

## SHORT- AND LONG-TERM EFFECTS OF THREE FISHERY-MANAGEMENT TOOLS ON DEPLETED FISHERIES

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### ABSTRACT

Marine reserves have come under criticism because of the short-term fishery losses likely to be associated with them. At the same time, marine reserves have been touted as a tool to rehabilitate depleted populations, at least for species with relatively limited adult movement. I used models to compare the short- and long-term fisheries consequences of three fishery-management tools for depleted populations. These management tools included temporary closure of the entire fishery, minimum size limits, and marine reserves. I compared them using three short-term indices: magnitude of initial drop in catches relative to those prior to new management, years until catches reached prior levels, and cumulative loss during those years. I examined a single long-term index: long-term sustainable yields. Results highlighted the potential of reserves as an efficient and effective fishery-management tool for species that will remain within reserve boundaries. Reserves created few short-term losses beyond those associated with other management measures, yet produced the highest stable catch levels. Moreover, peak catches with reserves occurred with less restriction than peak catches with other management measures. These results were consistent across two species that matured before entering the fishery but did not apply to one species that was fished while immature. In that latter case, minimum size limits produced more substantial benefits than reserves could. Nevertheless, these analyses suggest that a wide range of circumstances exist where reserves, if properly designed to minimize adult spillover while allowing abundant larval transport, can maximize fisheries harvests with a minimum of total restrictions.

Fisheries on wild stocks are in decline worldwide. According to a United Nations report, two-thirds of all commercial fisheries are fished beyond capacity or in danger of becoming so (Food and Agriculture Organization, 1995). In the United States, one-third of assessed species are classified as overfished or approaching overfished. More disturbingly, we do not know the status of over 60% of all managed stocks in the United States (National Marine Fisheries Service, 1998). It is particularly disturbing that these failures have occurred in industrialized countries like the United States with resources for science, management, and enforcement. In countries lacking these resources, many traditional fishery-management tools are simply unavailable.

In response to this growing problem, fisheries managers are showing a willingness to consider a broader range of tools. These nontraditional tools include no-take marine reserves, areas where fishing is prohibited indefinitely. Closed-area management is not a new idea; theory and examples stretch back decades (e.g., Beverton and Holt, 1957). Field studies have shown repeatedly that the number and average size of fished species increase within these closed areas, as does the total number of species (Roberts and Polunin, 1991; Dugan and Davis, 1993; Rowley, 1994; Bohnsack, 1996; and references therein). These changes alone offer fishery benefits by protecting vulnerable species and serving as insurance against management failure. In addition, theory suggests that reserves, if properly designed, can increase overall fish catches despite the loss of fishing area and can stabilize annual fluctuations in catches (Sladek Nowlis and Roberts, 1999, and references therein). These theoretical benefits remain unproven in the field, although existing

studies are suggestive (Alcala and Russ, 1990; McClanahan and Kaunda-Arara, 1996; Russ and Alcala, 1996).

Recently, reserves have received a great deal of attention, more than other forms of fishery management. The Plan Development Team (1990) qualitatively compared reserves to other management tools. The present study is an attempt to do so quantitatively. I assessed three management tools: marine reserves (protection as a function of space), minimum size limits (protection as a function of size), and temporary closures (protection as a function of time). Specifically, I used computer models to assess the ability of these tools to promote recovery of depleted populations. Because models are necessarily simplifications of reality, the results here would benefit from field testing.

## METHODS

I built three population models, one for each of three fishery management tools: temporary closure of the entire fishery, minimum size limits that reduced fishing mortality for the smallest fish, and no-take marine reserves. These models were mathematically based computer simulations and were general enough to adapt to a variety of fishery species. Using life-history parameters and management strategies, each model generated a time series of predicted catches, as biomass caught per year. These series were first run to stability without new management. Then the new management tool was enacted and the series continued until catches had restabilized.

All models shared fundamental biological assumptions (see Appendix for a detailed mathematical formulation), including a stock-recruitment relationship in which the rate of recruitment was evenly distributed and a function of the adult population biomass averaged over the entire management area. This relationship was density dependent (negative exponential) and assumed that forces such as competition or predation limited recruitment at high densities. All models were deterministic and included no built-in environmental variability. I also assumed that, before new management, each population had been reduced by fishing to population densities lower than those that would produce maximum sustainable yields, and I will refer to these conditions as depleted.

Each model had additional assumptions based on the specific management tool being tested. In the model examining temporary closures of the entire fishery, I assumed that, upon reopening, fishing rates were equal to those before closure. This assumption may not be met. If the closure is successful and produces a build-up in fish biomass, effort is likely to increase dramatically upon reopening, but the closure may also drive some fishermen and women out of business or into other fisheries, reducing effort after reopening. Because of these uncertainties, I chose an assumption of no overall effect.

In the model examining minimum size limits, I assumed that enactment of the limits eliminated fishing mortality for the smallest fish in the fishery and that this loss did not displace fishing effort to the still-available larger fish. These assumptions imply that the limits reduced fishing effort. It would in fact be sensible, in light of the depleted status of the populations I examined, for managers to enact regulations designed to reduce fishing effort in this way, probably by reducing quotas when size limits are raised.

In the model examining no-take marine reserves, I assumed that adult fish did not cross reserve boundaries, remaining for life in the areas where they settled. Conversely, I assumed that larvae dispersed widely across the reserve boundary, creating equal rates of settlement in all areas. These movement assumptions will not fit every species in every circumstance, but they are reasonable for the many species with restricted adult movement and long-lived larvae, including invertebrates and bottom-associated fish, especially if the system of reserves is designed with adult and larval movement patterns in mind (Sladek Nowlis and Yoklavich, 1998; Sladek Nowlis and Roberts, 1999). For this model, I further assumed that fishing mortality in the remaining fishing grounds did not change as a result of the closure. As in the previous model, I assumed that any tendency toward a compen-

satory increase in effort outside the reserves would be counteracted by additional regulations, such as reductions in quotas. Some examples suggest, however, that effort per unit fishing area need not change with the creation of even a very large reserve, even in the absence of additional management measures (McClanahan and Kaunda-Arara, 1996).

The assumptions were designed to create comparable levels of restriction across management tools. Specifically, a 20% reserve would protect that proportion of the fish population and thus lead to a 20% reduction in catch during the first year. Similarly, a 20% size limit would allow the smallest 20% of the catch biomass to escape, producing a 20% reduction in catch during the first year. I could not create a comparable level of restriction for temporary closures, and instead equated a 10-yr closure with 100% restriction. Future extensions of this work will consider additional management tools, including general effort reductions, which affect all size classes equally, and seasonal/temporal closures in which the fishery alternately opens and closes.

I monitored the models' performances in both short-term and long-term catches (biomass caught per year). Each model run began at an arbitrary point and ran until catches stabilized, at which point the new management tool was enacted. The models continued to monitor catches until they restabilized (Fig. 1). To compare strategies, I had to simplify these catch histories into measures of short- and long-term performance. I used the standard long-term sustainable yield as a measure of long-term performance, a technique used by several other modeling studies of marine reserves (Sladek Nowlis and Roberts, 1999, and references therein). Because no such standard index exists to measure short-term performance, I examined three: magnitude of initial drop in catches relative to those prior to new management, years until catches reached prior levels, and cumulative loss during those years. This approach is consistent with but not identical to those taken previously (e.g., by Hightower and Grossman, 1987; Sladek Nowlis and Roberts, 1997). Together, these measures provided a good indication of, respectively, the magnitude of initial catch losses, the duration of losses, and the total loss.

To compare these results generally, I produced grids of graphs that allowed visual comparison of the effects of four factors simultaneously. Within each grid, dependent variables included:

- Species, including *Panulirus penicillatus*, the Red Sea spiny lobster; *Haemulon plumieri*, the white grunt; and *Epinephelus guttatus*, the red hind. The species varied in two key respects: population growth rate ( $\lambda = 1.08, 1.16$ , and  $1.31$ , respectively) and whether they entered the fishery before (the lobster) or after (the red hind and white grunt) sexual maturity. Parameter values for all of these species are listed by Sladek Nowlis and Roberts (1999);
- Fishing rates, at three levels from lightly to heavily depleted;
- Management tool, including no-take marine reserves, minimum size limits, and temporary closures; and
- Degree of restriction, ranging from 0 to 1 (0 represented no new restriction and 1 represented complete, permanent closure). Temporary closures were handled slightly differently, as explained earlier in the methods.

Each graph shows results for a different measure of performance, including long-term sustainable catch levels, initial losses (as a proportion of the catch prior to new management), number of years until catches exceeded those prior to new management, and the cumulative amount of catch loss during those years (expressed in terms of annual catch rates prior to new management).

## RESULTS

Each run of the models generated a catch history, starting with the last year before new restrictions were enacted (year 0) and running until catches had reached stability. These catch histories had a characteristic shape for most runs; catches initially dropped as a result of the new restrictions and eventually rose to levels higher than those prior to new management (Fig. 1). The magnitude and duration of the initial drop, as well as level of

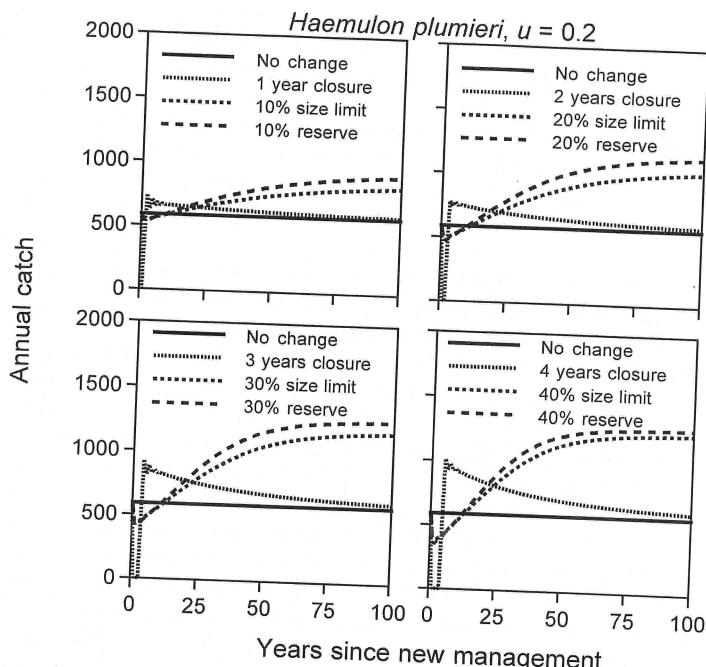


Figure 1. Examples of catch histories. These were for the white grunt, *Haemulon plumieri*, with adult fish facing 20% annual fishing mortality rate ( $u = 0.2$ ). Each graph represents a different degree of restriction and shows annual catches graphed over 100 yrs from when new management measures were enacted. The model actually ran longer before catches stabilized. All histories show initial declines after enactment of new management, followed by some kind of recovery. The magnitude and duration of initial losses and the level of ultimate recovery depended on the species, fishing mortality rate, and the type and degree of restriction.

stable catch, varied from run to run depending on several key variables, including the species, fishing rate, and degree and type of restriction (Fig. 1). When I examined the effect of these factors on the catch profile characteristics, some interesting patterns emerged.

Stable catch rates did not change with the duration of temporary closures. Instead, catches always returned to their pre-enactment levels (Fig. 2), as would be expected because effort after reopening matched levels prior to the closure. In contrast, when size limits or no-take reserves were used, catches increased unless restrictions were excessive. For example, when reserves were used for the white grunt at a fishing mortality rate of  $u = 0.2$ , catches peaked at a reserve size encompassing approximately 30% of the management area (Fig. 2). If reserves encompassed more than 80%, catches never rebuilt to pre-enactment levels. Size limits had a similar effect, but peak catches occurred under greater restrictions (e.g., 45% for the white grunt at a fishing mortality rate of  $u = 0.2$ ), and peak yield was slightly lower than that for reserves (Fig. 2). These patterns were consistent across all fishing rates for both the white grunt and the red hind (Fig. 2), but the spiny lobster, which was fished before reaching maturity, showed greater catch increases under size limits than through no-take reserves (Fig. 2).

Initial catch losses were directly related to the definition of restriction, so the results were very straightforward. These initial losses showed the same pattern for all species

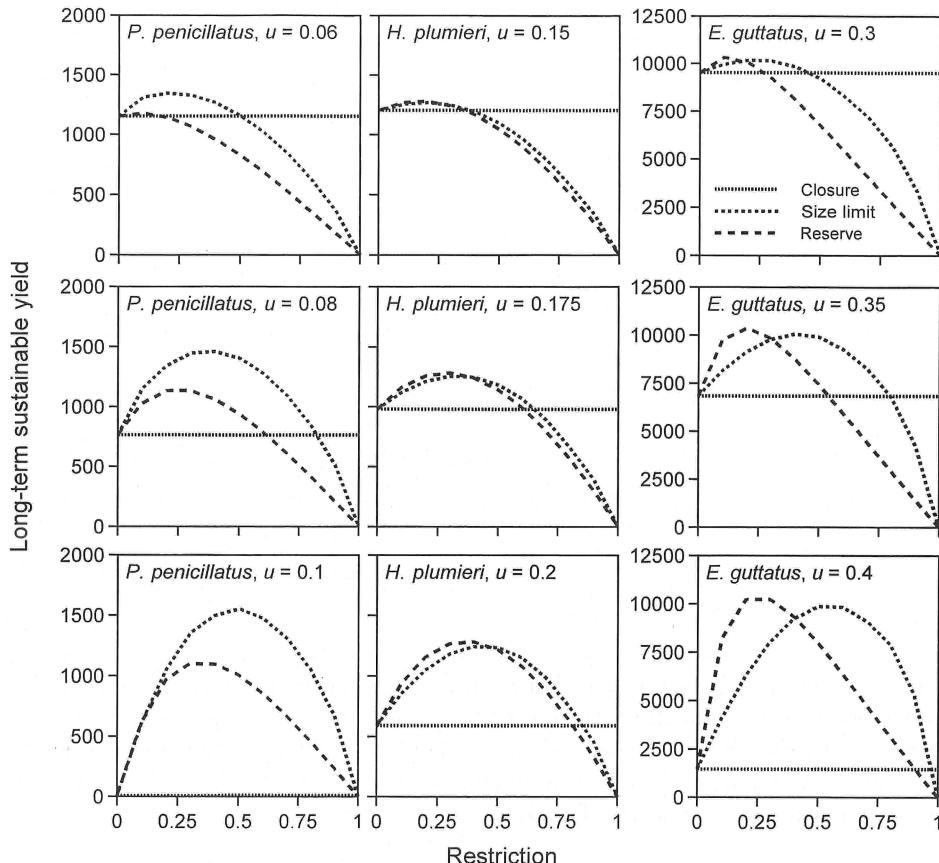


Figure 2. Long-term sustainable yields. Enhancement, or the degree to which the stable yields exceeded those prior to new management, depended on the species, fishing mortality rate, and the type and degree of restriction. Enhancements were greatest at high fishing rates but required greater restrictions under these conditions. For the two species in which individuals were not vulnerable to fishing until they had reached sexual maturity (the white grunt, *Haemulon plumieri*, and the red hind, *Epinephelus guttatus*), reserves produced greater enhancements than size limits and did so under lighter restrictions. For the spiny lobster, *Panulirus penicillatus*, reserve enhancements peaked under lighter restrictions, but this peak was lower than that resulting from size limits. Temporary closures did not produce any long-term catch benefits because of the lack of new restrictions at the reopening of the fishery.

and all fishing rates. Under temporary closures, the entire first-year catch was lost, whereas under size limits or no-take reserves, an amount of catch equal to the degree of restriction was lost (Fig. 3).

The duration of loss was more complex. In all cases, losses due to temporary closures lasted as long as the closure. When the fishery was reopened, fish biomass was higher than before new management, so catches exceeded those prior to new management in the first year of fishing (Fig. 4). Reserves and size limits took longer to show overall enhancements (Fig. 4). For the spiny lobster, losses lasted longest with no-take reserves (Fig. 4). For the other two species, duration of loss was roughly equal for reserves and size limits (Fig. 4).

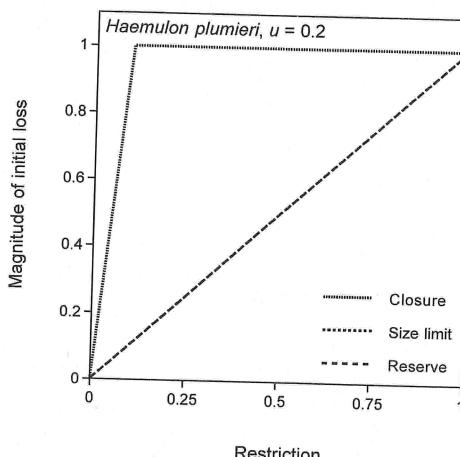


Figure 3. Magnitude of initial loss. The magnitude of initial loss depended on the type and degree of restriction but was consistent across all species and fishing mortality rates. For temporary closures, the entire catch was lost during the time of closure. For size limits and reserves, losses were equal to the degree of restriction because restriction was defined in terms of initial loss (the lines for these two measures overlapped completely).

Cumulative losses synthesize the magnitude and duration of losses, and thus might be the best of the short-term measures. For all three species, cumulative losses were greater for temporary closures than for size limits or no-take reserves at equivalent degrees of restriction. For the white grunt and red hind, size limits and marine reserves showed equivalent cumulative losses (Fig. 5). For the spiny lobster, size limits produced fewer cumulative losses than no-take reserves (Fig. 5).

#### DISCUSSION

Except where juveniles were vulnerable to fishing, marine reserves provided greater long-term catches and did so with less restriction than other management tools. Reserves allowed a subgroup of fish to grow much larger and achieve high reproductive output. In contrast, minimum size limits freed small fish from fishing mortality for a relatively short period of time. The increased reproductive output from these small fish did not compare with the output from larger individuals within reserves. Temporary closures did not provide any long-term catch enhancements.

In the short term, reserves also fared well. Initial losses were less severe with reserves and size limits than with temporary closures, although recovery times were longer. Cumulatively, reserves fared as well, per unit restriction, as any other tool for species where only adults were fished. For species where juveniles were susceptible to fishing, reserves did have higher cumulative losses than size limits at similar levels of restriction. However, reserves produced maximal benefits with less restriction than size limits. Because reserves required fewer restrictions and because losses increased with degree of restriction, reserves are likely to entail more modest short-term losses than other management tools, at least for species where juveniles are not fished. For species where juveniles are caught, these results suggest that size limits might be a better first choice as a tool for

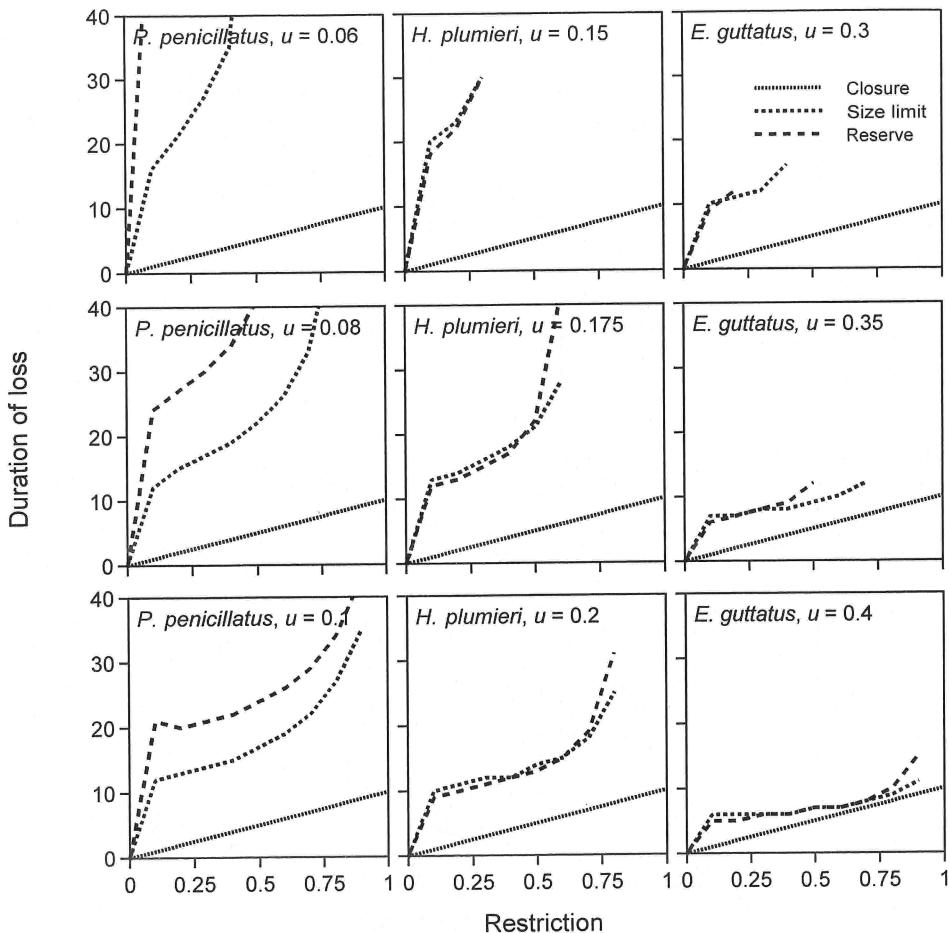


Figure 4. Duration of loss. The duration of initial loss depended on the species, fishing mortality rate, and the type and degree of restriction. Losses were generally greatest for the slowest-growing species and at the lowest fishing mortality rates. For the two species in which individuals were not vulnerable to fishing until they had reached sexual maturity (the white grunt, *Haemulon plumieri*, and the red hind, *Epinephelus guttatus*), cumulative losses were similar with reserves and size limits; both were somewhat higher than losses due to temporary closure of the entire fishery. For the spiny lobster, *Panulirus penicillatus*, losses were greatest with reserves and least for temporary closure of the entire fishery.

rehabilitating depleted populations. For species where juveniles are not caught, reserves may provide the greatest long-term benefits with the fewest short-term losses.

These results are fundamentally affected by the assumptions of the models, which are in turn central to the questions I was interested in asking and to the results I present. Had I assumed no relationship between adult density and recruitment, there would have been little incentive in the model environment to conserve adult fish. Consequently, the best strategies would have involved more substantial fishing pressure and less conservation. An assumption that the environment was variable would also have had significant, albeit less predictable, effects on the models' catches. We know from previous studies that reserves can stabilize annual catches (Lauck et al., 1998; Sladek Nowlis and Roberts, 1999)

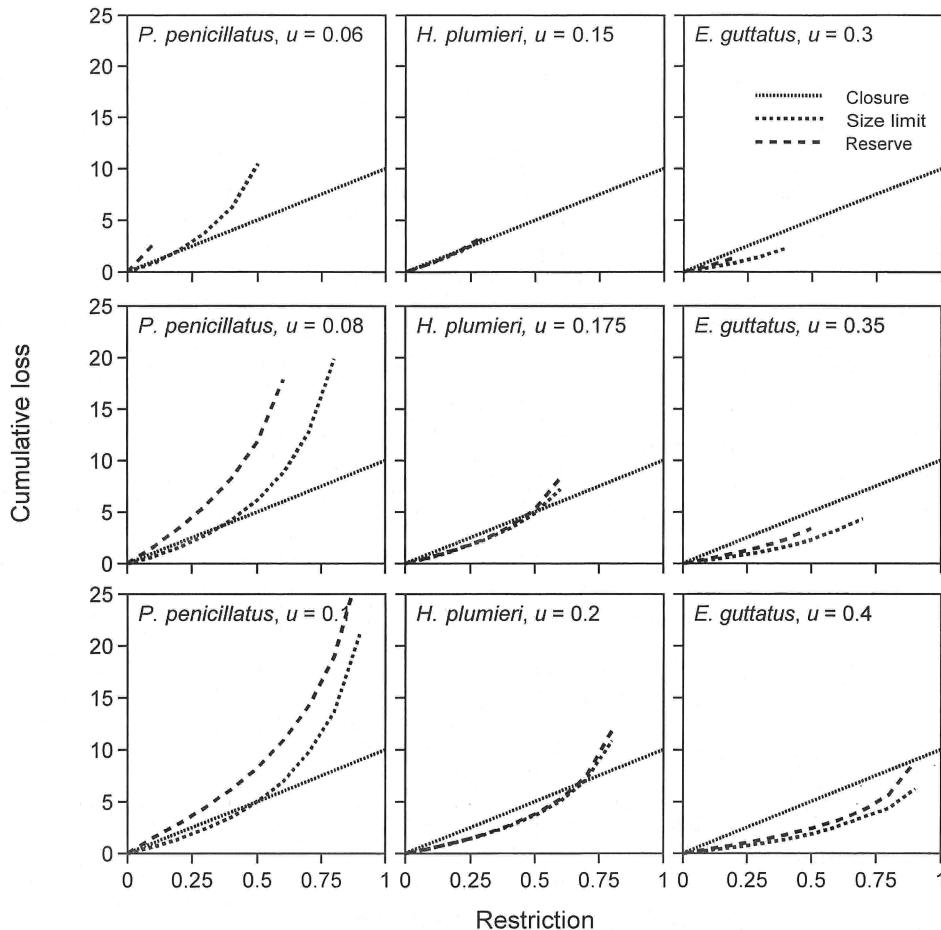


Figure 5. Cumulative loss. The cumulative loss depended on the species, fishing mortality rate, and the type and degree of restriction. Losses were generally greatest for the slowest-growing species and at the lowest fishing mortality rates. For the two species in which individuals were not vulnerable to fishing until they had reached sexual maturity (the white grunt, *Haemulon plumieri*, and the red hind, *Epinephelus guttatus*), cumulative losses were similar for reserves and size limits; both were somewhat lower than losses due to temporary closure of the entire fishery. For the spiny lobster, *Panulirus penicillatus*, losses were greatest with reserves and least with size limits unless restrictions were severe.

in a fluctuating environment, in part because they maintain a richer age structure than other tools. Thus, the assumption of a deterministic environment probably underestimated the benefits from reserves. Had I assumed that populations were underfished or fished to maximum capacity, no additional fishery restrictions would have been necessary to increase catches, and none of the new management tools would have fared well.

The assumptions specific to each model also affected the results. In the model examining temporary closures, I assumed that fishing rates upon reopening were equal to those prior to closure. Had I assumed that fishing rates increased, as might be expected because of the incentive of the increased biomass, the long-term results from this management strategy would have been even more dismal. Alternatively, if the closure drove some fish-

ermen and women out of business, and effort upon reopening was therefore lower than before closure, the long-term results would have looked more favorable.

In the model examining size limits, my assumption that the smallest fish in the fishery escaped fishing mortality biased results favorably toward this management tool. In reality, size limits via prohibitions against landings or even gear restrictions may not allow all smaller individuals to escape, if smaller individuals are simply caught and thrown back dead. The further assumption that effort did not increase on the larger fish biased results favorably toward this management tool in the long run but against it in the short run. If larger fish did become more vulnerable to fishing, some of the initial costs would be offset, but long-term benefits would be lost.

In the model examining reserves, the assumption that larvae crossed reserve boundaries freely but that adults never crossed them may bias results in favor of this management strategy in the long run but against it in the short run. If a reserve leaks adults, it will be less effective at protecting them; catches will remain higher in the short run, but reserves will not provide as many benefits in the long run. However, growing evidence regarding fish movement patterns suggests that these assumptions may be widely applicable, even for fish that are not thought of as bottom associated (Holland et al., 1996). As with the size-limit model, I assumed that fishermen and women did not increase their efforts on the remaining available fish. In reality, effort may increase outside the reserve because of displacement, or it may decrease if the establishment of reserves provides other economic incentives. Real-life examples suggest that effort can remain relatively constant outside reserves (e.g., McClanahan and Kaunda-Arara, 1996). If effort were to become more concentrated, reserves would provide more favorable short-term results but poorer long-term results.

These results provide several important lessons for managers. First, they demonstrate that, when a population is depleted and juveniles are vulnerable to fishing, management that eliminates fishing mortality for juveniles is highly effective. This lesson is common sense: fisheries should not be based on immature individuals.

Second, reserves show excellent promise as a management tool for rebuilding depleted populations. According to the results of this study, reserves can provide higher catches with less restriction and lower initial losses than other management tools. In order to reap these benefits, managers must design reserves to satisfy the basic assumptions of the model presented here, namely that larval fish cross reserve boundaries, that adult fish do not, and that reserve networks are designed on a scale that corresponds to the larval-dispersal capacity of the species of interest. This lesson may gain importance over the next several years. Many fisheries are depleted in the United States (National Marine Fisheries Service, 1998) and worldwide (Food and Agriculture Organization, 1995). Our growing awareness of overfishing is likely to steer us toward rebuilding tools, and reserves show great promise.

Finally, reserves are not a panacea. The positive results demonstrated here were based on the assumption that fishing mortality stayed constant in the remaining fishing grounds after the creation of the reserves. To meet this assumption, managers would offer incentives not only to reduce overall effort temporarily but also to limit future increases. Reserves do show promise as a tool but are only one of many in the toolbox of a good manager. This study also demonstrated the utility of size limits, and additional tools may be useful for maintaining and rebuilding productive fisheries.

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## APPENDIX

**SIZE STRUCTURE.**—Populations were divided into size categories by the same technique used by Sladek Nowlis and Roberts (1999). The concept is to select the size range encompassed by each size class so that year 0 fish have exactly a 100% chance of growing one size class during their first year. This categorization avoids the potential difficulty caused by fish that grow more than one size class in any year.

**DENSITY DEPENDENCE.**—Density dependence was incorporated into survivorship of size/year 0 fish during their first year. I used the negative exponential function

$$n_{x,t+1} = v_0 n_0 e^{-n_{0,t}/K} \quad \text{Eq. 1}$$

where  $n_{x,t}$  is the density of size  $x$  fish in year  $t$ ,  $v_0$  is the density-independent survival rate for size 0 fish, and  $K$  is a measure of the carrying capacity for the population. Other factors could also affect the density of size 1 fish, including fishing mortality and contributions from size 1 fish that survived the year but did not grow to size 2. Equation 1 merely illustrates the form of density dependence used in this modeling exercise.

**GENERAL POPULATION MODEL.**—This discrete-time model was projected yearly according to the equation:

$$n_{t+1} = F(Xn_t) \quad \text{Eq. 2}$$

where  $n_t$  is a vector representing the density of the population, by size class, at time  $t$ ;  $X$  is a matrix representing nonfishing life history factors as defined below; and  $F$  is a diagonal matrix whose elements represent the probability that members of each size class survive fishing for a year.  $X$  has three sets of elements: top row, diagonal, and below-diagonal. The top-row elements represent fecundities (as defined in Sladek Nowlis and Roberts, 1999) multiplied by the probability that eggs survive to become new recruits (see Sladek Nowlis and Roberts, 1999). The below-diagonal elements represent the probability for each size class that individuals survive natural mortality and grow to the next size class. The diagonal elements represent the probability for each size class that individuals survive natural mortality yet grow insufficiently to move to the next size class.

**TEMPORARY CLOSURES.**—The fishing matrix,  $F$ , was replaced by an identity matrix,  $I$ , for the years of the closure. Otherwise this model functioned like the general model.

**SIZE LIMITS.**—The fishing matrix,  $F$ , was modified in this model. Starting with the smallest size class vulnerable to fishing,  $x$ , the fishing mortality rate  $f_{xx}$  was reduced either until it became zero or such that the catch from the stable population size prior to new management had been reduced to the desired restriction level. If reducing  $f_{xx}$  to zero was insufficient to achieve the desired catch reduction, the program iteratively ran the same process with the next larger size class,  $x+1$ , and so on until the desired reductions were achieved. This new fishing matrix  $G$  was substituted for  $F$  in Equation 2 and used for the remainder of the model run.

**MARINE RESERVES.**—This model used the same assumptions and techniques as the model presented by Sladek Nowlis and Roberts (1999). The population was partitioned proportionately between the reserve and the remaining fishing grounds. The remaining fishing

population followed Equation 2, as did the reserve population but with an identity matrix, I, substituted for the fishing mortality matrix, F. The two populations interacted solely through larval dispersal. Size 0 population densities were calculated from a weighted average of the population densities in the reserve and fishing area. For a more detailed description, see Sladek Nowlis and Roberts (1999).