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CHAPTER SIX

Marine Protected Area Networks in California, USA

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Abstract

California responded to concerns about overfishing in the 1990s by implementing a network of marine protected areas (MPAs) through two science-based decision-making processes. The first process focused on the Channel Islands, and the second addressed California's entire coastline, pursuant to the state's Marine Life Protection Act (MLPA). We review the interaction between science and policy in both processes, and lessons

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learned. For the Channel Islands, scientists controversially recommended setting aside 30–50% of coastline to protect marine ecosystems. For the MLPA, MPAs were intended to be ecologically connected in a network, so design guidelines included minimum size and maximum spacing of MPAs (based roughly on fish movement rates), an approach that also implicitly specified a minimum fraction of the coastline to be protected. As MPA science developed during the California processes, spatial population models were constructed to quantify how MPAs were affected by adult fish movement and larval dispersal, i.e., how population persistence within MPA networks depended on fishing outside the MPAs, and how fishery yields could either increase or decrease with MPA implementation, depending on fishery management. These newer quantitative methods added to, but did not supplant, the initial rule-of-thumb guidelines. In the future, similar spatial population models will allow more comprehensive evaluation of the integrated effects of MPAs and conventional fisheries management. By 2011, California had implemented 132 MPAs covering more than 15% of its coastline, and now stands on the threshold of the most challenging step in this effort: monitoring and adaptive management to ensure ecosystem sustainability.

Keywords: California, MPA, Channel Islands, Population models, Science, Process, Planning

1. INTRODUCTION

California responded to rising global concerns regarding the effects of overfishing on marine ecosystems in the 1990s by implementing a network of marine protected areas (MPAs). Here, we describe that effort in terms of the ecological setting, the initial concerns, the enabling legislation, the planning process, and the concurrent development of the science of MPAs. We synthesize the various kinds of success achieved, the challenges in the process, and the potential for the future. Our intent is to provide an example for other future MPA processes of how science interacted with the legal, social, ecological, and economic aspects throughout the implementation process based on our experiences as scientists involved in this process. We base our exposition on the relevant scientific data, as well as on the century-long history of the science of marine resource management. In particular, we take note of the scientific developments taking place over the lifetime of the implementation process in California and how the structure of that process influenced the degree to which science informed the MPA design. The California process was groundbreaking in many ways, not least of which was the goal of developing a functional *network* of ecologically connected MPAs, as opposed to a collection of multiple MPAs designed independently of one another. As such our summary of the process pays particular attention to the science of MPA network design.

1.1. Physical and biological context

The marine environment of California is defined by the contrast between the warm-temperate/subtropical southern region (from the Mexican border to Point Conception, with biota derived from the San Diegan biogeographical province) and the cold-temperate northern region (north of Point Conception, with biota belonging to the Oregonian region; Horn et al., 2006; Figure 6.1A). The northern region is heavily influenced by the equatorward-flowing California Current, a highly productive Eastern Boundary Current. High productivity is driven by spring upwelling winds, which are more prominent to the north of Point Conception (Checkley and

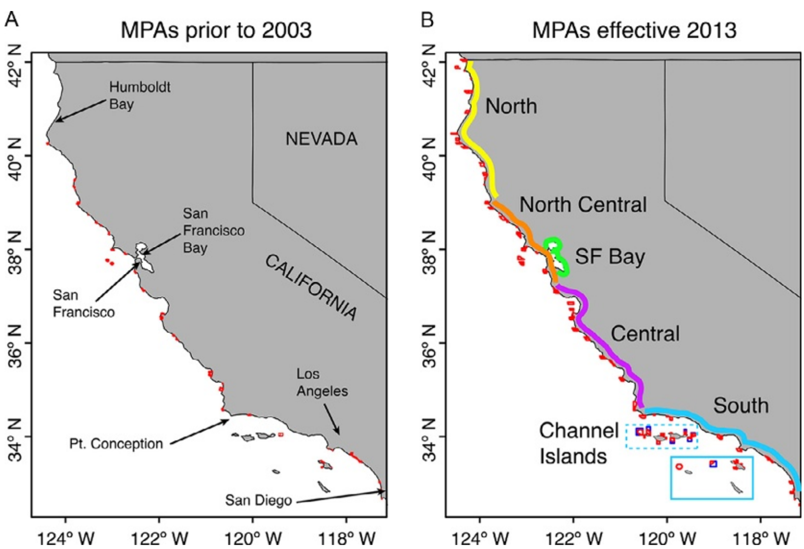

Figure 6.1 Map showing MPAs in California prior to 2003 (A) and those in place as of 2013 (B). MPAs administered by the state of California are red (black in the print version), and MPAs in U.S. territorial waters that are administered federally are outlined in blue (light grey in the print version). Estuarine MPAs are not shown. Coloured highlighting in (B) indicates the different Study Regions utilized in the design process that implemented the Marine Life Protection Act (MLPA). The dashed blue line (dashed light grey line in the print version) indicates the Channel Islands; MPAs were designed for those islands as part of a design process separate from the MLPA, and were not modified during the MLPA process for the South Coast region. The Channel Islands MPAs became effective in 2003, and MPAs designed under the Marine Life Protection Act (MLPA) became effective region-by-region between 2007 and 2013.

Barth, 2009). These winds vary from year to year, and productivity is also modulated on longer time scales by atmospheric and oceanographic conditions throughout the eastern North Pacific (Di Lorenzo et al., 2013). The continental shelf is narrower north of Point Conception and the shallow (<30 m depth) seafloor is primarily rocky reefs with kelp forests (comprises *Macrocystis pyrifera* and *Nereocystis leutkeana*) subject to frequent disturbance (Carr and Reed, 2015; Graham et al., 2008; Reed et al., 2011). In contrast, the California Current remains further offshore south of Point Conception, and nearshore surface waters are influenced more by warmer recirculating flow from the California Current and the poleward-flowing Davidson Current (the latter also extends northward past San Francisco Bay as a surface current in the winter; Hickey, 1998; Bray et al., 1999). In addition to warmer water, the southern region has a shallow, broad continental shelf and several large offshore islands and supports more persistent kelp (only *M. pyrifera*) forests than are found in the north (Carr and Reed, 2015; Graham et al., 2008; Reed et al., 2011). There are key ecological differences among the islands and the mainland (Ebeling et al., 1980). In particular, the northwestern most Channel Islands (San Miguel, Santa Rosa, and San Nicolas Islands) lie at the boundary between the bioregions, with cooler waters, more frequent disturbances, and a mix of San Diegan and Oregonian species (Hamilton et al., 2010; Pondella et al., 2005). Further south and east, the islands experience warmer waters and less frequent disturbances. The mainland coast south of Point Conception is more heavily influenced by human activities (e.g. ports, hardened coastlines, intake and discharge of power plants, recreational fishing, and urban runoff from the Los Angeles and San Diego metropolitan areas). In general, mainland south coasts are sandy with interspersed low relief rocky reefs, whereas the offshore islands contain primarily high relief rocky habitat and less turbid water (Pondella et al., 2005).

California has a Mediterranean climate, with wet winters and dry summers. Freshwater flow into the ocean is greater in the north, with several rivers forming large estuaries (e.g. San Francisco Bay, Humboldt Bay), although river damming has reduced both overall river outflow and variability during the twentieth century (Hanak et al., 2011; Hundley, 2001).

1.2. Context: History of fisheries management and conservation in California

The move to MPAs in California was influenced by the state of marine resource management from local to global levels in the late 1990s. Globally, there was growing concern for the high fraction of global fisheries that were reported overfished (e.g. Botsford et al., 1997; Hutchings, 2000; more recently reviewed by Worm et al., 2009). Reports of this fraction ranged from about 20% to more than 60%, depending on whether fully exploited fisheries were included in the overfished category (Mace, 2001). A second, related growing global concern was that marine resource management was falling short because of its focus on single species, ignoring the more extensive *ecosystem* effects of fishing (e.g. Botsford et al., 1997; Pikitch et al., 2004). The proposed solution was a more holistic, *ecosystem-based* approach that included the effects of (a) interactions among multiple species, (b) incidental take of nontarget species, (c) impacts on essential fish habitat, (d) the changing physical environment, and (e) the socioeconomic consequences of ecosystem status and marine ecosystem services. MPAs were considered to be an ecosystem-based management tool because they can protect both the physical (geomorphological, water quality) and biotic components of ecosystems from fishing and other anthropogenic impacts (Murray et al., 1999).

The local context was influenced by historical events of the previous several decades, as far back as the dramatic collapse of the California fishery for Pacific sardine (*Sardinops sagax*) around 1950 (Ueber and MacCall, 2005). A second fishery collapse occurred later in that decade with the decline of the central California Dungeness crab (*Metacarcinus magister*) fishery in 1958 (Botsford, 1981; Wild and Tasto, 1983). Other management crises followed in subsequent decades. In the 1980s, scientists and managers became aware that the history of California's abalone (*Haliotis* spp.) fishery was a prime example of serial depletion, leading to the near extinction of several species (Karpov et al., 2000). The many rockfish (*Sebastes* spp., Scorpaenidae) species off the California coast, ranging from nearshore reefs to the continental slope, went from being a concern as an under-utilized resource in the 1970s to having several species declared overfished in the 1990s (Love et al., 1998; Ralston, 1998). Interspersed among these declines was the rapid development in the 1980s of a fishery for the red sea urchin (*Strongylocentrotus franciscanus*) in northern California followed by a dramatic decline in catch (Botsford et al., 2004) as well as large increases in live-finfish fisheries (CDFG, 2002; Starr et al., 2002). Leet et al. (2001) provide a comprehensive survey of the status of California marine resources at that time.

This awareness of the vulnerability of California's marine resources set the context for improved management. It was coupled with an increasing conservation sentiment among California citizens, initiated in part by the effects of an oil spill in 1969 in the Santa Barbara region. These sentiments operated in the economic context of California's diverse modern economy (at least the eighth largest in the world since 1970s), with dominant entertainment, information technology, tourism and agricultural sectors, in contrast to a relatively small commercial fishing sector, and an economically more significant recreational fishing sector (Kildow and Colgan, 2005). Prior to the recent new MPAs (the subject of this chapter), there were only scattered, small, single-purpose MPAs in the state (McArdle, 1997; Figure 6.1A), accompanied by areas of excluded public use near military bases.

California fisheries are managed either by (a) the state of California (for species occurring only out to 3 nautical miles (nm; 5.56 km) offshore, the boundary of state waters within the United States), (b) the regional council of the federal management system, the Pacific Fishery Management Council (for species occurring from 3 to 200 nm, the U.S. territorial waters within the Exclusive Economic Zone), or (c) jointly by state and federal authorities.

1.3. Context: The state of fisheries and conservation science

By the late 1990s, the science of fisheries management around the globe had developed from concerns over declines in fishery catch in the early part of the twentieth century, to a standard procedure of calculating maximum sustainable yield (MSY) for a number of fisheries beginning in the 1950s, on to a gradual realization that simply seeking MSY would not be sufficient (Botsford, 2013). Concerns over the ineffectiveness of a simple MSY approach began to arise in the 1970s (Larkin, 1977), which ultimately led to development of a precautionary approach to fishery management in the early 1990s (FAO, 1996; Garcia, 1996; Mangel et al., 1996). The precautionary approach emphasized frequent observation of fisheries (e.g. biomass, age structure or catch), and comparison of these to reference points (i.e. predetermined values of those variables), with consequent responses by management, such as changes in allowable catch. These reference points included target reference points, which were essentially management goals similar to the earlier maximization of yield, and limit reference points, which were intended as critical limits to guard directly against overfishing and population collapse. Federal fisheries management in the U.S. operated under the Fisheries Conservation and Management Act (1976), which included specific attention to the potential for overfishing in its 1996 reauthorization as the Magnuson–Stevens Fisheries Conservation and Management Act (Restrepo and Powers, 1999; Rosenberg et al., 1994).

Parallel to the development of the reference point concept, a better understanding of the critical features of fish population dynamics emerged and largely supplanted the earlier approaches (e.g. logistic models, surplus production models) originally used to develop the MSY concept (Botsford, 2013). This new understanding centred on the realization that the key to persistence in marine populations is the maintenance of sufficient lifetime spawning to allow each adult to replace itself with a new recruit within its lifetime (i.e. remaining above a critical replacement threshold). Initial comparisons to empirical information on population collapses suggested that preserving 35% of unfished lifetime spawning would be a safe hedge against collapse (Clark, 1991; Mace and Sissenwine, 1993). Unfortunately, this 35% replacement level was too low for Pacific coast rockfishes, leading to overfishing (Clark, 2002; Ralston, 2002), and management has subsequently used more conservative replacement limits. For many fisheries, this limit is 40%. If the fishing mortality rate is high enough to cause lifetime reproduction to fall below the critical replacement limit (e.g. 35% or 40%) in the United States, the stock is declared to be undergoing overfishing. If the spawning stock biomass falls below a certain fraction of the unfished biomass (usually 40%), the stock is also declared to be overfished (Restrepo et al., 1998).

By the late 1990s, the federal fisheries management process in the United States had evolved to its current form (Fluharty, 2000). It generally involves a decision-making process in regional councils (e.g. http://www.pcouncil.com/), based on stock assessments involving population models fit to fishery data and fishery independent data, to determine periodically (annually in many cases) the amount of catch that should be taken. The stock assessments and technical aspects of decisions made by these councils are reviewed by a group of scientists called the 'Scientific and Statistical Committee'.

As the science of fishery management was maturing, conservation advocates and some fisheries biologists began to argue that fisheries could be managed more cautiously, and ecosystems could be better protected by reducing fishing effort to zero in designated protected areas, rather than attempting to control the overall level of fishing (Murray et al., 1999). These recommendations called for single protected areas, as well as 'networks' of protected areas; collections of protected areas linked by larval dispersal that replenish one another and the fished populations between them. There was also a growing realization among scientists that a decision-making process for management by MPAs would require new scientific understanding to predict their benefits and costs. For the most part, the models being used in conventional fisheries management did not consider how populations varied over space; they were concerned with temporal variability only. To manage populations using networks of MPAs, there would be a need to know (1) how many MPAs are required, how large they should be, and where they should be placed to ensure the persistence of multiple species and (2) how does fishery yield in management by MPAs compare to yield with conventional control of effort? These questions were only beginning to be addressed when the decision-making process for California's MPAs began in the late 1990s.

The effort to develop the science of marine reserve design and assessment was kick-started by a scientific working group at the National Center for Ecological Analysis and Synthesis (NCEAS) in Santa Barbara in 1998, and many of the seminal papers on the topic emerged from that group (Lubchenco et al., 2003 and references therein). With regard to the first questions (how many, how large, and where?), earlier population models had suggested that it was best to place an MPA in a 'source' location (e.g. an upstream reef in an archipelago) so that planktonic larvae spawned inside the MPA could seed populations in other patches (e.g. Crowder et al., 2000; but see Gaines et al., 2003; Hastings and Botsford, 2006 for potential drawbacks to this approach). Botsford et al. (2001) approached the question from a perspective more relevant to the California coast: a long, linear coastline with a network of evenly spaced MPAs, and relatively sedentary fish or invertebrate species that disperse widely as larvae. Analysis of their simple, strategic model (as opposed to a more detailed 'tactical' model of a specific location) showed that populations could persist in one of two ways: (1) in single MPAs that were at least as wide as the average dispersal distance of larvae (termed *self-persistence*) or (2) in a network of smaller MPAs covering an adequate fraction of the coastline. This mode of persistence was termed *network persistence* because even when individual MPAs within the network are too small to sustain themselves independently, larval connectivity among them allows the population distributed across the entire network to be sustained (White et al., 2010a). The minimum fraction of the coastline that must be protected to achieve network persistence was determined to be biologically related to the critical replacement threshold described above in a single-population context under conventional, non-spatial fishery management. Under the idealized assumption that fishing removed all reproduction outside MPAs (i.e. the 'scorched earth' assumption), the minimum fraction in MPAs necessary for network persistence would be equal to the critical replacement threshold from non-spatial population dynamics, presumed generally to be equal to 35% or 40%. When the amount of fishing outside the protected areas did not reduce reproduction to zero, the minimum fraction of coastline required for population persistence would be less. Also, the presence of alongshore flows transporting larvae would require higher fractions in reserves (Botsford et al., 2001). Later research would build on these basic results, further examining their sensitivity to such factors as alongshore currents, retention zones, and adult movement (Gaines et al., 2003; Kaplan, 2006; Moffitt et al., 2009; White et al., 2010a), but the central concept has proven highly influential. In particular, the second way of achieving the population persistence requirement was in part the inspiration for the idea that one could formulate general guidelines for the *size and spacing* of MPAs, and the idea that 35% of the coastline must be protected for MPAs to be effective (see Gaines et al., 2010). That percentage has been cited frequently as a theoretical requirement, while in reality the threshold actually would be less with less than scorched earth fishing outside the MPAs, and would depend on the settler–recruit relationship of a particular species, adult movement, and alongshore currents. It is not a general rule (Botsford et al., 2001; Kaplan and Botsford, 2005; Moffitt et al., 2009, 2011; White, 2010; White et al., 2010a).

With regard to the second question of differences in yield between MPAs and conventional management, analyses of simple, strategic models had shown that management by MPAs and conventional management by limiting catch or effort were essentially equivalent in the sense that under particular conditions, the potential yields from each would be equal (Hastings and Botsford, 1999; Mangel, 1998). These results implied that if a fishery were well managed (e.g. at MSY), adding MPAs would diminish yield because fishable area would be diminished (Holland and Brazee, 1996). However, if the fishery were overharvested beyond MPA boundaries, then MPAs could actually enhance fishery yields (Holland and Brazee, 1996; Sladek Nowlis and Roberts, 1999) and the enhancement would be greatest for networks of many small reserves (essentially maximizing the number of boundaries across which fish could spill over; Hastings and Botsford, 2003; Neubert, 2003).

2. ESTABLISHMENT OF MPAs IN CALIFORNIA

With the exception of the few individual MPAs established in *ad hoc* ways over the decades preceding the 1990s, two primary efforts in California led to implementation of science-guided networks of MPAs in California. The first effort was focused on the Channel Islands off southern

California (Figure 6.1), and the second concerned a statewide network of protected areas. We describe these processes here.

2.1. Channel Islands marine protected areas

In 1998, a group of fishermen, managers and other citizens who were concerned about declining fishery resources such as abalone, lobsters, and nearshore rockfishes, approached the California Fish and Game Commission with a proposal to a set aside areas for protection in the northern Channel Islands, bounding the Santa Barbara channel (CDFG, 2003; Osmond et al., 2010; Figures 6.1B and 6.2D). The Channel Islands region is complex from a planning perspective because of overlapping management and political jurisdictions as well as variable environmental and ecological conditions. Eleven federal, state, and local agencies have some jurisdiction in the planning region (Airamé et al., 2003). While both the Channel Islands National Marine Sanctuary (CINMS) and the Channel Islands National Park (CINP) overlap around the northern Channel Islands, neither agency regulates commercial or recreational fishing. The California Department of Fish and Wildlife (CDFW; previously the California Department of Fish and Game, CDFG, prior to 2013) manages all fisheries in state waters (within 3 nm (5.6 km) of shore), while the California Fish and Game Commission (an appointed body) has authority to set all state fishery regulations, including the creation of MPAs.

At the same time, the CINMS was beginning the process of updating its management plan and consideration of marine reserves was included as part of this plan. Rather than address the issue separately, the CDFG and the Channel Islands Sanctuary Advisory Council joined efforts in 1999 to create the Marine Reserves Working Group (MRWG), which included federal and state agencies, commercial and recreational fishermen, environmentalists, and other members of the Santa Barbara community (Bergen and Carr, 2003; CDFG, 2003; Figure 6.3). Additionally, two advisory panels were created to assist the work of the MRWG. A Science Advisory Panel (SAP) was tasked with assembling and evaluating ecological, physical and environmental data and a Socioeconomic Panel was formed to evaluate both recreational and commercial industries in the Channel Islands (Airamé et al., 2003). The MRWG developed several goals for marine reserves in the Channel Islands (Table 6.1; Airamé et al., 2003).

The MRWG, together with professional facilitators and the advisory panels, planned and debated for 3 years. While the MRWG was able to agree

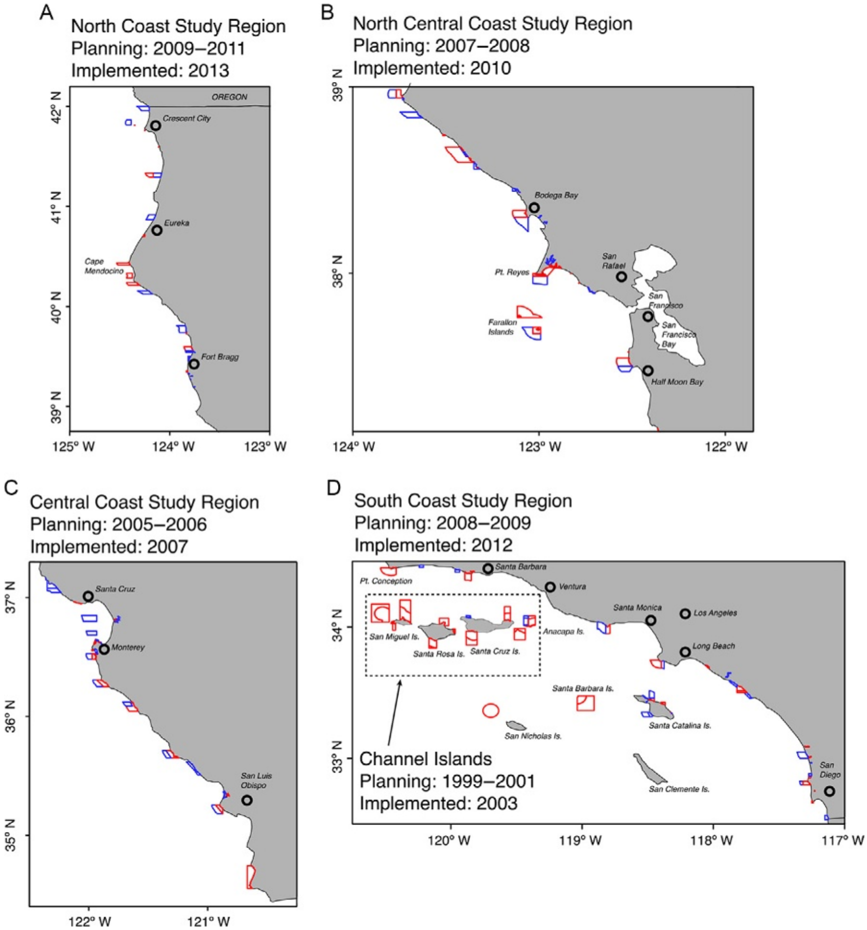

Figure 6.2 Maps of MPAs created in each of the MLPA Study Regions. No-take MPAs (most are State Marine Reserves) are outlined in red (light grey in the print version); limited-take MPAs (most are State Marine Conservation Areas) are outlined in blue (dark grey in the print version). Small dots, particularly in (A), are small special closures surrounding marine mammal haulout locations.

on overarching goals for the MPA network, the group dissolved in 2001 without reaching a consensus on the design of a potential MPA network, essentially ending the public process at that point (Helvey, 2004; Osmond et al., 2010). Following this, the superintendent of the CINMS and the Marine Region Manager of the CDFG developed a compromise solution that reflected the work of the MRWG and the advisory panels. This compromise plan, along with five other plans, was submitted to the

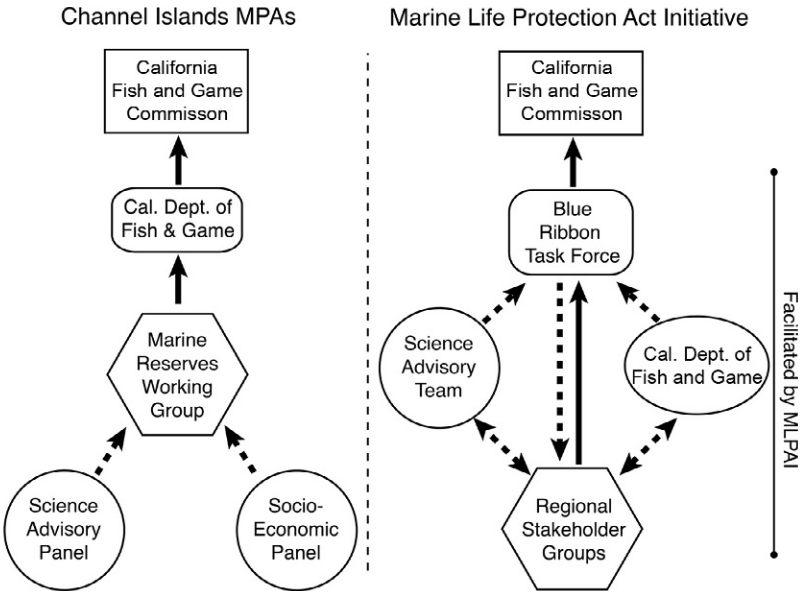

Figure 6.3 Flowchart of MPA design and decision making in the Channel Islands MPA process and the Marine Life Protection Act Initiative. Dashed arrows indicate flows of information (e.g. scientific guidelines); solid arrows indicate flows of MPA network proposals. Circles and ovals enclose groups providing scientific or regulatory guidance; hexagons enclose groups that originated MPA proposals based on guidelines; rounded rectangles enclose intermediate decision-making groups that refined and recommended proposals; rectangles enclose the final decision making and regulatory body.

California Fish and Game Commission. Ultimately, the compromise plan was approved by the Commission in 2002 and a network of MPAs (primarily marine reserves that allowed no commercial or recreational fishing) in state waters was implemented in April 2003. The compromise plan did include reserves which extended into federal waters but since the CINMS had no authority to manage fishing or other activities, formal protection was not extended until a separate, federal regulatory process was completed in 2007 (Osmond et al., 2010).

The science-based guidelines for reserve network design in the Channel Islands are detailed in Airamé et al. (2003) and briefly described here (Table 6.2). Taking both conservation and fisheries goals into account, the SAP recommended that 30–50% of the CINMS should be protected. Values this high were controversial. They were a collective professional judgement based on consideration of marine reserve literature, federal

Table 6.1 Summary of goals for marine protected areas established in the California Channel Islands (goals developed by the Marine Reserves Working Group; see Airamé et al., 2003) and along the entire California coastline (goals specified in the Marine Life Protection Act; see Kirlin et al., 2013)

Goal category	Channel Islands	Marine Life Protection Act
Ecosystem biodiversity	• Protect representative and unique marine habitats, ecological processes and populations of interest in the CINMS[a]	**1.** Protect the natural diversity and abundance of marine life and the structure, function and integrity of marine ecosystems
Sustainable fisheries	• Achieve sustainable fisheries by integrating marine reserves into fisheries management	**2.** Help sustain, conserve and protect marine life populations, including those of economic value, and rebuild those that are depleted
Economic viability	• Maintain long-term socioeconomic viability while minimizing short-term socioeconomic losses to all users and dependent parties	
Education	• Foster stewardship of the marine environment by providing educational opportunities to increase awareness and encourage responsible use of resources	**3.** Improve recreational, educational and study opportunities provided by marine ecosystems that are subject to minimal human disturbance, and manage those uses in a manner consistent with protecting biodiversity
Natural and cultural heritage	• Maintain areas of visitor, spiritual and recreational opportunities which includes cultural and ecological features and their associated values	**4.** Protect marine natural heritage, including protection of representative and unique marine life habitats in California waters for their intrinsic value
Management		**5.** Ensure that California's MPAs have clearly defined objectives, effective management measures and adequate enforcement, and are based on sound scientific guidelines
Network design		**6.** Ensure that the MPAs are designed and managed, to the extent possible, as a component of a statewide network

[a]Channel Islands National Marine Sanctuary.

Table 6.2 Science guidelines developed by the MLPA Science Advisory Teams for the design of MPA networks

	MPA design guideline	Design objective	Scientific rationale
Habitat representation	Every 'key' marine habitat should be represented in the MPA network	Protect the diversity of species that live in different habitats	Based on observed relationships between habitat type and marine community composition
Habitat replication	'Key' marine habitats should be replicated in multiple MPAs across large environmental gradients or geographic divisions	Protect the diversity of species that live in different ecological regions and geographical areas	Based on observed transitions in community composition across environmental gradients and geographic divisions
MPA size	• MPAs should extend from the intertidal zone to the offshore limit of state jurisdiction (5.56 km) • MPAs should have an alongshore span of 5–10 km (minimum) or 10–20 km (preferred)	• Accommodate the movements of individuals across depth zones • Protect populations of mobile organisms	Based on the reported movement scale of marine organisms, particularly adult fishes
MPA spacing	MPAs should be placed within 50–100 km (or less) of each other	Facilitate dispersal and connectedness among MPAs by benthic fish and invertebrates	Based on the reported movement scales of the larval stages of fish and invertebrates

Modified from Saarman et al. (2013).

fisheries management, dispersal rates and emerging fisheries in a qualitative way (PFMC, 2001). This differed from the population dynamic analyses described in this chapter in that 'No systematic assessments of populations within the CINMS were completed by the science panel' (PFMC, 2001).

The northern Channel Islands are situated in a complex geographical region with a strong environmental gradient across a relatively short geographic distance (see Section 1.1; also see Hamilton et al., 2010).

Consequently, the SAP defined three ‘bioregions’ and recommended that at least one, but preferably four, reserves be located in each bioregion.

The SAP combined all available information on substrate type, bathymetry and dominant macroalgal communities to characterize the habitats in order to ensure protections of each habitat type (Airamé et al., 2003). The SAP used information on species of concern or commercial importance to weight the importance of particular habitats. With this information, potential reserve configurations were generated using Sites v. 1, an analytical tool for planning regional-scale reserve networks (Andelman et al., 1999; Possingham et al., 2000). This program was precursor to the now widely used Marxan program (Ball et al., 2009) which identifies an efficient set of sites that collectively represent specified amounts of habitats, populations, or other features identified by the SAP (Airamé et al., 2003). These programs differ from the population dynamic, bioeconomic models in the MLPA process in that they do not calculate where populations of different species will actually persist based on spatial population dynamics (White et al., 2014; also see Section 2.3).

The network of MPAs finally implemented in the Channel Islands including Federal waters contained 21% of the CINMS waters in 11 state marine reserves (no commercial or recreational fishing allowed) and two conservation areas (where some types of fishing were allowed; Figures 6.1B and 6.2D).

2.2. Marine Life Protection Act

The second MPA effort in California applied to the whole state, and was initiated by conservation groups lobbying the legislature to obtain passage of legislation called the Marine Life Protection Act (MLPA) in 1999 (Osmond et al., 2010). This law directed the state to redesign its tiny collection of MPAs (0.2% of state waters) to meet six goals (summarized in Table 6.1). These goals were quite general, and even though the law was the enabling legislation for the MPAs, they contained few specific operational metrics. The goals were concerned with protection at the *ecosystem* level, but they did require the state to help sustain, conserve and protect marine life *populations* (Goal 2). They contained considerable ambiguities (e.g. what does it mean exactly ‘to protect natural diversity’, and what is a ‘statewide network’ of MPAs?). The MLPA had two other important requirements: (1) that it makes use of the best readily available science and (2) that after implementation, the MPAs be monitored and subject to adaptive management.

Enactment of the MLPA was not accompanied by sufficient funding to implement such a far-reaching decision-making process, one that would change marine fishery management throughout the State. This limited funding led to problems in early implementation efforts (Weible, 2008). In the first attempt, the CDFG formed a committee of marine scientists to suggest locations, configurations, and boundaries for MPAs throughout state waters. These proposed maps were presented at public meetings in 2001 as a starting point for discussion of the implementation of MPAs, but a strong negative reaction by stakeholders to already-developed maps led to the immediate failure of this approach. A second attempt a year later added statewide regional stakeholder groups (RSGs) and paid facilitators to the volunteer scientists. That attempt was also deemed inadequately funded, and was halted in the spring of 2003 (Gleason et al., 2010; Kirlin et al., 2013), although it foreshadowed some of the components of the later process that eventually succeeded.

In 2004, an agreement was struck between the state government agencies and a private foundation, the Resource Legacy Fund Foundation (funded by conservation-minded philanthropic foundations[1]), to fund a decision-making process to implement the MLPA. This process was to be controlled by an organization known as the MLPA Initiative (MLPAI). MLPAI staff included some state agency personnel and contractors with expertise in facilitation, spatial planning, geographic information systems, and policy analysis.

The planning process initiated and managed by the MPLAI divided the California coast into five Study Regions (Figures 6.1B and 6.2), and conducted the design process sequentially in each region, converting a statewide design problem into a sequence of regional-scale processes. Within each Study Region, the MLPAI appointed a RSG, and a Science Advisory Team (SAT), both based in part on nominations by interested citizens within each region. The RSGs comprises representatives of various constituencies (e.g. commercial and recreational fishing sectors, conservation groups, education and research sectors, interested state and federal agencies, tribal governments, and others[2]). There was also a Blue Ribbon Task Force (BRTF) appointed by the state Secretary of Resources in consultation with the Governor's office. The BRTF comprises four to five individuals with highly regarded experience in policymaking processes, although not necessarily in marine or fisheries conservation. The BRTF was responsible for overseeing

[1] http//www.resourcesllegacyfund.org/.
[2] https://www.dfg.ca.gov/marine/mpa/centralcoast_rsg.asp.

the integrity of the process to ensure it moved forward in a timely manner and was true to the goals of the MLPA (e.g. pushing for consensus among stakeholders, ensuring the RSGs strove to meet the science guidelines while recognizing the socioeconomic trade-offs in each region). The BRTF was responsible for winnowing lists of proposed plans emerging from the RSG, eventually submitting a short list of potential plans (usually including a 'consensus plan' preferred by the BRTF) to the California Fish and Game Commission. In addition, the CDFG provided feedback to the BRTF, SAT, and RSG on the regulatory and logistical feasibility of networks proposed by the RSGs, eventually submitting their recommended network proposal to the Commission in parallel with the BRTF (Figure 6.3). The Commission made the final decision on all MPA designs as the controlling authority for fishery regulations in state waters.

During the planning process for the first Study Region (Central Coast), the MLPAI and SAT developed the MLPA Master Plan (CDFG, 2008). This document dictated the detailed procedures of the MPA design process led by the MLPAI, and translated the somewhat vague policy goals of the MLPA into more specific, ecologically based objectives and design guidelines. The Master Plan was also approved by the Fish and Game Commission, and was used by the RSG and SAT in the development and evaluation of MPA network proposals in the first, and subsequent, Study Regions.

Within each Study Region, planning began with the MLPAI and CDFG preparing a Regional Profile that described the ecology, human uses and economics of the particular marine Study Region. Based on that profile and general MPA design principles, the SAT developed a series of region-specific scientific guidelines, presumably consistent with the Master Plan (Table 6.2). The RSG then began the process of developing a range of alternative, proposed spatial configurations of MPAs (Figure 6.4). Various subgroups of the RSG, with specific perspectives (e.g. favouring either conservation, recreation, commercial fishing, tribal, or other considerations) and staff support, were encouraged to develop collaborative, consensus proposals. External groups were also allowed to submit plans for consideration. Draft MPA plans were submitted to the SAT, who evaluated how well each plan met the scientific guidelines codified in the MLPA Master Plan. An iterative process followed, with the BRTF providing advice on the SAT-evaluated draft plans, the RSG then revising those plans and resubmitting them to the SAT. After three to four such rounds the BRFT submitted its recommendations to the Fish and Game Commission.

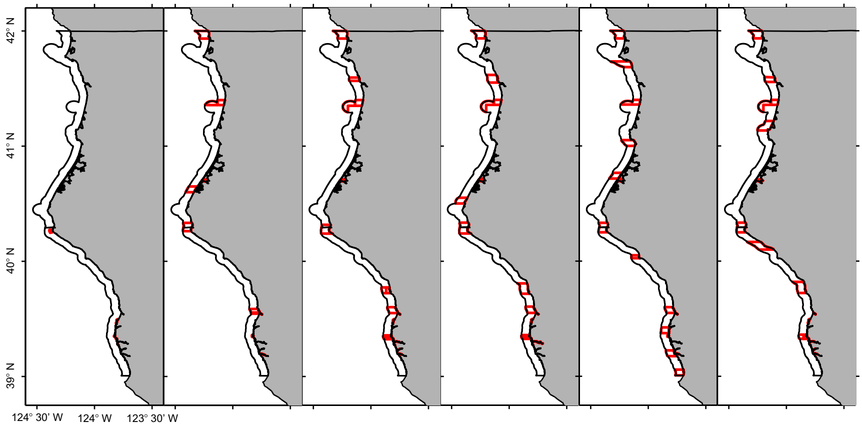

Figure 6.4 As an example, a representative range of proposed MPA networks during the first round of the North Coast design process. Proposed MPA boundaries are indicated in red (light grey in the print version); boundaries of California state waters are indicated in black. The leftmost panel shows the MPAs that existed in the region prior to the MLPAI process. Each proposed network contains a mixture of no-take and limited-take MPAs but these regulatory differences are not indicated on the figure.

This planning process was implemented first in the Central Coast Study Region beginning in 2003, and the MPAs in that region took effect in 2007. The planning process then moved to the North Central Coast (planning initiated 2007, implementation of MPAs 2010), the South Coast (initiated 2008, implementation 2012), and finally the North Coast (initiated 2009, implementation 2013; Figures 6.1B and 6.2). Across all four Study Regions, the MLPAI process created or expanded 124 MPAs, covering 16% of state waters; of these 61 (9.4% of state waters) were no-take State Marine Reserves, no-take State Marine Conservation Areas, or no-take State Marine Recreational Management Areas (in the latter fishing is prohibited but waterfowl hunting is permitted). The remaining MPAs were designated limited-take State Marine Conservation Areas or limited-take State Marine Parks (Gleason et al., 2013b).

As the planning process proceeded across the Study Regions, the MLPAI gradually improved outreach and interactions with stakeholders (Fox et al., 2013a,b; Sayce et al., 2013) by broadening the range of scientific expertise included on the SAT. In addition, new scientific tools were brought to bear on the process, including economic analyses, increasing consideration of spatially explicit, mathematical population models (Kaplan et al., 2006, 2009; Moffitt et al., 2009; White et al., 2010b, 2013a), and a Web-based spatial planning interface ('MarineMap', which later evolved into 'SeaSketch', Merrifield et al., 2013).

There were considerable differences among Study Regions in the stakeholder community, ranging from large groups of recreational fishermen and recreational water-users (kayakers, surfers, etc.) in the South to predominantly commercial fishing interests in the North Central and North Study Regions, with a large presence of Native American tribal stakeholders in the North (Fox et al., 2013a; Sayce et al., 2013). The stakeholders also became more involved in the process and more organized in their opposition or support, particularly after fishers for spot-prawn (*Pandalus platyceros*) abstained from the planning process in the Central Coast, resulting in some fishermen having all of their fishing grounds included in no-take MPAs.

Planning for the fifth Study Region (San Francisco Bay) had not yet begun fully when Governor Arnold Schwarzenegger left office in 2011, and as a new administration took office the political will and funding for the MLPAI process dissipated (particularly given the number and diversity of regulatory institutions and complicated stakeholder relationships in that bay). As of this writing, an MPA planning process has not begun for San Francisco Bay beyond an initial science review and considerations for the application of the network design guidelines for that region.

2.3. Scientific guidelines in the MLPA planning process

In each Study Region, as region-specific or additional science considerations emerged, regional science advisory teams (SATs) developed design guidelines in addition to those codified in the Master Plan. The intent of the guidelines was to ensure that MPAs would meet the statutory requirements of the MLPA, which required translating vague statutory language (e.g. 'preserve biological diversity') into an ecological and operational context (e.g. 'ensure that all habitat types were represented inside at least two MPAs in each Study Region'). The guidelines included recommendations for local habitat representation (what area of each key habitat should be included across the network of MPAs), habitat replication (how many MPAs in a Study Region should include each habitat type), and the minimum size and maximum spacing between MPAs (Saarman et al., 2013; Table 6.2). Eventually, guidelines were also developed for the minimum area of a habitat represented within an MPA that is required for an MPA to contribute to the spacing guidelines (i.e. network) for that habitat. There were also non-specific guidelines that each MPA should extend from the shore all the way to the 3 nm boundary of state waters (in order to accommodate cross-shore movements of fishes), and nonscientific guidelines promulgated by the CDFG enforcement division suggesting that MPAs have straight-line boundaries and be aligned with natural landmarks. These guidelines were formalized in the MLPA Master Plan document and used by the RSG in drafting proposed MPA networks. These draft network plans were then evaluated by the SAT as to how well each proposal met the guidelines.

In addition to evaluating how well MPA network proposals met these design guidelines above, the SAT also assessed the degree to which each proposed MPA intersected with locations relevant to other types of marine spatial planning. These included seabird foraging areas and rookeries, marine mammal haulouts, and regions affected by discharge from streams with high contaminant loads or wastewater outfalls. There was some debate among the SAT as to whether MPAs should be designed to avoid locations impacted by contamination, so they are more 'pristine' or whether they should target impacted locations in order to leverage improvements in water quality in the future. Similarly it was unclear how relevant seabird and marine mammal habitats were to MPA planning because those species were largely already protected by separate federal and state statues (e.g. the Marine Mammal Protection Act) and potentially move large distances. In general, these assessments had little bearing on the final configuration of MPA networks.

An additional aspect of the SAT's evaluation of MPA network proposals was characterizing the impact of specific extractive activities permitted in limited-take MPAs. Proposed networks typically included both no-take reserves and multiple types of limited-take MPAs (Figure 6.2). To evaluate limited-take areas, the SAT developed a protocol for characterizing the level of protection (LOP) afforded by each specific permitted activity, depending on the gear type used, ecosystem role of the targeted species, and other considerations (Saarman et al., 2013; Figure 6.5). In the SAT evaluations, the degree to which an MPA network proposal satisfied the scientific design guidelines was reported in terms of those LOPs; for example, a proposal might satisfy the size and spacing requirements if all MPAs with at least a 'Moderate–Low' LOP (some activities that will alter community structure are permitted) were counted along with MPAs with higher LOPs, but not if only MPAs with a 'High' LOP (no or very little extraction) were counted, for example, of two proposed networks that similarly met the size and spacing guidelines, the proposal comprises MPAs with higher levels of protection was considered to better meet the science guidelines and goals of the MLPA.

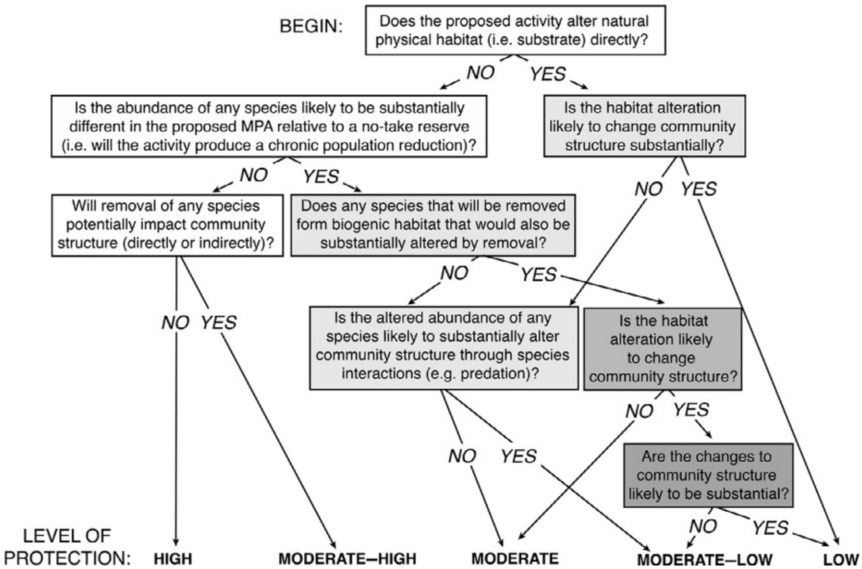

Figure 6.5 Decision tree flowchart used by the Science Advisory Team to determine the 'Level of Protection' afforded by an activity (e.g. fishing using a specific gear and target species) proposed to be allowed in a limited-take MPA. *Adapted from Saarman et al. (2013).*

2.3.1 Size and spacing guidelines

The size and spacing guidelines formulated by the SAT during the decision-making process for the first Study Region (Central Coast) were perhaps the most influential action taken by the scientists involved in the MLPA. The guidelines stated that MPAs should span at least 5–10 km in extent along the coastline, but that spanning 10–20 km along the coastline would be preferred (CDFG, 2008). This size guideline was based on qualitative examination of available information on home range sizes of California species (see CDFG, 2008 for references). The spacing guideline was that MPAs should be separated by no more than 50–100 km. This spacing guideline was based on information regarding larval dispersal distances (see CDFG, 2008 for references). Thus, although the guidelines did not specify a fraction of the coastline or habitat to be placed in reserves, the initial SAT at the outset effectively specified that between 5% (5 km MPAs spaced 100 km apart) and 28% (20 km MPAs spaced 50 km apart) of the coastline should be placed in MPAs. Recall from above that the percentage of coastline in no-take MPAs was 9.4%, while the percentage including limited-take MPAs was 16.0% (Gleason et al., 2013b). Note that these percentages refer to the entire coastline in a Study Region, not the percentages of particular habitats or species ranges. The size and spacing guidelines were not linked to particular habitat types or distributions; habitat-specific protection was instead addressed by separate guidelines for habitat representation and replication (Table 6.2).

There was not uniform agreement among the scientists on the Central Coast SAT that specifying *a priori* size and spacing guidelines was the best approach. Some members argued that a more comprehensive evaluation of all of the relevant factors affecting persistence of the fish populations present would lead to better performance of MPAs in the end. The state of the science at the time of the initial discussion (2003) was that the spatial configuration of MPAs required to ensure persistence of a species was known to depend on the larval dispersal distance of the species as well as how heavily that species was being fished outside the MPAs (Botsford et al., 2001). Moreover, population modeling tools that could calculate how specific MPA configurations, larval dispersal distances, and different levels of fishing would affect the spatial pattern of species abundance were under development, and would soon allow more direct evaluation of the effects of proposed spatial configurations of MPAs on fish populations (Kaplan et al., 2006, 2009; Moffitt et al., 2009; White et al., 2010b, 2013a). Some members of that initial SAT agreed to support the size and spacing guidelines as only a first step representing the best available science at that time, and the guidelines were incorporated into the Master Plan. Although the Master Plan was proposed as a 'living document' that could change as the best available science evolved (Kirlin et al., 2013), in practice it was deemed not possible to remove or fundamentally alter the primacy of size and spacing guidelines included in the original Master Plan as the 'best available science' improved. This was because (1) formally updating the Master Plan would require action by the Fish and Game Commission and (2) the MLPAI was concerned about components of the evaluation for some Study Regions differing from those used in the other Study Regions. Additionally, some SAT members argued that because spatially explicit population model evaluations could only be performed for certain species with adequate information, adopting size, and spacing guidelines without explicit calculations for any species was a conservative buffer against uncertainty about the response of the full suite of affected species. Consequently, the size and spacing guidelines remained the key component of the evaluation of proposed MPAs, even as more comprehensive modeling evaluations became available and were also used by the SAT in evaluating MPA network proposals (see Section 2.3.2).

Another characteristic of size and spacing guidelines noted by scientists familiar with decision making in natural resource problems was that by effectively specifying how much of the coastline to set aside in MPAs at the beginning of the process, they limited the scope for later decision making (Osmond et al., 2010). The size and spacing guidelines effectively limited the 'decision-making space' being considered by the SAT, the BRTF, and ultimately by the California Fish and Game Commission. However, even in the final designs implemented by the Fish and Game Commission not all of the MPAs in the network met the 'preferred' size and spacing requirements (i.e. 20 km MPAs spaced 50 km apart).

The size and spacing guidelines were not the only factor that constrained the design of MPA configurations by the stakeholders. Requirements for representation of minimum areas of key habitats, particularly rare habitats, and that those habitats be replicated in multiple MPAs throughout at Study Region (Saarman et al., 2013) effectively ensured that MPAs would be placed in certain key locations. Together, these requirements led to RSG groups proposing alternative MPA proposals that were largely quite similar to each other, particularly once plans that failed to meet the scientific guidelines were winnowed out in early evaluation stages (see modeling results below).

2.3.2 Population models and fisheries

Another characteristic of the scientific evaluation of proposed MPA networks that raised questions among the SAT members was the decision made by the MLPAI in the first region to ignore the relationship of the proposed MPAs to fisheries and their management outside the MPAs. This went so far as MLPAI staff directing the population modelers not to use the word 'sustainability' to describe population status, because it implied the MLPA decision making was related to sustainable fishery management. Scientists knew by that time that including the level of fishing outside the MPAs was necessary to predict the effects of the proposed MPAs on persistence of fish populations of various species (e.g. as noted above in Botsford et al., 2001). From the point of view of the MLPAI, however, this was a legal issue involving whether the implementation of the MLPA was required to interact with the implementation of a new law changing the way that California fisheries were managed, the Marine Life Management Act (MLMA) (Fox et al., 2013c).

The views of the MLPAI on the value of including fishery information changed in response to the publication of a population modeling study that addressed proposed MPAs in the Central Coast Region (Walters et al., 2007). These authors concluded that (1) movement of adult fishes could lead to lower fish abundance in MPAs, (2) population persistence in MPAs depended critically on fishery management outside the MPAs, (3) the size and configuration of MPAs had little impact on population dynamics, and (4) the MPAs were unlikely to benefit key fish species. While other modelers associated with the MLPA process noted potential flaws in that paper (see comments in Moffitt et al., 2009), the MLPAI began to support population modeling more formally after that. It was decided at that time that two groups, one at the University of California Davis and one at the University of California Santa Barbara should each formulate population models and report the effects of fishing on MPA performance and the effects of MPAs on fishery catch. These two modeling efforts converged on similar model structures and assumptions and produced similar results and were eventually folded into a single joint effort (White et al., 2013a). Many stakeholders and SAT members initially resisted inclusion of the models in the decision-making process, in large part because of debate about whether the model should assume that future fishing outside MPAs should be assumed to be at sustainable levels or unsustainable levels (White et al., 2013a). In the end, the models did not supplant the primary role of the size and spacing guidelines in the decision making. These guidelines were based on the assumption that current fishery management provided little protection against overfishing (Gaines et al., 2010; MRWG SAP, 2001).

The population models developed under the MLPA initially were extensions of the original modeling approach taken by Botsford et al. (2001) in the sense that they assumed the California coastline was essentially linear and that larval dispersal could be approximated by a symmetrical, spatially homogenous dispersal kernel (e.g. Figures 6.6A–D). That type of model was used to advise the design process in the Central Coast and North Central Coast regions, but in the South Coast and North Coast regions, the modeling groups developed two-dimensional models with finer (1 km^2) spatial resolution, and used results from Lagrangian simulations of larval dispersal in ocean circulation models (Drake et al., 2011; Mitarai et al., 2009) to obtain connectivity matrices for the population models (e.g. Figures 6.6E–H; White et al., 2013a). These models afforded much finer-scale assessments of the likely performance of individual MPAs (White et al., 2013a), and later analysis showed that they could have guided the planning process to network designs with higher fish biomass and higher fishery yields than those obtained by following the SAT's more general guidelines (Costello et al.,

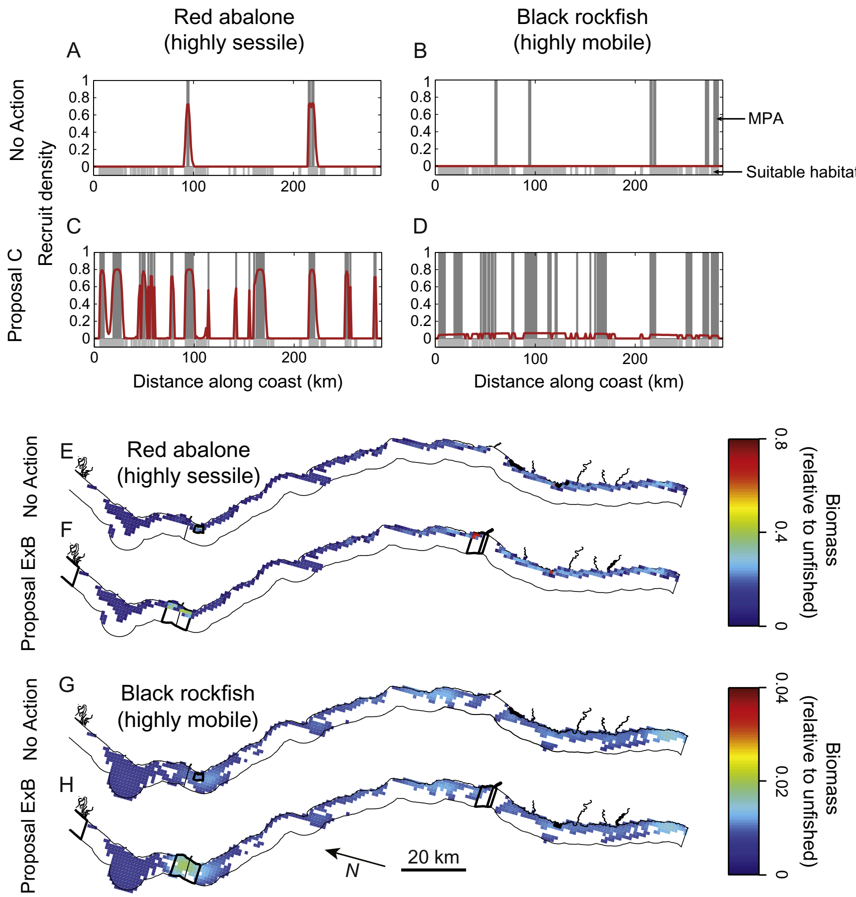

Figure 6.6 Representative results showing differences in the response of species with different larval dispersal distances to alternative MPA network proposals. In (A–D), a population model that approximated the North Central Coast Study Region as a linear coastline predicted that red abalone (*Haliotis rufescens*) would have self-persistent populations within MPAs in either the sparse 'No Action' proposal (with only previously existing MPAs; (A) or the conservation-oriented Proposal C (C). By contrast, black rockfish (*Sebastes melanops*) were predicted to have network persistence only in Proposal C (D). In (E–H), similar results for the same two species are displayed for the higher-resolution two-dimensional model used in the North Coast Study Region, for either the No Action alternative or the conservation-oriented Proposal ExB. In the North Coast, black rockfish populations were sustained by network persistence by the North Central Coast MPAs to the south of the Study Region, even in the No Action scenario (G). For model details, see White et al. (2010b, 2013a). In all of these examples, the populations were presumed to be overfished (i.e. lifetime reproduction was below the critical replacement level).

2010; Rassweiler et al., 2014). However, even these models had a key limitation: in the absence of estimates of present-day population density of species along the coastline, it was not possible to initialize the models to make short-term predictions, and only long-term equilibrium abundances could be forecast. This would prove to be an obstacle to using the models to guide short-term assessment and adaptive management of the MPAs (White et al., 2011; see Section 3).

Although the models became more sophisticated over the course of the MLPA process, the basic way that the modeled populations responded to MPA network designs and fishing did not change. The models developed to predict population responses to network designs evolved in complexity, from early modeling with an assumed straight coastline, and an assumed shape of a larval dispersal kernel for the Central Coast Study Region, along with results of later modeling with real coastlines and bottom topography, and larval transport from a circulation model for the North Coast Study Region (Figure 6.6). In the former models (Figures 6.6A–D), a short distance disperser, red abalone (*Haliotis rufescens*), persists in some locations even without additional MPAs while under Proposal C, this species persists wherever there is both suitable habitat and an MPA, but not elsewhere. The black rockfish (*Sebastes melanops*), a species with long larval dispersal distances and a large home range, does not persist anywhere under the assumed level of overfishing, and even with substantial area in MPAs, does not persist at a very high level (however, this highly mobile species was predicted to persist at higher biomass under lower levels of fishing; White et al., 2010b). Using the more sophisticated circulation model (Figures 6.6E–H) in a different region, the results are not as dramatically different between species and proposed plans, but the benefits of MPAs for both species can be clearly seen.

Because this form of graphical results (e.g., Figure 6.6) from the population models involved too much detail for MLPA decision-making groups, the results for a number of different proposals were summarized (Figure 6.7). Stakeholders and decision-makers could see how the conservation value (total biomass of all model species) and the economic value (total fishery yield) of each proposed MPA network varied with the different fractions of coastline in MPAs, at different levels of fishing outside the MPAs. As the total area in MPAs increased from the plan with the lowest fraction to the highest fraction, conservation value never decreased, and often increased. However, fishery yield increased with MPA area only in the case when it was assumed that overfishing was occurring outside MPAs. When it was assumed that fishing levels outside were at the level producing

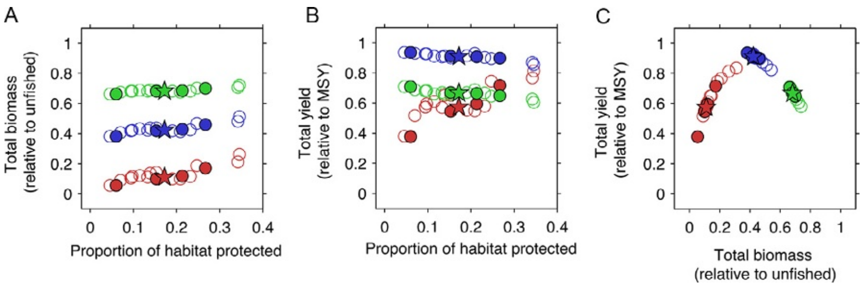

Figure 6.7 Example of the summary analysis of proposed MPA networks by the UC Davis bioeconomic model (White et al., 2013a). Each point is the result for one proposed South Coast Study Region MPA network, evaluated for one model species (the California sheephead, *Semicossyphus pulcher*). Panels show (A) equilibrium biomass in the Study Region as a function of the proportion of shallow rocky reef habitat (<30 m depth) in the Study Region protected by MPAs where sheep head fishing would be prohibited; (B) equilibrium fishery yield as a function of the proportion of habitat protected; and (C) equilibrium biomass as a function of equilibrium fishery yield. Biomass (kg) is expressed relative to the equilibrium unfished biomass estimated by the model, and yield (kg) is expressed relative to the maximum sustainable yield (MSY) for that species in the absence of MPAs, as estimated by the model. Because future fishing levels are unknown, results are shown for three different possible levels of harvest outside of MPA boundaries: unsustainable (the fishery would collapse without MPAs; red symbols (grey in the print version)), MSY-type (the fishery would be at or near MSY without MPAs; blue symbols (dark grey in the print version)), and conservative (the fishery would be below MSY without MPA (because of low fishing); green symbols (light grey in the print version)). Open symbols indicate proposals generated in the first two rounds of the design process, closed symbols indicate proposals from the final round, and the star indicates the 'preferred' proposal selected by the Blue Ribbon Task force. See White et al. (2013a), for model details.

MSY or less, fishery yield declined with increasing area in MPAs (White et al., 2013a).

These results for the South Coast Study Region also provide an example of how the range of proportions of habitat protected by proposed MPAs contracted as the deliberation among proposals proceeded (Figure 6.7). The original proposals (Figure 6.7, open circles) ranged from 0.05 to 0.35, then contracted to a range of 0.07 to 0.28 (Figure 6.7, solid circles) by the final round of decisions for the BRTF. The plan eventually chosen by the California Fish and Game Commission (Figure 6.7, star) was midway between these.

Viewing the conservation and economic values together (Figure 6.7C) indicates whether there are trade-offs involved or a win-win situation. When the species are overfished, total yield and total biomass increase together. When the fishing level is at or below that causing MSY, fishing yield declines as total biomass increases.

The science of population modeling for MPAs developed rapidly during the period of the MLPA process. Kaplan et al. (2006, 2009) showed how differences in larval dispersal distances among species, and spatial heterogeneity in the distribution of habitat would affect the persistence and spatial distribution of different species with alternative MPA network proposals for the Central Coast Study Region. Moffitt et al. (2009) then showed how accounting for the additional effect of movement of adult fish within home ranges reduces the effectiveness of MPAs. This model achieved the capability to quantitatively evaluate population persistence considering the combination of adult and larval movement, as the SAT had done qualitatively in developing the size and spacing guidelines. Accordingly, Moffitt et al. (2011) next analyzed the effects of size and spacing guidelines in a way that also accounts for the level of fishing and the spatial configuration of the MPAs. Because the effects of proposed MPA networks on populations depends on the intensity of fishing outside MPA boundaries (which is highly uncertain at the local scale of MPAs), White et al. (2010b) used population models to perform a decision analysis, evaluating likely MPA performance over a probability distribution of different 'states of nature' (Hilborn and Walters, 1992) representing different levels of fishing. The results illustrated how recommendations for MPA design could depend on decision-makers' assumptions about the effectiveness of conventional fisheries management in the future (fewer and smaller MPAs would be recommended under optimistic assumptions about conventional management), but this type of analysis was not adopted by the MLPAI.

2.3.3 Economic assessments

Although economic considerations were not among the statutory goals of the MLPA (Table 6.1), the MLPAI and BRTF recognized that the fishing communities in each Study Region faced potentially substantial economic costs from new MPAs. Consequently, the SAT in each region also considered assessments of the economic costs of lost fishing grounds in each MPA network proposal. Scholz et al. (2004, 2011) describe the details of the analysis used. Essentially, in each Study Region they surveyed a stratified sample of participants in important commercial and recreational fisheries as to the extent and relative stated importance of their fishing grounds, as well as a number of demographic and operating cost variables. The aggregated data were then used to calculate the percentage of fishing grounds closed under each MPA network proposal, as well as the minimum first order economic losses due to that closure. These assessments were highly valued by RSG members and the BRTF because it was one of the few SAT analyses (along with the bioeconomic population models) that centred on the economic costs of MPAs rather than the potential ecological benefits. An important limitation to the economic analysis was its static nature: it implicitly assumed that fishing grounds inside MPAs were a complete loss; i.e., there was no way to account for the potential increase in biomass inside MPAs that could eventually 'spill over' and sustain fishery yields. Thus the analysis reflected only the initial short-term costs of closing fishing grounds; this was a mirror image of the limitation of the population modeling, which could project long-term equilibrium outcomes but not short-term trajectories (White et al., 2013a).

2.3.4 What species were likely to benefit?

The science guidelines codified in the Master Plan (Table 6.2) operated on the assumption that by setting aside a certain fraction of habitat area, species would persist at higher levels within those areas, and ecosystems would be preserved in a more natural state. Consequently, the list of 'species likely to benefit' from the MPAs assembled by the SAT in each region, as required by the MLPA, typically included any species that could be taken in a fishery, or that might benefit from reduced disturbances or habitat improvements inside MPAs. Such lists did not reflect species differences in harvest pressure present, larval dispersal patterns, adult movement rates, or other life history characteristics that were known or predicted to affect the response of species to MPAs (Botsford et al., 2001; White et al., 2010b, 2011). In general, it should be reasonable to expect any fished species to increase in abundance to some degree after protection in an MPA (assuming that fishing mortality exceeds any negative effects of the MPA on predator–prey and competitive interactions among fished species), but the compilation of a broad, unranked list contributed to the expectation that there should be -the-board increases in fish abundance after MPA implementation. This turned out not to be the case: preliminary *post hoc* assessments show that not all fished species have increased, and some have increased much more than others (Hamilton et al., 2010; see Section 3.1).

3. IMPACTS OF THE MPAs

Because the Channel Island MPAS were implemented in 2003, and the first region of the MLPA was implemented in 2007, there has been a relatively short time for impacts to occur, be observed and be interpreted through analysis, especially as regard potential network benefits. Moreover, there have been very few studies conducted to evaluate social or economic impacts for either of these networks.

3.1. Ecological impacts

The ecological impacts of the Channel Islands MPAs are more apparent because of the longer time since implementation. One key development has been the realization that removing the confounding effects of biogeographic and physical factors is key to detecting effects (Hamilton et al., 2010). Results after 5 years showed large increases for several fished finfish species, but curiously no net change in abundance for other fished species (e.g. the recreationally fished kelp bass, *Paralabrax clathratus*) possibly due to environmentally driven failures in larval recruitment in the years after implementation and the short time scale involved. Kay et al. (2012a) also documented an increase in catch-per-unit-effort (CPUE) and the size of spiny lobsters (*Panulirus interruptus*) inside of MPAs, but not outside. Moreover, inside the MPAs, lobster CPUE increases with distance from the MPA boundary, which could imply spillover, but could also be due to poaching (Kay et al., 2012b).

For the MLPA MPAs, there is a mandated periodic 5-year review and evaluation process. This has been completed for the Central Coast Study Region, and is underway for the North Coast Study Region. The monitoring effort is managed jointly by the California Ocean Science Trust (CalOST) and CDFW (OST, 2013). It began early enough to be considered baseline monitoring and continuing monitoring is planned. The monitoring effort is directed at measuring ecosystem-level effects, but it includes single-species population outcomes. Initial evaluations for the Central Coast Study Region have been mixed, with some species showing increases and others showing decreases; potentially a result of high variability in environmental conditions and larval recruitment (OST, 2013). This initial report is of limited utility in that reported results of monitoring are not accompanied by associated measures of uncertainty, such as confidence limits.

3.2. Fishery impacts

As noted above, a method for assessing the loss of preferred fishing grounds based on interview data was developed during the implementation of the MLPA MPAs (Scholz et al., 2011; White et al., 2013a). These are short-term, worst case cost projections that do not account for long-term trends (positive or negative) and they have not been tested since the implementation of the MPAs.

Currently, several economic studies of the response of fishing are in place to track the effects of the MPAs on fishing, but it is too early to describe extensive results (OST, 2013). The results of a survey of fishermen reporting how many were affected by the implementation of the MPAs are reported in OST (2013).

3.3. Interface with fisheries organizations

There has been some coordination with fisheries management regarding the Channel Islands MPAs. The Pacific Fisheries Management Council, the regional federal management body, reviewed the Channel Islands implementation process in 2001 (PFMC, 2001). The Channel Island MPAs were originally implemented in State waters only (i.e. out to 3 nm from shore), and later extended into federal waters (i.e. out to 200 nm from shore).

Although California's MLMA, passed near the same time as the MLPA specifically recognized the MLPA as a means by which the state could move toward a more ecosystem-based approach to fisheries management, there has been no formal consideration for integration of the MPA networks into the state's approach to fishery management (CDFG, 2001, 2002). As noted above (e.g. Figure 6.7), the population models indicated that fishery yields would decline with increasing area in MPAs if the fishing effort was that producing MSY, or less. However, the MLPA process concluded that this should not be accounted for because future fishing levels were highly uncertain.

However, two studies have explored the application, or potential application, of these reserves for informing stock assessments. Schroeter et al. (2001) demonstrated the application of reserves in evaluating the fishery status of the warty sea cucumber (*Parastichopus parvimensis*) in the northern Channel Islands. Similarly, Babcock and MacCall (2011) explored the application of reserves for stock assessments for a suite of nearshore California fishes.

3.4. Social impacts

There were some strong negative responses to MPAs by fishermen. The strongest was the response by fishermen in northern California to the early attempt to implement MPAs (see Section 2). Later, there was strong resistance by fisherman in the Southern California Study Region (including lawsuits seeking to enjoin the implementation of the MPAs; Fox et al., 2013a), and resistance by Native Americans in the northern California Region (see Fox et al., 2013c for details). Nonetheless, the MLPAI process was purposefully inclusionary and iterative, and strove to ensure that all stakeholder groups had opportunities to voice their views (Fox et al., 2013c; Sayce et al., 2013).

3.5. Enforcement and its effectiveness

The importance of enforcement to management with MPAs is widely appreciated, but the deleterious effect of violations of MPAs on monitoring and adaptive management may not be as well appreciated. The presence of poaching in MPAs can render the task of assessing the protective effects of MPAs almost impossible. One remedy that can reduce that effect is carefully keeping records of violations, and, if possible, their biological effects. This point is underscored in the history of California's MPAs. Recorded levels of poaching was one of the potential reasons for the lack of a significant difference in fish density between reserve and non-reserve sites Hopkins Marine Life Refuge (one decade old) and Pt. Lobos Marine Reserve (two decades old), both in Central California (Paddack and Estes, 2000).

CDFW is the agency responsible for enforcement of the MPAs. They patrol by boat, and can respond to poaching in progress. Records of violations are kept, and presumably will be available for analyses associated with adaptive management of the MPAs. Between 2008 and 2011 (4 years) between 3 and 16 violations of MPAs occurred per year in the central coast region (OST, 2013).

4. OVERVIEW: LOOKING AHEAD

4.1. What was achieved?

In the Channel Islands a contentious, early decision-making process led to the implementation of 13 MPAs in state waters, which were eventually extended to federal waters (Figures 6.1B and 6.2D). These covered 21% of the CINMS waters.

The Channel Islands process likely influenced the development of the MLPA by calling attention to the effects of stakeholder involvement and a strong role for science-based guidelines. One important difference between the Channel Islands process and the MLPA is that in the Channel Islands, local community members initiated an *ad hoc* process that grew into a joint state and federal partnership, but without overarching legislation to drive the process (Osmond et al., 2010). Though there exists no formal state-sponsored monitoring to evaluate the impacts of the Channel Island reserves separately from the South Coast Study Region, independent academic (Partnership for Interdisciplinary Studies of Coastal Oceans) and federal (Channel Islands National Park Service) studies continue.

A number of achievements were accomplished under the MLPA. The law was passed, and 124 MPAs were implemented through a public decision-making process. A number of publications are now available focused on how implementation of the law was successfully accomplished (Gleason et al., 2013a and references therein). It has not yet been demonstrated that the central goal of the MLPA, i.e., improvement of the sustainability of California's coastal ecosystem, has been accomplished. That will require implementation of the adaptive management of the MPAs, which includes, as a first step, evaluation of monitoring data to determine whether they are 'working'. As we have noted in this chapter, adaptive management following implementation is a requirement of the MLPA. Baseline monitoring has been accomplished and a monitoring framework is under development. So far monitoring of abundance and size distributions of key species, both inside and outside of the MPAs has been accomplished over the 7 years since implementation in the Central Coast Region. A meeting organized by the CalOST in February 2013 celebrated proposed indications of success (OST, 2013). However, the results presented at that meeting did not include an account of uncertainty in estimates (e.g. confidence limits). Additional time and analysis will be required to assess the performance of these MPAs more definitively. Work in progress by the authors on direct assessment of potential increase of abundance and mean size of three species, inside and outside of three MPAs in the Central Coast Region indicate abundance and sizes have not increased. Population modeling of expected population responses of these three species, accounting for observed levels of recruitment variability and local estimates of fishing mortality, indicate that it is too early to expect to detect positive indications that these MPAs have had the desired effect.

4.1.1 Resistance to global change

California faces a number of specific, identified threats from climate change, and the predicted responses of species indicate that California's MPAs will provide some resilience to their effects. Two key design traits of the MLPA network underpin the potential for the network to buffer the effects of climate change on species and communities; the depth range of individual

MPAs and the spacing between MPAs scaled to larval dispersal distances (Carr et al., 2010). Increases in sea surface temperatures and thermal stratification cause increased vulnerability of species to thermal stress at shallow depths. Whether stress-related or reflecting thermal preferences, species populations find thermal refuge in deeper cooler waters (e.g. Dulvy et al., 2008). California MPAs that extend from the intertidal to the outer edges of the continental shelf provide protection for species as populations shift to deeper depths. Another phenomenon associated with climate change is an overall warming of ocean temperatures and a concomitant latitudinal (poleward) shift in species ranges (e.g. Perry et al., 2005; Pinsky et al., 2013; Poloczanska et al., 2013), including a poleward shift in intertidal species over a 30-year period documented in California (Barry et al., 1995). Larval transport is an important mechanism by which species shift distributions along the coast (Gaylord and Gaines, 2000). Spacing and larval connectivity among MPAs can allow species to track changes in water temperature while maintaining protection afforded by MPAs by shifting from one protected area to another as their ranges shift along the coast.

The adaptive management scheme specified in the MLPA also contributes to the resilience to global change. The MPAs will be sampled for monitoring every 5 years, tendencies for species to shift distribution can be detected and accounted for by moving boundaries if desired.

There are also indications that both the magnitude and seasonal timing of upwelling in the California Current large marine ecosystem have changed and will continue to change (Bakun, 1990; Garcia-Reyes and Largier, 2010; Snyder et al., 2003). In addition to the proposed long-term, gradual changes, there have been a number of episodic changes in physical conditions that have affected marine populations, including a period during which upwelling began later in the year near 2005 (Barth et al., 2007), and occasional periods of anoxia at various locations (Chan et al., 2008; Grantham et al., 2004). The most recently identified effect of increasing CO_2 is the observation that upwelled waters are becoming increasingly acidic (Feely et al., 2008).

The evaluations of proposed MPAs throughout the MLPA implementation indicate there will likely be some amelioration of the effects of climate change and ocean acidification through the increase in lifetime reproduction implied by the increase in biomass with area in MPAs in virtually all of the population model results. The results of population modeling during the MLPA process showed that increasing coverage in MPAs would not cause a decline in biomass, and in many examples it would cause an increase in biomass. This increase in biomass was due to an increase in lifetime reproduction in the affected species provided by the proposed MPAs, lifting them further above their critical replacement level. This increase in replacement provides a buffer against both long-term decline in population productivity and occasional episodic low rates of survival or productivity. A MPA implemented nearby in Mexico provides a clear empirical example of this increase in resilience provided by MPAs (Micheli et al., 2013). There the greater potential for reproduction in the protected population was sampled and the consequences for sustained settlement during a hypoxic period were directly observed.

5. FUTURE REQUIREMENTS

The task remaining in the management of California's new MPAs (i.e. implementing and executing their adaptive management) is arguably the most important part of the Channel Islands MPA and the MLPA efforts, both from the perspective of local resource management and the global need for information regarding the performance of MPAs. One of the initial steps required was to reanalyze the population model responses to MPAs, focusing on the short-term, transient population response, rather than the long-term effects used in the decision making for the MLPA. The general expectations of fish population transient responses to a removal of fishing mortality have been described (White et al., 2013b). In addition, how these responses would be detected from sampling over a range of temporal and spatial scales have been compared (Moffitt et al., 2013). Both of these results depend, of course, on the level of fishing to which the populations have been subjected prior to the implementation of the MPAs. Because fishery management commonly resolves variability in fishing only with coarse spatial resolution, the local fishing mortality rates affecting specific MPAs will need to be estimated from local size distributions. The observed population responses to implementation will also depend on annual variability in past recruitment, as well as measurement errors. We are in the process of evaluating the combined effects of these for a number of species and locations in the central coast region. The results indicate that it is currently too early to detect expected increases in population abundance or individual size, given the time scales and stochasticity inherent in the dynamics of these populations.

The fact that physical oceanographic conditions influence recruitment, individual growth, fecundity and mortality rates, as well as larval dispersal of species in a network of MPAs suggests that physical observations will be essential for interpretation of the monitoring of the biological status of fish species in a network of MPAs. Carr et al. (2011) explained how monitoring of the effects of MPAs on fish size and abundance, as well as the relative make-up of the species composition in the fish community could benefit from monitoring of specific physical oceanographic variables by recently developed ocean observation systems (e.g. in the United States, the various OOSs).

A framework plan for future monitoring in support of adaptive management of MPAs in California is currently under development through a collaboration between the CDFW and California's Ocean Science Trust. This plan is aimed at the ecosystem level, as were the MPA implementation efforts described herein. However, it correctly seeks operational information at the population level, i.e., species densities and size distributions. From the material presented in this chapter, it appears that if this planned monitoring occurs, it would provide the information needed for adaptive management only if it (a) made use of the population results regarding transient responses to link MPA effects to monitoring observations and (b) provided the information necessary to allow the adaptive management program to account for uncertainty. The former would be required to connect life histories and MPA designs to the observations, as required for adaptive management (i.e. for asking whether the observations match the 'predictions'?). The latter would be vital, simply put, to guarantee that the adaptive management would be based on statistically significant results. The need for both of these is especially acute in this case because of the complexity of this kind of resource management, and the nascent nature of our understanding of it.

5.1. Could it have been achieved differently/more effectively?

Not surprisingly, there is a range of opinions regarding whether implementation of these MPAs should have been done differently. These opinions depend largely on one's view of the ultimate goals of the MLPA, and more generally, the role of science in resource decision making.

Whether the size and spacing guidelines should have played such a dominant role is a central question. They were formulated on a qualitative basis as 'rules of thumb', statements formulated to facilitate the formulation of initial spatial configurations for proposed MPAs (Carr et al., 2010). Such rules serve a useful purpose in expediting broadly based decision making, and these certainly played that role in the MLPA, as did the similar specification of the fraction to be placed in MPAs in the Channel Islands implementation.

The problem with these guidelines was that they came to be very influential in the design process, and were treated as rigorous scientific requirements, which they were not. That is, there is not a quantitative basis for saying that the MPAs would be optimal in any sense, if they satisfy those guidelines. As can be seen in Figure 6.8, greater area in MPAs will protect more species, and even a figure such as that one depends critically on the amount of fishing assumed. Analyses such as these with population dynamic models could have been used to create more realistic lists of species expected to benefit.

The size and spacing guidelines were based on the intuitive ideas that (1) MPAs should be big enough for some species home ranges to be contained

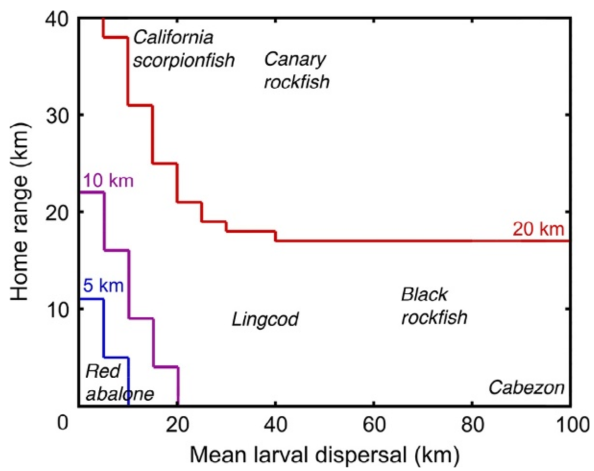

Figure 6.8 Illustration of the nonlinear relationship between MPA size and spacing and the persistence of marine populations. Results are derived from a population model of idealized species living on an infinite linear coastline with homogenous habitat. Model species had different combinations of larval dispersal distance (expressed as the standard deviation of a symmetrical dispersal kernel) and adult home range size (expressed as the diameter of the home range). The coastline had no-take MPAs spaced 50 km apart, and with sizes of 5, 10, or 20 km wide (alongshore dimension). Movement scale combinations interior of each contour lines indicates the range of species that would persist within different size and spacing configurations. Approximate estimates of adult and larval movement scales for several California species are shown for comparison. For 5 or 10 km MPAs, only 10% or 20% of the coastline respectively, in is MPAs, and species only persist if movement is low enough to allow self-persistence. For 20 km MPAs, the total MPA area (40% of coastline) is sufficient for network persistence, protecting a much greater range of movement combinations. Thus gradual increases in MPA size (or decreases in MPA spacing) can yield abrupt jumps in the protection afforded by the MPAs. *Adapted from Moffitt et al. (2011).*

within them and (2) MPAS should be close enough together for the larvae from one MPA to reach other MPAs. When implementation of MPAs under the MLPA began, these rules were probably the best way to begin drawing lines on maps. But at about that same time, scientists were discovering that (a) the dependence of persistence of populations on size and spacing change completely with different amounts of fishing outside the MPAs and (b) the dependence of persistence on size and spacing was not linear, as presumed by these rules of thumb (Botsford et al., 2001; Hastings and Botsford, 1999; Mangel, 1998; Moffitt et al., 2011; Figure 6.8).

Use of the size and spacing guidelines narrowed the range of decisions that could be made by the RSGs, the SATs, the BRTFs, and the California Fish and Game Commission (Osmond et al., 2010). The question of whether scientific advice should have played that role in the decisions is certainly a reasonable one. Should the role of scientific advice in resource decision making be merely to provide the best possible estimate of the consequences of various policy actions, or should scientists be specifying the policy actions? Such specification of scientific advice would be regarded by some as bordering on advocacy of a specific policy, rather than just the provision of objective scientific advice regarding the consequences of a specific policy.

Adoption of the size and spacing guidelines led to the early misunderstanding that the effects of MPAs on populations and ecosystems did not depend on the level of fishing outside of MPAs, hence fishing could be ignored in the MLPA process. This changed with the publication of Walters et al. (2007), after which there was greater acceptance of population modeling that accounted for the controlling effects of fishing and individual movement rates. However, the effects of fishing levels on MPA responses were never specifically accounted for in the decision-making process, even though a decision analysis based on the uncertainty in fishing rates was developed (White et al., 2010b). The effects of MPAs on fishing were only minimally accounted for through the economic analysis of Scholz et al. (2011). Moreover, thus far the monitoring of fishing activities outside MPAs at relevant spatial scales across California's network has not been implemented.

Additional support for the broad application of the size and spacing guidelines was recently proposed by Gaines et al. (2010), who presented the same qualitative argument that MPA size should depend on home range and MPA spacing should depend on larval dispersal distances, but also related the resulting suggested fraction in MPAs to the results from Botsford et al. (2001). They noted that their conclusion that one-third of the coastline should be in MPAs was close to the 35% result in Botsford et al. (2001). That reference by Gaines et al. (2010) was appropriately qualified by noting that it was the result obtained for the case in which fishing removes all of the fish outside the MPAs (i.e. the 'scorched earth assumption'). However, use of the value of 35% was justified nonetheless by stating that we needed to assume complete fishing removal to guard against that eventuality since there was so much overfishing in the world. Making such an assumption eliminates consideration of how much fishing there is (or should be), and supplants policy decisions regarding the level of precaution that should taken, narrowing the scope for decision to be made by the BRTF and the California Fish and Game Commission.

Ignoring the effects of MPAs on fishing (i.e. possibly reduced fishery yield), and the effects of fishing on MPAs (possibly lower MPA fractions needed) are not recommendations that we would make for future MPA efforts. In fact, an increase in fishery yield is often the promise associated with implementation of MPAs. Our recommendation would be to attempt to determine which situation applies in Figure 6.7B: overfished (red symbols), fished at MSY (green symbols) or fished less than MSY (blue symbols).

Adopting size and spacing guidelines with primacy over explicit consideration of species movement rates and level of fishing through population modeling in the implementation of the MPAs of the MLPA limited efforts to meet two requirements of the act: (1) the requirement for the use of the best available science and (2) the requirement for adaptive management. As the science of quantitative assessment of MPAs developed, new scientific advances could not be incorporated into the MLPA decision making because that would be perceived as changing the rules between Study Regions. Lack of quantitative, population-specific results of the expected benefits of the MPAs limited the ability to perform an essential element of adaptive management: comparison with the predicted benefits (and costs) of the management action. This removed explicit consideration of major uncertainties such as those in fishing levels and larval dispersal, as well as consideration of another societal input, fishing outside the MPAs. There is broad appreciation of the value of adaptive management in combating uncertainty to avoid management failures in biological resources (Parma and NCEAS, 1998). California now has the opportunity to move forward with the adaptive management of these newly implemented MPAs in a way that accounts for uncertainty and fishing through population modeling and monitoring (e.g. Moffitt et al., 2013; White et al., 2013b).

6. SUMMARY

In summary, California implemented a network of 132 MPAs, covering more than 15% of its coastline, in a science-based, stakeholder-inclusive process. The success of this outcome was driven by the passage of a law, substantial funding by philanthropy, qualitatively justified size and spacing guidelines, and paying minimal attention to fishery interactions (Fox et al., 2013c; Osmond et al., 2010). The goals of the Channel Islands and the MLPA processes go beyond mere implementation of MPAs, but rather concern the ultimate effects of those MPAs on California's coastal ecosystem. Whether these goals are met will depend strongly on the outcome of future monitoring and adaptive management.

In spite of the fact that future attempts to implement MPAs will likely not have the same financial resources as California (see Gleason et al., 2013b for an accounting of costs), and may not have the same fishery infrastructure, California's experience may be valuable. It seems that some level of evaluation of the interactions of proposed MPAs with locally fished species will be possible and worthwhile, rather than simply choosing a fraction of coastline to be placed in MPAs. This more comprehensive approach will enable (a) direct interaction with management of the fishing outside the MPAs and (b) direct integration of MPA design into an MPA monitoring and evaluation program, achieving true adaptive management. That type of integrated management approach is necessary for the science of MPAs to proceed.

REFERENCES

Airamé, S., Dugan, J.E., Lafferty, K.D., Leslie, H., Mcardle, D.A., Warner, R.R., 2003. Applying ecological criteria to marine reserve design: a case study from the California Channel Islands. Ecol. Appl. 13 (1), S170–S184, Supplement.

Andelman, S., Ball, I., Davis, F., Stoms, D., 1999. Sites V 1.0. An Analytical Toolbox for Designing Ecoregional Conservation Portfolios. Manual Prepared for The Nature Conservancy. National Center for Ecological Analysis and Synthesis, Santa Barbara, CA, USA.

Babcock, E.A., MacCall, A.D., 2011. How useful is the ratio of fish density outside versus inside no-take marine reserves as a metric for fishery management control rules? Can. J. Fish. Aquat. Sci. 68, 343–359.

Bakun, A., 1990. Global climate change and the intensification of coastal ocean upwelling. Science 247, 198–201.

Ball, I.R., Possingham, H.P., Watts, M., 2009. Marxan and relatives: software for spatial conservation prioritisation. In: Moilanen, A., Wilson, K.A., Possingham, H.P. (Eds.), Spatial Conservation Prioritisation: Quantitative Methods and Computational Tools. Oxford University Press, Oxford, UK, pp. 185–195.

Barry, J.P., Baxter, C.H., Sagarin, R.D., Gilman, S.E., 1995. Climate-related, long-term faunal changes in a California rocky intertidal community. Science 267, 672–675.
Barth, J.A., Menge, B.A., Lubchenco, J., Chan, F., Bane, J.M., Kirincich, A.R., McManus, M.A., Nielsen, K.J., Pierce, S.D., Washburn, L., 2007. Delayed upwelling alters nearshore coastal ocean ecosystems in the northern California current. Proc. Natl. Acad. Sci. U.S.A 104, 3719–3724.
Bergen, L.K., Carr, M.H., 2003. Marine reserves: how can science best inform policy? Environment 45 (2), 8–19.
Botsford, L.W., 1981. The effects of increased individual growth rates on depressed population size. Am. Nat. 117 (1), 38–63.
Botsford, L.W., 2013. Maximum Sustainable Yield. Oxford Bibiolographies Online, http://dx.doi.org/10.1093/OBO/9780199830060-0071.
Botsford, L.W., Castilla, J.C., Peterson, C.H., 1997. The management of fisheries and marine ecosystems. Science 277, 509–515.
Botsford, L.W., Hastings, A., Gaines, S.D., 2001. Dependence of sustainability on the configuration of marine reserves and larval dispersal distance. Ecol. Lett. 4, 144–150.
Botsford, L.W., Campbell, A., Miller, R., 2004. Biological reference points in the management of North American sea urchin fisheries. Can. J. Fish. Aquat. Sci. 61, 1325–1337.
Bray, N.A., Keyes, A., Morawitz, W.M.L., 1999. The California current system in the Southern California bight and the Santa Barbara Channel. J. Geophys. Res. 104, 7695–7714.
California Department of Fish and Game (CDFG), 2001. The Master Plan: A Ruide for the Developmnst of Fishery Management Plans. http://www.dfg.ca.gov/marine/masterplan.asp.
California Department of Fish and Game (CDFG), 2002. Nearshore Fisheries Management Plan, Chapter 2: Background. http://www.dfg.ca.gov/marine/nfmp/.
California Department of Fish and Game (CDFG), 2003. History of the Channel Islands Marine Reserve Working Group Process. https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=31320.
California Department of Fish and Game (CDFG), 2008. Marine Life Protection Act Master Plan for Marine Protected Areas. http://www.dfg.ca.gov/marine/mpa/masterplan.asp.
Carr, M.H., Reed, D.C., 2015. Shallow rocky reefs and kelp forests. In: Mooney, H., Zavaleta, E. (Eds.), Ecosystems of California. UC Press, Berkeley.
Carr, M.H., Saarman, E., Caldwell, M.R., 2010. The role of 'rules of thumb' in science-based environmental policy: California's Marine Life Protection Act as a case study. Stanf. J. Law Sci. Policy 2, 1–17.
Carr, M.H., Woodson, C.B., Cheriton, O.M., Malone, D., McManus, M.A., Raimondi, P.T., 2011. Knowledge through partnerships: integrating marine protected area monitoring and ocean observing systems. Front. Ecol. Environ. 9, 342–350.
Chan, F., Barth, J.A., Lubchenco, J., Kirincich, A., Weeks, H., Peterson, W.T., Menge, B.A., 2008. Emergence of anoxia in the California current large marine ecosystem. Science 319, 920.
Checkley, D.M., Barth, J.A., 2009. Patterns and processes in the California current system. Prog. Oceanogr. 83, 49–64.
Clark, W.G., 1991. Groundfish exploitation rates based on life history parameters. Can. J. Fish. Aquat. Sci. 48, 734–750.
Clark, W.G., 2002. $F_{35\%}$ revisited ten years later. N. Am. J. Fish. Manag. 22, 251–257.
Costello, C., Rassweiler, A., Siegel, D., De Leo, G., Micheli, F., Rosenberg, A., 2010. The value of spatial information in MPA network design. Proc. Natl. Acad. Sci. U.S.A 107, 18294–18299.
Crowder, L.B., Lyman, S.J., Figueira, W.F., Priddy, J., 2000. Source-sink population dynamics and the problem of siting marine reserves. Bull. Mar. Sci. 66, 799–820.

Di Lorenzo, E., Combes, V., Keister, J.E., Strub, P.T., Thomas, A.C., Franks, P.J.S., Ohman, M.D., Furtado, J.C., Bracco, A., Bograd, S.J., Peterson, W.T., Schwing, F.B., Chiba, S., Taguchi, B., Hormazabal, S., Parada, C., 2013. Synthesis of Pacific Ocean climate and ecosystem dynamics. Oceanography 26, 68–81.

Drake, P.T., Edwards, C.A., Barth, J.A., 2011. Dispersion and connectivity estimates along the U.S. west coast from a realistic numerical model. J. Mar. Res. 69, 1e37.

Dulvy, N.K., Rogers, S.I., Jennings, S., Steizenmuller, V., Dye, S.R., Skjoldal, H.R., 2008. Climate change and deepening of the North Sea fish assemblage: a biotic indicator of warming seas. J. Appl. Ecol. 45, 1029–1039.

Ebeling, A.W., Larsen, R.J., Alevzion, W.S., 1980. Habitat groups and island-mainland distribution of kelp-bed fishes off Santa Barbara, California. In: Power, D.M. (Ed.), Multidisciplinary Symposium on the California Islands, Santa Barbara Museum of Natural History.

Feely, R.A., Sabine, C.L., Hernandez-Ayon, J.M., Ianson, D., Hales, B., 2008. Evidence for upwelling of corrosive 'acidified' water onto the continental shelf. Science 320, 1490–1492.

Fluharty, D., 2000. Habitat protection, ecological issues, and implementation of the sustainable fisheries act. Ecol. Appl. 10, 325–337.

Food and Agriculture Organization (FAO), 1996. Precautionary approach to capture fisheries and species introductions. Elaborated by the technical consultation on the Precautionary Approach to capture fisheries (including species introductions), 6–13 June 1995, Lysekil, Sweden. FAO Tech. Guidelines Responsible Fish. No 2.

Fox, E., Miller-Henson, M., Ugoretz, J., Weber, M., Gleason, M., Kirlin, J., Caldwell, M., Mastrup, S., 2013a. Enabling conditions to support marine protected area network planning: California's Marine Life Protection Act as a case study. Ocean Coast. Manag. 74, 14e23.

Fox, E., Poncelet, E., Connor, D., Vasques, J., Ugoretz, J., McCreary, S., Monie, D., Harty, M., Gleason, M., 2013b. Adapting stakeholder processes to region-specific challenges in marine protected area network planning. Ocean Coast. Manag. 74, 24e33.

Fox, E., Hastings, S., Miller-Henson, M., Monie, D., Ugoretz, J., Frimodig, A., Shuman, C., Owens, B., Garwood, R., Connor, D., Serpa, P., Gleason, M., 2013c. Addressing policy issues in a stakeholder-based and science-driven marine protected area network planning process. Ocean Coast. Manag. 74, 34e44.

Gaines, S.D., Gaylord, B., Largier, J.L., 2003. Avoiding current oversights in marine reserve design. Ecol. Appl. 13, S32–S46.

Gaines, S.D., White, C., Carr, M.H., Palumbi, S., 2010. Designing marine reserve networks for both conservation and fisheries management. Proc. Natl. Acad. Sci. U.S.A 107, 18286–18293.

Garcia, S., 1996. The precautionary approach to fisheries and its implications for fishery research, technology and management: an updated review. FAO Fish. Tech. Pap. No. 350/2.

Garcia-Reyes, M., Largier, J., 2010. Observations of increased wind-driven coastal upwelling off central California. J. Geophys. Res. 115, C04011.

Gaylord, B., Gaines, S.D., 2000. Temperature or transport? Range limits in marine species mediated solely by flow. Am. Nat. 155, 769–789.

Gleason, M., McCreary, S., Miller-Henson, M., Ugoretz, J., Fox, E., Merrifield, M., McClintock, W., Serpa, P., Hoffman, K., 2010. Science-based and stakeholder-driven marine protected area network planning: a successful case study from north central California. Ocean Coast. Manag. 53, 52–68.

Gleason, M.G., Kirlin, J., Fox, E., 2013a. California's marine protected area network planning process: introduction to the special issue. Ocean Coast. Manag. 74, 1–2.

Gleason, M.G., Fox, E., Vasques, J., Whiteman, E., Ashcraft, S., Frimodig, A., Serpa, P., Saarman, E., Miller-Henson, M., Kirlin, J., Weber, M., Caldwell, M., Ota, B., Pope, E., Wiseman, K., 2013b. Designing a network of marine protected areas in California: achievements, costs, lessons learned, and challenges ahead. Ocean Coast. Manag. 74, 90–101.

Graham, M.B., Halpern, B.S., Carr, M.H., 2008. Diversity and dynamics of California subtidal kelp forests. In: McClanahan, T.R., Branch, G.M. (Eds.), Food Webs and the Dynamics of Marine Reefs. Oxford University Press, New York, NY, pp. 103–134.

Grantham, B.A., Chan, F., Nielsen, K.J., Fox, D.S., Barth, J.A., Huyer, A., Lubchenco, J., Menge, B.A., 2004. Upwelling-driven nearshore hypoxia signals ecosystem and oceanographic changes in the northeast Pacific. Nature 429, 740–754.

Hamilton, S.L., Caselle, J.E., Malone, D., Carr, M.H., 2010. Incorporating biogeography into evaluations of the Channel Islands marine reserve network. Proc. Natl. Acad. Sci. U.S.A 107, 18272–18277.

Hanak, E., Lund, J., Dinar, A., Gray, B., Howitt, R., Mount, J., Moyle, P., Thompson, B., 2011. Managing California's Water from Conflict to Reconciliation. Public Policy Institute of California, San Francisco, CA.

Hastings, A., Botsford, L.W., 1999. Equivalence in yield from marine reserves and traditional fisheries management. Science 284, 1537–1538.

Hastings, A., Botsford, L.W., 2003. Comparing designs of marine reserves for fisheries and for biodiversity. Ecol. Appl. 13, S65–S70.

Hastings, A., Botsford, L.W., 2006. Persistence of spatial populations depends on returning home. Proc. Natl. Acad. Sci. U.S.A 103, 6067–6072.

Helvey, M., 2004. Seeking consensus on designing marine protected areas: keeping the fishing community engaged. Coast. Manag. 32, 173–190.

Hickey, B.M., 1998. Coastal oceanography of western North America from the tip of Baja California to Vancouver Island. In: Robinson, A.R., Brink, K.H. (Eds.), In: The Sea: The Global Coastal Ocean: Regional Studies and Syntheses, vol. 11. John Wiley, New York, pp. 345–393.

Hilborn, R., Walters, C.J., 1992. Quantitative Fisheries Stock Assessment and Management: Choice, Dynamics and Uncertainty. Chapman and Hall, New York, USA.

Holland, D.S., Brazee, R.J., 1996. Marine reserves for fisheries management. Mar. Resour. Econ. 11, 157–171.

Horn, M.H., Allen, L.A., Lea, R.N., 2006. Biogeography. In: Allen, L.G., Horn, M.H. (Eds.), Ecology of Marine Fishes: California and Adjacent Waters. University of California Press, Berkeley, CA, pp. 3–25.

Hundley, N., 2001. The Great Thirst: Californians and Water, a History. University of California Press, Berkeley, CA.

Hutchings, J.A., 2000. Collapse and recovery of marine fishes. Nature 406, 882–885.

Kaplan, D.M., 2006. Alongshore advection and marine reserves: consequences for modeling and management. Mar. Ecol. Prog. Ser. 309, 11–24.

Kaplan, D.M., Botsford, L.W., 2005. Effects of variability in spacing of coastal marine reserves on fisheries yield and sustainability. Can. J. Fish. Aquat. Sci. 62, 905–912.

Kaplan, D.M., Botsford, L.W., Jorgensen, S., 2006. Dispersal per recruit: an efficient method for assessing sustainability in marine reserve networks. Ecol. Appl. 16, 2248–2263.

Kaplan, D.M., Botsford, L.W., O'Farrell, M.R., Gaines, S.D., Jorgensen, S., 2009. Model-based assessment of persistence in proposed marine protected area designs. Ecol. Appl. 19, 433–448.

Karpov, K., Haaker, P.L., Taniguchi, I., Rogers-Bennett, L., 2000. Serial depletion and the collapse of the California abalone fishery. Can. Spec. Publ. Fish. Aquat. Sci. 130, 11–24.

Kay, M.C., Lenihan, H.S., Guenther, G.M., Wilson, J.R., Miller, C.J., Shrout, S.W., 2012a. Collaborative assessment of California spiny lobster population and fishery responses to a marine reserve network. Ecol. Appl. 22, 322–335.
Kay, M.C., Lenihan, H.S., Kotchen, M.J., Miller, C.J., 2012b. Effects of marine reserves on California spiny lobster are robust and modified by fine-scale habitat features and distance from reserve borders. Mar. Ecol. Prog. Ser. 451, 137–150.
Kildow, J., Colgan, C.S., 2005. California's ocean economy: Report to the Resources Agency. National Ocean Economics Program, State of California.
Kirlin, J., Gleason, M., Ashcraft, S., Caldwell, M., Fox, E., Harty, M., Miller-Henson, M., Ota, B., Weber, M., Wiseman, K., 2013. California's Marine Life Protection Act Initiative: supporting implementation of legislation establishing a statewide network of marine protected areas. Ocean Coast. Manag. 74, 3e13.
Larkin, P.A., 1977. An epitaph for the concept of maximum sustainable yield. Trans. Am. Fish. Soc. 106, 1–11.
Leet, W.S., Dewees, C.M., Klingbeil, R., Larson, E.J., 2001. California's Living Marine Resources: A Status Report. California Resources Agency, California Department of Fish and Game, Sacramento, CA, USA.
Love, M.S., Caselle, J.E., van Buskirk, W., 1998. A severe decline in the commercial passenger fishing vessel rockfish (*Sebastes* spp.) catch in the southern California Bight 1980–1996. CalCOFI Rep. 39, 180–195.
Lubchenco, J., Palumbi, S.R., Gaines, S.D., Andelman, S., 2003. Plugging a hole in the ocean: the emerging science of marine reserves. Ecol. Appl. 13, S3–S7.
Mace, P.M., 2001. A new role for MSY in single-species and ecosystem approaches to fisheries stock assessment and management. Fish Fish. 2 (1), 2–32.
Mace, P.M., Sissenwine, M.P., 1993. How much spawning per recruit is enough? Can. Spec. Publ. Fish. Aquat. Sci. 120, 110–118.
Mangel, M., 1998. No-take areas for sustainability of harvested species and a conservation invariant for marine reserves. Ecol. Lett. 1, 87–90.
Mangel, M., Talbot, L.M., Meffe, G.K., Agardy, M.T., Alverson, D.L., Barlow, J., Botkin, D.B., Budowski, G., Clark, T., Cooke, J., et al., 1996. Principles for the conservation of wild living resources. Ecol. Appl. 6, 338–362.
Marine Reserves Working Group Science Advisory Panel, 2001. Draft Summary of the Joint Meeting of the Science Advisory Panel of the Marine Reserves Working Group/Science and Statistical Committee ad hoc Marine Reserve Committee. http://www.pcouncil.org/bb/2001/1101/Ex_F.1.c_Supp_MRWG_Sci_PanRep_Nov2001BB.pdf.
McArdle, D.A., 1997. California Marine Protected Areas. California Sea Grant College System, La Jolla, California, Publication No. T-039.
Merrifield, M., McClintock, W., Burt, C., Fox, E., Gleason, M., Serpa, P., Steinback, C., 2013. MarineMap: a web-based platform for collaborative marine protected area planning. Ocean Coast. Manag. 74, 67e76.
Micheli, F., Saenz-Arroyo, A., Greenley, A., Vazquez, L., Espinoza Montes, J.A., Rossetto, M., De Leo, G.A., 2013. Evidence that marine reserves enhance resilience to climatic impacts. PLoS One 7, e40832.
Mitarai, S., Siegel, D.A., Watson, J.R., Dong, C., McWilliams, J.C., 2009. Quantifying connectivity in the coastal ocean with application to the Southern California Bight. J. Geophys. Res. 114, C10026.
Moffitt, E.A., Botsford, L.W., Kaplan, D.M., O'Farrell, M.R., 2009. Marine reserve networks for species that move within a home range. Ecol. Appl. 19, 1835–1847.
Moffitt, E.A., White, J.W., Botsford, L.W., 2011. The utility and limitations of size and spacing guidelines for designing marine protected area (MPA) networks. Biol. Conserv. 144, 306–318.

Moffitt, E.A., White, J.W., Botsford, L.W., 2013. Accurate assessment of marine protected area success depends on metric and spatiotemporal scale of monitoring. Mar. Ecol. Prog. Ser. 489, 17–28.

Murray, S.N., Ambrose, R.F., Bohnsack, J.A., Botsford, L.W., Carr, M.H., Davis, G.E., Dayton, P.K., Gotshall, D., Gunderson, D.R., Hixon, M.A., Lubchenco, J., Mangel, M., MacCall, A., McArdle, D.A., Ogden, J.C., Roughgarden, J., Starr, R.M., Tegner, M.J., Yoklavich, M.M., 1999. No-take reserve networks: sustaining fishery populations and marine ecosystems. Fisheries 24, 11–25.

Neubert, M.G., 2003. Marine reserves and optimal harvesting. Ecol. Lett. 6, 843–849.

Ocean Science Trust (OST), 2013. State of the Central California Coast: Results from Baseline Monitoring of Marine Protected Areas from 2007–2012. http://oceanspaces.org/sites/default/files/cc_results_report.pdf.

Osmond, M., Airame, S., Caldwell, M., Day, J., 2010. Lessons for marine conservation planning: a comparison of three marine protected area planning processes. Ocean Coast. Manag. 53, 41–51.

Pacific Fishery Management Council, 2001. Exhibit F.1: Status of Marine Reserve Proposals for the Channel Island National Marine Sanctuary. http://www.pcouncil.org/resources/archives/briefing-books/november-2001-briefing-book/#marine.

Paddack, M.J., Estes, J.A., 2000. Kelp forest fish populations in marine reserves and adjacent exploited areas of central California. Ecol. Appl. 10, 855–870.

Parma, A.M., NCEAS Working Group on Population Management, 1998. What can adaptive management do for our fish, forests, food, and biodiversity? Integr. Biol. 1, 16–26.

Perry, A.L., Low, P.J., Ellis, J.R., Reynolds, J.D., 2005. Climate change and distribution shifts in marine fishes. Science 308, 1912–1915.

Pikitch, E.K., Santora, C., Babcock, E.A., Bakun, A., Bonfil, R., Conover, D.O., Dayton, P., Doukakis, P., Fluharty, D., Heneman, B., Houde, E.D., Link, J., Livingston, P.A., Mangel, M., McAllister, M.K., Pope, J., Sainsbury, K.J., 2004. Ecosystem-based fishery management. Science 305, 346–347.

Pinsky, M.L., Worm, B., Fogarty, M.J., Sarmiento, J.L., Levin, S.A., 2013. Marine taxa track local climate velocities. Science 341, 1239–1242.

Poloczanska, E.S., Brown, C.J., Sydeman, W.J., Kiessling, W., Schoeman, D.S., Moore, P.J., Brander, K., Bruno, J.F., Buckley, L.B., et al., 2013. Global imprint of climate change on marine life. Nat. Clim. Chang. 1, 919–925.

Pondella, D.J., Gintert, B.E., Cobb, J.R., Allen, L.G., 2005. Biogeography of the nearshore rocky-reef fishes at the southern and Baja California islands. J. Biogeogr. 32, 187–201. http://dx.doi.org/10.1111/j.1365-2699.2004.01180.x.

Possingham, H., Ball, I., Andelman, S., 2000. Mathematical methods for identifying representative reserve networks. In: Ferson, S., Burgman, M. (Eds.), Quantitative Methods for Conservation Biology. Springer-Verlag, New York, pp. 291–305.

Ralston, S., 1998. The status of federally managed rockfish on the US West Coast. In: Yoklavich, M.M. (Ed.), Marine Harvest Refugia for West Coast Rockfish: A Workshop, pp. 6–16, NOAA-TM-NMFS-SWFSC-255, La Jolla, CA.

Ralston, S., 2002. West coast groundfish harvest policy. N. Am. J. Fish Manag. 22, 249–250.

Rassweiler, A., Costello, C., Hilborn, R., Siegel, D.A., 2014. Integrating scientific guidance into marine spatial planning. Proc. R. Soc. B281, 20132252.

Reed, D.C., Rassweiler, A., Carr, M.H., Cavanaugh, K.C., Malone, D.P., Siegel, D.A., 2011. Wave disturbance overwhelms top-down and bottom-up control of primary production in California kelp forests. Ecology 92, 2108–2116.

Restrepo, V.R., Powers, J.E., 1999. Precautionary control rules in US fisheries management: specification and performance. ICES J. Mar. Sci. 56, 846–852.

Restrepo, V.R., Thompson, G.G., Mace, P.M., Gabriel, W.L., Low, L.L., MacCall, A.D., Methot, R.D., Powers, J.E., Taylor, B.L., Wade, P.R., Witzig, J.F., 1998. Technical Guidance on the Use of Precautionary Approaches to Implementing National Standard 1 of the Magnuson-Stevens Fishery Conservation and Management Act. NOAA-TM-NMFS-F/SPO-31.
Rosenberg, A., Mace, P., Thompson, G., Darcy, G., Clark, W., Collie, J., Gabriel, W., MacCall, A., Methot, R., Powers, J., Restrepo, V., Wainwright, T., Botsford, L., Hoenig, J., Stokes, K., 1994. Scientific review of Definitions of Overfishing in U.S. Fishery Management Plans, NOAA Tech. Memo. NMFS-F/SPO-17, 205 p.
Saarman, E., Gleason, M., Ugoretz, J., Airamé, S., Carr, M., Fox, E., Frimodig, A., Mason, T., Vasques, J., 2013. The role of science in supporting marine protected area network planning and design in California. Ocean Coast. Manag. 74, 45–56.
Sayce, K., Shuman, C., Connor, D., Reisewitz, A., Pope, E., Miller-Henson, M., Poncelet, E., Monie, D., Owens, B., 2013. Beyond traditional stakeholder engagement: public participation roles in California's statewide marine protected area planning process. Ocean Coast. Manag. 74, 57e66.
Scholz, A.J., Bonzon, K., Fujita, R., Benjamin, N., Woodling, N., Black, P., Steinback, C., 2004. Participatory socioeconomic analysis: drawing on fisher men's knowledge for marine protected area planning in California. Mar. Policy 28, 335e349.
Scholz, A., Steinback, C., Kruse, S., Mertens, M., Silverman, H., 2011. Incorporation of spatial and economic analysis of human-use data in the design of marine protected areas. Conserv. Biol. 25, 485e492.
Schroeter, S.C., Reed, D.C., Kusher, D.J., Estes, J.A., Ono, D.S., 2001. The use of marine reserves in evaluating the dive fishery for the warty sea cucumber (Parastichopus parimensis) in California, U.S.A. Can. J. Fish. Aquat. Sci. 58, 1773–1781.
Sladek Nowlis, J., Roberts, C., 1999. Fisheries benefits and the optimal design of marine reserves. Fish. Bull. 97, 604–616.
Snyder, M.A., Sloan, L.C., Diffenbaugh, N.S., Bell, J.L., 2003. Future climate change and upwelling in the California current. Geophys. Res. Lett. 30, 1823.
Starr, R.M., Cope, J.M., Kerr, L.A., 2002. Trends in Fisheries and Fishery Resources Associated with the Monterey Bay National Marine Sanctuary from 1981–2000. California Sea Grant College Program, University of California, San Diego, La Jolla, CA, p. 156. www.montereybay.noaa.gov/research/techreports/fisherytrends.pdf.
Ueber, E., MacCall, A., 2005. The rise and fall of the California sardine empire. In: Glantz, M. (Ed.), Climate Variability, Climate Change, and Fisheries. Cambridge University Press, Cambridge UK, pp. 31–48.
Walters, C., Hilborn, R., Parrish, R., 2007. An equilibrium model for predicting the efficacy of marine protected areas in coastal environments. Can. J. Fish. Aquat. Sci. 64, 1009–1018.
Weible, C.M., 2008. Caught in a Maelstrom: implementing California marine protected areas. Coast. Manag. 36, 350–373.
White, J.W., 2010. Adapting the steepness parameter from stock-recruit curves for use in spatially explicit models. Fish. Res. 102, 330–334.
White, J.W., Botsford, L.W., Hastings, A., Largier, J.L., 2010a. Population persistence in marine reserve networks: incorporating spatial heterogeneities in larval dispersal. Mar. Ecol. Prog. Ser. 398, 49–67.
White, J.W., Botsford, L.W., Moffitt, E.A., Fischer, D.T., 2010b. Decision analysis for designing marine protected areas for multiple species with uncertain fishery status. Ecol. Appl. 20, 1523–1541.
White, J.W., Botsford, L.W., Baskett, M.L., Barnett, L.A.K., Barr, R.J., Hastings, A., 2011. Linking models and monitoring data for assessing performance of no-take marine reserves. Front. Ecol. Environ. 9, 390–399.

White, J.W., Scholz, A.J., Rassweiler, A., Steinback, C., Botsford, L.W., Kruse, S., Costello, C., Mitarai, S., Siegel, D., Drake, P.T., Edwards, C., 2013a. A comparison of approaches used for economic analysis in marine protected area planning in California. Ocean Coast. Manag. 74, 77–89.

White, J.W., Botsford, L.W., Hastings, A., Baskett, M.L., Kaplan, D.M., Barnett, L.A.K., 2013b. Transient responses of fished populations to marine reserve establishment. Conserv. Lett. 6, 180–191.

White, J.W., Schroeger, J., Drake, P.T., Edwards, C.A., 2014. The value of larval connectivity information in the static optimization of marine reserve design. Conserv. Lett. http://dx.doi.org/10.1111/conl.12097.

Wild, P.W., Tasto, R.N. (Eds.), 1983. Life History, Environment, and Mariculture Studies of the Dungeness Crab, Cancer Magister, with Emphasis on the Central California Fishery Resource. California Department of Fish and Game Fish Bulletin, Sacramento, CA, USA, p. 172.

Worm, B., Hilborn, R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennings, S., Jensen, O.P., Lotze, H.K., Mace, P.M., McClanahan, T.R., Minto, C., Palumbi, S.R., Parma, A.M., Ricard, D., Rosenberg, A.A., Watson, R., Zeller, D., 2009. Rebuilding global fisheries. Science 325, 578–585.