

Chapter 18

Common Oversights in the Design and Monitoring of Ecosystem-Based Management Plans and the Siting of Marine Protected Areas



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Introduction

Nearly 200 million tons of fish, invertebrate and macroalgae are extracted every year from wild or cultured populations at sea, which is considered essential for human food security and health (FAO 2020), and it is expected to increase in importance for human life in the next decades (Naylor et al. 2021). There is no doubt that the scale of these extractive activities is large enough to be causing negative impacts on marine life. This pressure, coupled with the stress caused by a changing climate, is pushing many ecosystems and their services to the brink of collapse. This is certainly the case for the southeastern Pacific. Since the 1960s, the industrial and coastal fisheries along the nutrient-rich waters of the Humboldt Upwelling Ecosystem (HUE) and the smaller Galápagos upwelling have provided a sizable fraction of the world landings of fish biomass, but many of the fished resources

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show signs of over-exploitation or outright collapse (Salas et al. 2011; IFOP 2014; Vinueza et al. 2014). As a result, there is an urgent need for scientists to provide guidelines to preserve ecosystem services, foster sustainability and restore exploited populations.

Unfortunately, the scientific information to achieve this grand goal will never be sufficiently long, extensive or complete to provide a definitive ‘how-to’ user manual for managers and conservation agencies. Facing the impossibility of scientifically assessing coastal fisheries using traditional fishery protocols and recognizing the inadequacies of the ‘resource-focused’ approaches, which disregard the wider effects of fisheries, especially the human dimensions of sustainability, scientists and practitioners have in the last decades turned to more integrated and holistic approaches. In this context, the implementation of Marine Protected Areas (MPAs) and the development of Ecosystem Based Management (EBM) programs have been largely endorsed by the scientific community, and are being prioritized around the world (Charles 2001; Douvere and Ehler 2009), and in the Galápagos Islands in particular (Vinueza et al. 2014; Castrejón et al. 2014; Walsh and Mena 2016), where artisanal fisheries exert strong pressure on coastal resources (Fig. 18.1).

It is clear that there will be no sustainability of coastal ecosystems and their services without an effective and explicit inclusion of humans and their social structures as the main players within managed ecosystems. Yet, resource governance structures, whether polycentric or monocentric, even when well-matched to social systems (Cáceres et al. 2022), do not guarantee fishery sustainability, and may still lead to fishery degradation if they don’t consider the scales and spatial dependencies

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Fig. 18.1 The fish market on the main street of Santa Cruz Island open to tourists and locals

of metapopulation dynamics. Indeed, a rather stationary approach has dominated thinking when it comes to spatial management and the siting or spatial design of no-take MPAs. Below, we illustrate why the consideration of metapopulation principles is critical if we are to sustainably manage exploited ecosystems using spatial strategies. We highlight the advances in ecological and oceanographic sciences that make possible to realistically model connectivity and provide an example of why they should be implemented in the Galapagos Marine Reserve (GMR).

It is also clear that, if the fabric of nature that maintains fish biomass is slowly degrading, failure to consider the wider effects of fisheries on the ecological web, including non-exploited species, could mean that apparent long-term sustainability of even 'well-managed' fisheries is only a short-term illusion (Travis et al. 2014). Coastal spatial management and integrated coastal management, which are considered as fundamental to Ecosystem Based Management programs (Douvere and Ehler 2009), are conceived as a response to overcome the shortcomings of mono-specific and resource-focused-management. Consequently, well-designed monitoring programs, including baseline information and time series, are essential and integral to any EBM programs (Edgar et al. 2004; Day 2008). Many have criticized varied aspects of EBM approaches (see Murawski et al. 2007, and references therein). Our goal here is not provide another criticism to EBM or MPAs but to note that the state of knowledge and capabilities in ecological and oceanographic sciences today, make it possible to overcome some of the shortcomings.

Including Connectivity in Zoning, Siting of MPAs and Assessment of Performance of EBM Programs

Spatial management approaches and spatially integrated coastal management – the foundations of the EBM approach – focus on managing the multiple human uses of designated areas or ‘zones’ that are assigned to varied, overlapping or non-overlapping uses. Since the designation of no-take MPAs is another spatial management strategy to improve conservation and increase resilience to fisheries, it has become part of most if not all EBM programs. The approach attempts to best match and find best tradeoffs solutions to stakeholders’ spatial priorities and interests. For varied reasons, from the point of view of the species subjected to fisheries the spatial zoning approach has been static, rarely considering the inherent underlying dynamics of spatially structured populations imposed by dispersal in an advective environment.

The absence of connectivity principles in spatial management plans may owe in part to lack of suitable information, but also to unawareness of frameworks that allow metapopulation dynamics to be reconciled with habitat suitability. Advances over the last decade suggest that this oversight may now be overcome. Firstly, non-equilibrium metapopulation theory, which focuses on timescales that approach management and conservation goals, has been well-developed in the last decade and can provide useful guidelines even under limited metapopulation information (Aiken et al. 2007; Aiken and Navarrete 2014; Aiken and Navarrete 2020; Aiken and Navarrete 2011; Williams and Hastings 2013; Mari et al. 2017; Farrell and Ioannou 1996). This body of knowledge comes to complement the more traditional, stability approaches to metapopulations which have been reviewed in the context of spatial management and marine protected areas (Hastings 2014; Botsford et al. 2001; Gaines et al. 2010). Aiken et al. (2023) provide an example of how principles of population persistence and growth in reactive systems, e.g. those exhibiting transient dynamics at timescales matching scales of management, can be applied to guide restoration of seagrass *Zoostera mulleri* in Port Curtis, Australia, even with incomplete information about dispersal.

Secondly, our capacity to realistically simulate larval dispersal has improved considerably. Numerical 3D models of the coastal circulation can be run at high enough resolution to capture important meso-scale and sub-mesoscale oceanographic processes, including the complex coastal boundary layer (Capet et al. 2008; Dauhajre et al. 2019), sourcing realistic boundary conditions from global simulations that assimilate ever more complete observations of the ocean state, over spatial extents large enough to encompass larval dispersal. In addition, a dense network of Lagrangian drifting buoys provide a unique empirical insight into ocean dispersal (Álvarez-Noriega et al. 2020; Jönsson and Watson 2016; Aiken and Navarrete 2020).

But why should we include connectivity principles in spatial management and conservation? Is it not enough to secure governance in the exploitation of resources and that EBM destination zoning or MPA’s are being accepted and complied by stakeholders? Metapopulation theory for spatially structured populations has shown

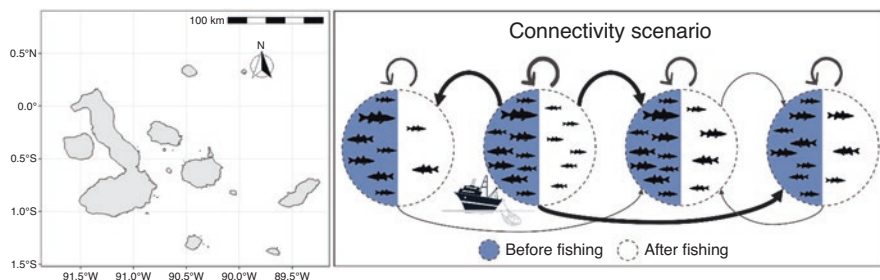


Fig. 18.2 Schematic representation of connectivity and spatial management that includes zones where fishing is allowed and zones where fishing is prohibited (no-take MPAs). The scheme could represent the Galapagos Marine Reserve Management Plan for the Galapagos Marine Reserve presented in the map. In the example, arrows and their thickness represent direction and strength of dispersal by propagules among the four different zones or local populations. Initially, before fishing (blue section of the local populations), fish biomass is high everywhere and especially in the zone where fishing is allowed. After fishing (white section of the local populations) and as expected, fish biomass, and fish size, are significantly reduced in the fished zone. But fish biomass is also reduced in protected areas (MPAs) because their productivity or population replenishment depends, to varying degrees, on the influx of new propagules from the fished population. In the example, fish biomass in the first zone on the left is drastically reduced because of its dependence on in fluxes from the fished zone upstream, even though fish size in that MPA may not have been altered

the importance of the character of the dispersal process on population dynamics, coexistence, and persistence. The general principle being that mid-term persistence of spatially-structured populations, under natural conditions or when subjected to anthropically imposed mortality (e.g. fisheries, Fig. 18.1), depends on local dynamics and connectivity (Hastings 2014). Figure 18.2 illustrates a hypothetical, but realistic scenario of fishing and protected zones connected by dispersal. Four findings, well founded in theory suffice to demonstrate why basic principles of connectivity must be considered when planning destination zones, siting no-take MPAs, or assessing the performance of these spatial measures.

Local population productivity, *i.e.*, the replenishment of benthic populations within any EBM or MPA status, depends on the spatial structure of dispersal and emerging patterns of connectivity. Over most scales of zoning and no-take MPAs, complete retention and self-recruitment cannot be assumed (Lett et al. 2015) and therefore, 'effective reproduction', *i.e.* recruitment of dispersing propagules to local populations within the zone, depends from upstream populations. Using local knowledge of high historical productivity of exploited species at designated sites is an important piece of information, but it only indicates that propagules arrive at high rates at a given site, not where they come from (Burgess et al. 2014; Lett et al. 2015).

The resilience of populations to fishing mortality in fishing access areas depends on the patterns of connectivity and dependencies to other distant fishing grounds and to zones under fishing protection. In other words, the same fishing mortality exerted in a population, and even if the same biomass is being removed, can have

widely different effects on local and metapopulation persistence if the area exploited is a source or a sink of propagules, and how the footprint of that site is in the entire metapopulation (Aiken et al. 2023; Aiken and Navarrete 2020). Some fishing grounds may be having much larger effects in the metapopulation than others.

Mid-term persistence of populations in no-take protected areas largely depends on the spatial pattern of sources and sinks (Roughgarden and Iwasa 1986; Salomon et al. 2010; Botsford et al. 2001). In other words, a well-protected and enforced no-take MPA may see populations of some species vanish if propagule sources outside the protection are reduced (Aiken et al. 2011; Dedrick et al. 2021).

Entire metapopulation persistence (extinction), especially under intensive fishing mortality, can be highly reliant on a few sites or localities that are still sustaining transient population growth (Aiken et al. 2023; Mari et al. 2017). Those sites can be very distant, beyond the areas our management plans attempt to protect (Dedrick et al. 2021). Protecting those sites, and not necessarily the apparent high-productivity areas within islands, must be a priority.

Thus, establishing the EBM zoning and siting no-take areas based solely on patterns of abundance, or habitat suitability, or tradeoffs among stakeholders' needs and goals, and ignoring recruit fluxes between populations, may result in less-than-expected benefits of the management plan, even if correctly enforced (Fig. 18.2). Efforts should thus be made to incorporate some of these basic principles when assessing the existing Galápagos Marine Reserve Management Plan (GMRMP) and when siting small no-take MPAs along the coast.

Why the Galápagos Islands Is an Ideal System and Why We Should also Look at Continental Populations?

As identified by theorists such as Ilka Hanski, islands provide ideal testing grounds for metapopulation theory, as a highly punctuated connectivity can accentuate processes that are difficult to observe when dispersal barriers are more diffuse. Although the shallow and intertidal habitats of the Galápagos are distant some 800 km from the south American west coast, some degree of connectivity between the two is likely for species with pelagic larval durations of weeks. Importantly, this connectivity is predominantly unidirectional. Figure 18.3 illustrates the probability of passive drifting from and to the Galápagos, derived from 45 years of data from the Global Drifter Program (Elipot et al. 2022) following the method of Aiken and Navarrete (2020). As such, the Galápagos Islands likely receive recruits from the mainland via the swift and constant south equatorial current, but the return journey is far less likely to occur (Fig. 18.3). This strongly biased dispersal can have profound consequences for persistence and coexistence in the Galápagos that must be considered within management strategies (Aiken and Navarrete 2011, 2014, 2020; Aiken et al. 2023). As a consequence of strongly biased westwards advection in the equatorial Pacific, a species with a planktonic larval dispersal time on the order of

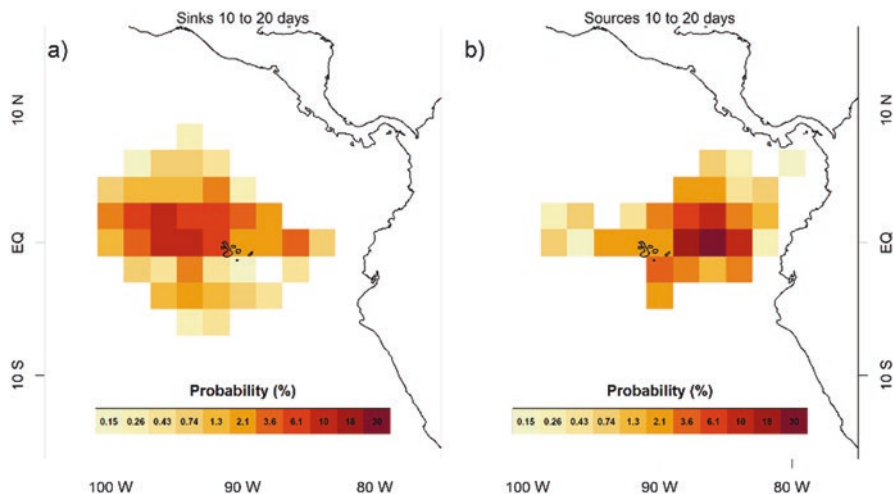


Fig. 18.3 Probability of drifting (a) from the Galapagos Islands and (b) to the Galapagos Islands, from the continent, with the period of 10–20 days that simulates larval pelagic development times for many invertebrate and fish species. Probabilities are based on 45 years of data from the Global Drifter Program data, using the method presented in Aiken and Navarrete (2020)

multiple weeks may exhibit weak self-recruitment, and potentially depend upon subsidization by continental populations. This situation determines the metapopulation dynamics, with the result that large population fluctuations that are unrelated to local conditions can occur simply due to variations in larval supply from remote source populations. Clearly in such a case persistence of the species in the Galápagos depends, to some extent, on the level of protection afforded the continental source populations.

The reactive dynamics of long dispersers in the Galápagos has additional consequences for coexistence, *i.e.*, local diversity of species. Taking a pair of species as example, if one species were the dominant when competing for a local resource, but the subordinate had a shorter pelagic larval duration and hence more stable recruitment, “reactive coexistence” would be possible, whereby dispersal driven fluctuations in recruit supply are sufficient to allow persistence of the subordinate when the dominant is subject to biased dispersal. The subordinate survives by occupying a “dispersal niche” (Aiken and Navarrete 2014), being the inferior competitor in terms of using habitat resources, but using a dispersal strategy that is more robust. Again, co-oscillation in the populations of these two competitors, and even local extinction of one or the other, could be driven by variations in the recruit supply of the dominant alone, driven by either larval production rates in the continental source populations, the protected areas within the Galápagos archipelago, or an unfavorable ocean circulation.

The Importance of Species Interactions in Monitoring and Assessing EBM and MPAs

The implementation of EBM programs to coastal habitats has been difficult to say the least. Probably the best example of a successful coastal EBM program, although not exempt of pitfalls and limitations, comes from the Great Barrier Reef (Day 2008). This program, however, manages an oligotrophic, tropical marine ecosystem adjacent to a relatively low density and wealthy human population, and as such is not necessarily suitable to be applied in other contexts. The implementation of the Galápagos Marine Reserve Management Plan (GMRMP) provides a more realistic example for implementing EBM and no-take MPA's in highly productive coastal fisheries, such as the ones in upwelling ecosystems. Castrejón and Charles (2013), Castrejón et al. (2014), Reck (2014) and Edgar et al. (2004) provide excellent accounts of the history, definition process, problems and tradeoffs in the implementation of EBM in the Galápagos Marine Park. Several assessments of the performance of the GMRMP have been conducted, mostly on the co-management system, structure of socio-economic systems, governance (Cáceres et al. 2022; Mestanza-Ramón et al. 2019; Heylings and Bravo 2007; Castrejón and Charles 2013) and, to a lesser extent, the biodiversity components (Edgar et al. 2008).

Adaptive monitoring is an essential step of any EBS and coastal spatial management program. Selected monitoring targets typically include the fished species subjected to the spatial management, integrated or composite community variables (e.g. species richness), as well as 'sentinel' species that may indicate or integrate somehow the state of the ecosystem (e.g. marine iguanas, sharks, (Vinueza et al. 2014; Castrejón and Charles 2013; Edgar et al. 2008)). However, we argue that without the aid of a model that can help managers anticipate which species in the ecosystem will respond to management policy, monitoring programs will misrepresent or entirely miss the effects of the management plan and fail to attribute causes of biodiversity change to the implemented policy. Sentinel or abundant species may be useful to monitor for many reasons (e.g. indicators of climate change, pollution, etc.), but may not be the species that respond to a given fishery management policy. Monitoring all biodiversity of species that interact in a marine ecosystem and that transform biomass to finally sustain fisheries, is virtually impossible too. All standardized quantitative assessments of biodiversity (e.g. subtidal visual transects, quadrats, destructive samples, eDNA) are systematically biased towards or against small, rare, cryptic, large, highly mobile, infrequent, etc. species. Thus, efforts to assess the diversity of co-occurring species that can be affected by management are colossal and unsustainable over time.

Indeed, propagation of management-induced alterations through the ecological web cannot be assumed to be linear, proportional or even in the same direction to the magnitude of the alteration. The problem of propagation of species interactions was brightly captured in Yodzis's 'indeterminacy of ecological interactions' over 30 years ago (Yodzis 1988). Basically, alteration of one species biomass will have positive, negative or negligible effects on the abundance and persistence of other species that are not directly connected to the target one, potentially driving them to

extinction or to a pest status (*e.g.* release from top-down control). Therefore, identifying which species in the community will be positively or negatively affected by removing biomass (fishing), is not possible without consideration of the type and intensity of species interactions.

Attributing Biodiversity Change to Management

For coastal regions with abundant ecological information, such as the Galápagos Marine Reserve (Riofrío-Lazo et al. 2021), models of intermediate to high complexity can be constructed to represent the ecological web with sufficient realism to simulate scenarios for different fisheries impacts (Ávila-Thieme et al. 2021; Riofrío-Lazo et al. 2021) and different levels of compliance to set policies (Navarrete et al. *in press*). Diverse multi-species or multi-component modeling strategies exist (Yodzis 2001; Pauly et al. 2000). Riofrío-Lazo et al. (2021) have implemented an Ecopath with Ecosim (EwE, Pauly et al. 2000) that could be the basis for the type of quantitative-qualitative modeling illustrated in Fig. 18.4. However, the allometric trophic web models (ATN), based on bioenergetic

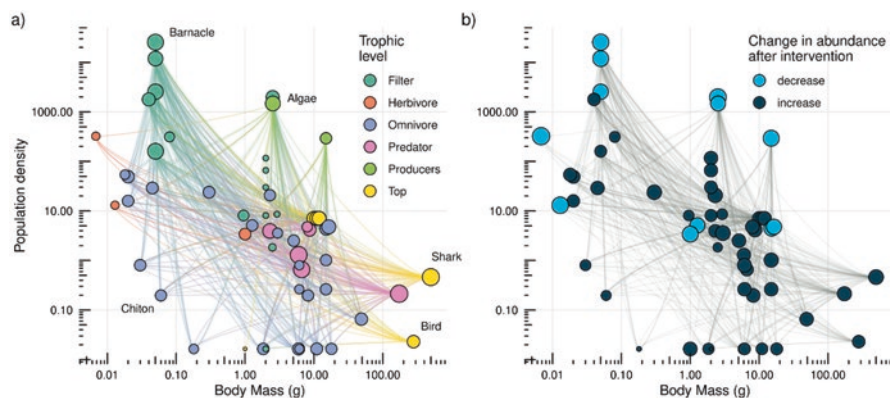


Fig. 18.4 (a) Schematic representation of an ecological web representing consumption type of interactions (food web) among 60 species (nodes) from all trophic levels, indicated by the different color nodes. Names of some nodes are indicated for reference. The links representing trophic interactions among nodes are taken from the rocky shore intertidal food web of central Chile (Kéfi et al. 2012) and therefore represent a realistically complex ecological system. The nodes and links are ordered in the Abundance (Population density) and Body Mass plane to show how small-bodied and moderate abundance species can participate in the fabric of nature and be affected by fisheries and management policies. Many of these species are not included in standardized monitoring programs. (b) Shows the change in abundance of species that takes place after a single species is fished in the system. Both, significantly positive (blue nodes) and significantly negative (black nodes) changes are observed with respect to the unfished system, and they appear in different sectors of the Abundance-Body Size spectrum. These nodes are ‘interaction-indicator’ species that should be included in spatial management and Ecosystem Based Management monitoring programs, which allows managers to attribute biodiversity changes to policy

equations for biomass transfer, present some advantages over EwW because part of the parameterization can be done from basic allometric principles (Martinez 2020; Brose 2010). The more mechanistic basis for this modeling framework is also more amenable to the inclusion of non-trophic interactions (Kéfi et al. 2012; Kefi et al. 2016), such as habitat provisioning that can be critically important for benthic coastal ecosystems.

Although quantitative results from multi-species models are highly dependent on parameters, it can be shown that qualitative outcomes to identify “interaction-indicator” species can be quite insensitive to a range of parameter values. Navarrete et al. (in press) showed that moderately complex models of species interactions can be used to link management policies and levels of compliance with set regulations, with biodiversity monitoring programs, through identifying which species most likely respond to the alteration of the biomass of fishery target species. Since the pathways of propagation of biomass and information are mostly defined by the topology of the interaction web, the identities of the species being affected by harvesting are relatively insensitive to parameter values (Fig. 18.4).

Concluding Remarks

Providing useful and effective guidelines to improve the sustainability of exploited ecosystems and conserving biodiversity, grounded on solid scientific findings, is a great challenge. Here, we illustrate how the consideration of basic principles of dispersal and connectivity in spatial planning and EBM, as well as multi-species models for assessing ecosystem-wide consequences of management policies, can be integrated in ecosystem and spatial management programs. This is especially relevant in areas where non-compliance with regulations is compounded by the gross shortage of resources and infrastructure of management agencies, such as most of South America and GMR productive upwelling ecosystems.

In Galápagos and Chile, high levels of non-compliance are rampant in most fisheries (Castrejón and Charles 2013; Fernández et al. 2020; Oyanedel et al. 2018, 2020). Fishers must effectively participate in the conservation and MPA siting process to reduce non-compliance. This requires maximizing the returns per unit area (stock fraction) that is set aside, and demonstrating that the loss of fishable biomass provides longer-term benefits. This can be aided using transient metapopulation theory and improved circulation models as illustrated here.

Decision makers are usually pushed to protect zones and siting no-take MPAs in areas that have less or no value for fishers and other stakeholders (Edgar et al. 2004) rather than areas where existing biological information would advise. They usually assess the effectiveness of a zoning system as if the sole creation of a protected area, anywhere in a metapopulation, was sufficient to improve fishery resilience and productivity. Many empirical studies suggest that the effectiveness of MPAs, even

within their limits, can depend on several factors (Edgar et al. 2014). As discussed above, the effect on adjacent fishing areas and population persistence strictly depend on connectivity and this must be considered when assessing effectiveness of spatial management.

Precisely because a spatial management plan results from trade-offs among multiple needs and objectives (Douvere and Ehler 2009; Castrejón and Charles 2013), adaptive and carefully designed monitoring is essential for its success from the perspective of exploited marine populations. All-encompassing biodiversity monitoring is unfeasible in all real ecosystems and economically unsustainable over time. Multi-species dynamic models can be used to guide and focus monitoring programs to those ‘interaction-indicator’ species that most likely will respond to management policies and to lack of compliance with those regulations.

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