
A Decision Framework for the Adaptive Management of an Exploited Species with Implications for Marine Reserves

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Abstract: *Marine reserves have both conservation and fishery benefits. Nevertheless, there are no general criteria about when and where to establish new reserves, how to evaluate their efficacy, and how to conduct adaptive management to achieve conservation goals. We applied a decision-theory framework to optimally allocate conservation resources between improving data on population status and establishing a reserve for species conservation. Our goal was to maximize reserve benefits given the constraints of a population growth rate that would permit sustainability of resources. We illustrate our decision framework with a retrospective analysis of a 7-year time series on abundance of the leopard grouper (*Mycteroperca rosacea*) in the Sea of Cortés, Mexico. We used the lower bound of the distribution of the population growth rate (λ) as a decision rule for determining how many years of monitoring are needed to detect reserve effects. We determined the minimum time frame needed to estimate λ based on a stated level of risk tolerance for four sites. As expected, the coefficient of variation for the λ declined with the number of years of data. This increased precision with additional years of data resulted from the high degree of annual variability in the system. Where populations were slow to respond to reserves, more data were needed to detect a positive λ value. For the leopard grouper case study, confidence in the estimate of λ increased with the number of years of data. Our decision framework may be used to identify the minimum number of years of data needed before a management decision about reserve establishment could be made that is reasonably likely to meet its management objectives.*

Keywords: adaptive management, decision theory, leopard grouper, marine reserve, Sea of Cortés, species monitoring

Un Marco de Referencia para Decisiones para el Manejo Adaptativo de una Especie Explotada con Implicaciones para Reservas Marinas

Resumen: *Las reservas marinas tienen beneficios para la conservación y las pesquerías. Sin embargo, no existen criterios generales para dónde y cuándo establecer nuevas reservas, para evaluar su eficacia y para llevar a cabo manejo adaptativo para alcanzar sus metas de conservación. Aplicamos un marco de referencia de teoría de decisiones para asignar recursos para la conservación de manera óptima entre mejorar los datos sobre el estatus poblacional y el establecimiento de una reserva para la conservación de especies. Nuestra meta fue maximizar los beneficios de la reserva dadas las limitaciones de una tasa de crecimiento poblacional que pudiera permitir la sustentabilidad de los recursos. Ilustramos nuestro marco de decisiones con un análisis retrospectivo de una serie de tiempo de 7 años sobre la abundancia de *Mycteroperca rosacea* en el Mar de Cortés, México. Utilizamos el límite inferior de la distribución de la tasa de crecimiento poblacional (λ) como una regla de decisión para determinar cuántos años de monitoreo se requieren para detectar efectos de las*

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reservas. Determinamos el tiempo mínimo requerido para estimar λ basado en un nivel predeterminado de tolerancia de riesgo en 4 sitios. Como se esperaba, el coeficiente de variación (CV) de la tasa de crecimiento poblacional λ declinó con el número de años con datos. Este incremento en la precisión con datos de años adicionales fue causado por una gran variabilidad anual en el sistema. Donde las poblaciones respondían lentamente a las reservas, se requerían más datos para detectar un valor positivo de λ . Para el estudio de *M. rosacea*, la confianza en la estimación de λ incrementó con el número de años con datos. Nuestro marco de referencia para la toma de decisiones puede ser utilizado para identificar el número mínimo de años con datos que se requieren antes de tomar una decisión de manejo que tenga una probabilidad razonable de alcanzar los objetivos de manejo.

Palabras Clave: Mar de Cortés, manejo adaptativo, monitoreo de especies, *Mycteroperca rosacea*, reserva marina, teoría de decisiones

Introduction

Although there is wide recognition among marine conservation biologists that ecosystem-based management (EBM) is essential to effective conservation (Christensen et al. 1996), conservation practitioners have been hard-pressed to implement EBM (Pikitch et al. 2004; Arkema et al. 2006). One approach that has been advocated is the establishment of marine-protected areas (MPAs) to allow biomass of targeted species and their communities to recover (McClanahan & Graham 2005; Russ & Alcala 2005). Properly implemented MPAs that include no-take reserves may improve fishery yield outside the reserves because reserves can generally increase reproduction within their boundaries and thus may produce spillover (from hundreds of meters to kilometers) and act as a source of larvae for replenishment of fished areas (tens of kilometers away, DeMartini 1993; Gell & Roberts 2003). Nevertheless, few tools exist to determine how and when marine reserves will fulfill conservation and fishery goals. There are no general formulae to decide how many and what kind of reserves should be created to achieve specific goals (Botsford et al. 2003; Roberts et al. 2003; Sladek Nowles & Frielander 2005). In addition, most existing reserves do not have explicit goals and expectations, and temporal changes in the populations are seldom monitored (Gerber et al. 2005a, 2005b). Therefore, it is difficult for decision makers to prioritize the establishment of marine reserves and for reserve managers to evaluate the efficacy of the reserves (Grafton & Kompas 2005). Hence, management and policy vis-à-vis conservation of marine resources is generally inconsistent and based on political issues rather than on the realities of the resources they are intended to manage and conserve.

Monitoring is critical to the assessment of the effectiveness of a reserve and for the adaptation of management and monitoring strategies to new information (Walters & Holling 1990; Field et al. 2004; Gerber et al. 2005a). Nevertheless, the types of data collected (e.g., increases in biomass, size and age structure of populations within reserves; Halpern 2003) do not provide a basis for man-

agement decisions. Of greater importance is the magnitude of the effect size and whether that effect meets model predictions or stakeholders' expectations (Gerber et al. 2005a). Despite the widespread recognition of the importance of monitoring and adaptive management to enhance marine reserve efficacy, few monitoring studies have led to an adjustment of management. Adaptive management is essential to enhance the efficacy of reserves because, although the initial design of a reserve may be suboptimal, key decision variables can be changed later depending on the information gained from monitoring and evaluation (Grafton & Kompas 2005).

The Sea of Cortés, Mexico, is an ideal study system for exploring the intersection between monitoring and management of marine reserves. The Sea of Cortés has long been subjected to intense fishing pressure. Consequently, populations of many marine species have been depleted, including sharks, large groupers, and sea turtles (Sala et al. 2002). To mitigate these impacts and improve traditional fisheries management, the creation of a network of marine reserves in the Sea of Cortés is being considered by Mexican governmental and nongovernmental organizations. A decision framework is needed as a tool for practitioners to optimally allocate resources to effectively meet reserve objectives. If the goal is to recover overexploited populations by increasing the area protected as no-take zones in the Sea of Cortés, it is important to provide a clear set of decision rules to monitor and manage reserves based on field data on populations of conservation concern. Conservation decisions are often the result of a laborious course of negotiation that rarely yields repeatable results. These ad hoc strategies are used to arrive at a decision for a specific situation, but cannot be generalized to apply to other situations. We propose instead the use of decision theory (Jeffrey 1983, 1992) to formalize the decision-making process associated with marine-reserve creation and management.

We devised a protocol to link monitoring data to management decisions and applied it to the Sea of Cortés, Mexico. Although MPAs are aimed at protecting ecological communities, their establishment rarely is based on

data for entire communities because such data are seldom available. Thus data for single species are relied on to determine where reserves should be established and how they should be monitored. As a case study of this general situation, we analyzed 7 years of monitoring data on the leopard grouper (*Mycteroperca rosacea*) near four islands in the Loreto Bay National Park, an MPA that is partially protected by fishing regulations. We used these data to determine the minimum number of years of data needed before a management decision about reserve establishment could be made that is reasonably likely to meet its management objectives.

Leopard groupers are an important resource for local commercial fisheries in the Sea of Cortéz. They are a ubiquitous medium-sized predator (Hobson 1968) and inhabit rocky bottoms down to 50 m in depth. Because predators such as leopard groupers exert top-down control on community structure and ecosystem function (Frieland & DeMartini 2002), protecting the leopard grouper means other species will likely be protected. Furthermore, because many grouper species exhibit ontogenetic shifts in habitat use, protection of leopard grouper habitat may be an effective way to protect other species in the community as well (Sluka & Sullivan 1996).

We used our decision protocol and our monitoring data to identify whether the Loreto Bay National Park has been successful in conserving leopard grouper populations. In the case that it has not been successful, we determined what areas in the park should be closed to fishing to allow for the recovery of populations of leopard groupers. To develop a systematic process for continually improving management decisions and practices by learning from the outcomes of management strategies (adaptive management), we also explored the extent to which data quantity influences decisions pertaining to reserve establishment.

Methods

Study System and Fish Surveys

The Loreto Bay National Park (LBNP) is in the southwestern Sea of Cortés (Fig. 1). The LBNP was established in 1996 by the Mexican federal government to protect the area from industrial trawling and seining. Nevertheless, commercial and recreational hook and line fishing is still permitted throughout the park (CONANP-SEMARNAT

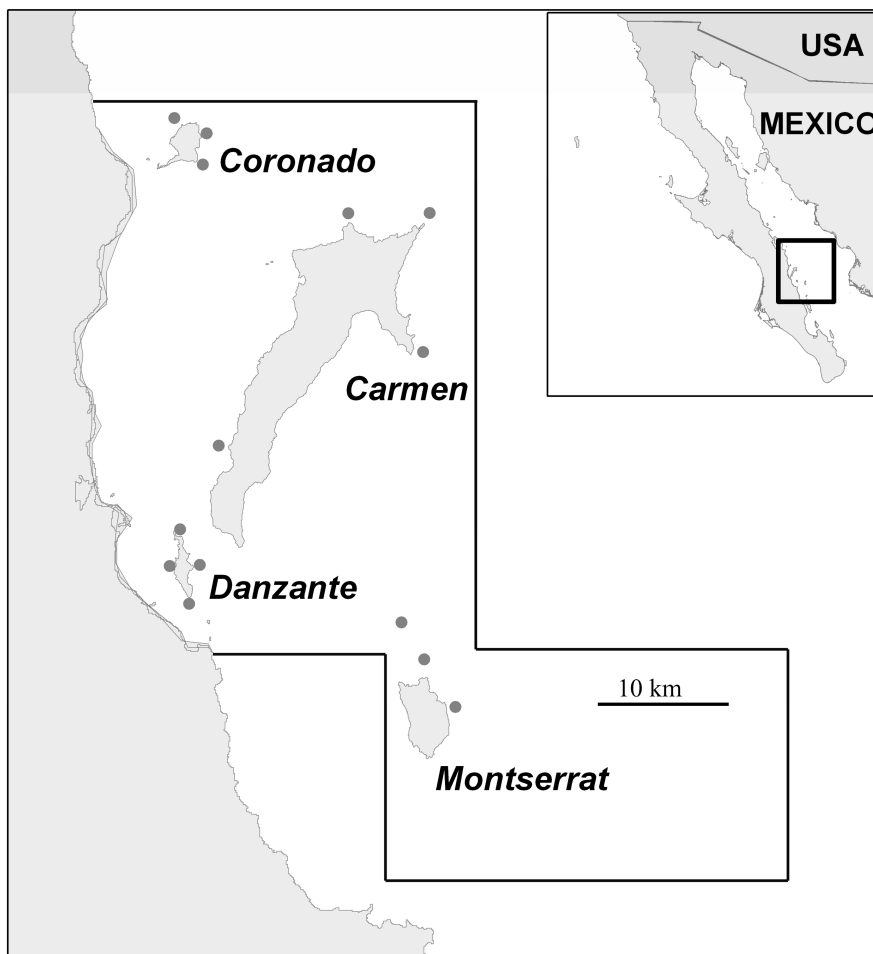


Figure 1. Location of Loreto Bay National Park (inset) and the four study sites: Carmen, Coronado, Danzante, and Montserrat. Sites of transect deployment are marked with circles.

2002). There are only two no-take areas in the park, which accounts for <1% of its surface area.

The leopard grouper is one of the most important fishery resources in the southern Sea of Cortés (Díaz-Uribe et al. 2001). Between 1998 and 2005, we studied the abundance of juvenile and adult leopard grouper in the LBMP in rocky bottoms at 5 and 20 m depth off the islands of Carmen, Coronado, Danzante, and Montserrat (Fig. 1). We chose depths of 5 and 20 m because a previous multivariate analysis showed that these depths harbor two distinct fish assemblages in the Sea of Cortés (Sala et al. 2002). Three to four sampling sites were located randomly near each island (Fig. 1) within the areas known to harbor rocky reefs of sufficiently large area to allow running a number of replicate transects. In September of each year, between 4 and 6, 50 × 5 m transects were surveyed at each site and depth. We used scuba diving and a standard visual belt-transect census method (e.g., Harmelin-Vivien et al. 1985) to quantify fish density (number of fish per transect). On every replicate transect the diver randomly selected a starting point and unreel a 50-m-long measuring tape, while swimming in a linear direction, and recorded the abundance and total length of all leopard groupers within the 5-m path. We visually estimated fish sizes of all groupers in the transects to the nearest 5 cm.

Estimating Population Parameters

For each year and site we estimated the discrete annual rate of population growth (λ) for leopard grouper based on size and depth-specific density estimates for each of our four study islands. Our demographic model included the two life stages for the leopard grouper that were found to be most important in determining population dynamics: (1) fish <35 cm (juveniles and subadults) and (2) fish >35 cm (adults) for the two depths described earlier (Wielgus et al. 2007). Each island was considered a distinct population based on available data on population structure and dispersal (Wielgus et al. 2007). Population growth rates were estimated with an “inverse estimation” procedure in which abundance data are used to calculate parameter values for the demographic model (Wood 1994, 1997; Caswell 2001; Wielgus et al. 2007). We used the inverse estimation method (instead of, e.g., simply using the change in counts between years) because it allows estimation of both demographic rates and λ (see Wielgus et al. 2007). Nevertheless, the specific method for estimating λ and CV is not a critical component of our decision framework; rather, a variety of approaches could be employed to estimate λ and CV within our decision framework.

We determined how estimates of the stochastic growth rate (λ_s , Caswell 2001) varied with sampling window and data quantity (Wielgus et al. 2007). We also estimated demographic schedules for different data samples for each island. Our goal was to evaluate how a “naïve” researcher

would estimate the mean and variance of λ for different sampling windows. We focused on two critical aspects of a population: population trend and variability in population trend. These parameters comprise both average tendencies to increase and decrease, and variability about these tendencies reflects variability in population growth rates (Gerber et al. 1999).

To characterize patterns of population growth, we first sampled data subsets from contiguous surveys to create progressively impoverished data sets (7, 6, 5, 4, 3, 2 years of fish-density estimates). For any given survey duration, n (out of N total years), we selected all possible permutations of continuous sets of survey data of n years. Thus, for the 7 years of available data, there were six samples of two consecutive years, five samples of three consecutive years, etc. (i.e., the number of permutations, $p = N - n + 1$)

To incorporate variability in growth rates, we estimated the variance in λ values for each survey duration for all permutations of sequential data combinations. Because the 7-year subset included only one permutation, we generated a variance estimate by jackknifing the seven estimates of λ . The jackknife estimation technique allows for variance estimation and bias correction (Haddon 2001). Jackknife replicates may be calculated from a data set of n values for n (noncontiguous) subsets of $(n-1)$ data points. For the 7-year subset, we estimated λ_s and generated 50% and 95% confidence intervals from the standard errors for seven subsets of six data points.

Developing a Decision Framework

There are two stages to our decision framework for reserve establishment: (1) identifying the number of years of baseline data needed before a management decision on closing an area or allowing fishing to continue is made (T), and (2) determining how many years of monitoring data are needed to document reserve effects (M) on the basis of estimates of λ (Fig. 2). We illustrate the application of our decision framework for stage 1, given that our data were collected before the reserve was established. In addition, we describe how a resource manager may use our resulting estimates of λ to adaptively manage reserves in the LBMP. Candidate sites are partially protected through gear regulations, but they are not entirely no-take areas. Our decision scheme provides a tool with which to determine which candidate sites should be closed to fishing.

If $\lambda_\rho < 1$ or $t < T$ years of prerreserve data are available, a reserve is established until $\lambda_\rho > 1$ for M years; if $\lambda_\rho > 1$ for T years, no reserve is needed but monitoring continues (ρ is level of risk tolerance and t is the number of years of data available). If in subsequent years $\lambda_\rho < 1$, this population decline would trigger the establishment of a reserve. The T and M may be determined based on the time frame needed to estimate λ such that $\lambda_\rho > 1$. We used our estimates of λ and variance to estimate the

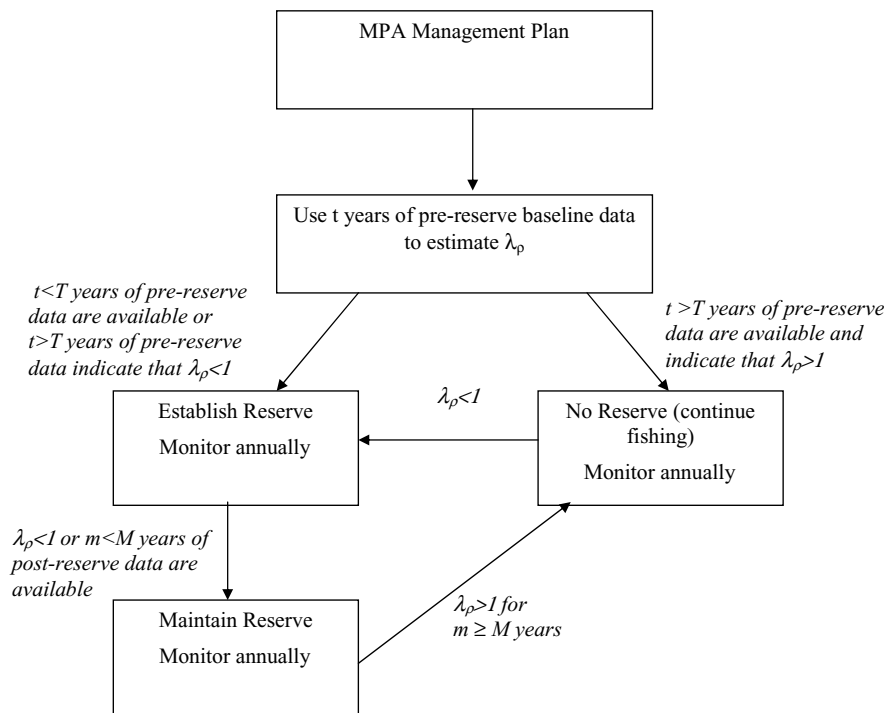


Figure 2. Decision framework illustrating an approach to management and monitoring of marine protected areas (MPAs) (T , number of years of data needed for a decision to establish a reserve; t , number of years of data available; M , number of years of data needed to detect the effects of a reserve; m , number of years of data available; and λ_ρ , the level of risk tolerance for estimated population growth rate).

coefficient of variation (CV) for λ as a criterion to identify the number of years of data needed before a management decision could be made. In this example CV is used as a basis for the first decision regarding T needed for desired levels of precision in λ . We then show how estimates of λ and associated CIs are influenced by data availability.

The CV reflects variation in λ from year to year due to changes in environmental conditions and variation due to measurement error. This value provides an indication of how much variability there is in annual estimates of λ for each site and hence how reliable monitoring data are in determining effects of reserves. This approach takes into account the inherent variability in λ , even if a mean is highly accurate. Although the CV could also vary as a result of changes in fishing mortality, fishing effort in Loreto did not change significantly during our field study (Wielgus et al. 2007). For each site a data subset with a high CV value is one for which one can conclude very little about population growth because it is driven by high variability. Ideally, the selection of an acceptable CV value should be determined by policy makers based on levels of risk aversion. Nevertheless, scientists should recommend appropriate CV values based on risk to help policy makers understand risk and to avert the selection of risk-prone approaches that maximize short-term profits.

For the sake of illustration for our case study, we relied on guidelines provided by the U.S. National Marine Fisheries Service to determine acceptable removal levels for marine mammals. In general, acceptable levels of risk should be determined by policy makers based on risk aversion and species status. For our case study a CV < 0.3 corresponded to a higher level of potential removals than

a CV > 0.3 (Wade & Angliss 1997). We similarly assumed that a CV describing variation in λ must be < 0.3 to accurately assess the status of a population before a reserve is established (T) and to detect the effects of a reserve (M). For example, a CV of 0.3 from prereserve baseline data allows determination of the number of years of data needed for a decision to establish a reserve. If the CV exceeds 0.3, more data are needed before a decision is made, regardless of the value of λ_ρ . Our approach is precautionary in that if $t < T$ years of prereserve data are available, a reserve is established until $m < M$ years of postreserve data are available (Fig. 2). To illustrate how management decisions would vary for alternate CV values, we also considered CV values of 0.5 and 0.8 (Fig. 3).

If a population is determined to be declining based on our estimated mean and variance in λ , the decision would be to establish a no-take zone and continue monitoring. If the population is determined to be increasing for $t > T$ years, fishing may continue with annual monitoring. To incorporate uncertainty in λ resulting from annual variation estimated on the basis of the methods described earlier, one may consider the lower bound of a distribution for λ (λ_ρ), where ρ is the level of risk tolerance, as a decision rule. In practice, establishing ρ may be accomplished by using simulation results to determine how risk-averse management has to be to achieve a given management goal.

For each site one may determine the minimum time frame needed to estimate λ such that $\lambda_\rho > 1$. This estimate is conditional on the population having a positive λ . We are focused on reducing uncertainty to the level where λ is significantly positive. Under this scheme if $\lambda_\rho < 1$, a

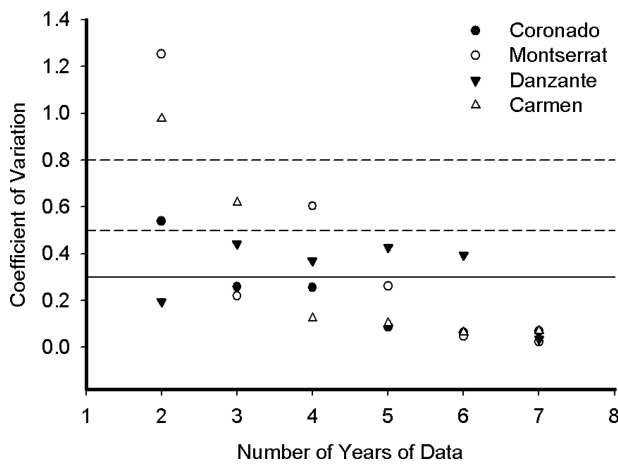


Figure 3. Coefficients of variation for population growth rates (λ) for data subsets of number of years of data. This graph can be used to identify the number of years of data needed to establish a reserve (T) for each site. Three reference lines are shown for CV thresholds for defining T : 0.3, 0.5, and 0.8.

reserve is established until $\lambda_\rho > 1$. If $\lambda_\rho > 1$ no reserve is needed but monitoring occurs at a reduced frequency. If in subsequent years λ_ρ declines below 1, a reserve is established. As an example we considered the lower 5th and 25th percentiles of λ ($\lambda_{0.05}$, $\lambda_{0.25}$) as threshold values for decisions regarding reserve monitoring; nevertheless, this value ideally should be determined by resource managers and policy makers and be based on how risk-averse management has to be to achieve a given management goal. Depending on the risk that a manager considers tolerable based on risk associated with various management outcomes, other percentile values could be considered. Thus, the cost of monitoring and management may be reduced when populations are confidently determined to be increasing, and the traditional use of resources by local communities can be allowed to resume.

Results

We used the lower bound of the distribution of λ (λ_ρ) as a decision rule for determining how many years of monitoring are needed to detect reserve effects (M , Fig. 2). For each site we determined the minimum time frame needed to estimate λ such that $\lambda_\rho > 1$, where ρ is the level of risk tolerance. Where populations were slow to respond to reserves, more data were needed to detect a positive λ value. Our decision framework helped identify the minimum number of years required to detect positive population growth (Fig. 3).

Our grouper data can be used to identify T for each island (Fig. 3). As expected, the CV declined with the number of years of data. This increased precision with

additional years of data resulted from the high degree of annual variability in the system. With fewer years of data, estimates of λ were highly variable (imprecise). For the 0.3 threshold for CV, 7 years of data were needed for all sites, but 5 years of data were enough for all sites, with the exception of Danzante. For a CV of 0.5 and 0.8, 5 and 3 years of data, respectively, were needed to determine T for all sites.

For the leopard grouper case study, confidence in the estimate of λ increased with the number of years of data, and the optimal decision was the one in which $\lambda_\rho > 1$ (Fig. 4). To incorporate uncertainty, we considered the lower 5th and 25th percentiles ($\lambda_{0.05}$, $\lambda_{0.25}$) as threshold values for decisions regarding reserve monitoring. Because variation in λ is different for each site, the CV also varies for each site. In general, the average λ declined with number of years of data, although this pattern did not hold for all years and all sites (Fig. 4).

Discussion

One of the many goals of marine reserves is to help manage fisheries threatened by decreasing yield and unsustainable harvest in the face of uncertainty regarding the relationship between environmental conditions and net recruitment in the population of interest (Botsford et al. 2003). The ideal marine reserve project combines management schemes and associated monitoring based on knowledge of the natural system. To accomplish this, baseline data are needed to understand how anthropogenic and environmental factors interact to produce observed trajectories. By establishing a reserve it may become possible to tease apart anthropogenic from environmental effects. In turn this information should inform management based on predetermined trigger values for changes in management and monitoring. Although such a scheme would be ideal, establishing a marine reserve is often viewed as an achievement in itself, and resources for monitoring are usually limited (Gerber et al. 2005a).

For the leopard grouper, our results indicate that short-term monitoring may overestimate the population growth rate of target species. This suggests that sampling over just a few years may yield an overly optimistic picture of the status of the population. Not only did shorter data subsets result in an overestimate of λ but precision in our estimation of λ increased with the number of years of data. In the context of our decision scheme, the high degree of uncertainty and the overly optimistic estimate of λ for smaller data subsets underscore the importance of establishing a reserve in the face of uncertainty.

Our results also highlight the importance of adequate baseline data for candidate sites for marine reserves and support the idea of precautionary management and the need for reducing type II error when dealing with

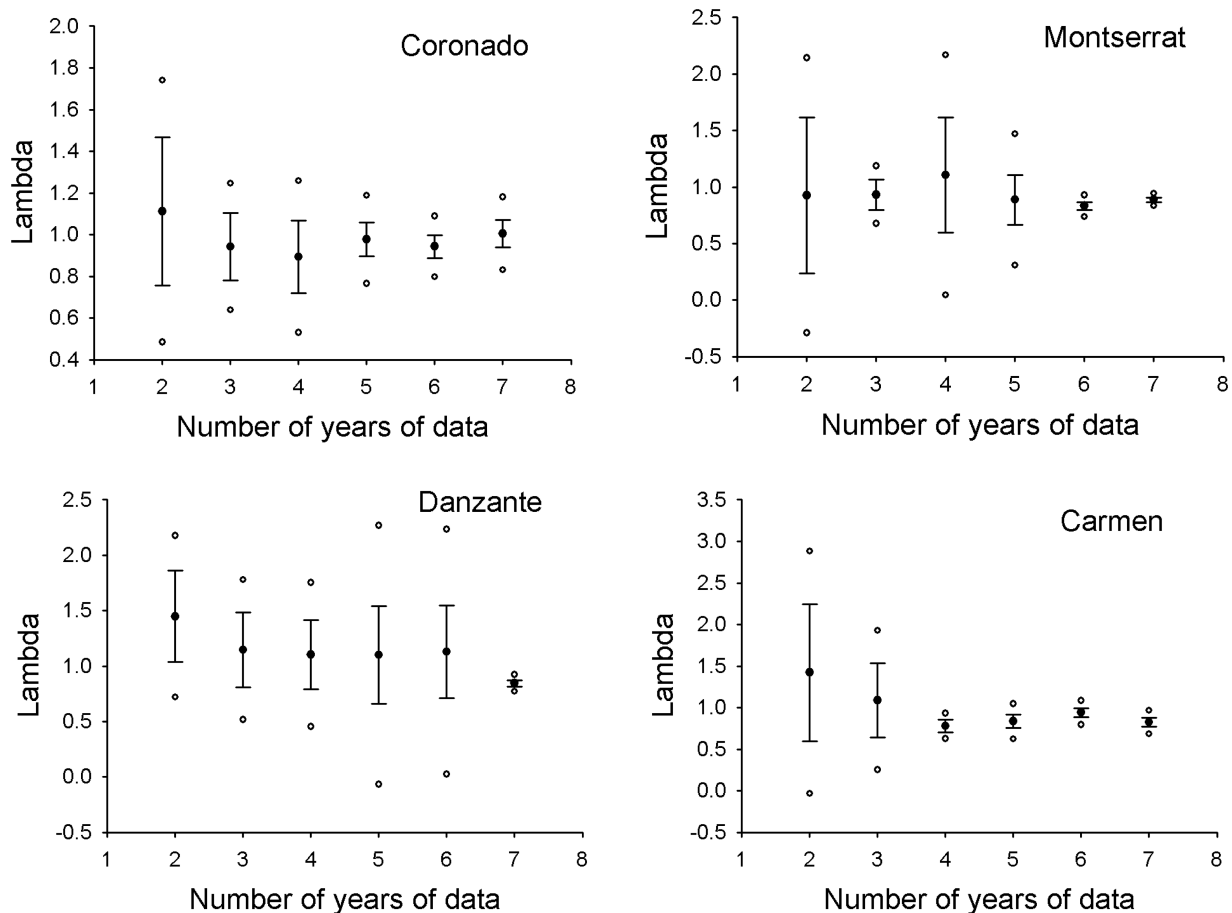


Figure 4. The population growth rate (λ) value (closed circle), 50% and 95% CI in λ (whiskers and open circles, respectively) for subsets of number of years of data. The lower 50% CI indicates $\lambda_{0.25}$ and the lower 95% CI indicates $\lambda_{0.05}$. The λ values are estimated from empirical data using the approach of Wielgus et al. (2007).

management decisions (Dayton 1998). It is interesting to note that patterns of variability in population growth differed between sites (e.g., Danzante), suggesting the need for site-specific monitoring plans. Our results agree with those of Regan et al. (2005) and Halpern et al. (2006), who also conclude that incorporating rather than avoiding uncertainty will increase the chances of successfully achieving conservation and management goals.

Our results are not without caveats. First, the inverse estimation method we used does not perform well with only 2 years of data (Wielgus et al. 2007). In addition, our variance estimate for the 7-year data set was obtained with a different approach than all other data sets (i.e., jackknife vs. standard variance estimate), which could yield inconsistent estimates if data are biased. The model could also be extended to explicitly examine the extent to which estimates of mean and variance of λ exhibit annual fluctuations (e.g., by considering random 2-year data combinations). The application of our results to other systems should consider alternate values for the CV threshold in light of the type of surveys being conducted and the species being considered. Finally, although we focus

on the length of time necessary to get a good estimate of pre- and postreserve population status, another way to increase power to detect change is to increase the number of replicate sites being monitored.

Our decision framework provides reserve managers in the Sea of Cortés and elsewhere with a practical decision tool for adaptive reserve management. More broadly, our framework applies to marine reserves and to any spatial management intervention or treatment. Our leopard grouper example illustrates how managers can use baseline data on size structure to estimate population growth for candidate sites and update this estimate every year as new data are collected. Based on this simple approach to decision making that relies on baseline data on fish density, a manager would (1) estimate population growth and variability in growth and (2) use these estimates as metrics for deciding how to allocate resources to monitoring and management. In particular, a general application of our approach includes six steps: (1) choose surrogate species; (2) choose candidate sites; (3) estimate the cost of monitoring, enforcement, loss of revenue from closing areas to fishing; (4) monitor local population size, size

structure, or other applicable population metrics; (5) estimate λ and CV with applicable method; and (6) apply decision rule and adapt management strategy.

The value of this approach is that it takes into account the inherent variability in λ , even if a mean is highly accurate. Although variance in λ generally declines with data quantity, there is always some inherent variability in λ due to process error (Figs. 3 & 4). For some sites the mean changes very little but the variance changes dramatically for different data subsets. This suggests that even with unlimited data, there will still be some variability in λ , reflecting underlying year to year changes in the system. When a reserve is created, the variability in population growth rate may be reduced based on reduced interseasonal and interannual fluctuations in fish biomass (e.g., Francour 1994) because the unfished assemblages become increasingly dominated by predators with lower turnover rates (Russ et al. 2005) as opposed to smaller species with higher turnover and likelihood of strong fluctuations (Hsieh et al. 2006). Thus, the time frame for detecting significant changes after protection may depend largely on postreserve variability.

Marine reserves have been designed in an ad hoc fashion with little reliance on explicit decision rules. One of the overriding problems with incorporating scientific information into reserve management and monitoring is that science makes up only one component in the decision-making process. Furthermore, the level of uncertainty present in scientific information provides a number of complexities for decision making. We believe that the integration of formal decision analysis with conservation science is a potentially powerful approach to implementing solutions to marine conservation problems in the Sea of Cortés and elsewhere.

Decision makers need new tools that will delineate the consequences of policy decisions, from both ecological and socioeconomic points of view. It is essential that rigorous monitoring programs be established and that biological and socioeconomic indicators be measured through time so that the efficacy of marine reserves can be evaluated and the size of reserves can be expanded if conservation goals are not achieved. Marine reserves should then be conceived as experiments in which initial hypotheses (e.g., the abundance of target species will increase in marine reserves) are put forth, experimental treatments (monitor the reserves and unprotected areas nearby) are established, and hypotheses (evaluation of reserve efficacy) are tested (Walters & Holling 1990). If hypotheses are rejected (i.e., the goals of the reserves are not fulfilled), then one needs to determine why and try an alternate strategy. In such a case this would mean understanding why reserves were not effective (e.g., lack of enforcement, too small an area protected) and determining what policy (management) changes need to be implemented to achieve the goals. These changes in policy

would have consequences (costs and benefits) for both marine resources and humans.

Our case study deals with a single target species that is vulnerable to fishing. Nevertheless, marine reserves are created mainly to conserve representative samples of ecosystems, including hundreds or thousands of species. Solutions that work for one species might not work for other species and selection of several surrogate species with a broad range of life history characteristics will increase the potential effectiveness of reserves in protecting entire ecosystems. Nonetheless, our general approach could be used to evaluate the efficacy of reserves at the ecosystem level based on measures of change in ecosystem indicators or based on a community viability analysis. Finally, our approach need not be limited to marine reserves. It may be of general use for establishing repeatable and scientifically defensible methods for management and monitoring.

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