non-enforcement results in lower incomes to fishers, income gained from poaching in these areas contributes to overall social welfare. Further research into the social benefits and costs of poaching in non-enforced marine management zones is needed to investigate this possibility further.

A limitation of the model presented here is that the relationship between resource abundance and fishing effort is not considered (Arreguín-Sánchez 1996). If species' abundance goes down, it is likely that species' 'catchability' will also reduce (Gordon 1954), resulting in higher costs to fishers. Accounting for differences in fishing effort among management zones, and impacts of these fishing efforts on local species' abundance, would contribute to a more realistic analysis of the economic motivations of fishers in the study area.

5. Conclusion

The key finding of this analysis is that accounting for the benefits and costs of enforcement will enable policy makers to more efficiently manage marine areas. This analysis has shown that the highest-priority areas for enforcement are those with the lowest transaction costs of enforcement, by virtue of being close to population centres. On the other hand, even in relatively distant areas, benefits of enforcement to marine species' abundance in the Chilean central region exceed the costs. Enforced-TURF areas were shown to be the best management strategy to maximise fisher revenue. Note that this result was conditional on market price; when the market price of Loco decreases, enforced-TURF areas are less attractive. Finally, the model results show that the current spatial allocation of managed areas in the region is inefficient. Fisher revenue can be increased by modifying this spatial allocation.

Our results diverge from the observed behaviour of fisher associations, who have chosen not to enforce limitations on take in parts of the TURF system that are more costly to monitor. Further work to understand this divergence would be beneficial. It may be that fishers under-estimate the benefits of enforcement, suggesting that a training program, or perhaps temporary subsidies for enforcement, may be beneficial. Alternatively, it may be that existing fisher organisations lack the capacity, the authority or the structures needed to enforce all TURF areas, suggesting the need for efforts to enhance the organisations. Finally, it may be that we have under-estimated the transaction costs of enforcement in our model, and so over-estimated the net benefits of enforcement.

Acknowledgements

This research was conducted with the support of funding from the Australian Research Council Centre of Excellence for Environmental Decisions. SG acknowledges the financial support of Fondecyt grant #1120103 and Centro de Conservación Marina Iniciativa Cientifica Milenio of the Ministerio de Economia, Fomento y Turismo P10-033F.

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