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Drivers and trends in catch of benthic resources in Chilean TURFs and surrounding open access areas

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ABSTRACT

Beginning in the 1990's, Chile implemented an extensive Territorial User Rights for Fisheries (TURFs) network that now comprises nearly 1,000 TURFs. This network provides a rare opportunity to examine spatial and temporal trends in TURF use and impacts on surrounding open access areas (OAAs). In this analysis, landings of keyhole limpet (Fissurella spp.), kelp (Lessonia spp.) and red sea urchin (Loxechinus albus) were used to estimate catch-per-unit effort (CPUEs) and catch-per-unit area (CPUAs) indices inside and outside TURFs by fishing cove. For these species, CPUEs and CPUAs in 2015 were significantly higher inside TURFs. However, temporal trends analyzed with a linear mixed effects model indicate that CPUAs inside TURFs have been significantly decreasing since 2000 for keyhole limpet, red sea urchin and for loco (Concholepas concholepas), while in OAAs this measure only decreased for limpet. An elastic net regression was used to better explain catches in OAAs during 2015, including a variety of variables related to the characteristics and activity of proximal TURFs. Results indicate that exogenous factors unrelated to TURF management were the primary drivers of catches in OAAs during 2015 but that factors related to proximal TURFs appear to have a slight negative impact that grows over time. Collectively, these results indicate that while TURFs are associated with higher catch rates than surrounding OAAs, catch rates appear to be decreasing over time and, though limited, the impact of TURFs on surrounding OAAs may be negative. These findings suggest a need for a more nuanced and dynamic approach to spatial management on benthic resources in Chile.

1. Introduction

Spatial property rights can eliminate many common pool externalities that plague fisheries, thereby better incentivizing sustainable and profitable resource use (Beddington et al., 2007; Cancino, 2007; Costello et al., 2008). Specifically, Territorial User Rights for Fisheries (TURFs) is a management tool that grants individuals or groups exclusive access to harvest resources within an area (Christy, 1982). TURFs have been associated with biological, ecological and economic benefits in several small-scale fisheries (Castilla and Fernández, 1998; Gelcich et al., 2008a, 2012; Defeo et al., 2016). During the last decade, TURFs have been promoted as a general approach to tackling the negative impacts of open access fishing (Wilen et al., 2012; Kratz and Block, 2013; FAO, 2014, Nguyen Thi Quinh et al., 2017), particularly for unassessed fisheries in developing countries that often suffer from overexploitation (Costello et al., 2012). However, the full impacts of TURFs on fisheries sustainability, including long-term trends in catch rates and impacts beyond TURFs boundaries, are not yet fully understood (Orensanz et al., 2005; Aburto and Stotz, 2013; Aburto et al., 2014; Gelcich et al., 2019). As the implementation of individual quotas and marine protected areas has been found to have unintended impacts on unregulated subpopulations and habitats (referred to here as "management spillover"; Hilborn et al., 2004; Murawski et al., 2005; Asche et al., 2007; Branch, 2009; Abbott and Haynie, 2012), similar effects might be expected from other area- or rights-based management and conservation instruments, including TURFs. To our knowledge, the influence of the implementation of TURFs on surrounding areas has not yet been assessed (Nguyen Thi Quinh et al., 2017) despite the fact that the spatial dynamics of most fisheries exceed the scale of an individual TURF. This study looked at the long-term changes in catch and catch rates (i.e., catch per unit effort, CPUEs, and catch per unit area, CPUAs) inside and outside TURF managed areas and also evaluated the possibility of management spillover.

In Chile, the implementation of TURFs was a reaction to the collapse of the economically important artisanal fishery for the muricid snail *Concholepas concholepas* in the 1980s (known in Chile as *loco,* elsewhere as the false abalone) (Bernal et al., 1999). The fast recovery of the high valued *loco* stocks in initial TURFs increased demand for further TURF development along the entire Chilean coast throughout the 2000s. In 2017, there were 957 officially designated Chilean TURFs implemented as part of a national TURF policy (Fishery and Aquaculture Law n° 18, 1991). According to the Chilean Fisheries Authorities, the primary

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Abbreviations

CPUA	Catch per unit of area
CPUE	Catch per unit of effort
OAA	Open access area
TURF	Territorial user right for fisheries

objectives of Chilean TURFs are to "ensure the sustainability of artisanal fishing through the assignment of natural banks", and to "maintain and increase the biological productivity of benthic resources" (SUBPESCA, 2003). This TURF network constitutes the dominant form of spatial management of benthic resources in Chile and is the largest worldwide, covering about 1,500 km^2 (though only about half of these 957 TURFs are currently operative). Known in Chile as "Área de Manejo y Explotaciones de Recursos Bentónicos" (Management Areas for the Exploitation of Benthic Resources; AMERB), this system grants exclusive fishing rights to legally constituted fishing organizations for the exploitation of benthic resources in defined portions of the seabed – usually adjacent to a *caleta* or artisanal fishing cove (Aburto et al., 2013). Each TURF has species-specific quotas proposed by the fishing organization and approved by the Undersecretary of Fisheries. Artisanal fisher organizations have to comply with a series of regulations, such as establishing a baseline study, management plan, and regular stock assessments, for which they have to contract technical assistance from specialized environmental and/or fisheries consultants (Gelcich et al., 2008b). TURFs are interspaced with open access areas (OAAs) where seasonal closures and limits on catch size are used, but entry, within-season effort, and total catch are not restricted. The Chilean TURFs system was initially (i.e., from the 1990s to the 2000s) successful and associated with positive ecological and economic benefits, such as the recovery of *loco* stocks, increased species richness inside TURFs, and increased welfare and economic revenues (Castilla and Fernández, 1998, Defeo and Castilla, 2005; Gelcich et al., 2008a, 2012). OAAs produced the majority of catch and fishing revenues however. While income from TURFs was largely supplemental, believed to represent 7%–41% of total incomes (Romero et al., 2016), it was thought to play an essential role in securing fishers' livelihoods (Aburto et al., 2013; Van Holt, 2012;).

Though ecological conditions appear to have improved within TURFs (Castilla and Fernández, 1998; Gelcich et al., 2012), TURF profitability is thought to have declined over the last decade (Gelcich et al., 2017). The development of abalone aquaculture in Asia has negatively influenced international demand for *loco*, leading to a reduction in exports from Chile to Asia (from 2,400 mt in 1993 to less than 1,000 mt in 2013), and a drop in the price of *loco* (Chávez et al., 2010; Castilla et al., 2016). Furthermore, the cost of TURF maintenance, which includes assessment, enforcement, and surveillance, is thought to have increased (based on perception surveys; Gelcich et al., 2009, Gelcich et al., 2017). Assessments are typically conducted by private environmental consultants, whose fees have increased in part because of the relatively small number of such companies available in Chile (Gelcich et al., 2009; Davis et al., 2015). Additionally, extensive illegal fishing (González et al., 2006; Andreu-Cazenave et al., 2017; Oyanedel et al., 2017) suggests that local fishing organizations must dedicate significant time and resources to enforcement in TURFs. Though the Chilean government recognizes that there is poaching activity and, in theory, is responsible for apprehending and penalizing poachers, in practice the responsibility of detecting poaching in TURFs often falls on fishing organizations. Many fishers now indicate they do not have enough capacity (i.e. resources and time) for surveillance of their TURFs and consider "government punishment of poachers to be ineffective" (Moreno and Revenga, 2014; Davis et al., 2015; Biggs et al., 2016). Thus, the combined influence of a lower price for *loco* and presumed increased maintenance costs, with a reduced enforcement capacity, have likely increased variability in financial returns and decreased the profitability of TURFs (Chávez et al., 2010; Gelcich et al., 2010, Gelcich et al., 2017). In fact, in recent years (roughly 2010–2017), fishers appear to be relying on TURFs less than initially (i.e., 1990s–2000s) and TURF exploitation now represents a smaller fraction of fishers' overall incomes (Gelcich et al., 2017). This has coincided with an observed increase in exploitation of OAAs (de Juan et al., 2017) and substantial illegal fishing of *locos* (Andreu-Cazenave et al., 2017). Reduced incentives for the exploitation of a TURF could either result in its abandonment (San Martín et al., 2010, Gelcich et al., 2017), its maintenance for purposes other than fishing such as market access or social empowerment (Cancino et al., 2007; Zúñiga et al., 2008; Aburto et al., 2013; Rosas et al., 2014, Gelcich et al., 2017), or its maintenance at a lower but still positive level of profitability.

Potential positive or negative interactions between maintained TURFs and surrounding OAAs are unknown. The large TURF system of Chile offers opportunities to explore the consequences of spatial management on fisheries in surrounding areas. TURFs are expected to secure fisheries harvests within their boundaries and provide incentives for sustainable use of surrounding fishing grounds (Christy, 1982). Recent studies in the Chilean system of TURFs have shown higher potential egg production of two benthic species (the limpet *Fissurella latimarginata* and the red sea urchin *Loxechinus albus*) within TURFs than under an open access scenario (67% and 52% higher, respectively) (Blanco et al., 2017; Fernández et al., 2017), suggesting the potential to enhance fishing opportunities both inside and outside TURFs. Negative impacts of TURFs and other entry-restriction management and conservation tools beyond their limits are less well known. Management spillover consisting of effort displacement from high-regulation TURFs to lower-regulation areas outside TURFs (analogous to the "fisheries squeeze effect" in the context of marine protected areas; Attwood and Bennett, 1995; Bohnsack, 2000; Halpern et al., 2004) could be expected to occur, potentially deteriorating opportunities in surrounding fishing grounds. Recent reductions in TURF profitability may provide increased incentives for TURF users to increase fishing effort in OAAs, possibly further eroding the sustainability and profitability of Chilean coastal fisheries in these areas.

The primary goals of this study were to analyze catch and catch rates within and outside of TURFs to document any trends and interactions that might impact the ability of the TURF system to meet the objectives of ensuring sustainability and increasing biological productivity of benthic fishery resources. Specifically, we first examined and compared CPUE and CPUA indices (catch rates) between TURFs and adjacent OAAs by fishing cove in 2015 for three important target species (keyhole limpet (*Fissurella spp.*), kelp (*Lessonia spp.*) and red sea urchin (*Loxechinus albus*)). Second, temporal dynamics in TURF and OAA catch rates were investigated by looking at time series of CPUAs calculated for each management area by fishing cove and year. Finally, to assess if catch rate differences between TURFs and adjacent OAAs observed in 2015 were related to TURF implementation, a penalized regression model was developed to explain catch in OAAs. The explanatory variables examined in the model were either related to proximal TURFs' characteristics and activity (e.g., TURF age, TURF area fraction, TURF fishing effort), or additional geospatial variables related to the spatial extent and context of OAAs (e.g., coastline length, local productivity, proximity to urban areas).

2. Methods

2.1. Data

National data on catch and effort by fishing cove were obtained from the governmental agency SERNAPESCA (National Fisheries Service). Artisanal fishers are required to report landings by species, weight and origin (i.e., TURF or OAA; Moreno and Revenga, 2014). TURF geographical layers were obtained from the governmental agency

SUBPESCA (Undersecretary of fisheries). Fishing coves considered for the study (Fig. 1) had at least one designated TURF assigned to a fishers' organization (referred to here as a functioning TURF; i.e., an operative TURF with a use agreement and quota in place or a stand-by TURF for which a quota has been assigned in the last 4 years, but monitoring has not been conducted by the due date, Appendix 1).

The artisanal benthic fisheries of Chile target a variety of species, including crustaceans, mollusks, sea urchins, tunicates and several species of seaweed (Gelcich et al., 2010). Catch data were obtained from landings reports, focusing on the most important benthic resources targeted in TURFs. The primary target resource inside TURFs is the *loco*, which has the highest commercial value (beach sale value: 11,647 US$/mt; landings: 2,255 mt in 2011) (Moreno and Revenga, 2014). *Loco* extraction is banned in OAAs, and, therefore, only catches from inside TURFs were analyzed for this species. Kelps (comprising the *Lessonia nigrescens* species complex, *Lessonia trabeculata*, *Macrocystis pyrifera* and *Macrocystis integrifolia*) and the red sea urchin (*Loxechinus albus*) are the largest landed benthic resources ranked by weight (landings: ~300,000 mt and 31,901 mt for kelp and sea urchin, respectively, in 2011). We also considered catches of keyhole limpets (comprising *Fissurella spp.*, *Fissurella costata*, *Fissurella cumingi*, *Fissurella latimarginata*, *Fissurella picta*, and *Fissurella maxima*), another economically important benthic resource (beach sale value: 2,354 US$/mt; landings: 1,785 mt in 2011). Individual catch reports from 2000 through 2015 for these four main exploited benthic resources were aggregated by fishing cove and month (an individual harvester could report catch several times in a month), and distinguished by their origin (i.e., inside or outside TURFs). Catches in OAAs (i.e., outside TURFs) included catches gathered from artisanal boats or from the shore.

The number of active harvesters in 2016 per fishing cove was also obtained from SERNAPESCA (most recent estimation, note that the number of fishers for 2015 was not available). Individuals who have not operated for the last three successive years were removed from the national registry. Chilean law distinguishes four categories of artisanal harvesters: 1) Divers, who manually extract mollusks, crustaceans or echinoderms, or spearfish for reef fish, usually operating from a boat; 2) Collectors, who harvest or collect seaweeds from the shore; 3) Fishers, who are captains or crew members of an artisanal boat, from which they operate with nets, including trammel nets, long lines, and hand lines; and 4) Ship owners, who are limited to one or two artisanal boats, defined as 18 m or less in length, and 50 tons or less. The different categories are not mutually exclusive. Effort was estimated in terms of the number of divers (for *loco*, limpet and sea urchin exploitation) or number of collectors (for kelp exploitation) registered in a fishing cove and able to exploit the resource. Fishers' organizations that are granted a TURF can only be comprised of licensed artisanal harvesters. However, not all licensed artisanal harvesters are part of a fishers' organization. Therefore, effort "inside" TURFs only considered licensed harvesters who were also registered in the corresponding fishers' organization, while effort "outside" TURFs considered all licensed harvesters registered in a particular fishing cove. A small number of harvesters (about 10%) were licensed in one fishing cove but associated with fishing organizations in different fishing coves. To avoid overestimating effort per fishing cove, the contribution of an individual harvester to effort in a cove was calculated by equally dividing one unit of effort (i.e., one harvester) among the different fishing coves with which the harvester was associated.

Fishing area estimates, for both TURFs and OAAs in each cove, were calculated using different data and proxies. TURF areas were obtained through a Google Earth layer publicly available on the SUBPESCA website for 2016. Total fishing ground polygons (comprising TURFs and OAAs) were created per fishing cove based on sailing time and bathymetry (Appendix 2). Buffer zones of 17 km (alongshore cutoff) around fishing coves were produced in ArcGIS to represent total accessible fishing grounds for each cove. The 17-km cutoff was based on the average distance from the fishing cove center to fishing grounds potentially visited as determined by artisanal fisher survey results (Ruano-Chamorro et al., 2017). These 17-km buffers were then intersected with a bathymetric polygon consisting of the area between 0 and 20 m depth. The offshore width of these polygons was based on a typical maximum harvest depth of 20 m (González et al., 2006). These alongshore and offshore cutoffs are similar to those used by Castilla (1994) and Aburto et al. (2009) which applied an offshore limit of 30 m

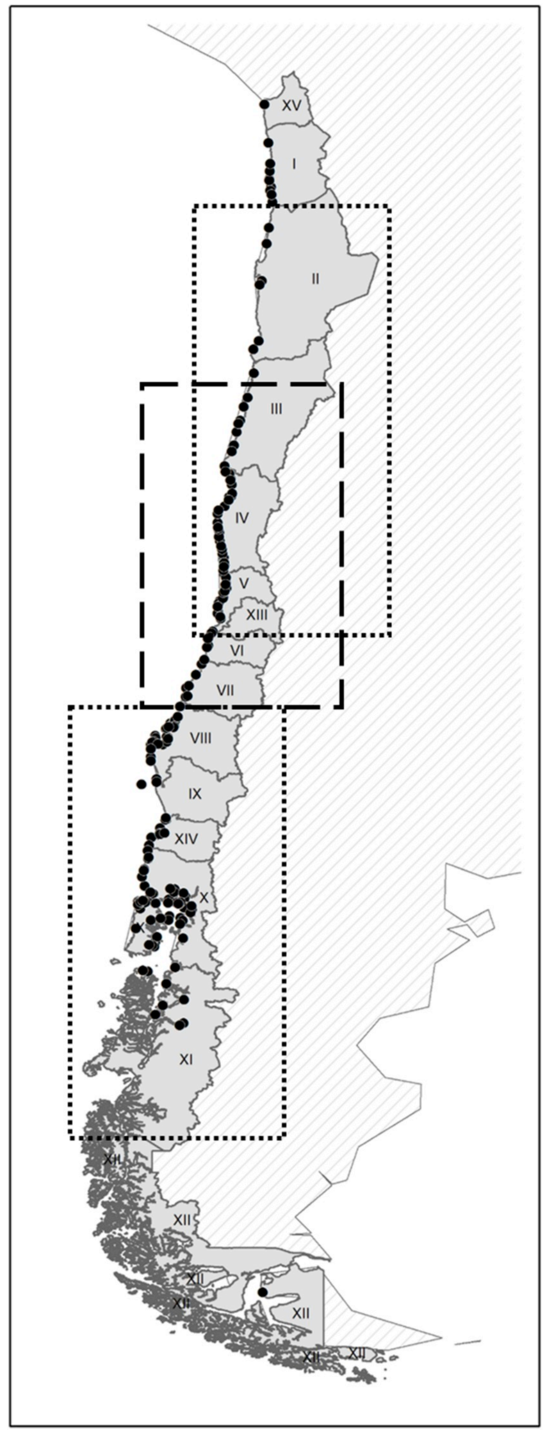

Fig. 1. Administrative regions of Chile. Fishing coves included in the comparison of 2015 catch rates and mixed effect models are represented with the black dots. The elastic net regressions only consider fishing coves within the northern regions II, III, IV, V and within the southern regions VIII, IX, XIV, X and XI (dotted rectangles). The 20 m isobaths layer was only available from central Chile (dashed rectangle from 27° to 36°).

and an alongshore cutoff of 15 km based on travel distance with one full tank of gas. The 20-m isopleth was only available for central Chile (from 27° to 36°, Fig. 1) whereas a 100-m isopleth was available for the whole Chilean coast (source GEBCO). Estimates for the areas of the 0–20 m fishing ground depth range were derived from the areas of 0–100 m depth range using multiple linear regression (see Appendix 2 for details). Finally, estimates of OAA areas were calculated as total area of fishing grounds minus assigned TURFs areas.

2.2. Catch rate comparisons between TURFs and OAAs in 2015

Annual catches divided by the number of active months (several species are only landed during part of the year) of keyhole limpet, kelp, and red sea urchin were used to estimate CPUEs and CPUAs per fishing cove for 2015 (most recent complete year for catch data at the time of the study, SERNAPESCA). *Loco*'s estimates were not compared since its extraction is banned in OAAs, and, therefore, only catches from inside TURFs were available. CPUEs and CPUAs were differentiated by their origin, i.e., catches inside TURFs or in OAAs, and then compared to one another to determine differences in fisheries productivity. CPUEs for each fishing cove were calculated as the catches inside or outside TURFs divided by the adjusted number of divers (or collectors) (i.e., after having adjusted this number to account for harvesters associated with multiple fishing coves) inside or outside TURFs, respectively. The number of licensed harvesters in 2016 was the best available effort proxy for estimating CPUEs in 2015 even though this is a crude estimate as it is unknown how many trips each individual took. CPUAs for each fishing cove were calculated as the catches inside or outside TURFs divided by the total assigned TURF area (inside) or the estimated OAA area (outside). For each group of species, differences between CPUEs and CPUAs inside and outside TURFs were tested for statistical significance using a nonparametric Wilcoxon signed-rank test. Reporting rates from TURFs and OAAs could differ given higher enforcement capacity within TURFs (Ruano-Chamorro et al., 2017). We therefore calculated what catch in OAAs would have to be for catch rates in OAAs to equal those in TURFs (assuming full reporting in TURFs), and then deduced the misreporting rate in OAAs it would imply for each species and catch rate metric.

2.3. Temporal analyses of CPUAs inside and outside TURFs

CPUAs of *loco*, keyhole limpet, kelp, and red sea urchin were analyzed over time to investigate temporal performance of TURFs and OAAs over the last two decades. Fisheries data was only available at the scale of an entire fishing cove, prohibiting differentiation between multiple TURFs associated with a single fishing cove. Estimated OAA areas from 2016 were adjusted over years according to implemented TURFs' area for that year and fishing cove (implementation year of TURFs were available from the SUBPESCA data). Complementary temporal analysis of CPUE trends was not feasible as the annual number of fishers was not available at the fishing cove scale. Changes in CPUA over time may reflect changes in biomass, changes in fishing effort, or changes in spatial management. If biomass were improving inside TURFs, CPUAs in these areas might be expected to increase over time. Conversely, if TURFs displaced fishing effort into OAAs, CPUAs in OAAs might be expected to decrease due to overfishing (but may increase initially as increased effort fishes down stocks). Additionally, a fishing cove can have several TURFs (up to 15 managed areas, but on average three). If the initial TURF implemented in a given fishing cove was located in the best habitat (Wilen et al., 2012), then fishing coves with multiple TURFs might experience sequential reductions in CPUAs. Finally, as catch depends on effort, it is also possible that changes in CPUA reflect changes in fishing effort over time (e.g., CPUA reductions arising due to reduced fishing effort independent of any changes in fish stocks).

A linear mixed effects model (i.e., model 1) was used to estimate the temporal trend and the effect of the number of TURFs per fishing cove on CPUAs inside and outside TURFs:

$$\ln(CPUA_{s,i,t,a}) = \beta_1\, year_t + \beta_2\, N_{TURFi,t} + \delta_i + \varepsilon_{s,i,t,a} \quad (1)$$

In (1), the dependent variable is the log-transformed CPUA for species s, observed in the fishing cove i, for year t, in area a (inside or outside TURFs). β_1, β_2 are the unknown coefficients of the fixed effects variables year (from 2000 to 2015) and N_{TURF}, the number of functioning (i.e., operative or stand by) TURFs per fishing cove for each year, respectively. δ_i is a random effect for fishing cove i, to control for heterogeneity across fishing coves and $\varepsilon_{s,i,t,a}$ is the error term.

To further disentangle the effects of time and number of TURFs per fishing cove on CPUAs, an additional linear mixed effects model (i.e., model 2) was developed without the variable N_{TURF}. Model 2 included a subsample of 57 fishing coves (29% of the 196 coves considered in this study) that have had a constant number of TURF(s) for at least 10 years.

Statistical estimation of coefficients was performed in R (R Core Team, 2018) with the lme4 package (Bates et al., 2015). Marginal and conditional coefficients of determination, r_m^2 and r_c^2, respectively, were estimated with the MuMIn package (Nakagawa and Schielzeth, 2013).

2.4. Elastic net regression model

The catches of keyhole limpet, kelp and red sea urchin from OAAs in 2015 were examined to assess the impact of adjacent TURF characteristics and activity. Chile is divided into 15 administrative regions; fisheries for each of the species groups considered in this analysis generally occur in only a subset of these regions (Appendix 3). As the great majority of limpet and kelp catch occurred in the northern regions of Chile (specifically regions II, III, IV and V) and the great majority of sea urchin catch occurred in the southern region (specifically, regions VIII, IX, XIV, X and XI), data for species-specific analyses were limited to these northern and southern zones (see Section 3.1 for details regarding the basis for selecting these zones).

A regularized linear regression model, the elastic net regression (Zou and Hastie, 2005, see Appendix 4 for model development), was developed to explain catch per cove in OAAs, including explanatory variables either related to proximal TURFs' characteristics and activity, or related to geospatial context (e.g., area and coastline length) and number of fishers targeting a given species. This model uses a penalized maximum likelihood method that allows a large number of variables to be included with relatively few observations and prevents over-fitting issues prevalent in more common Ordinary Least Square (OLS) or stepwise regression methods (Friedman et al., 2010; Morozova et al., 2015). The algorithm accomplishes variable selection by constraining the sum of the magnitudes of normalized coefficients. A shrinkage penalty is included in the objective function; it "shrinks" the effect of unimportant variables to select the simplest and most accurate model. Two different values of the regularization parameter controlling the strength of the shrinkage were considered; only results from the less restrictive regularization are shown here (see Appendix 5 for results with the more restrictive regularization, i.e., a larger penalty that leads to models with a smaller number of predictors with non-zero coefficients).

The response variables, i.e., catches in OAAs for limpet, kelp, and sea urchin, were log-transformed before centering. We considered catch as the dependent variable instead of CPUEs and CPUAs because we preferred a model including both effort and area as explanatory variables simultaneously.

Given that effort displacement and any resulting ecological and social impacts are dynamic processes, TURFs established for longer periods might be expected to have more significant effects on catches outside of TURFs. In order to assess these temporal effects, elastic net models included the variables number of years since the implementation of a TURF (*Age_TURF*) and number of years since the establishment of the associated fishers' organization (*Age_Organization*) (source

SUBPESCA). In theory, a fisher's organization is established before a TURF is implemented, but in some instances (~30% of our fishing coves), the organization had changed over time or several TURFs had merged or been split leading to the TURF being implemented before the associated fishers' organization. Since several fishers' organizations can operate in each cove and a fishing cove can have several TURFs, each associated with one fisher's organization, the average and maximum values were calculated for both *Age_TURF* and *Age_Organization*. Spatial aspects of TURF use were captured by the variables *N_TURF, Area_Fraction* and *Area_OA* which measured the number of functioning TURFs per fishing cove, the fraction of the total estimated fishing ground managed as TURFs and the total area of open access grounds, respectively. The potential effects of fishing effort displacement should be greater in fishing grounds with more TURFs and/or proportionately larger TURFs or smaller OAAs. Fishing effort was included through the variables *Harvesters_All*, *Harvesters_per_OAA* and *Harvesters_per_TURFs,* respectively, the total number of divers (or collectors) in OAAs, the number of divers (or collectors) in OAAs divided by the OAAs area, and the number of divers (or collectors) inside TURFs divided by the TURFs area. The predictions are that catch in OAAs should increase with the total number of divers (or collectors) and decrease with the number of divers (or collectors) per unit of area. Finally, the number of fisher's organizations per fishing cove, *N_ORG*, was used as another proxy for local effort levels and fisheries involvement.

Data on additional geospatial variables related to the spatial extent and context of OAAs were also obtained to include in analyses of catch for each species. Coastline length was calculated for fishing grounds adjacent to a fishing cove to capture differences in coastal habitats (e.g., straight along beach and sinuous along cove leading to short and long coastline lengths, respectively). Fractured coastlines with many small inlets are expected to be more favorable for sea urchin productivity (Lawrence, 2006) whereas linear beaches may represent regions of wide continental shelf where unproductive sandy habitat is more common. As proximity to urban areas might impact exploitation rates and other human pressures on benthic resources, a binary variable was included to indicate if a fishing cove was within 50 km of one of the ten biggest cities of Chile (source Instituto Nacional de Estadísticas). We also identified fishing coves close to fishing ports, since increased market access could trigger higher effort and catches. Thus, if a fishing cove was within 50 km of one of the forty major fishing ports of Chile, total landings by weight (comprising algae, fish, mollusk, crustacean, other) from these proximal fishing ports were summed together and associated with this cove; if no fishing ports were within 50 km, this variable was set to zero. Finally, CPUEs for each species group (i.e., *loco*, keyhole limpet, kelp and sea urchin) within TURFs were included as proxies for local abundance conditions. Abbreviations, definitions and units for all variables included in the elastic net regression are given in Table 1.

Model parameters were estimated with the glmnet algorithm in R (Friedman et al., 2010; R Development Core Team, 2018). A bootstrapping process, randomly sampling the data with replacement, was used to re-estimate the model 10,000 times. Coefficient means ($\overline{\beta}$), standard errors (σ_β) and probabilities of inclusion for each regression coefficient were calculated following bootstrap iterations. We considered "highly important" predictors to be those with coefficients retained in at least 80% of the bootstrap iterations; "important" predictors to be coefficients retained in 60–80% of the iterations; and "moderately important" to be coefficients retained in 40–60% of the iterations. Elastic net log-linear regression coefficients were transformed into percent changes in catch for a given change in the predictor variable using the following formula: $\%\Delta y = 100\cdot(e^{\beta\cdot\Delta x} - 1)$.

OLS models using either the full set of independent variables (OLS_all), using only TURF related variables (OLS_TURF), using only geospatial context variables (OLS_Geo) or using only variables selected by the elastic net model (OLS_elastic) were also run for comparison with the elastic net outputs. P-values for the coefficients of each explanatory factor in the OLS models were adjusted utilizing the Dunn-Šidák correction method for multiple statistical tests (Šidák, 1967; Ury, 1976). We considered the possibility of spatial heterogeneity in catch reporting by examining OLS model residuals using Studentized Breusch-Pagan tests.

3. Results

3.1. Regional description of the system

We analyzed 196 fishing coves with a total of 478 functioning TURFs in this study. Average TURF size was 1.5 km^2 (ranging from 0.01 km^2 to 39 km^2). Average total TURF area per fishing cove was 4 (± 7.3) km^2 while average OAA area per fishing cove was 82 (± 29) km^2. Limpet and kelp catch in the northern regions (i.e., regions II, III, IV, V) accounted for 81.3% and 84.8% of total national catch of each species group, respectively. Contrarily, 95.1% of sea urchin catch and 77.6% of *loco* catch were landed in the southern regions (i.e., regions

Table 1
Response variable and predictor abbreviations and definitions for the elastic net model.

Variable	Definition
Y	Log-transformed, centered catches for species s in OAA areas per fishing cove (mt)
Age_TURF_mean	Average time since the different TURFs implementation per fishing cove (yr)
Age_TURF_max	Maximum time since the oldest TURF implemented per fishing cove (yr)
Age_Organization_mean	Average time since the different fishers' organizations implementation per fishing cove (yr)
Age_Organization_max	Maximum time since the oldest fishers' organization implemented per fishing cove (yr)
N_TURF	Number of TURFs per fishing cove
N_ORG	Number of fishers' organizations per fishing cove
Area_OAA	Open access areas per fishing cove (km^2)
Area_OAA^2	Open access areas per fishing cove (km^4)
Area_Fraction	TURF areas divided by total fishing ground (TURF areas + OAA areas) (%)
Harvesters_All	Outside effort, or all licensed divers (or collectors) per fishing cove (divers or collectors)
Harvesters_per_OAA	Outside effort divided by the OAA areas per fishing cove (km^{-2})
Harvesters_per_TURF	Inside effort divided by the TURF areas per fishing cove (km^{-2})
Limpet_per_diver	Catches of limpet inside TURF divided by inside effort (mt/diver)
Kelp_per_collector	Catches of kelp inside TURF divided by inside effort (mt/diver)
Urchin_per_diver	Catches of sea urchin inside TURF divided by inside effort (mt/diver)
Loco_per_diver	Catches of *loco* inside TURF divided by inside effort (mt/collector)
Coastline_length	Length of coast adjacent to the fishing cove (km)
Landings_port	Total landings (algae, fish, mollusk, crustacean, other) of fishing port(s) within 50 km, if any (mt)
Urban_area	Fishing cove is within 50 km to one of the ten biggest cities [a] (1|0)

[a] Antofogasta, Arica, Conception, Iquique, Puerto Montt, Punta Arenas, San Antonio, Serena, Valdivia, Valparaiso.

VIII, IX, XIV, X and XI,Fig. 1, Appendix 3). The contrasting landing patterns were accompanied by differences in TURFs' size. TURF average area per fishing cove was higher and more variable in southern regions ($4.9 \pm 9.3\,km^2$) than in the northern region ($3.2 \pm 3.1\,km^2$), and OAA average sizes associated with each fishing cove were larger in southern regions ($90.5 \pm 29.2\,km^2$) than in northern regions ($58.0 \pm 103.0\,km^2$) (Table 2). The sizes of OAAs were consistently larger than those of TURFs, however the ratio between OAA and TURF size was similar between the north and the south. In terms of effort, the number of divers (or collectors) that could fish in OAAs was higher than the number that could fish in TURFs, with this difference being larger for fishing coves in the south (Table 2).

3.2. Catch rate comparisons between TURFs and OAAs

CPUE and CPUA values for 2015 for each fishing cove were compared by their origin, i.e. inside or outside TURFs (Fig. 2). CPUEs for limpet were observed to be higher inside TURFs ($p = 0.01$). However, CPUEs were not significantly different between the two origins for kelp ($p = 0.36$) and sea urchin extraction ($p = 0.34$), though their corresponding medians were higher inside TURFs. For each of the three groups of species, CPUAs were significantly higher inside TURFs ($p = 2.1 \times 10^{-5}$ for limpet, $p = 5.4 \times 10^{-6}$ for kelp, and $p = 1 \times 10^{-3}$ for sea urchin). Overall, median catch rates were at least 75% higher inside TURFs (Table 3). With regard to catch rate values across species, limpet and red sea urchin were caught at similar rates in terms of metric tonnes per month per unit effort/area, whereas kelp was caught at a much higher rate, and *loco* was caught at an intermediate rate. Assuming perfect reporting within TURFs, equal catch rates between TURFs and OAAs imply 70%–99% of catch from OAAs would be unreported. Higher catch rates observed in TURFs therefore appear to be robust to catch misreporting.

3.3. Temporal mixed effects analysis of area catch rates

Linear mixed effects models revealed that CPUAs had decreased significantly over time inside TURFs, with rates of decrease of 7.8%, 4%, and 4.8% per year for *loco*, limpet, and sea urchin, respectively ($p < 0.05$, Table 4). For all species groups, CPUAs also significantly decreased inside TURFs as the number of TURFs implemented in a fishing cove increased (between 10 and 29% decrease in CPUA per additional TURF implemented, $p < 0.05$, Table 4). Effects of the temporal driver *Year* were weaker in OAAs (Table 5). Only CPUAs for limpet significantly decreased in OAAs over years (4.7% decrease in CPUA/year, $p < 0.05$, Table 5). Interestingly, CPUAs for kelp increased significantly outside of TURFs over time (3% increase per year, $p = 0.02$, Table 5) whereas there was no temporal trend inside TURFs. The number of TURFs did not have any effect on CPUAs in OAAs for any of the species groups considered. Predicted values of CPUAs inside TURFs from 2000 to 2015 were consistently higher than predicted values of CPUAs within OAAs (Fig. 3). When the models were restricted to just the subset of fishing coves having a constant number of TURFs (i.e., model 2), CPUAs were found to decrease significantly over time inside TURFs for *loco* (8.5% decrease per year), and outside TURFs for limpet (2.5% decrease per year, $p < 0.05$, Tables 4 and 5). Differences between conditional r_c^2 and marginal r_m^2 show that 40%–80% of variability is due to spatial heterogeneity across fishing coves (Tables 4 and 5).

3.4. Elastic net regression of OAA catch

3.4.1. OLS and elastic net regressions comparison

Catch of limpet, kelp, and sea urchin in OAAs were examined to resolve the effect of TURFs on adjacent areas in 2015 (*loco* is not included in OLS and elastic net regressions since its extraction is banned in OAAs). OLS models were inconclusive, yielding no significant predictors of catches outside TURFs though a considerable proportion of the variances were explained (adjusted $r^2 = 0.31$ for limpet, 0.55 for kelp, and 0.34 for sea urchin, Table 6). Geospatial variables were found to explain a greater amount of variance than TURF variables in OLS models for all species. Elastic net regression models explained similar proportions of variance as the OLS models, but with fewer variables (adjusted $r^2 = 0.31$ for limpet, 0.55 for kelp, and 0.51 for sea urchin, Table 6).

3.4.2. predictors selected by the elastic net regression of OAA catch

Contrasting results from the elastic net regression model were found for the three groups of species, with different predictors selected by the penalized model in explaining OAA catches (Table 7). All "highly important predictors" retained to explain catch in OAAs for the three species groups were related to the geospatial context. The predictor *Urban_Area* was selected in 83.69% of the 10,000 bootstraps when modeling limpet catches and 99.73% of the bootstraps when modeling kelp extraction, being the strongest identified driver of catch outside of TURFs in both cases. This predictor exhibited a negative relationship with catches outside TURFs for both species groups, with lower catches in the OAAs for limpet (33% decrease for coves within 50 km to urban areas compared to those far from urban areas) and kelp (72% decrease) in fishing coves close to urban areas. Additional predictors for catch of limpet outside TURFs included *Area_OAA²* (selected in 60.35% of cases) and *Area_fraction* (selected in 41.86% of the bootstraps; definitions of predictors in Table 1). There was a 0.08% reduction in limpet catch per $10\,km^2$ of additional OAA area and a 1.5% reduction per 1% increase in the fraction of the total area that is TURF.

Several variables were found to be important predictors of kelp catches in OAAs. *Loco* CPUEs inside TURF (*Loco_per_diver*) was a highly important, positive predictor of outside catches of kelp and was included in 91.11% of the bootstraps (42% increase of kelp catch in OAAs for every additional 1 mt catch of *loco* per diver within the TURF, with the average *loco* catch being 0.46 mt *loco*/diver). Similarly, higher catch rates of kelp inside TURFs were associated with higher catches of kelp outside (1.4% increase of kelp catch outside a TURF for every additional 1 mt catch of kelp per collector within the TURF, with the average kelp catch being 8.6 mt kelp/collector). Counterintuitively, lower catches of kelp outside of TURFs were associated with fishing coves that had larger OAAs (11% decrease in catch for every additional $10\,km^2$ of OAA area; *Area_OAA* and *Area_OAA²* were selected in at least

Table 2
Average size of TURFs and OAAs with associated average harvesters effort for fishing coves considered in each region. Standard deviations are indicated in parentheses. Northern regions consist of regions II, III, IV, V while southern regions include regions VIII, IX, XIV, X and XI. The number of fishing coves included for each region is given by N.

Region	TURF		OPEN-ACCESS AREA	
	Area (km^2)	Effort (individual)	Area (km^2)	Effort (individual)
Northern $N = 63$	3.2 (± 3.1)	25.1 (± 25.4)	60.8 (± 16.5)	33.1 (± 34.5)
Southern $N = 114$	4.9 (± 9.3)	37.0 (± 57.1)	90.5 (± 29.2)	58.0 (± 103.0)

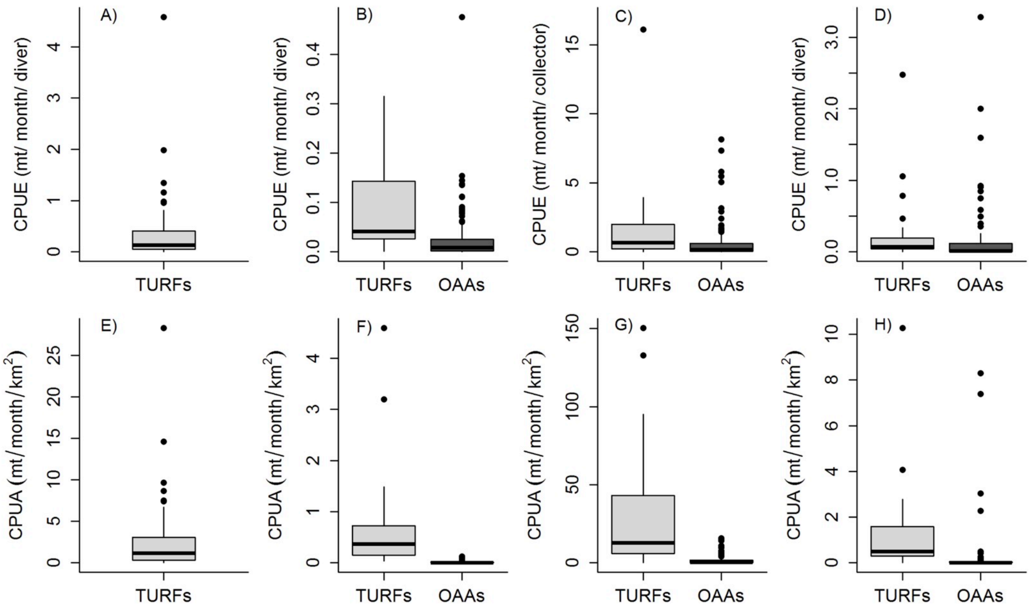

Fig. 2. Boxplots of 2015 CPUEs (A, B, C, D) and CPUAs (E, F, G, H) for the four species *loco* (A,E) keyhole limpet (B, F), kelp (C, G), and red sea urchin (D, H) by fishing coves, differentiated by catch origin inside or outside TURFs (i.e. OAA). *Loco*'s extraction is banned in OAAs.

80% of bootstraps). Lower catches of kelp outside of TURFs were associated with fishing coves that had older TURFs (e.g., for every year increase in *Age_TURF_max*, there is a ~8% decrease in catch; *Age_TURF_mean* and *Age_TURF_max* were included in 73.03% and 84.40% of the bootstraps) and fishing coves with a higher fraction of fishing grounds managed as TURFs (0.8% decrease for every 1% increase in the fraction of area designated as TURFs; *Area_fraction* was selected in 47.36% of models). Finally, lower catches of kelp in OAAs were observed in fishing coves with several fishers' organizations (4.4% decrease for every additional organization). The model indicates that OAAs with higher catches of kelp tended to be smaller, outside of urban centers, in areas with productive *loco* fisheries, and have fewer, younger, and proportionately smaller proximate TURFs.

Higher catches of sea urchin in OAAs were associated with fishing coves that have longer coastline lengths (13% increase in catch for every additional 10 km of coastline). This predictor was highly important in explaining catch of sea urchin (selected in 98.98% of cases). A decrease of sea urchin catch in OAAs was observed in fishing coves that had older TURFs (3% decrease in catch for every additional year since TURF implementation). The related variables *Age_TURF_max* and *Age_TURF_mean* were included in 47.3% and 47.0% of the bootstraps, respectively.

4. Discussion

We evaluated temporal and spatial trends in catch and catch rates for TURFs and OAAs in Chile. This study is the first to consider fishing coves all along the Chilean coast to understand the TURF system in its entirety (TURFs and their surrounding areas) over two decades. Though increased CPUEs inside of TURFs compared to OAAs has been demonstrated in previous literature (Castilla and Fernández, 1998; Gelcich et al., 2012; Defeo et al., 2016), most studies have focused on small-scale projects in specific regions of the country. The most spatially extensive study was based on a systematic literature review of the effects of TURFs on ecosystem services in Chile considering 268 study sites all along the Chilean coast (Gelcich et al., 2019). It showed that TURFs sustain biodiversity and all typologies of ecosystem services (i.e., supporting, provisioning, regulating and cultural services), but stressed a lack of studies addressing potential negative or unpredicted consequences of TURFs and a need to better understand changes over time (Gelcich et al., 2019). Our study expands the scale of previous analyses, focusing on the comparison between TURFs and OAAs, and shows that median catch rates (CPUAs and CPUEs) of benthic resources were at least 75% higher inside TURFs than in surrounding areas. To the extent that these catch rates are indicators of biomass, this result points out that Chilean TURFs appear to align with their main objectives in 2015, i.e. "ensure the sustainability of artisanal fishing through the assignment of natural banks" and "maintain and increase the biological productivity of benthic resources". However, our study also indicates that catch rates have been steadily declining within TURFs and that TURFs may impact catch levels in surrounding OAAs, both of which are potential risks to system sustainability.

Three possible mechanisms could produce higher CPUASs and CPUEs in TURFs: 1) recovered biomass could have built up and improved catch rates within TURFs over time, 2) TURFs could have been implemented in areas of better habitat and higher quality grounds, and/or 3) effort displacement following the implementation of TURFs could have degraded OAAs over time. CPUAs and CPUEs of *loco*, keyhole limpet, kelp, and red sea urchin in TURFs and OAAs were analyzed to investigate differences between areas and over time. Additionally, catch

Table 3
2015 median catch rates (i.e., catch per unit of effort and catch per unit of area) from inside TURFs and OAAs (i.e., outside TURFs) for each of the four species groups. CPUE is given in mt/month/harvester. CPUA is given in mt/month/km^2. % Diff. is the percentage difference between median catch rates from the two areas.

	Loco	Limpet			Kelp			Sea urchin		
	Inside	Inside	Outside	% Diff.	Inside	Outside	% Diff.	Inside	Outside	% Diff.
CPUE	0.13	0.04	0.01	75.00	0.66	0.16	75.75	0.07	0.01	85.71
CPUA	1.09	0.36	0.001	99.72	12.85	0.39	96.96	0.49	0.01	97.96

Table 4
Results of the linear mixed effect models estimating log-transformed CPUAs for *loco*, limpet, kelp, and sea urchin inside TURFs. Model 2 only considers a subsample of fishing coves that have a constant number of TURF(s) for at least ten years. Significance is denoted by: p < 0.001 = '***', p < 0.01 = '**', p < 0.05 = '*', p < 0.1 = '.'. Coefficients were transformed in the text into percent changes in CPUA for a given change in the predictor variable using the following formula: $\%\Delta y = 100{\cdot}(e^{\beta\Delta x} - 1)$. Marginal and conditional coefficients of determination are respectively given by r_m^2 and r_c^2. Number of observations and number of fishing coves included for each model are respectively given by n and N.

		Loco		Limpet		Kelp		Sea urchin	
Model	Predictor	Coeff.	p-value	Coeff.	p-value	Coeff.	p-value	Coeff.	p-value
1	Intercept	1.19	1.23E-19 ***	0.37	0.01 *	3.63	5.32E-18 ***	0.82	0.01 *
	Year	−0.08	3.46E-25 ***	−0.04	2.80E-4 ***	−0.02	0.45	−0.05	0.03 *
	N_{TURF}	−0.11	1.19E-4 ***	−0.29	3.07E-10 ***	−0.34	4.31E-7 ***	−0.22	4.10E-3**
	r_m^2	*0.10*		*0.22*		*0.25*		*0.12*	
	r_c^2	*0.68*		*0.64*		*0.65*		*0.72*	
	n	*1,077*		*473*		*203*		*220*	
	N	*138*		*78*		*49*		*50*	
2	Intercept	178.18	2.47E-13 ***	21.22	0.49	138.30	0.16	53.13	0.53
	Year	−0.09	2.71E-13 ***	−0.01	0.49	−0.07	0.17	−0.03	0.53
	r_m^2	*0.08*		1.38E-3		*0.03*		2.81E-3	
	r_c^2	*0.59*		*0.63*		*0.43*		*0.67*	
	n	*372*		*160*		*52*		*65*	
	N	*49*		*31*		*13*		*16*	

of keyhole limpet, kelp, and red sea urchin in OAAs was investigated to resolve any impacts of proximal TURFs. Our findings indicate that CPUAs and CPUEs are consistently larger inside TURFs but that CPUAs have been decreasing in TURFs over time and also with the number of TURFs implemented by fishing cove. Further, a weak negative impact of proximal TURFs on catches in OAAs was also found. This evidence appears to provide the strongest support for the hypothesis that TURFs were selectively implemented in the best fishing grounds since catch rates are higher inside TURFs throughout our data, yet declining over time and with the addition of new TURFs. Additionally, the small negative effect of proximal TURFs of OAA catches could result from effort displacement and suggests management spillover. Declining catch rates over time within TURFs does not appear to support the hypothesis that catch rates are improved in TURFs due to a recovery of biomass. As we were only able to calculate CPUAs over time, this finding could result from consistent reductions in effort. Nationally, however, the number of registered divers has been constant while the number of collectors has increased over the last decade (Appendix 6, Sernapesca 2015). It is not clear how average fishing effort by registered harvesters (e.g., number of trips/harvester) may have changed over this period, and TURFs may now be used less intensively. Interestingly, for some species, CPUAs were found to have been decreasing in fishing coves that have had a constant number of TURF(s) for at least 10 years, indicating that the observed temporal change in CPUA is not only due to selective implementation of TURFs, but possibly due to changes in the local environment or the intensity of fishing effort.

Exogenous geospatial factors (e.g., coastline, OAA areas, urban areas) were the main drivers explaining variability of catches in OAAs across fishing coves for 2015 (based on selection in elastic net regressions and the greater amount of variance explained in the OLS analyses including just these variables, Table 5). Geospatial predictors always had a higher percentage of inclusion when compared to TURF management-related predictors (Table 6). The negative relationship between catches of limpet and kelp in OAAs and proximity to urban centers could be due to higher historical fishing pressure and deteriorated environments in more populated urban areas. Additionally, catch of kelp, a lower value product, in OAAs could also be higher in rural areas where there are fewer economic opportunities and thus lower opportunity costs for fishers. Fishing coves with longer coastline lengths seem to support higher catches of sea urchin, suggesting environmental

Table 5
Results of the linear mixed effect models estimating log-transformed CPUAs for limpet, kelp, and sea urchin outside TURFs (OAAs). There is no result for *loco* as it is not exploited in OAAs. Model 2 only considers a subsample of fishing coves that have a constant number of TURF(s) for at least ten years. Significance is denoted by: p < 0.001 = '***', p < 0.01 = '**', p < 0.05 = '*', p < 0.1 = '.'. Coefficients were transformed in the text into percent changes in CPUA for a given change in the predictor variable using the following formula: $\%\Delta y = 100{\cdot}(e^{\beta\Delta x} - 1)$. Marginal and conditional coefficients of determination are respectively given by r_m^2 and r_c^2. Number of observations and number of fishing coves included for each model are respectively given by n and N.

		Limpet		Kelp		Sea urchin	
Model	Predictor	Coeff.	p-value	Coeff.	p-value	Coeff.	p-value
1	Intercept	−5.87	1.40E-16 ***	−2.63	8.03E-19 ***	−5.23	2.48E-16 ***
	Year	−0.05	4.57E-9 ***	0.03	0.03*	-8.01E-3	0.41
	N_{TURF}	0.02	0.61	−0.03	0.66	6.87E-3	0.85
	r_m^2	*0.01*		2.21E-3		2.33E-4	
	r_c^2	*0.60*		*0.81*		*0.75*	
	n	*1,433*		*933*		*1,155*	
	N	*179*		*148*		*161*	
2	Intercept	44.93	0.05*	4.49	0.92	−23.49	0.45
	Year	−0.03	0.02*	-3.01E-3	0.89	8.82E-3	0.57
	r_m^2	4.84E-3		*2.59E-5*		3.29E-4	
	r_c^2	*0.55*		*0.65*		*0.66*	
	n	*534*		*336*		*394*	
	N	*54*		*49*		*52*	

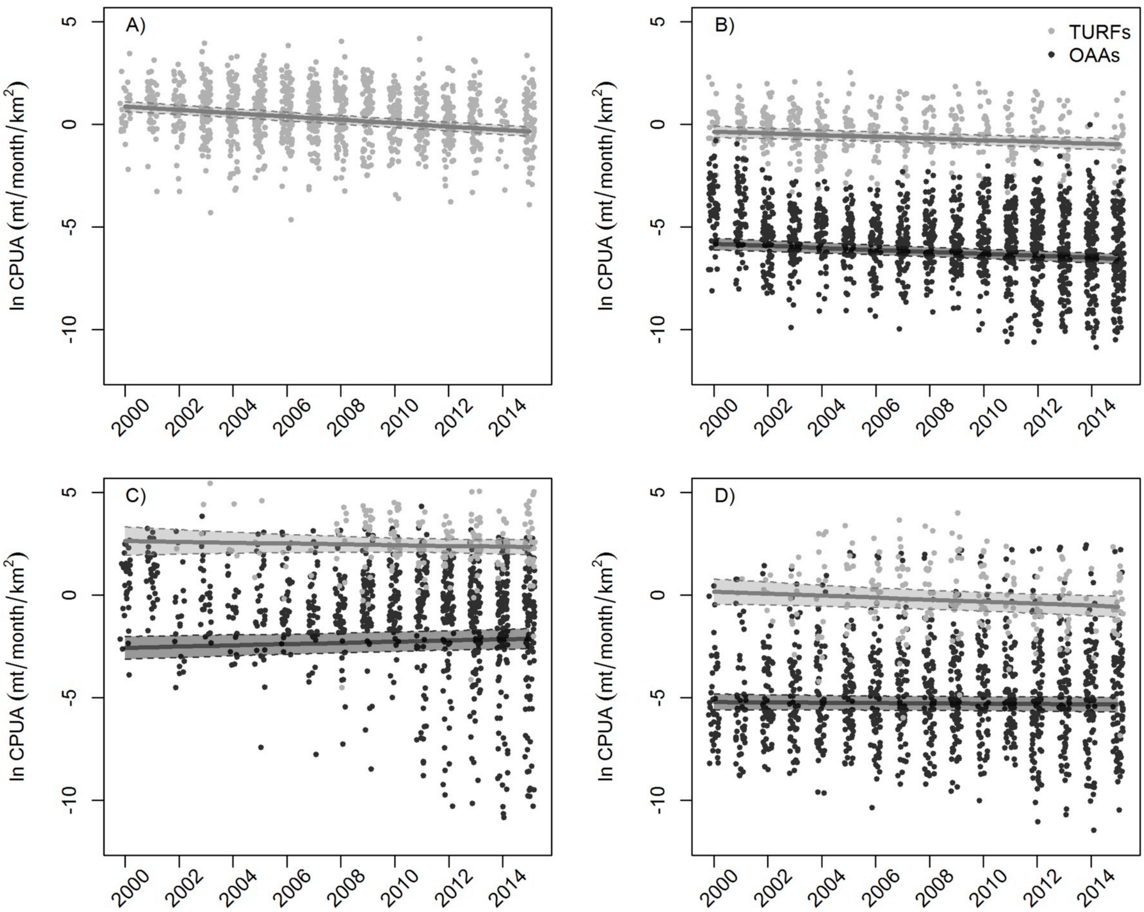

Fig. 3. Observed catch per unit of area (mt/month/km^2) values for the four species *loco* (A), keyhole limpet (B), kelp (C), and red sea urchin (D) used in the mixed effect models. Light grey dots are CPUAs from inside TURFs, dark grey dots are CPUAs from OAAs. Predicted value and standard errors for a given year is given by the straight line and shaded area.

Table 6
Change in OLS variance explained with specific variables: OLS_all includes all initial predictors, OLS_TURF includes only TURF related predictors, OLS_Geo includes only geospatial context predictors and OLS_elastic includes only predictors selected by the elastic net model.

		OLS_all	OLS_TURF	OLS_Geo	OLS_elastic
Limpet	Adj.r^2	0.31	0.12	0.25	0.31
	r^2	0.54	0.28	0.37	0.36
Kelp	Adj.r^2	0.55	0.26	0.42	0.55
	r^2	0.71	0.40	0.52	0.65
Sea Urchin	Adj.r^2	0.34	0.33	0.40	0.51
	r^2	0.70	0.52	0.55	0.58

factors related to coastline complexity may be the principal drivers for sea urchin abundance and availability. The effect of TURFs on catches in OAAs was especially weak for limpet and sea urchin (TURF-related predictors selected for 40–50% of bootstraps). However, when predictors related to TURFs' characteristics and activity (i.e., time since TURF registration or fisher organization implementation, and fraction of TURF area) were retained in the models, they consistently displayed a negative relationship with OAA catches.

Several aspects of the Chilean TURF system and available data are worth mentioning to provide additional context and inform interpretation of results. First, this study only considered fishing coves with at least one functioning TURF (operative or stand by) in 2016. Gelcich et al., (2017) revealed that about 40% of TURFs are inactive or currently abandoned in Chile. TURFs that have been abandoned would have increased OAAs, inferring that CPUA values could be lower in OAAs than actually observed (but possibly higher within TURFs). Second, it is possible that temporal dynamics and interactions between TURFs and OAAs may have changed over time. Our analysis began in 2000, however TURF management commenced in the early 1990s and approximately 18% of the TURFs considered here were initiated prior to 2000. Further analysis and investigation are needed to determine temporal changes and management interactions during the first decade of TURF management. Finally, TURFs are a management tool typically used to achieve sustainable fisheries and resource extraction within their boundaries (Christy, 1982; Aceves-Bueno and Halpern, 2018), though it is possible that some TURFs in Chile are maintained today for non-extractive purposes. For example, Chilean TURFs have been argued to build leadership and social cohesion among fishers (Rosas et al., 2014; Gelcich et al., 2019) and may offer benefits for conservation or restoration of benthic habitats (Gelcich et al., 2008a; Blanco et al., 2017; Fernández et al., 2017). Non-extractive social or ecological benefits arising from maintained TURFs in Chile are not considered here but are important areas for future research.

While this analysis was able to discern broad temporal and spatial trends by evaluating catch and catch rates across 196 fishing coves over two decades, the available data was generally coarse and requires consideration for potential biases. Recent studies have shown that misreporting can be a problem in officially reported catches (Oyanedel et al., 2017; Ruano-Chamorro et al., 2017), particularly with respect to *locos* (official catch is thought to only account for 14–30% of total *loco* extraction in Chile). As this research was primarily focused on relative trends and comparisons among catch and catch rates in OAAs and TURFs, misreporting was considered to only be problematic if it were

Table 7
Results of the elastic net regression model estimating catches for limpet, sea urchin and kelp outside the TURFs according to λ_{min}, the value that minimizes the cross-validation MSE which yields the most accurate model. Only predictors that were selected for at least 40% of the 10,000 bootstraps are shown in this table and they are ranked according to their importance (i.e., higher percentage of inclusion in the model). Elastic net mean coefficients were returned on the original scale here but they were transformed in the text into percent changes in catch for a given change in the predictor variable using the following formula: $\%\Delta y = 100 \cdot (e^{\beta \Delta x} - 1)$. OLS normalized coefficients are unitless. Number of observations for each model is given with n.

	Predictor	% inclusion	Sign	Elastic net coefficient	OLS normalized coefficient
Limpet	Divers_All [a]	100	+	0.02	0.99
n = 54	Urban_Area	83.69	–	0.40	0.50
	Area_OAA^2	60.35	–	8.15E-5	0.58
	Area_fraction	41.86	–	0.02	0.72
Kelp	Collectors_All [a]	100	+	2.31E-3	0.93
n = 54	Urban_Area	99.73	–	1.28	0.63
	Loco_per_diver	91.11	+	0.35	0.47
	Area_OAA	90.48	–	0.01	0.51
	Area_OAA^2	84.77	–	1.03E-4	2.10E-3
	Age_TURF_max	84.40	–	0.08	0.16
	Kelp_per_collector	78	+	0.01	0.29
	Age_TURF	73.03	–	0.05	0.33
	Age_Organization_max	59.74	–	0.07	0.16
	Area_fraction	47.36	–	7.53E-3	0.18
	N_ORG	41.80	–	0.05	0.48
Sea Urchin	Divers_All [a]	100	+	0.01	1.37
n = 36	Coastline_length	99.67	+	0.01	1.86
	Limpet_per_diver	62.45	–	4.84[b]	0.48
	Age_TURF_max	47.34	–	0.04	0.56
	Age_TURF	46.96	–	0.03	0.02

[a] The shrinkage penalty was set to 0 for the variable Divers_All and Collectors_All (instead of 1 for other variables), forcing this variable to be included in the model.
[b] This large effect is driven by two outliers. Removing this predictor did not qualitatively change the results.

non-uniform over space or time or differing between TURFs and OAAs. We examined the possibility of spatial heterogeneity in catch reporting by examining OLS model residuals using Studentized Breusch-Pagan tests and found no evidence of heterogeneous error variances across observations ($p > 0.05$, Appendix 7). Additionally, higher catch rates observed in TURFs appeared to be robust to catch misreporting. Estimation of OAAs and fishing effort were based on a number of assumptions regarding fishing behavior. The negative relationship found between catches of kelp and limpet and OAA size appears counter-intuitive: higher catches outside of TURFs were observed in fishing coves with smaller OAAs. It is possible that total fishing ground boundaries based on average travel distance (Ruano-Chamorro et al., 2017) and bathymetry were too liberal and thus OAA areas were over-estimated in some instances (e.g., coastline complexity and wave exposure might limit sailing of small boats and the effective fishing area). Future research could incorporate fishers' mobility among proximal fishing coves in fishing effort estimates, though it would require extensive field studies to determine the appropriate spatial range of effort. Finally, though CPUE values were found to be lower in OAAs, this metric relies on a crude estimate of effort as information on the number of trips or dive durations was not available. Nevertheless, consistency between CPUE and CPUA measures (metrics were found to be positively correlated in all areas) suggests that our CPUE values were a reasonable reflection of catch rates around fishing coves.

Various factors related to local governance could further explain low CPUAs and CPUEs observed in OAAs as well as the decrease of CPUAs observed over time. Such variables could include leadership, organizations' degree of cooperation, government support and governance network structure. A social-ecological-system framework (Ostrom, 2007) was found to be useful for examining these variables and associated institutional regimes in Mexico and Costa Rica (Basurto et al., 2013; García Lozano and Heinen, 2016). This type of analysis would require extensive fieldwork, and, therefore, the spatial scale of such analysis would likely be considerably smaller than that used in this study. Nevertheless, application of such an approach to the Chilean context represents an important avenue for future work that could enhance our understanding of the interaction between institutional factors and successful TURFs-based fisheries management.

Many countries are transitioning marine resource management from common property systems towards rights-based approaches (e.g., individual transferable quotas, catch shares, or TURFs), driven by concerns related to sustainability and resource stewardship (Orensanz et al., 2005, Nguyen Thi Quinh et al., 2017). Although the influence of MPAs on surrounding areas and fisheries sustainability are now well known, enhancing biomass through larval export and adult spillover (Gell and Roberts, 2003; Harrison et al., 2012) or negatively impacting surrounding unprotected waters through "fishery squeeze" and/or "fishing the line" behavior (Kellner et al., 2007; Caveen et al., 2014; Abbott and Haynie, 2012), the impacts of TURFs on surrounding areas have been poorly documented. This study contributes to a better understanding of management spillover between TURFs and OAAs. Whereas the impacts of TURFs appeared weak in this study, possibly growing over time given the negative relationship with TURF age variables, CPUEs and CPUAs were significantly lower in OAAs. This finding suggests that OAAs, whose total area is more than 50 times larger than grounds currently managed as TURFs, may be substantially degraded and overfished. Several authors have suggested that resources in OAAs might be heavily exploited and even depleted (González et al., 2006; Orensanz and Parma, 2010; Andreu-Cazenave et al., 2017; de Juan et al., 2017; Oyanedel et al., 2017; Ruano-Chamorro et al., 2017). Interestingly, our results do not show significant temporal declines in OAA CPUAs, suggesting either shifts in effort over time or that OAAs were depleted prior to 2000. The research presented here suggests that TURFs could place additional burden on already heavily fished OAAs. The current fisheries management regime in Chile includes limited assessment or monitoring of OAAs. It appears important that more attention be focused on OAAs, and on the system as a whole. By knowing that TURFs affect fisheries in OAAs, stocks outside managed areas may be more effectively controlled, provided that existing harvest controls outside of TURFs (i.e., bans, minimum legal size) are better enforced.

The Chilean TURF network is the largest worldwide, has been extensively studied and may, therefore, provide useful guidance for countries or regions transitioning toward rights-based approaches. For example, many Latin America countries have similar spatial management policies for small-scale fisheries (Mexico, Brazil, Costa Rica, Ecuador, Galapagos) and also share similar capacities for enforcement, dependence on a few high-value benthic species, extended OAAs, and co-management regimes (da Silva, 2004; Beitl, 2011; Defeo et al., 2016, García Lozano and Heinen, 2016). Determining whether or not unintended impacts of TURFs on OAAs, similar to those found here, exist in these regions is an important area for future research.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ocecoaman.2019.104961.

References

Abbott, J.K., Haynie, A.C., 2012. What are we protecting? Fisher behavior and the unintended consequences of spatial closures as a fishery management tool. Ecol. Appl. 22, 762–777. https://doi.org/10.1890/11-1319.1.

Aburto, J., Stotz, W., 2013. Learning about TURFs and natural variability: failure of surf clam management in Chile. Ocean Coast Manag. 71, 88–98. https://doi.org/10.1016/j.ocecoaman.2012.10.013.

Aburto, J., Thiel, M., Stotz, W., 2009. Allocation of effort in artisanal fisheries: the importance of migration and temporary fishing camps. Ocean Coast Manag. 52, 646–654. https://doi.org/10.1016/j.ocecoaman.2009.10.004.

Aburto, J., Gallardo, G., Stotz, W., Cerda, C., Mondaca-Schachermayer, C., Vera, K., 2013. Territorial user rights for artisanal fisheries in Chile – intended and unintended outcomes. Ocean Coast Manag. 71, 284–295. https://doi.org/10.1016/j.ocecoaman.2012.09.015.

Aburto, J.A., Stotz, W.B., Cundill, G., 2014. Social-ecological collapse: TURF governance in the context of highly variable resources in Chile. Ecol. Soc. 19, 2. https://doi.org/10.5751/ES-06145-190102.

Aceves-Bueno, E., Halpern, B.S., 2018. Informing the design of territorial use rights in fisheries from marine protected area theory. Mar. Ecol. Prog. Ser. 596, 247–262. https://doi.org/10.3354/meps12571.

Andreu-Cazenave, M., Subida, M.D., Fernandez, M., 2017. Exploitation rates of two benthic resources across management regimes in central Chile: evidence of illegal fishing in artisanal fisheries operating in open access areas. PLoS One 12, e0180012. https://doi.org/10.1371/journal.pone.0180012.

Asche, F., Gordon, D.V., Jensen, C.L., 2007. Individual vessel quotas and increased fishing pressure on unregulated species. Land Econ. 83, 41–49. https://doi.org/10.3368/le.83.1.41.

Attwood, C.G., Bennett, B.A., 1995. Modelling the effect of marine reserves on the recreational shore-fishery of the South-Western Cape, South Africa. Afr. J. Mar. Sci. 16, 227–240. https://doi.org/10.2989/025776195784156458.

Basurto, X., Gelcich, S., Ostrom, E., 2013. The social–ecological system framework as a knowledge classificatory system for benthic small-scale fisheries. Glob. Environ. Chang. 23, 1366–1380. https://doi.org/10.1016/j.gloenvcha.2013.08.001.

Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Stat. Softw. 67, 1–48. https://doi.org/10.18637/jss.v067.i01.

Beddington, J.R., Agnew, D.J., Clark, C.W., 2007. Current problems in the management of marine fisheries. Science 316, 1713–1716. https://doi.org/10.1126/science.1137362.

Beitl, C.M., 2011. Cockles in custody: the role of common property arrangements in the ecological sustainability of mangrove fisheries on the Ecuadorian coast. Int. J. Commons 5 (2), 485–512. http://doi.org/10.18352/ijc.285.

Bernal, P.A., Oliva, D., Aliaga, B., Morales, C., 1999. New regulations in Chilean fisheries and aquaculture: ITQ's and territorial users rights. Ocean Coast Manag. 42, 119–142. https://doi.org/10.1016/S0964-5691(98)00049-0.

Biggs, D., Amar, F., Valdebenito, A., Gelcich, S., 2016. Potential synergies between nature-based tourism and sustainable use of marine resources: insights from dive tourism in territorial user rights for fisheries in Chile. PLoS One 11. https://doi.org/10.1371/journal.pone.0148862.

Blanco, M., Ospina-Álvarez, A., González, C., Fernández, M., 2017. Egg production patterns of two invertebrate species in rocky subtidal areas under different fishing regimes along the coast of central Chile. PLoS One 12, e0176758. https://doi.org/10.1371/journal.pone.0176758.

Bohnsack, J.A., 2000. A comparison of the short-term impact of no-take marine reserves and minimum size limits. Bull. Mar. Sci. 66, 635–650.

Branch, T.A., 2009. How do individual transferable quotas affect marine ecosystems? Fish Fish. 10, 39–57. https://doi.org/10.1111/j.1467-2979.2008.00294.x.

Cancino, J.P., 2007. Collective Management and Territorial Use Rights: the Chilean Small-Scale Loco Fishery Case. ProQuest.

Castilla, J., 1994. The Chilean small-scale benthic shellfisheries and the institutionalization of new management practices. Ecol. Int. Bull. 47–63.

Castilla, J.C., Fernández, M., 1998. Small-scale benthic fisheries in Chile: on co-management and sustainable use of benthic invertebrates. Ecol. Appl. 8, S124–S132.

Castilla, J.C., Espinosa, J., Yamashiro, C., Melo, O., Gelcich, S., 2016. Telecoupling between catch, farming, and international trade for the gastropods *Concholepas concholepas* (loco) and Haliotis spp. (abalone). J. Shellfish Res. 35, 499–506. https://doi.org/10.2983/035.035.0223.

Caveen, A., Polunin, N., Gray, T., Stead, S.M., 2014. The Controversy over Marine Protected Areas: Science Meets Policy. Springer.

Chávez, C., Dresdner, J., Quiroga, M., Baquedano, M., Gonzalez, N., Castro, R., 2010. Evaluación socio-económica de la pesquería del recurso loco asociada al régimen de áreas de manejo, como elemento de decisión para la administración pesquera. (No. Informe Final. Proyecto FIP 2008-31).

Christy, F.T., 1982. Territorial Use Rights in Marine Fisheries: Definitions and Conditions. UN Food & Agriculture Organisation.

Costello, C., Gaines, S., Lynham, J., 2008. Can catch shares prevent fisheries collapse? Science 321, 1678–1681. https://doi.org/10.1126/science.1159478.

Costello, C., Ovando, D., Hilborn, R., Gaines, S.D., Deschenes, O., Lester, S.E., 2012. Status and solutions for the World's unassessed fisheries. Science 338, 517–520. https://doi.org/10.1126/science.1223389.

da Silva, P.P., 2004. From common property to co-management: lessons from Brazil's first maritime extractive reserve. Mar. Policy 28, 419–428. https://doi.org/10.1016/j.marpol.2003.10.017.

Davis, K.J., Kragt, M.E., Gelcich, S., Burton, M., Schilizzi, S., Pannell, D.J., 2015. Why are Fishers not enforcing their marine user rights? Environ. Resour. Econ. 1–21. https://doi.org/10.1007/s10640-015-9992-z.

de Juan, S., Gelcich, S., Fernández, M., 2017. Integrating stakeholder perceptions and preferences on ecosystem services in the management of coastal areas. Ocean Coast Manag. 136, 38–48. https://doi.org/10.1016/j.ocecoaman.2016.11.019.

Defeo, O., Castilla, J.C., 2005. More than one bag for the World fishery crisis and keys for Co-management successes in selected artisanal Latin American shellfisheries. Rev. Fish Biol. Fish. 15, 265–283. https://doi.org/10.1007/s11160-005-4865-0.

Defeo, O., Castrejón, M., Pérez-Castañeda, R., Castilla, J.C., Gutiérrez, N.L., Essington, T.E., Folke, C., 2016. Co-management in Latin American small-scale shellfisheries: assessment from long-term case studies. Fish Fish. 17, 176–192. https://doi.org/10.1111/faf.12101.

FAO, 2014. The State of World Fisheries and Aquaculture 2014. Food and Agriculture Organization of the United Nations, Rome.

Fernández, M., Blanco, M., Ruano-Chamorro, C., Subida, M.D., 2017. Reproductive output of two benthic resources (*Fissurella latimarginata* and *Loxechinus albus*) under different management regimes along the coast of central Chile. Lat am jaquat res 45, 391–402. https://doi.org/10.3856/vol45-issue2-fulltext-14.

Friedman, J., Hastie, T., Tibshirani, R., 2010. Regularization paths for generalized linear models via coordinate descent. J. Stat. Softw. 33. https://doi.org/10.18637/jss.v033.i01.

García Lozano, A.J., Heinen, J.T., 2016. Identifying drivers of collective action for the Co-management of coastal marine fisheries in the gulf of Nicoya, Costa Rica. Environ. Manag. 57, 759–769. https://doi.org/10.1007/s00267-015-0646-2.

Gelcich, S., Godoy, N., Prado, L., Castilla, J.C., 2008a. Add-on conservation benefits of marine territorial user rights fishery policies in central Chile. Ecol. Appl. 18, 273–281. https://doi.org/10.1890/06-1896.1.

Gelcich, S., Kaiser, M.J., Castilla, J.C., Edwards-Jones, G., 2008b. Engagement in co-management of marine benthic resources influences environmental perceptions of artisanal Fishers. Environ. Conserv. 35, 36–45. https://doi.org/10.1017/S0376892908004475.

Gelcich, S., Godoy, N., Castilla, J.C., 2009. Artisanal Fishers' perceptions regarding coastal co-management policies in Chile and their potentials to scale-up marine biodiversity conservation. Ocean Coast Manag. 52, 424–432. https://doi.org/10.1016/j.ocecoaman.2009.07.005.

Gelcich, S., Hughes, T.P., Olsson, P., Folke, C., Defeo, O., Fernández, M., Foale, S., Gunderson, L.H., Rodríguez-Sickert, C., Scheffer, M., others, 2010. Navigating transformations in governance of Chilean marine coastal resources. Proc. Natl. Acad. Sci. 107, 16794–16799. https://doi.org/10.1073/pnas.1012021107.

Gelcich, S., Cinner, J., Donlan, C.J., Tapia-Lewin, S., Godoy, N., Castilla, J.C., 2017. Fishers' perceptions on the Chilean coastal TURF system after two decades: problems, benefits, and emerging needs. Bull. Mar. Sci. 93. https://doi.org/10.5343/bms.2015.1082.

Gelcich, S., Fernández, M., Godoy, N., Canepa, A., Prado, L., Castilla, J.C., 2012. Territorial user rights for fisheries as ancillary instruments for marine coastal

conservation in Chile. Conserv. Biol. 26, 1005–1015. https://doi.org/10.1111/j.1523-1739.2012.01928.x.
Gelcich, S., Martínez-Harms, M.J., Tapia-Lewin, S., Vasquez-Lavin, F., Ruano-Chamorro, C., 2019. Comanagement of small-scale fisheries and ecosystem services. Conserv Lett 12, e12637. https://doi.org/10.1111/conl.12637.
Gell, F.R., Roberts, C.M., 2003. Benefits beyond boundaries: the fishery effects of marine reserves. Trends Ecol. Evol. 18, 448–455. https://doi.org/10.1016/S0169-5347(03)00189-7.
González, J., Stotz, W., Garrido, J., Orensanz, J.M., Parma, A.M., Tapia, C., Zuleta, A., 2006. The Chilean TURF system: how is it performing in the case of the loco fishery? Bull. Mar. Sci. 78, 499–527.
Halpern, B.S., Gaines, S.D., Warner, R.R., 2004. Confounding effects of the export of production and the displacement of fishing effort from marine reserves. Ecol. Appl. 14, 1248–1256. https://doi.org/10.1890/03-5136.
Harrison, H.B., Williamson, D.H., Evans, R.D., Almany, G.R., Thorrold, S.R., Russ, G.R., Feldheim, K.A., van Herwerden, L., Planes, S., Srinivasan, M., Berumen, M.L., Jones, G.P., 2012. Larval export from marine reserves and the recruitment benefit for fish and fisheries. Curr. Biol. 22, 1023–1028. https://doi.org/10.1016/j.cub.2012.04.008.
Hilborn, R., Stokes, K., Maguire, J.-J., Smith, T., Botsford, L.W., Mangel, M., Orensanz, J., Parma, A., Rice, J., Bell, J., Cochrane, K.L., Garcia, S., Hall, S.J., Kirkwood, G.P., Sainsbury, K., Stefansson, G., Walters, C., 2004. When can marine reserves improve fisheries management? Ocean Coast Manag. 47, 197–205. https://doi.org/10.1016/j.ocecoaman.2004.04.001.
Kellner, J.B., Tetreault, I., Gaines, S.D., Nisbet, R.M., 2007. Fishing the line near marine reserves in single and multispecies fisheries. Ecol. Appl. 17, 1039–1054. https://doi.org/10.1890/05-1845.
Kratz, B., Block, W.E., 2013. Privatize to save the fish. World Futures Rev. 5, 256–265. https://doi.org/10.1177/1946756713500025.
Lawrence, J.M., 2006. Edible Sea Urchins: Biology and Ecology. Elsevier.
Moreno, A., Revenga, C., 2014. The System of Territorial Use Rights in Fisheries in Chile. The Nature Conservancy, Arlington, Virginia, USA.
Morozova, O., Levina, O., Uusküla, A., Heimer, R., 2015. Comparison of subset selection methods in linear regression in the context of health-related quality of life and substance abuse in Russia. BMC Med. Res. Methodol. 15. https://doi.org/10.1186/s12874-015-0066-2.
Murawski, S.A., Wigley, S.E., Fogarty, M.J., Rago, P.J., Mountain, D.G., 2005. Effort distribution and catch patterns adjacent to temperate MPAs. ICES J. Mar. Sci. 62, 1150–1167. https://doi.org/10.1016/j.icesjms.2005.04.005.
Nakagawa, S., Schielzeth, H., 2013. A general and simple method for obtaining R^2 from generalized linear mixed-effects models. Methods Ecol Evol 4, 133–142. https://doi.org/10.1111/j.2041-210x.2012.00261.x.
Nguyen Thi Quynh, C., Schilizzi, S., Hailu, A., Iftekhar, S., 2017. Territorial use rights for fisheries (TURFs): state of the art and the road ahead. Mar. Policy 75, 41–52. https://doi.org/10.1016/j.marpol.2016.10.004.
Orensanz, J.M., Parma, A.M., 2010. Chile: territorial use rights successful experiment? Samudra Rep. 55.
Orensanz, J.M., Parma, A.M., Jerez, G., Barahona, N., Montecinos, M., Elias, I., 2005. What are the key elements for the sustainability of "S-fisheries"? Insights from south America. Bull. Mar. Sci. 76, 527–556.
Ostrom, E., 2007. A diagnostic approach for going beyond panaceas. Proc. Natl. Acad. Sci. 104, 15181–15187. https://doi.org/10.1073/pnas.0702288104.
Oyanedel, R., Keim, A., Castilla, J.C., Gelcich, S., 2017. Illegal fishing and territorial user rights in Chile. Conserv. Biol. 32. https://doi.org/10.1111/cobi.13048.
R Core Team, 2018. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria Available online at: https://www.R-project.org/.
Romero, P., Grego, E., Ariz, L., Figueroa, L., 2016. Contribución de las Áreas de Manejo de recursos bentónicos al nivel socioeconómico de los pescadores artesanales de la macro zona centro sur de Chile, Sudamérica. Cienc. Mar. 13.
Rosas, J., Dresdner, J., Chávez, C., Quiroga, M., 2014. Effect of social networks on the economic performance of TURFs: the case of the artisanal fishermen organizations in Southern Chile. Ocean Coast Manag. 88, 43–52. https://doi.org/10.1016/j.ocecoaman.2013.11.012.
Ruano-Chamorro, C., Subida, M.D., Fernández, M., 2017. Fishers' perception: an alternative source of information to assess the data-poor benthic small-scale artisanal fisheries of central Chile. Ocean Coast Manag. 146, 67–76. https://doi.org/10.1016/j.ocecoaman.2017.06.007.
San Martín, G., Parma, A.M., Orensanz, J.L., 2010. The Chilean experience with territorial use rights in fisheries. Handbook of marine fisheries conservation and management 24, 324–337.
Šidák, Z., 1967. Rectangular confidence regions for the means of multivariate normal distributions. J. Am. Stat. Assoc. 62, 626–633. https://doi.org/10.1080/01621459.1967.10482935.
Ury, H.K., 1976. Comparison of four procedures for multiple comparisons among means (pairwise contrasts) for arbitrary sample sizes. Technometrics 18, 89–97. https://doi.org/10.2307/1267921.
Van Holt, T., 2012. Landscape influences on Fisher success: adaptation strategies in closed and open access fisheries in southern Chile. Ecol. Soc. 17, 28. https://doi.org/10.5751/ES-04608-170128.
Wilen, J.E., Cancino, J., Uchida, H., 2012. The economics of territorial use rights fisheries, or TURFs. Rev. Environ. Econ. Policy 6, 237–257. https://doi.org/10.1093/reep/res012.
Zou, H., Hastie, T., 2005. Regularization and variable selection via the elastic net. J Roy Stat Soc B 67, 301–320. https://doi.org/10.1111/j.1467-9868.2005.00503.x.
Zúñiga, S., Ramírez, P., Valdebenito, M., 2008. Situación socioeconómica de las áreas de manejo en la región de Coquimbo, Chile. Lat am jaquat res 36, 63–81. https://doi.org/10.4067/S0718-560X2008000100005.
Subpesca, 2003. is an Administrative document and should be: Documento de Difusión No. 1. Áreas de Manejo y Explotación de Recursos Bentónicos. Departamento de Coordinación Pesquera y Departamento de Pesquerías de la Subsecretaría de Pesca. Available at: http://www.subpesca.cl/publicaciones/606/articles-9758_documento.pdf.