A mathematical optimisation model for the management of coastal ecosystem services conditioned by biophysical connectivity dynamics

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Abstract

In coastal areas there is a strong interdependence of humans and ecosystems, implying that management approaches need to ensure the provision of services to society. We developed a spatial optimization model that explores how the provision of ecosystem services varies along the coast under different management scenarios. Our approach assumes that the baseline structure of biological communities largely relies on large-scale connectivity. Therefore, based on the baseline potential of a site to provide services, conditioned by biophysical connectivity, the model explores the optimal allocation of protected areas. This approach was tested in central Chile, where the biological productivity is tightly linked to the hydrodynamics driven by the Humboldt current, through biophysical connectivity of early-life stages. In this region small-scale fisheries are regulated through management areas that alternate with open access fisheries. Some management areas have been shown to enhance abundance of target species and richness of associated communities, however, there are evidences for over-exploitation of fishery resources. By using this scenario as case study, we explored how the current coastal ecosystem could be spatially managed to optimize the provision of ecosystem services. The model evidenced that the current positioning of management areas could be maintained, but an additional 10% of the coast should be no-take area. Importantly, by considering all ecosystem properties prioritized by end-users, the model improved the spatial representativeness. The novelty of this approach is that it develops a spatial optimization model that embeds the dynamism of biophysical connectivity to assess the provision of ecosystem services within a network of protected areas.

1. Introduction

Worldwide, coastal and marine ecosystems are being fragmented, subjected to biodiversity loss and to the over-exploitation of many key marine resources as a result of multiple and cumulative human impacts (Crain et al., 2009; Worm et al., 2006). Addressing this global phenomena requires effective implementation of sustainable marine resource management and the conservation of ecosystems (Adams et al., 2004; Hutton & Leader-Williams, 2003). International panels like the Convention on Biological Diversity (United Nations, 1992) have responded by requiring member countries to protect 10% of their coastal and marine areas by 2020 (UNEP, 2010). According to Aichi target 14, by 2020, ecosystems that provide essential services and contribute to health, liveli-
hoods and well-being, must be restored and safeguarded, while accounting for the needs of women, indigenous and local communities, and the poor and vulnerable. To achieve this ambitious target, there is a need for integrated management plans that treat ecosystems and humans as a coupled, interacting system. In the coastal and marine realm, it is thus essential that the coast is approached as a spatial continuum that benefits society, or fails to do so, depending on how it is managed (Collins et al., 2011). Therefore, coastal conservation plans must be designed to allow for connectivity between spatially distinct areas to ensure population, genetic, community and/or ecosystem connectivity (Smith & Metaxas, 2018). In the marine environment, spatial connectivity is key for the architecture of ecosystems, as it affects the species as well as ecosystem productivity, dynamics and resilience (Hidalgo et al., 2017) and, therefore, the ecosystem ability to provide services to society (Carr et al., 2017). Yet, spatially and temporally dynamic connectivity amongst marine populations has rarely been incorporated into spatial planning, including MPA design criteria (Leslie, 2005). Instead, potential connectivity is usually incorporated through a static designation of MPA size and space between protected units (Smith & Metaxas, 2018).

There are different types or scales of ecological spatial connectivity (e.g., population connectivity, genetic connectivity, ecosystem connectivity), but, in general terms, ecological spatial connectivity refers to biological and biophysical processes that connect areas at large spatial scales and that is crucial for the persistence of marine communities (Carr et al., 2017). In the marine realm, large-scale connectivity is mediated through the dispersal of early-life stages of marine organism, that are predominantly sedentary as adults in demersal and benthic environments. Larval connectivity patterns are driven by biophysical processes that are characterized by strong temporal and spatial variability (Ospina-Alvarez et al., 2018). This variability is likely to have important consequences for the flow of ecosystem services to societies, and therefore, is an essential input for the configuration of marine reserve networks (Fovargue et al., 2017). Systematic conservation planning has been widely applied to inform decision makers on the spatial configuration of management units (Margules & Pressey, 2000). However, most approaches for conservation planning focus on maximizing the representation of habitat types as a proxy for biodiversity, without taking into account the key role of ecological connectivity (Pressley et al., 2007; Magris et al., 2015). This is largely due to the complexity of incorporating dynamic spatio-temporal processes into spatial optimization algorithms, while spatial connectivity structures marine communities with a strong inter- and intraannual variability Ospina-Alvarez et al. (2018). The spatial configuration of marine protected areas must account for large-scale biological connectivity, mainly driven by the pelagic stages of marine organisms. Recent models (e.g., White et al., 2014) incorporate connectivity parameters in spatial optimization software (e.g., MARXAN), including self-retention and several connectivity network metrics (e.g., betweenness centrality, degree centrality, or meta-population lifetime) to identify sites most relevant for meta-population persistence (Magris et al., 2015; Kruelck et al., 2017). But these are static metrics that establish ecological guidelines for reserve network design. Moreover, there is generally a trade-off in maximizing either persistence or spillover in reserve networks (Chollett et al., 2017).

On the other hand, it is widely acknowledged that ecosystem service research needs to be incorporated into decision-making to improve the outcomes, and acceptability, of marine conservation strategies. ‘Uses’ and ‘activities’ are familiar concepts for marine resource managers, but the social benefits associated with a given use are often overlooked (Cornu et al., 2014). In fact, advances in this field have been slow (Bennett et al., 2015; Daily et al., 2009). A current scientific challenge is to provide evidence of how optimal configurations of protected areas can maximize the flow of services to society. To obtain positive social responses to conservation and management actions, it is imperative to incorporate societies’ perceptions and priorities into conservation proposals (de Juan et al., 2017; Martín-López et al., 2012). By doing so, it is possible to take an integrated approach to protecting ecosystems and optimizing the benefits they provide to society. This approach is an improvement on traditional approaches based on resource use. Ecosystem services, as the benefits society obtain from ecosystems, are reliant, to a large extent, on a suit of ecosystem processes and functions, including the diversity of biological communities and habitats (Kremen, 2005). However, this link entails complexity as many services are generated by different combinations of ecosystem processes,
interacting over different space and time scales (Kremen, 2005). Consequently, ecosystem services are not always demanded in the same location as the ecological processes that underpin them (van Jaarsveld et al., 2005; Rodríguez et al., 2006). In future management scenarios, accounting for the flow of ecosystem services between management units will be crucial to ensure ecosystems can contribute to the well-being of societies in the long term (Potts et al., 2014).

Particularly in coastal areas, connections between habitat patches are crucial to maintain the resilience of populations (Ospina-Alvarez et al., 2018) and ensure the flow of services to society (Carr et al., 2017), e.g., the provision of food by a productive area (Lester et al., 2013; Chollett et al., 2017). In this research, we develop a spatial optimization model that incorporates a biophysical connectivity model, dynamic in space and time (Ospina-Alvarez et al., 2018), the quantification of ecosystem services provision by subtidal communities (de Juan et al., 2015), and users’ preferences for coastal ecosystem attributes (de Juan et al., 2017). The novelty of this approach is a spatial optimization model that relies on biological variables, conditioned by biophysical connectivity, to optimize the provision of ecosystem services along a coastal area. Importantly, the spatial optimization model differs from decision supports tools, that generally require static information on connectivity (e.g., MARXAN, Chollett et al., 2017), in that it embeds the connectivity matrices in the optimization algorithm. Our final objective is to introduce a novel optimization approach that has potential as a practical tool for advancing the large-scale integrated management of coastal and marine ecosystems.

The optimization model is tested in central Chile, which serves as an ideal location for this research for several reasons. First, the importance of the area’s hydrological system, which is driven by the Humboldt current (Thiel et al., 2007). Second, the current management regimen is dominated by areas locally managed by the small-scale fisheries associations (Gelich et al., 2012). Finally, data on subtidal coastal ecosystems is available, with estimations of ecosystem service provision in these areas (de Juan et al., 2015) and the preferences by the principal users (de Juan et al., 2017). In this region, coastal fisheries principally exploit benthic resources and are regulated through a well-developed system of TURFs, known as Management and Exploitation Areas for Benthic Resources (hereafter management areas) (Fernández & Castilla, 2005). This well-established system of management areas is currently being reformed using coastal planning approaches and incorporating CBD recommendations for protected area implementation. Traditional fisheries in this region coexist with increasing demands for recreational space (Godoy et al., 2010), particularly in the central coast where there is high population density (de Juan et al., 2017). The coexistence of traditional and new uses, combined with the current management transformations, is the perfect scenario to test a novel spatial optimization approach that incorporates the dynamics of marine populations and social preferences for ecosystem services into spatial planning decisions.

2. Methods

2.1. Model Domain

The study area is within the Eastern Boundary Upwelling Ecosystem of the south-east Pacific (Strub et al., 1998; Thiel et al., 2007), and the variability in hydrographic conditions is driven by the south-east Pacific atmospheric anticyclone that provides increasingly steady upwelling favorable winds with distance North. The principal hydrodynamic driver is the Humboldt current flowing northwards approximately 200 km off the coast, and a more coastal current (Chilean coastal current) that responds to upwelling-favorable wind forcing and is essential for the transport of coastal released gametes (Ospina-Alvarez et al., 2018).

The model domain extends through the central coast of Chile (31.57°S to 36.00°S), with a spatial resolution of 2 latitudinal km (Fig.2.1). The model has been designed for the biophysical system where small-scale fisheries generally operate, and where most of the existing data has been obtained: the rocky intertidal and subtidal coastal strip. Shallow subtidal rocky habitats along the coast of central Chile are characterized by kelp forests (Lessonia trabeculata) and a diverse array of mobile macro-invertebrates that predate or graze over benthic communities (Pérez-Matus et al., 2017). Some of these invertebrates are subjected to fisheries exploitation by divers or coastal gatherers: the principal species (p) are the Chilean abalone (p1, C. concholepas), key-hole limpet (p2, Fissurella spp.) and red sea urchin (p3, Loxechinus albus).
Figure 1: Study area maps showing the proportion of rocky shore substrate per unit area at left and the proportion of managed areas (M) on rocky shore substrate at right.
Within the model domain there are 75 operative management areas for the exploitation of benthic resources (hereafter M), covering ca. 31% of the coastal area. The availability of suitable habitat for the three key target species, i.e. rocky shores, was obtained by characterizing the habitat type (rock vs. sand) using high resolution satellite photography (Fig. 2.1, left). Also, the spatial distribution of the fishing regimen was characterized based on the presence or absence of M along the coast with data available from the Sub-secretary of Fisheries (SUBPESCA; Fig. 2.1, right). All digitalization was conducted with the software QGis 3.4. (see detailed methodology in Blanco et al., 2017). The combination of these two sources of data provided information on the effective M per cell (i.e., proportion of rocky coast covered by M), while the remaining proportion of rocky coast in each cell is allocated to open access areas (hereafter O). Note that the model domain, despite encompassing ca. 5 latitudinal degrees, is restricted to rocky shores as the habitat of the species objective of this study, which corresponds to approximately 40% of the model domain.

2.2. Biological data

In order to estimate the gamete supply ($L$) of the three target species, biological surveys were conducted in the subtidal area between 2012 and 2014. Four sites were selected within the model domain to collect samples and assess adult size and gonad investment of key-hole limpet and red sea urchin (Blanco et al., 2017). Published technical reports providing data in the area were used to estimate the size and gonad investment of Chilean abalone ($n = 2900$). An additional set of samples obtained in seven sites within this domain, including M, O and no-take areas (hereafter N), were used to assess adult density (de Juan et al., 2015). These parameters allowed the estimation of potential fecundity. At each site, paired M-O or N-O sites were sampled to assess differences in the biological parameters driven by the fishing regimen (de Juan et al., 2018). The potential egg production per cell was estimated based on potential fecundity per unit of rocky area per fishing regime (Blanco et al., 2017). The resulting total eggs produced in a cell, escalated by the percentage of available habitat and rated by the percentage of M/N/O, is the gamete supply per species in a cell (Blanco et al., 2019).

2.3. Biophysical connectivity

The ocean velocity fields were obtained from the Hybrid Coordinate Ocean Model (HYCOM) analysis (Chassignet et al., 2007), a data assimilating forecast of the global ocean circulation of sufficiently high resolution to accurately reproduce the mesoscale processes that dominate ocean variability. Then, a Spatially Explicit Individual Based Model (SEIBM) was used to simulate the dispersal of early life stages of the three target species. The SEIBM was coupled with the 3D hydrodynamic model using a customized version of the open source modeling tool ICHTHYOP (Lett et al., 2008). The code was used to simulate trajectories of the early life stages from velocity fields in a 3D hydrodynamic model. The coupled model took into account the Pelagic Larvae Duration of the species: 90 days for Chilean abalone, 20 days for red-sea urchin and 10 days for key-hole limpet. During this time, individual larvae are subjected to advective and diffusive processes that condition their alongshore, cross-shore and their vertical movement in the water column following Diel Vertical Migration (DVM) behavior. We used an adjusted connectivity score that incorporates seasonality: summer and winter for Chilean abalone, and spring and summer for key-hole limpet and red sea urchin. This is necessary because the species exhibit reproductive peaks in these seasons. The spawning locations were identical for all species and homogeneous along the coast, and the optimal number of particles to be released was established as 20,000 per day, each five days (i.e., 120,000 particles per month). For a detailed description on the methodology see Ospina-Alvarez et al. (2018).

Several connectivity matrices were produced from the larval dispersal model, and one matrix per dispersal event was extracted. Potential connectivity was defined as the probability of larval transport from a spawning site $j$ to a destination location $i$. The matrices obtained were square matrices with as many rows and columns as there are coastal sites in the region of study. The columns $j$ represented the sources and the rows $i$ represented the destinations. The intersection $jxi$ define the matrix cells and record information about the probability of connection between each pair of locations. As the matrices were square-type, the diagonal represented local retention. Therefore, these matrices represented the potential connectivity, defined as the probability of larval transport from a spawning site $j$ to a destination location $i$, and contained...
Table 1: Number of gametes emitted per % of rock in each cell per species according to management.

| Gamete supply       | Open Access (O) | AMERB (M) | No-Take (N) |
|---------------------|-----------------|-----------|-------------|
| Loco (p1)           | 0.230           | 0.83      | 1           |
| Limpet (p2)         | 0.126           | 0.83      | 1           |
| Sea urchin (p3)     | 0.350           | 0.83      | 1           |

all needed information for metapopulation dynamics and time-varying processes (Aiken & Navarrete, 2014). Then, realized larval connectivity can be defined as the effective number of larvae that travel from j to i and the corresponding matrix \( H_{ijp} \) can be estimated for each species (p) using potential connectivity matrices weighted by relevant biological and environmental information, following Watson et al. (2010). Specifically, habitat availability (percentage of rocky and sandy coastline), management regimen (percentage of M, N and O) and adult abundance (as a factor determining egg and larval production) per each cell were used for weighting the potential matrix following Blanco et al. (2017).

In consequence, the model was structured in main five components:

- environmental envelope: proportion of rocky coast and prevailing management regimen \( z \in (O, M, N) \)
- source sites (i.e., j, spawning sites)
- destination sites (i, settlement sites)
- Gamete supply as a function of density, size, fecundity and sex ratio of adult individuals.
- potential larval connectivity: probability that a larvae spawned from any site, \( j_{1-n} \), will end up at \( i_{1-n} \).
- realized larval connectivity: matrix multiplies the probability that a larva will end up in cell, \( i_{1-n} \), by the number of larvae which were in the source cell, \( j_{1-n} \). The number of larvae released from \( j_{1-n} \) is a function of the gamete supply and the environmental envelope of the source cell (binary) and the number of larvae (variable) released from that cell. The number of larvae recruited is conditioned by the available habitat (% rock), the management regimen does not play a role as larvae can be transported to any area (Fig. 2.3).

2.4. Quantification of ecosystem services

The demand for ecosystem services in the study area was assessed through face-to-face interviews with the principal end users (see more details on the approach in de Juan et al. 2017). An intangible value of the coast prioritised by end users was the scenic beauty of the coast, identified by the absence of urban development on rocky coasts (de Juan et al., 2017). The proportion of the coast, within the model domain, that has not been modified by urban development was identified through Google Earth images. This information was overlapped with the characterization of rocky coast and provided the proportion of each cell characterized by the attribute scenic beauty. The tangible values prioritised by end users were the biological diversity and the fisheries productivity, which we realistically approached through estimations of gamete supply of the target species in a site. We estimated biological diversity as the average species richness \( (V_z) \) observed in the rocky subtidal under a variable fishing regimen. This information was collected by surveying benthic communities in a set of sites with paired M and O, and recording the average number of species under each fishing regimen, resulting in an average of 10.2 species in O, 11.7 in M, and 12.1 in N (see details on the methodology in de Juan et al. 2015). The gamete supply of the three principal target species emitted and received in a site \( (L_{ijp}) \) is the source of individuals to adult populations in the region. Based on the connectivity matrices described in the previous section, we can identify areas that are key to supply gametes to other areas along the coast. And sites that excel for their gamete supply are key to maintain the population of target species in the area.

2.5. Spatial optimization model

We use a spatial optimization model to identify the optimal configuration of restricted areas (including M and N) to maximize a set of ecosystem services in different
Figure 2: Larval connectivity networks for key-hole limpet, red sea urchin and Chilean abalone. The connection between two locations is represented as arcs (yellow-red color scale) and the recruitment intensity by dots (green-blue color scale).
scenarios, including: gamete supply of target species, biological diversity, and scenic beauty. The optimization exercise follows a step-by-step approach by first taking into account the gamete supply of target species; secondly incorporating biodiversity values; and third, adopting an integral approach by also considering the scenic beauty prioritized by end-users. The software used for all optimization procedures was the General Algebraic Modeling System (GAMS) 23.5.2 (available at https://www.gams.com) using the CPLEX solver. We considered a range of management scenarios, described below.

2.5.1. Management scenarios

Alternative management scenarios were tested to assess how changes in the spatial configuration of management areas conditions the flow of ecosystem services to society. The optimization model was also run to assess if the current management regimen in the central coast of Chile is optimal to maximize the provision of ecosystem services, by comparing the current situation with the outcome of an optimal allocation of M. The following scenarios and models were assessed:

Scenarios:

- A. Ecosystem service: Gamete supply.
- B. Ecosystem service: Gamete supply and diversity.
- C. Ecosystem service: Gamete supply, diversity and scenic beauty.

Models:

1. Maintains existing management conditions.
2. Final allocation is constrained to 100 % O.
3. Model is constrained by existing ratio of M and O areas (30:70), but is free to allocate these areas in space.
4. Model solution is constrained to include existing M areas, but freely allocates 10% of the rocky study area to N (from existing O areas).
5. Final allocation is constrained to 100 % N.

2.5.2. Spatial optimization

We use linear programming with positive decision variables \(X_{iz}\) for each cell \(i\) for each zone \(z\), to identify optimal zoning of each cell under each model condition (1-5) for scenario A, and for model 3 for scenarios B and C. \(X_{iz}\) will take a value of 1 if cell \(i\) is allocated to zone \(z\), and 0 otherwise. The objective function for scenario A incorporates larva connectivity: how many larva of each species will be emitted from source cell \(j\), and successfully settle in sink cell \(i\). All terms are as previously defined, and \(R_j\) describes the percentage of rock in each destination cell.

\[
\begin{align*}
(1) \quad & \text{max} \sum_{ij} X_{iz} \times L_{ipz} \times H_{jpz} \times \lambda_{pz} \times R_i \\
(2) \quad & \sum_{i} X_{iz} = 1
\end{align*}
\]

Subject to:

This first constraint (Eq. 2) specifies that each cell must be fully allocated – the sum of the proportion of each cell allocated to each zone must equal 1. The term \(\lambda_{pz}\) describes the probability of a gamete of each species settling in a cell under different management regimes. The value in each case is described in Table 1. The following are additional, model-specific constraints.

Model 3:

\[
\begin{align*}
(3) \quad & \sum_{i} R_i \times C_z = \sum_{j} X_{ij} R_j
\end{align*}
\]

This constraint specifies that the rocky area in each cell that is allocated to a specific zone must be equal to \(C_z\), which is a zone-specific area target that describes the rocky proportion of each cell that must be allocated to either zone M (0.3, Model 3) or N (0.1, Model 4).

The objective function for scenario B builds on the objective function of scenario A, including all constraints, but includes a term to describe the diversity value \(V_z\) specific to the management regime.

\[
\begin{align*}
(4) \quad & \text{max} \sum_{ij} X_{iz} \times L_{ipz} \times H_{jpz} \times \lambda_{pz} \times R_i \times V_z
\end{align*}
\]

The objective function for scenario C is the same as scenario B, but subject to an additional constraint: that \(N\) can only occur in cells in which at least 80% of the rocky area of the cell is undeveloped (no human construction).
2.5.3. Normalisation

We normalised all ecosystem service values $x \in (L_{ij}, V_{ij})$ between 0 and 1 following the equations below, so that ecosystem system service values were comparable, and contributed equally to the optimisation value.

\[
\begin{align*}
(5) \quad a &= \min(x) \\
(6) \quad b &= \max(x) \\
(7) \quad x_n &= \frac{(x - a)}{(b - a)}
\end{align*}
\]

3. Results

Across the management scenarios that focused on the provision of the gamete supply (scenario A), the highest objective function value was achieved under model 5—which allocated 100% of the study area to the N zone. The worst objective function values were achieved under the models restricted to the existing management allocation (model 1) or 100% O (model 2). Compared to these base-or worst-case models, allowing the optimization model to freely allocate the existing M and O allocation (model 3), resulted in a much improved objective function (17.49) (Table 2). Allowing an additional 10% of rocky area to be allocated from the O zone (model 4), did improve the objective function, but not by much (17.94). Models 3 and 4, which optimized the location of management zones, indicated a 3 and 5 fold improvement over the base-case (model 1) and worse-case (model 2) models respectively. However, under these models, the fisheries restricted areas (M and N) were concentrated in the central and northern sections of the model domain, implying poor spatial representativity (Fig.3, panel A3 and A4).

Scenario B incorporates biological diversity in addition to the gamete supply. Under this scenario, models 3 and 4 obtain objective values similar to these models under scenario A. However, models B3-4 distribute the area to be protected amongst a larger number of cells and the southern region gains representativity in the fishery restricted area network, despite most restricted areas still being concentrated in the central region (Fig.3, panel B3). In this case, the objective value for biological diversity is only slightly lower than the value obtained in scenario B5, with all the area allocated to N (Table 2).

The scenario C4, incorporating the scenic beauty of the sites in the spatial optimization exercise, obtains an objective value lower than in scenario B4. While the distribution of fishery restricted areas is similar to B4, there is an improved spatial representativity of N, with less concentration of N% area in a few cells as observed in B4 (Fig.3, panel C4). Importantly, this scenario incorporates all the variables prioritized by end-users.

4. Discussion

An integrated coastal management should take into account the human dimension of ecosystems that can be approached by the assessment of the benefits that ecosystems provide to society, i.e. ecosystem services (Costanza et al., 1997). However, this is not an easy task, as the provision of coastal ecosystem services relies on the structure and processes of biological communities (Kremen, 2005), and thus it is ultimately controlled by large-scale ecological processes like biophysical connectivity (Ayata et al., 2010; Morgan & Fisher, 2010; Cowen et al., 2006). Currently, there is a lack of quantitative approaches that can be applied to marine spatial planning exercises that can address the dynamism of large-scale ecological connectivity. To our knowledge, published approaches are based on heuristic guidelines on reserve size and spacing, maximizing connectivity amongst units (and not taking into account persistence) and including site-level metrics on connectivity (Beger et al., 2010; Chollett et al., 2017). These approaches do not consider the intrinsic dynamism of biophysical connectivity. To address this challenge, we introduce a quantitative methodological framework that considers the biophysical and human components in a holistic and connected way to explore how optimal configurations of protected areas can maximize the flow of services to societies. In addition, this framework aims to address the challenge of achieving a diversity of management objectives, ranging from the conservation of key species for local fisheries to societal preferences for ecosystem attributes.

The spatial optimization model has been designed for a biophysical system where small-scale fisheries operate, and where different sources of data were available: the rocky intertidal and subtidal, characterized by kelp forests in the central coast of Chile. The oceanography of the region coupled with the biological characteristics
| Scenario 1 | Scenario 2 | Scenario 3 | Scenario 4 | Scenario 5 |
|-----------|-----------|------------|------------|------------|
| $NT_0$    | 0%        | 0%         | 0%         | 100%       | 0%         |
| $M_0$     | 30%       | 30%        | 30%        | 0%         | 30%        |
| $O_0$     | 70%       | 70%        | 70%        | 0%         | 70%        |
| Current   | Current   | Current    | Ideal      | Current    |
| Worst     | Optimal   | Ideal      | Ideal      |
| $NT_1$    | 0%        | 0%         | 10%*       | 10%*       | 100%       |
| $M_1$     | 0%        | 30%        | 30%*       | 30%*       | 0%         |
| $O_1$     | 100%      | 70%        | 60%*       | 60%*       | 0%         |

Figure 3: The assessment of changes in the provision of ecosystem services in the transition from the current management scenarios to the optimal, ideal and worst scenarios.

Figure 4: Scenarios to allocate management or/and no take areas from an optimization model. A1; A3; and A4.
Figure 5: Scenarios to allocate management or/and no take areas from a optimization model. B3; B4; and C4
Table 2: Description of scenarios, models, assumptions and data inputs.

| Scenario | Model | Description | Objective function value | Larval connectivity value | Species diversity value |
|----------|-------|-------------|--------------------------|--------------------------|-------------------------|
| A 1      | Existing |             | 5.96                     | -                         | -                       |
| A 2      | 100% allocated to O |             | 3.45                     | -                         | -                       |
| A 3      | Existing proportion but optimal allocation [M: 0.3, O: 0.7] |             | 17.49                    | -                         | -                       |
| A 4      | Existing M allocation, but additional 10% of study area into N (area taken from O) [M: 0.3, O: 0.6, N: 0.1] |             | 17.94                    | -                         | -                       |
| A 5      | 100% allocated to N |             | 39.08                    | -                         | -                       |
| B 3      | Existing proportion but optimal allocation [M: 0.3, O: 0.7] |             | 16.95                    | 17.49                    | 51.21                   |
| B 4      | Existing M allocation, but additional 10% of study area into N (area taken from O) [M: 0.3, O: 0.6, N: 0.1] |             | 17.65                    | 17.74                    | 51.63                   |
| B 5      | 100% allocated to N |             | 39.08                    | 39.08                    | 55.99                   |
| C 4      | Existing M allocation, but additional 10% of study area into N (area taken from O) [M: 0.3, O: 0.6, N: 0.1] |             | 15.33                    | 15.43                    | 51.63                   |

of the individuals determine that larvae produced in a site are transported to another site and, depending on favorable environmental conditions (for example, suitable habitat for benthic species) settle and recruit to the population (Ospina-Alvarez et al., 2018). These mechanisms were assessed by coupling a hydrodynamic model with an individual-based model. To work with realistic scenarios, the coupled model took into account the Pelagic Larvae Duration of the species: 90 days for Chilean abalone, 20 days for red-sea urchin and 10 days for key-hole limpet. During this time, individual larvae are subjected to advective and diffusive processes that condition their along-shore, cross-shore and their vertical movement in the water column following Diel Vertical Migration (DVM) behavior. We used an adjusted connectivity score that incorporates seasonality: summer and winter for Chilean abalone, and spring and summer for key-hole limpet and red sea urchin. This is necessary because the species exhibit reproductive peaks in these seasons.

The model nourishes from a diversity of data sources: in-situ estimations of diversity and density of benthic invertebrates in M, N and O areas; laboratory experiments to identify biological parameters of the key species; hydrodynamic modeling; face-to-face surveys with end users; and the implementation of a Spatially-Explicit Individual Based Model (SEIBM) for the study of larval transport from spawning to settlement areas. The biophysical connectivity matrix was embedded in the spatial optimization algorithm to estimate the realized supply of gametes spatially-explicit within the model domain (Blanco et al., 2017). The different data sources were combined by first identifying the importance of larval connectivity for the supply of gametes of three key target species, to then estimate diversity of benthic communities and the scenic beauty of different sites along the coast. This approach allows taking into account the different attributes prioritized by end-users in the study region (de Juan et al., 2015, 2017): the availability of target species for the small-scale fisheries, the biological diversity in the coastal area, and the scenic beauty of the coast perceived as undisturbed rocky areas. Relying on this multi-dimensional data structure, the optimization algorithm allowed the exploration of the consequences of different scenarios of management on the flow of services to society. The scenarios were founded on the basis that the fisheries management regimen (M, N, O) will change the underlying diversity and biological community composition that, in turn, affects the gamete supply along the coast mediated by biophys-
ical connectivity. The different scenarios provided similar outputs, likely driven by correlation between variables, e.g. biodiversity and non-urbanized areas, which is a consequence of exploring the value of coastal areas already shaped by human intervention. Despite a limited variability in the output, the proposed model confirmed expectations as the most integrated approach, considering the three key services demanded by end-users in the study area, improved the geographical distribution of protected areas and the benefits obtained from the connected network of open access and fisheries restricted areas.

The first scenario (scenario A) that prioritized gamete supply, improved the objective function of the current scenario when the optimization exercise freely allocated the existing proportions of M and O areas. Closing an additional 10% of the coast to fisheries (change O to N), while maintaining current position of M did not improve the objective function much. These results suggest that to maximize gamete supply we could either maintain the current proportion of M area, but in different locations, or maintain the proportion and location of M, but transform 10% of O to N area. A negative aspect of scenario A is the poor representation of the southern sections of the model domain in the restricted area network and thus, fails to meet the representativeness criteria included in Aichi targets. Scenario B incorporated biological diversity and provided similar objective values than scenario A; however, the protected areas were distributed among a larger number of cells and the southern region gained representation. Importantly the objective value obtained in models 3 (freely allocating current proportions of M and O) and 4 (changing 10% of O to N) was similar to that obtained when allocating 100% of the area to N, indicating these are good strategies to maximize gamete supply and biodiversity. The increase of protected areas by allocating 10% of current O to N could achieve a maximum benefit with no alteration of current position and extension of M, therefore, preserving the fishermen territorial rights associated to these areas. The maintenance of M current positioning, with the option to further regulate restriction regimens within these areas, could avoid the consequences of fishing effort displacement, while avoiding fishermen confrontation due to closure of their fishing areas. However, a case by case study approach is recommended to explore alternatives of N positioning that minimizes the impact on end-users. Scenario C incorporated scenic beauty in the spatial optimization exercise, so it illustrated the multi-dimensionality of coastal ecosystems. Despite this scenario providing a lower objective value than scenario B, it improved the spatial representation of N. The spatially uneven distribution of restricted areas across scenarios is a limitation of the model, as it prioritizes regions where the rocky habitat for the target species is dominant. A further development of this model should incorporate key species and diversity values from the sandy subtidal areas in the region, so the habitat representativeness proposed by Aichi targets is considered.

Aichi Target 11 urges CBD Contracting Parties to go far beyond the structure and spatial arrangement, the number, size and coverage of MPAs (Woodley et al., 2012). Target 11 also requires that areas are “conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures…”.

Several studies have criticized the lack of analysis to assess the extent to which the representativeness of many sensitive habitats has been achieved within global MPAs. However, very few studies have highlighted the almost complete lack of information on effective biological connectivity between established areas. At the same time, representativeness and connectivity do not feature in attempts to achieve objective 11 of Aichi. Our current approach seeks to remediate this situation in a complex system of fishery restricted areas that are connected in a system highly controlled by the hydrodynamism of the Humboldt current (Ospina-Alvarez et al., 2018). However, the model was designed for the rocky coast, meaning that central region was consistently prioritized as a conservation area, as the southern section of the model domain is characterized by greater proportions of sandy coast. This result emphasizes that habitat representativeness is an important criteria for reserve network design. With the availability of high-resolution information on biological community structure, including diversity among habitat types, we should incorporate greater biological spatial variability. Currently, the model considers average biological metrics per habitat type and management regimen, shaped by the dynamism of the biophysical model on ecological connectivity. An additional issue that needs to be resolved is the incorporation of temporal variability in the optimization exercise. As such, the model should deal with temporal variability in the hydrodynamic connectiv-
ity, so that management decisions regarding allocation of areas will achieve long-term objectives. To address this seasonal and annual variability, the connectivity matrix should represent the historic average (ideally, more than 20 yrs.). Connectivity based on this average would identify sites that are consistently good larva supply sites and would thus provide a solid basis for formal spatial planning exercises.

The current optimization frequently overlapped priority sites with highly populated areas, which include large commercial harbors and polluted areas associated with hydrothermal plants. This overlap implies an unfeasible establishment of protected areas and highlights the need to take into account all the components of coastal ecosystems in the spatial optimization, where the human users, the biophysical environment and the spatial-temporal dynamism form a complex interactive matrix. Scenario C, that takes into account the scenic beauty of a site, largely overcomes this issue as, by prioritising sites that hold higher scenic beauty, it indirectly selects less urbanized sites. But, paradoxically, coastal spatial planning in Chile over the last decade has been guided by the demand for space to allocate development activities such as aquaculture or coastal development. Therefore, efforts have been concentrated in areas with conflict between uses, and there has been an absence of a systematic approach for coastal spatial planning. Most existing MPAs are recent and have been selected in accordance with available budgets and by a governance system characterized by multiple sectorial agencies, all related with marine conservation but not coordinated (Cárcamo et al., 2013). In consequence, the designation of MPAs and areas for specific uses have relied on geo-political decisions and not on scientific recommendations that incorporate knowledge on supply and demand for ecosystem services. These local issues, that complicate spatial planning approaches, are common to many countries that lack a legal framework for the management of the coastal and marine continuum (REF? Miriam?). In these scenarios, the identification of priority sites relying on multiple and interacting data sources is essential for decision making.

During the last decade, Chile has made advances in the establishment of MPAs. However, a remaining task is to consolidate an MPA network in the coastal area, particularly in the central region where coastal human-uses are concentrated and no-take areas will face fishermen opposition (Suman et al., 1999; Gelcich et al., 2009). From the twenty official MPAs in Chile, only five are effectively managed (Mora et al., 2006; Petit et al., 2018). In this context, the current exercise can assist prioritization of management efforts for existing MPAs and the identification of management areas that could incorporate a no-take section and/or be managed under special regimens to increase the enforcement level. This could be a feasible approach as densities of economically important macroinvertebrates and reef fish have not been significantly different between highly enforced management areas and no-take MPAs (Gelcich et al., 2012). In fact, the allocation of management areas was highly consistent across models and there is probably an historical explanation for this mechanism, as fishermen tend to select the most productive sites for the allocation of management areas. Therefore, some prioritized management areas, allocated in the best places to supply ecosystem services, could be important ancillary conservation instrument if they were well enforced.

The broad adoption of ecosystem service science in coastal management is limited by the paucity of data (Saunders et al., 2016; Cárcamo et al., 2013), therefore, models such as ours are designed to motivate the acquisition of information in complex socio-ecological coastal systems. These models should work in an adaptive way by gradually improving with the incorporation of new knowledge, including data on small-scale ecological configurations linked to the provision of services and, subsequently, identifying the value of those services to society. In this work, we explored priorities for management and conservation efforts by identifying optimal zoning on the Chilean coast to maximize a set of ecosystem services. Through the exploration of different scenarios, the optimization of management and conservation efforts considers the consequences of moving beyond the traditional target-species oriented approach (currently prevailing in the study area), to an integrated approach that incorporates ecosystem components and the benefits these provide to society. A novelty of this optimization model is the incorporation of the biophysical connectivity, as it is crucial for structuring marine communities, and conditions the gamete supply of target species in a site. Our spatial optimization exercise is not designed to produce recommendations regarding optimal reserve design in central Chile; this would require care-
ful consideration of potential uncertainties in our data input. However, by gathering high quality data, including habitat representativeness and temporal dimensions, outputs could guide the establishment of new MPAs and the allocation of economic resources to those areas relevant for the provision of ecosystem services, including the strengthening of monitoring and enforcement efforts. Additionally, by incorporating ecological connectivity in the spatial prioritization of marine resources, the impact of management activities would propagate beyond the region through the biophysical processes that support the provision of ecosystem services in coastal areas. Hence, we urge the compilation of spatially-explicit databases at regional scales to obtain more precise inferences on the processes that sustain resilient socio-ecological systems. Notwithstanding these data limitations, the spatial optimization approach introduced in this work is a highly novel approach that integrates multi-disciplinary information at multi-dimensional scales to explore optimal configurations of restricted areas along the coast that, ultimately, contribute to ecosystem conservation and social well-being.

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