MARINE PROTECTED AREAS AS A PRECAUTIONARY APPROACH TO MANAGEMENT

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ABSTRACT

Various sources of uncertainty have greatly impeded the effectiveness of traditional fisheries management to assure acceptable levels of sustainability of fisheries and species populations. Marine protected areas are receiving increasing consideration and show potential as a means of contributing to the sustainability of populations and guarding against fishery failures. Marine protected areas take advantage of the open population structure that characterizes most exploited benthic marine species by considering the pelagic dispersal of propagules and the patchy distribution of benthic habitat. Because protected areas have only recently been considered for west coast fisheries, because poorly designed reserves may be useless and possibly detrimental, and because optimal design criteria are not yet understood, incorporating the evaluation of empirically derived design criteria into the final implementation of protected area networks (i.e., adaptive management) is the only prudent approach.

INTRODUCTION

A number of recent articles review the potential value of marine protected areas (PDT 1990; Roberts and Polunin 1991; Rowley 1994; Agardy 1997; Bohnsack 1998; Murray et al., in press) and caveats regarding their design and evaluation (Carr and Reed 1993; Allison et al. 1998; Carr and Raimondi 1998). This paper is designed to (1) explain the ecological basis for the potential roles of marine protected areas (MPAs) as a mechanism contributing to sustainable fisheries; (2) describe how MPAs differ from the more traditional management strategy of stock assessment, quota forecasting, and catch adjustment; and (3) emphasize the importance of thoughtful design and evaluation. This paper complements the paper by Parrish (also in this symposium section) by summarizing reasons for the rapidly increasing interest in MPAs, along with some caveats regarding their limitations.

The increasing interest in marine protected areas as a complement to traditional management strategies stems from many growing concerns. First is the rapid increase in either fishery failures or dramatic declines in many fisheries globally as well as regionally along the west coast of North America. Examples from the west coast include the once very productive abalone fishery through-out southern California, the sea urchin fishery of northern California, and the dramatically declining stocks of lingcod and some rockfish.

Simultaneously, fisheries scientists have become keenly aware of the great difficulty in accomplishing, and uncertainty associated with, each of three components of traditional management: stock assessment, forecasting stock size, and catch adjustment. Accurate stock assessment is hindered both by the great and unpredictable natural variability in year class strength and stock size, which is characteristic of most exploited species, and by the logistical difficulties of estimating stock size, catch rates, and population parameters (Ludwig et al. 1993; Clark 1996; Botsford et al. 1997; Lauck et al. 1998). The problem is exacerbated by the usual paucity of resources (financial and human) to make such estimates and the statistical artifacts of fishery-dependent sampling (e.g., in most recreational finfish and most invertebrate fisheries).

Moreover, the environment continues to change in new and unpredictable ways. Natural long-term (e.g., decadal) shifts in oceanographic conditions have only recently been recognized by marine ecologists and fisheries biologists (Roemmich and McGowan 1995; Holbrook et al. 1997; Francis et al. 1998; McGowan et al. 1998; Steele 1998). In addition, anthropogenic effects develop or change unpredictably through time; they include new pollutants, global climate change, and new fisheries that can affect the sustainability of existing fisheries (e.g., multispecies fisheries, bycatch, and practices that alter critical habitat of existing fisheries).

Another component of uncertainty is the arbitrary or unpredictable behavior of the fishing industry, consumers, and the political arena of management decisions (e.g., the rapidly developed fishery for live fishes). The ability of managers to adjust catch rates promptly can be further jeopardized by the process of policy decisions that must encompass more than harvest recommendations from fisheries biologists.

Taken together, the natural, analytical, and social causes of uncertainty in projecting stock trends and adjusting fishery yields have prompted great concern and a more conservative approach to ensuring sustainability of marine resources (Parma et al. 1998; Shea et al. 1998).

LARVAL DISPERSAL AS A KEY TO SUSTAINABILITY

Fisheries ecologists have long recognized that life-history traits are key features for determining the potential sustainability of a fishery for any species (Clark 1991; Leaman 1991). The issue is complex because of the many life-history traits that influence a population's response to exploitation.

Historically, particular attention has been given to the intrinsic growth rate (r) of species populations, which can influence how rapidly a population recovers from exploitation (Pitcher and Hart 1982). Species with high intrinsic growth rates are typically characterized by short life spans, early age of reproduction, high per capita fecundity, and other traits that may contribute to high resiliency and the ability to rapidly rebound from exploitation. However, some populations characterized by high intrinsic growth rates have not sustained high rates of exploitation (e.g., sea palm, anchoveta, Pacific sardine, market squid).

Thus, other life-history traits must contribute to sustainability. Two critical traits are longevity and the difference between age of maturity and age of recruitment to the fishery (D_{MR}). The greater the longevity and D_{MR}, the greater the potential for a "storage effect" (sensu Warner and Chesson 1985). Most marine organisms are iteroparous, so adults can "store" opportunities for successful reproduction and recruitment from one favorable period to the next simply by surviving periods of unfavorable conditions. This is likely to be particularly important in temperate systems where episodic recruitment is common among species, presumably driven by variation in environmental conditions that contribute to the successful recruitment of a year class (Parrish et al. 1981; Leaman and Beamish 1984). The strength of the storage effect as a buffer against coincident periods of overexploitation and recruitment failure will depend (in part) on (1) lifespan, (2) D_{MR}, (3) the number of mature pre-exploitable individuals, (4) size-specific fecundity, and (5) degree of compensation (degree to which early mortality is density-dependent). For example, species with high fecundity and early age of maturity (high intrinsic growth rates) but very short life spans have little or no storage effect. Extreme cases are species that live only one or two years, mature, and become exploitable within the first or second year of life. Overexploitation of any year class in combination with poor recruitment conditions potentially depletes most adults and the reproductive potential of that population for many years into the future. Squid, anchovies, and pink salmon may be classic examples of species with high intrinsic growth rates but little storage effect. Thus, intrinsic growth rate is only one of several determinants of how a population will respond to exploitation.

Another critical life-history trait is the dispersal potential of reproductive propagules (spores, gametes, eggs, larvae) of a species. Many exploited marine species have life histories that include benthic juvenile and adult stages with very limited dispersal, but whose reproductive propagules can be dispersed great distances in the plankton. If juvenile and adult stages of a species move very little, increasing the dispersal potential of reproductive propagules dramatically influences the "openness" of a population. In general, offspring produced by a local adult population are dispersed from, and contribute little to replenishment of, that parental population. Rather, a parental population replenishes populations elsewhere, and its own replenishment depends on recruitment of larvae produced elsewhere. Such "open" populations are common among marine species, including almost all exploited coastal species. (Although some evidence for local retention of pelagically dispersed larvae exists for embayments and oceanic islands, there is little evidence of this for most commercially exploited species along the open coast.) The openness of a population depends on the spatial scale on which the population is defined. For a given dispersal potential, the smaller the spatial definition of a population, the more open that population (i.e., the more likely it is that larvae are dispersed from that population). At the other extreme, on the scale of the geographic range of a species, any population is closed. This suggests that a local open population can be harvested without reducing its replenishment rate, because replenishment is derived from larvae produced elsewhere. Likewise, local open populations protected from exploitation could export larvae to replenish exploited populations, acting as a buffer against overexploitation or possibly enhancing replenishment rates, relative to conditions in which all populations are exploited. However, the replenishment of the protected local population in this example depends on input from other, presumably unprotected, populations.

Taken together, intrinsic growth rate, longevity, and the dispersal potential of propagules contribute to our understanding of sustainability among representative west coast fisheries (table 1). We consider propagule duration as a proxy for dispersal potential of a species (Reed et al., in press). Species characterized by both low dispersal potential and low intrinsic growth rates are unlikely to sustain exploitation unless very carefully managed (probably best through traditional methods). Examples include most elasmobranchs, and the viviparous surf-perches, which have relatively low fecundity, late maturity (elasmobranchs), and whose young recruit directly to local parental populations. Less extreme examples of low intrinsic growth rate and low dispersal potential include two commercially exploited gastropods that produce planktonic larvae: abalone and the turban snail,

TABLE 1
Life-History Characteristics of 14 Representative Taxa Exploited Commercially or Recreationally along the Coast of California

Taxa	Fecundity	Age at maturity (yr)	Longevity (yr)[a]	Relative "r"[b]	Propagule[c] duration (days)
Elasmobranchs	Low	Late (5–8)	Long (20)	Low	Short (0)
Surfperch	Low	Early (1)	Long (8–10)	Low	Short (0)
Sea palm	High	Early (<1)	Short (1)	High	Short (0)
Abalone	High	Late (5–7)	Long (20)	Low	Short (7)
Turban snail	High	Early (2–3)	Long (10)	Low (?)	Short (7)
Rockfishes	High	Late (7–8)	Long (20–120)	Moderate	Long (30– >100)
Lobster	High	Late (7)	Long (10–15)	Moderate	Long (270)
Cancer crabs	High	Early (2–3)	Long (8–10)	High	Long (130–150)
Bay scallops	High	Early (<1)	Long (12–25)	High	Long (20–40)
Red sea urchin	High	Early (2–3)	Long (20)	High	Long (37)
Anchovies	High	Early (1)	Short (5)	High	Long (50–60)
Sardines	High	Early (1)	Short (5)	High	Long (50–60)
Market squid	High	Early (<1)	Short (1–2)	High	Unknown
Pacific oysters	High	Early (1)	Long (25–30)	High	Long (28)

[a]Longevity is defined as the average life span of adults.
[b]"r" refers to intrinsic growth rate.
[c]Propagule refers to the life-history phase that is postfertilization and premetamorphic to adult form.

Lithopoma (= *Astraea*) (Tegner 1993; table 1). Fisheries for at least abalone have been difficult to sustain in the United States. In these species, high fecundity is offset by high larval and juvenile mortality.

In contrast, many exploited species characterized by relatively low or moderate intrinsic growth rates have high larval dispersal. Representative species include some rockfishes of the genus *Sebastes*, and lobster. Their low or moderate intrinsic growth rates are caused by a balance between late maturity and high fecundity. Though several rockfish populations are at historically low abundances, some species have sustained exploitation. Past sustainability may be the consequence of both high dispersal potential and great longevity (i.e., storage effect).

High intrinsic growth rate alone does not insure sustainability. Species with a potential for a high intrinsic growth rate but only limited dispersal have also suffered from heavy local exploitation. One example is the sea palm (*Postelsia palmata*), an intertidal alga with high fecundity and early maturity but extremely limited spore dispersal. *Postelsia* was exploited for only a short period before it became evident that the level and timing of harvest was having devastating effects on local populations (Dawson and Foster 1982).

Finally, there are species with both high intrinsic growth rate and high dispersal potential. Representatives include historically heavily exploited species such as sea urchins and some pelagic species such as anchovies and sardines. Though fisheries for these species have experienced great variability, including marked declines in the face of heavy exploitation, they have generally tolerated exploitation for several decades. The dramatic variability in the size of these exploited stocks over time, particularly in those with both high intrinsic growth and high potential for propagule dispersal, is believed to be caused by variation in the conditions for dispersal and survival of the early pelagic stages (Sinclair 1988). This observation reinforces the importance of the pelagic dispersive stage to the persistence of local populations and points to a need to manage adult populations so that recruitment failures do not necessarily lead to a long-term collapse of the fishery.

We suggest that the potential for larval dispersal is related to, and can contribute importantly to, the local sustainability of exploitation of a species—perhaps as importantly as the intrinsic growth rate of a species' population. Though traditional fisheries management strategies have indirectly taken advantage of larval dispersal, it has not been incorporated into the design of management approaches. In contrast, marine protected areas focus on this critical life-history trait, with the intention of using it as a means to insure against the uncertainties of exploitation and to contribute to the replenishment (sustainability) of exploited populations outside reserves.

THE SPATIAL STRUCTURE OF BENTHIC MARINE POPULATIONS

The recent surge in the theoretical and conceptual development of spatially structured population and community models in terrestrial ecology is thought to be one of the most important conceptual advances in population ecology and conservation biology since McArthur and Wilson's theory of island biogeography. Terrestrial ecologists have recently come to recognize how spatial structure and movement of individuals among local populations influences the persistence of both local and regional collections of local populations (i.e., source-sink and metapopulation models; Pulliam 1988; Gilpin and Hanski 1997). This development has had a tremendous influence on recent approaches to managing human effects on terrestrial species and environments.

The patchy distribution of reef habitats (usually along discrete isobaths) and current regimes contribute to a spatial structure with high connectivity among local populations. Though some traditional management strategies have explicitly considered this spatial structure, particularly at very large spatial scales (Sinclair 1988; Hilborn and Walters 1992, chap. 5), many continue to assume that stocks are largely homogeneously distributed, or that the scale of fishing regulations (zones) is appropriate to the spatial scale of reef-associated populations. These assumptions are partly evident from the way fisheries data are collected and stocks are assessed for fisheries in which catch records are based on arbitrarily defined "blocks." Such blocks encompass unknown numbers of local reef-based populations whose larval replenishment can be largely reliant on populations from other blocks. Differences in yield among local populations within a block are not distinguished, nor is the extent and effect of exploitation on these populations. In particular, many recreational and some commercial fisheries for nearshore reef-associated species continue to be managed this way, and stock assessments are based on port landings and fishing logs.

The conceptual basis of marine protected areas explicitly recognizes the spatial structure of populations and the critical role of larval dispersal among local populations. The decoupling of local adult fecundity from the replenishment of a parental population implies that exploitation of a local population may not reduce the rate at which that parental population is replenished (with the critical exception of species whose recruitment is increased by the presence of adults, such as the red sea urchin). Likewise, protecting a local population may not enhance its rate of replenishment, especially if recruitment is density-dependent, but can affect the replenishment of populations to which its larvae disperse. Thus a protected area sufficiently large to encompass larval dispersal, or a network of protected areas that replenish one another via larval dispersal, not only preserves populations within the protected area, but may also contribute to the replenishment of exploited populations outside the area. The realization that replenishment, and hence sustainability, of protected populations can be largely reliant on larvae produced outside the protected area is the rationale for a network of protected areas, and for continued management of the exploited populations to guarantee a level of larval production high enough to prime protected populations.

HOW MARINE PROTECTED AREAS MAY CONTRIBUTE TO MANAGEMENT

By emphasizing the critical role of dispersal potential and the spatial structure of marine populations, marine protected areas may contribute to the sustainability of a fishery in many ways. The most comprehensive reviews of the various ways that marine protected areas may contribute to fisheries management and create nonfishery benefits have been made by the Plan Development Team (PDT 1990) and Bohnsack (1998). Of a list of 21 benefits (PDT 1990), we present a handful to convey the breadth of possible benefits.

- Protection of spawning stocks enhances rates of replenishment of exploited populations, thereby countering the effects of recruitment overfishing.

- Emigration of older juveniles and adults from protected populations to exploited populations increases the average size of harvested individuals, thereby countering growth overfishing.

- The spawning potential of large numbers of adults in protected populations buffers regional populations from environmental variability, hastening the replenishment and recovery of populations after perturbations (especially the combined effects of exploitation and natural disturbances).

- Eliminating fishing activities and local sources of pollution preempts destruction of essential fish habitat and provides pristine habitat as a baseline for research and for assessing the effects of tourism and other nonconsumptive activities.

- The genotypes of protected populations are subjected to natural selective factors that maintain natural size and social structure, age at maturity, and genetic diversity. Because their larvae disperse to exploited populations, protected populations may counter the genetic effects of exploitation. Moreover, the decoupling of local reproduction and recruitment may prevent the deleterious effects of inbreeding (e.g., depression and genetic drift) which have plagued reserve efforts for terrestrial species with closed populations.

THE IMPORTANCE OF DESIGN AND EVALUATION

When possible, the degree to which any approach to fisheries management and conservation meets its intended objectives should be evaluated. The costs of any approach that fails to meet its objectives may be catastrophic, depending on how much managers rely on the success of that approach. Because of the many possible objectives identified for marine reserves and the early stages of development of this approach, the effectiveness of reserves in meeting each objective is not entirely clear. Moreover, reserves are vulnerable to external perturbations (e.g., pollution, invasive species, climate change) that might impair their effectiveness. Therefore, developing approaches to evaluate the effectiveness of a reserve (for a given objective) and an overall framework for responding to an evaluation must be developed.

Some aspects of the uncertainty that plagues traditional approaches to fisheries management also pertain to marine protected areas. For example, if the intention is to measure a fishery's response to the establishment of a protected area, using traditional assessment data will continue to be problematic. The statistical power to detect changes in the trajectory of stock size over time is often depressingly low. Examples for west coast groundfish come from recent assessments of two rockfishes— bocaccio (*Sebastes paucispinis*) and canary rockfish (*Sebastes pinniger*)—and the lingcod (*Ophiodon elongatus*; PFMC 1996; fig. 1). An analysis of statistical power (Lenth 1987) indicates that the year-to-year variability in abundance of these species requires a 50% increase in lingcod abundance, a 100% increase in canary rockfish, and an increase of 500,000 bocaccio (200%) from 1996 to 2000 to detect a significant increase with acceptable levels of error ($\alpha = 0.10$) and power ($1 - \beta = 0.80$). Thus, other methods for assessing the consequences of a protected area must be developed.

Likewise, detecting increases in rates of larval replenishment (recruitment to benthic populations rather than to the fishery) may be difficult because of the dilution of recruits over vast areas of larval dispersal (also difficult is identifying areas of recruitment).

Nonetheless, methods are available for determining whether areas closed to fishing are effective at protecting populations and assemblages within a protected area (i.e., detecting increases in density and average size). Recently developed approaches to detecting genetic signatures, the genetic structure of marine populations, and patterns of larval dispersal seem more promising than previously thought (Doherty et al. 1995; Herbinger et al. 1997; Shaklee and Bentzen 1998; Bilodeau et al. 1999; Davies et al. 1999). Also, the elemental microchemistry of otoliths of fish and statoliths of invertebrate larvae may prove valuable for tracing the dispersal of larvae, identifying areas of larval sources, and constructing patterns of connectivity among local populations (Swearer et al., in press). Such efforts to determine patterns of larval dispersal are fundamental to both the design and evaluation of marine protected areas. Taken together, these methods can provide evidence of increased spawning stocks within reserves and the export of larvae to exploited stocks beyond reserve boundaries.

Given some level of uncertainty in the performance or evaluation of protected areas, designing them in a way that makes it possible to evaluate differences in design criteria seems well worth consideration. For example, protected areas that differ in size, spacing, local oceanographic features, or methods of enforcement could be established and monitored to determine how such differences influence the sustainability of protected populations or assemblages. Such evaluation requires well-designed
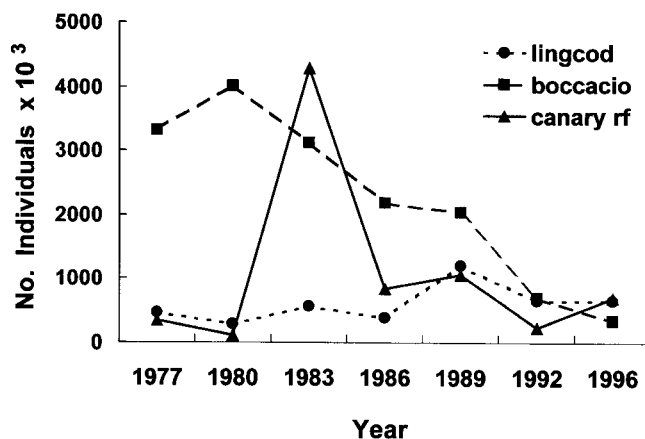

Figure 1. Estimated trends of abundance of three commercially exploited west coast groundfishes. Data are from fishery-independent triennial trawl surveys (PFMC 1996).

monitoring programs intended and funded for the many years necessary for effects of protection to be realized. Incorporating evaluation and the ability to respond to the result of such evaluation into reserve design (i.e., adaptive management) could foster the development of optimal designs for existing and future reserves.

Such an approach would benefit from replication of design criteria, but establishing independent replicate reserves of each criteria level may be difficult, given the potential size of reserves and the spacing necessary to assure independence. One alternative analytical approach is the before-after-control-impact (BACI) design often used for point source environmental impact assessment (Schmitt and Osenberg 1996). Instead of spatial control-impact comparisons, the BACI design calls for comparisons between reserves differing in design criteria before and after establishment. However, such statistical approaches are designed to assess the effects of only one criterion. Another valuable approach would be to compare the response of exploited species with that of unexploited species (controls) to distinguish reserve effects from those caused by natural environmental variation over time. This approach may be problematic if the responses of unexploited and exploited species are not independent of one another. As Parma et al. (1998) state, "Experiments are not free, but neither is ignorance; some short-term sacrifices may be worthwhile if they can lead to better management in the future."

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