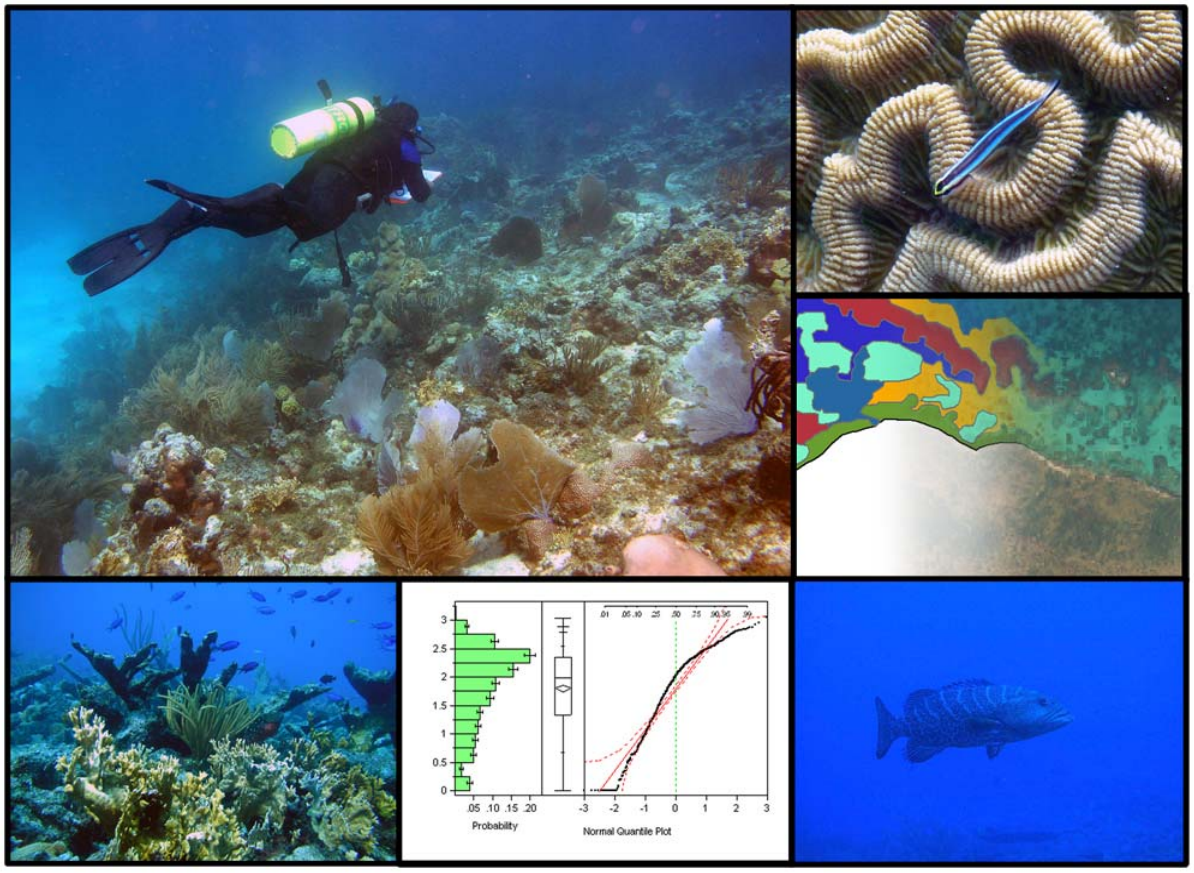
A GUIDE TO MONITORING REEF FISH IN THE NATIONAL PARK SERVICE'S SOUTH FLORIDA / CARIBBEAN NETWORK



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A Guide to Monitoring Reef Fish in the National Park Service's South Florida / Caribbean Network

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Executive Summary

Reef fishes are conspicuous and essential components of coral reef ecosystems and economies of southern Florida and the United States Virgin Islands (USVI). Throughout Florida and the USVI, reef fish are under threat from a variety of anthropogenic and natural stressors including overfishing, habitat loss, and environmental changes.

The South Florida / Caribbean Network (SFCN), a unit of the National Park Service (NPS), is charged with monitoring reef fishes, among other natural and cultural resources, within six parks in the South Florida - Caribbean region (Biscayne National Park, BISC; Buck Island Reef National Monument, BUIS; Dry Tortugas National Park, DRTO; Everglades National Park, EVER; Salt River Bay National Historic Park and Ecological Preserve, SARI; Virgin Islands National Park, VIIS). Monitoring data is intended for park managers who are and will continue to be asked to make decisions to balance environmental protection, fishery sustainability and park use by visitors. The range and complexity of the issues outlined above, and the need for NPS to invest in a strategy of monitoring, modeling, and management to ensure the sustainability of its precious assets, will require strategic investment in long-term, high-precision, multispecies reef fish data that increases inherent system knowledge and reduces uncertainty.

The goal of this guide is to provide the framework for park managers and researchers to create or enhance a reef fish monitoring program within areas monitored by the SFCN. The framework is expected to be applicable to other areas as well, including the Florida Keys National Marine Sanctuary and Virgin Islands Coral Reef National Monument. The favored approach is characterized by an iterative process of data collection, dataset integration, sampling design analysis, and population and community assessment that evaluates resource risks associated with management policies. Using this model, a monitoring program can adapt its survey methods to increase accuracy and precision of survey estimates as new information becomes available, and adapt to the evolving needs and broadening responsibilities of park management.

To conduct reef fish population and community assessments, monitoring programs must collect abundance and size-frequency distribution data for distinct fish taxa. Concurrently collected data on benthic habitat and water quality are desirable as well, and can be assimilated in a survey design to improve survey performance.

The method of measurement should establish a constant search area for a sample unit (e.g. transect, fixed radius cylinder) and obtain an accurate representation of the reef fish community within the sample unit, tempered by the time required to obtain the sample. The choice of

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measurement method depends on the species or species-complex, life history stages, and habitat chosen for sampling.

Underwater visual census methods are ideal for assessing reef fishes in the Florida Keys (e.g. DRTO) and Virgin Islands (e.g. BUIS, SARI, VIIS) because of prevailing good visibility, rugose habitats, and management concerns requiring the use of non-destructive census methods. The most well known visual census methods are the belt transect and the stationary visual census. Alternative methods must be used to census fish in turbid (e.g. BISC) or deep environments or when nighttime surveys are required.

A primary consideration of a monitoring program is to delineate the target population which will be monitored. For reef fish, this can be done by selecting an ecosystem area to be surveyed. For most monitoring purposes a sample of the population is used to infer the status of the population. There are many ways to select a sample from a population, but the more information available about a population, the easier it is to devise a selection method which provides accurate and precise survey estimates. A simple random sampling design is appropriate for situations where there is no spatial structure in the variance of investigated fish metrics or little information is available. Since fish metrics are typically heterogeneous, a stratified random sampling design will sample a fish population more effectively. Maps of environmental covariates, such as benthic habitat, at the appropriate spatial scales and spatial extent can be used to effectively divide the sampled population into strata.

The main goal of sample surveys is to obtain accurate, high-precision estimates of population and community metrics at a minimum of cost. The objective of sample design analysis is to determine the appropriate number of samples required to achieve enough precision in population and community metrics (e.g., species numbers-at-size, species composition) to understand ecological processes and to make management decisions. Iterative analysis of candidate survey design performance can be used to refine survey estimates and reduce sampling cost.

The range and types of statistical analyses that will be performed to assess the status and dynamics of reef fish populations and communities in National Parks depends on the specific management questions and resource goals to be addressed. These analyses utilize the range of fundamental survey data outlined above and recommended for collection in monitoring programs. These survey data are then used to generate multiple metrics for individual species, species-complexes or life history stages to assess status and trends of reef fish over time and in relation to specific sustainability metrics.

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A single standardized monitoring protocol is not advocated because of the variability in ecological condition, size, management capability, and available data among SFCN park units. This guide is meant to serve only as a framework. Three reef fish monitoring program case studies are provided which build upon the presented framework using park-specific data sets, management concerns, and local partnerships.

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1 Background and Objectives

1.1 Rationale

Reef fishes are conspicuous and essential components of coral reef ecosystems and economies of southern Florida (Johns *et al.* 2001; Ault *et al.* 2005a) and the United States Virgin Islands (USVI) (Hinkey *et al.* 1994; Mateo 1999, 2001). More than 500 species of reef fish including sharks, eels, flounders, gobies, puffers, groupers, parrotfishes, snappers, jacks, and damselfishes can be found in the reef ecosystems of the Florida Keys and USVI (Longley and Hildebrand 1941; Bohlke and Chaplin 1968; Clavijo *et al.* 1980; Munro 1983; Robbins and Ray 1986; Smith-Vaniz *et al.* 1995; Randall 1996; Bohnsack *et al.* 1999; Humann and DeLoach 2002).

Throughout Florida and the USVI, coral reef ecosystems are under threat from a variety of anthropogenic and natural stressors, including overfishing, habitat loss, and environmental changes. Over the past several decades, public use and conflicts over fishery resources in the two regions have increased sharply (Appeldoorn *et al*. 1992; Bohnsack *et al.* 1994; Leeworthy and Vanasse 1999), while catches from historically productive fishery stocks, especially the snapper-grouper complex, have declined (Allen and Tashiro 1976; Bohnsack *et al.* 1994; Ault *et al.* 1997, 1998, 2001, 2005a). In the Florida Keys, recent quantitative assessments of the multispecies reef fish community have shown that exploitation levels are very high, that many stocks are "overfished", and that overfishing has been clearly evident since the late 1970's (Ault *et al.* 1998, 2001, 2002, 2005ab). Throughout the region, there is evidence of loss of grouper spawning aggregations, reduced catch per unit effort, changed species composition of landings, and lower mean sizes and abundances of several assemblages (de Graaf and Moore 1987; Appeldoorn *et al*. 1992; Bohnsack *et al.* 1994; Ault *et al.* 1998, 2005b; Beets 1996ab; Garrison *et al.* 1998; Beets and Friedlander 1999; Beets & Rogers 2002; Beets and Muehlstein 2003). The impacts to the community are of great concern because fishing has depleted top trophic levels, shifted community structure and reduced the length and complexity of food webs that affect fishery resilience and prospects for their sustainability (Pauly *et al.* 2002; Ault *et al.* 2006). Several species, for example Goliath Grouper (*Epinephelus itajara*), Nassau Grouper (*Epinephelus striatus*) and Queen Conch (*Strombus gigas*), have been so depleted that they are now protected (Sadovy and Eklund 1999, Caribbean Fishery Management Council 1996).

The reef fish community has also been affected by a series of other anthropogenic and natural stressors. Coastal development influences reef fish through a plethora of negative impacts

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on reef fish habitats (Rogers and Beets 2001). Fish habitats are altered by dredging, boating, fishing gears, wetlands reclamation, beach renourishment, mangrove removal, and sea defense construction (Rogers *et al.* 1988; Rogers 1991; Beets 1996a; Lindeman and Snyder 1999; Rogers and Beets 2001; Wilber *et al.* 2003; Mumby *et al.* 2004; Chiappone *et al.* 2005). Additional consequences of development include changes to water quality from pollution, sedimentation, nutrient loading, freshwater inflows, and regional scale hydrodynamics (McIvor *et al.* 1994; MacDonald *et al.* 1997; Serafy *et al.* 1997; Mannoni 1999; Nemeth and Sladek-Nowlis 2001; Rogers and Beets 2001; Cowie-Haskell and Delaney 2003). These impacts may be exacerbated by intensification of hurricane activity and climate variability (Rogers *et al.* 1982; Hughes 1994; Edmunds 2002; Gardner *et al.* 2005; Pandolfi *et al.* 2005).

As coastal populations, tourism and fishing pressure continue to increase, park managers are being asked to make decisions to balance environmental protection, fishery sustainability, and visitor resource use. Balancing these conflicting uses is a complex issue, and requires an ecosystem-based management (EBM) approach. EBM considers knowledge and uncertainties in biotic, abiotic, and human components of the whole ecosystem in an attempt to balance societal objectives (Christensen *et al.* 1996; Larkin 1996; Schramm and Hubert 1996; Pikitch *et al.* 2004). These broad-spectrum objectives are set within a framework created by the NPS's universal mission to protect cultural and natural resources (Organic Act of 1916, USC title 16).

1.2 Management Domain

The National Park Service (NPS) through the South Florida / Caribbean Network (SFCN) is one of many administrative entities monitoring reef fishes and coral reef ecosystems in southern Florida and USVI. The SFCN is composed of four managed areas in southern Florida (Big Cypress National Park, BICY; Biscayne National Preserve, BISC; Dry Tortugas National Park, DRTO; Everglades National Park, EVER) and three in the USVI (Buck Island Reef National Monument, BUIS; Salt River Bay National Historic Park and Ecological Preserve, SARI; Virgin Islands National Park, VIIS) (**Figure 1**). Six of these areas are inhabited by reef fishes (BICY is excluded). The Virgin Islands Coral Reef National Monument (VICR) is not currently part of the SFCN, but is managed by the NPS and shares many ecological and management characteristics with VIIS, BUIS, and SARI. Information on location, date of establishment, cultural and natural resources, marine habitat area, and management zoning for each park is given on the NPS website (www.nps.gov). Site characterizations of fisheries resources and habitats have been carried out for

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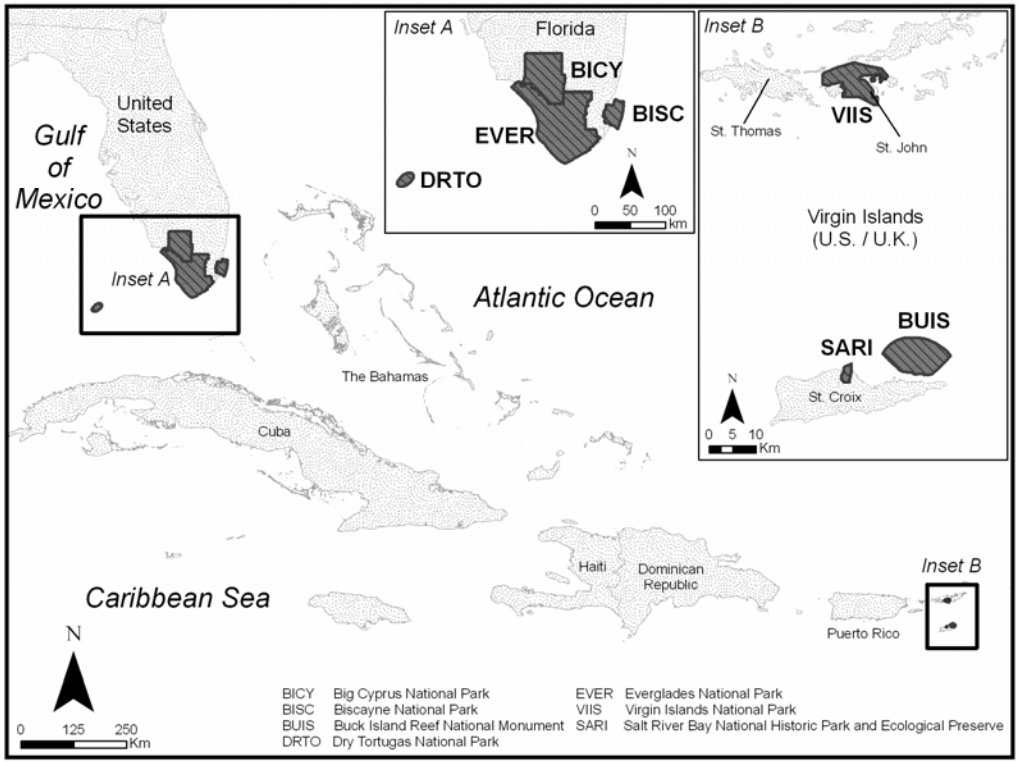

Figure 1: Map of the Caribbean region showing parks and monuments that are part of the National Park Service's South Florida / Caribbean Network.

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BISC (Ault *et al.* 2001) and DRTO (Schmidt *et al.* 1999; Ault *et al.* 2002), but only habitats for SARI (Kendall *et al.* 2005).

1.3 Objectives

The range and complexity of the issues outlined above, and the need for NPS to invest in a strategy of monitoring, modeling, and management to ensure the sustainability of its precious assets, will require strategic investment in long-term, high-precision, multispecies reef fish data that benefits EBM by increasing inherent system knowledge and reducing uncertainty. Objectives to meet the goals of park-specific EBM are to:

- Assess condition and changes in reef fish community diversity and its composition
- Assess condition and changes in the sustainability status of reef fishes under exploitation
- Assess anthropogenic impacts (e.g., fishing, sedimentation, pollution, etc.) on community dynamics
- Assess effectiveness of management strategies such as Marine Protected Areas (MPAs), fishing regulations, land and water uses, etc.
- Determine biological and physical processes that govern health and sustainability of reef fish resources

These objectives may be tailored by local resources and interested stakeholders as outlined in each park's General Management Plan, Resource Management Plan and/or Fisheries Management Plan. These plans set the management philosophy and direction for planning horizons up to 15-20 years in the future and are amended accordingly.

There is a clear need to link monitoring programs to meet management goals and objectives. Monitoring programs are comprised of an iterative process of data collection, dataset integration, design analysis, and population and community assessment that evaluates resource risks associated with management policies. **Figure 2** illustrates a conceptual model of an ideal reef fish monitoring program. The model is adaptive in two respects. First, as new information becomes available, the monitoring survey design used to collect data and assimilate data is tailored to increase accuracy and precision. Second, the monitoring process can adapt to the evolving needs and broadening responsibilities of EBM and NPS management plans. Adaptation need not be instantaneous or final. A reevaluation of the management objectives every 3-, 5- or 10-years can be sufficient to adapt to new management policies, shifted resource conditions, and a compilation of new data.

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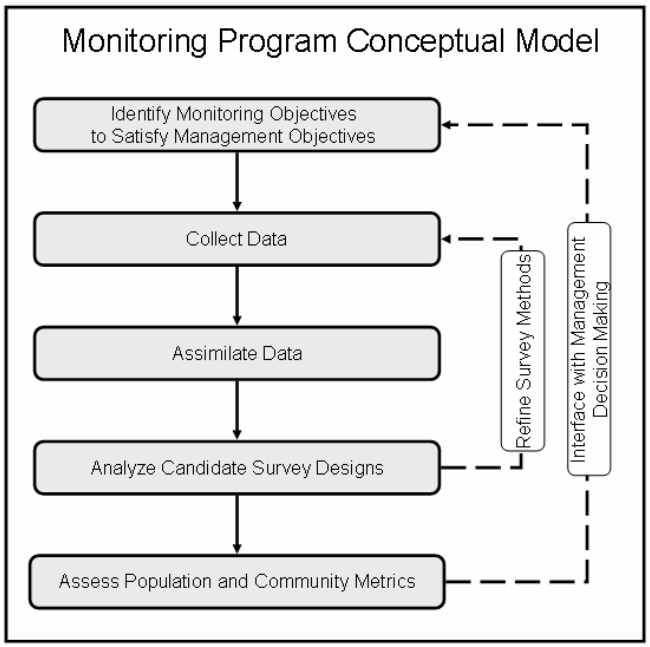

Figure 2: A conceptual model of an ideal reef fish monitoring program. The model illustrates the iterative process of data collection, dataset integration, design analysis, and population and community assessment that evaluates resource risks associated with management policies. Feedback loops critical to the iterative process are shown.

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1.4 Document Organization

This guide mainly addresses the first four components of the iterative monitoring program outlined in **Figure 2**. These components are organized in the following sections of this document:

1. Background and Objectives
2. Data to be Collected and Methods of Measurement
3. Population to be Sampled and Selection of the Sample
4. Candidate Sampling Design Analysis
5. Population and Community Assessments

In addition, three case studies are provided that illustrate existing monitoring programs within managed areas of the SFCN. The case studies are described further in Section 6.

1. Case Study A: Reef Fish Monitoring in Virgin Islands National Park and Buck Island National Monument, 2001-2005 (VIIS, BUIS)
2. Case Study B: Monitoring Reef Fish Assemblages inside Virgin Islands National Park and around St. John, US Virgin Islands, 1988-2000 (VIIS)
3. Case Study C: Assessment of Coral Reef Fishery Resources in Dry Tortugas National Park, 1999-2004 (DRTO)

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2 Data to be Collected and Methods of Measurement

2.1 Data to be Collected

To conduct reef fish population and community assessments, monitoring programs must collect several types of fundamental data on reef fishes:

- Identification to the lowest possible taxonomic classification (preferably to the species level) of each individual
- Abundance and size-frequency distribution of each taxa

Concurrently collected data on benthic habitat and water quality are desirable as well and can be assimilated in a survey design to improve survey performance. Complementary data often include:

- Depth
- Substrate composition
- Benthic floral/faunal composition
- Vertical rugosity
- Benthic habitat type (based on strata of a stratified sampling design)
- Temperature
- Salinity

2.2 Methods of Measurement

The method of measurement chosen for a survey should establish a consistent search area for a sample unit (e.g. transect, fixed radius cylinder) and obtain an accurate representation of the reef fish community within the sample unit, tempered by the time required to obtain the sample. Consistent search areas ensure comparability among sampling units and equal (or known) probability of sample unit selection. Search methods based solely on time are not recommended if sample units are defined by area, because they invalidate the equal selection property that is fundamental to using probability-based sampling designs.

The choice of the sampling method depends on the species or species-complex, life history stages, and habitat chosen for sampling. These choices are governed by the monitoring program goals. The following sections outline methods that take these considerations into account.

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2.2.1 Visual Census Methods

Underwater visual census methods are ideal for assessing reef fishes in the Florida Keys and Virgin Islands because of prevailing good visibility, rugose habitats, and management concerns requiring the use of non-destructive assessment methods. The most well known visual methods are the stationary visual census and the belt transect. The stationary visual census samples all reef fish within an imaginary cylinder of fixed radius (Bohnsack and Bannerot, 1986). Belt transects sample all fish within a rectangle of fixed width and length (Brock, 1954). Belt transects may be more appropriate when sites are characterized by low visibility, highly rugose habitats, or adjacent to mangroves because they place the diver closer to fish. In addition, the belt transect may be more effective in sampling small, cryptic species (e.g. Chaenposidae, Gobiidae, Labrisomidae). Case study A collects reef fish data using the belt transect method. Stationary visual census methods may minimize measurement bias attributed to diver movements and is superior for counting pelagics (e.g. Carangidae, Scombridae, Clupidae). Examples of the stationary visual census are provided in case studies B and C. The choice of a visual census method should maximize effectiveness (minimize bias) of the census given potential fish species and habitats among sample units. Some logistical factors that could improve survey performance (more samples per unit time) and reduce diver fatigue are use of Nitrox SCUBA and "live-boating" (boat driver remains on non-anchored vessel) at dive sites.

Alternative visual census methods which do not employ SCUBA diving and thus are free of SCUBA's depth constraints utilize underwater video cameras, but they are plagued by biases associated with species selectivity and inconsistent census area. Underwater stereo-video (Harvey and Shortis 1996) and baited video cameras (Willis and Babcock 2000) are two examples.

2.2.2 Non-Visual Census Methods

In general, statistical issues of capture efficiency and size-selectivity (e.g., MacLennan 1992; Gunderson 1993) are minimized using visual census methods; thus, they are typically preferred for coral reef ecosystems. However, not all fish species or individuals in the sampled area will be detected by visual methods. Alternative methods may be required in cases where: visibility is occluded by turbid waters or densely vegetated habitats; night-time sampling is most effective for target species; cryptic species are targeted; or depths exceed operational diving limits. In these situations, fish may be sampled more effectively using gear or poisons (ichthyocides) to capture fish. Two classes of gear have been used to sample reef fishes: active gear, such as trawls and seines

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that are towed or pulled to capture fish within a sample unit (e.g., Robblee and DiDomenico 1991; Sedberry and Carter 1993; Serafy *et al.* 1997; Ault *et al.* 1999); and passive gear, such as gill nets, traps, and pots that are assumed to sample a fixed unit area over a given unit time (e.g., Collins and Sedberry 1991; Hickford and Schiel 1995; Pratt and Fox 2001; Watson and Munro 2004).

Morphologically or behaviorally cryptic species will likely be underestimated by visual census methods (Smith-Vaniz *et al.* in press), and active and passive gear. Cryptic species can be quantified more-effectively using ichthyocides (e.g. rotenone) (Smith-Vaniz *et al.* in press; Ackerman and Bellwood 2000), but ichthyocides negatively impact the studied assemblage.

2.2.3 Measurement Biases

A sample will rarely provide an absolutely accurate measurement, but the mean of many unbiased samples will tend towards the mean of the population. A measurement bias occurs when the measurement process affects the measurements in such a way that the sample mean does not tend towards the population mean, but rather another (sometimes unknown) value. Common causes of measurement bias include inconsistent survey effort, diver behavior, species detectability, observer experience/training, and fish density. The primary reason fishery-independent data are sought for monitoring programs is because fishery data are plagued by measurement biases associated with differences in catch per unit effort. Thompson and Mapstone (1997) demonstrate observer bias can be considerable in underwater visual censuses and provide guidance for observer training to ameliorate its impact.

In general, it is assumed that the method of measurement chosen has accounted for issues in selectivity and minimizes the probability of non-detection of a species if it is present in the sampling area. Selectivity is defined as the probability that an individual will be detected (or retained) by the sampling method (gear) given that it is vulnerable (Gunderson 1993). While there is always the possibility that a species will not be detected at a site, despite being present, for most species and gear types this non-sighting probability diminishes with increasing animal size, thus demanding strategic choice of the methods of measurement. There are well-known statistical methods for correcting for size selectivity by gear or method in sampling surveys (e.g., Pope *et al.* 1975; Gunderson 1993). In addition, MacKenzie *et al.* (2002), Azuma *et al.* (1990) and Tyre *et al.* (2003) give some insights into correcting frequency of occurrence data when there are unequal selection probabilities amongst species.

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3 Population to be Sampled and Selection of the Sample

3.1 Population to be Sampled

A primary consideration of a monitoring program is to delineate the target population which will be monitored. For reef fish, this can be done by selecting an ecosystem area to be surveyed. This surveyed area in which ecological processes occur and management decisions will be made is known as the survey domain.

A distinction must be made between the target population (the population about which information is desired) and the sampled population. These populations should coincide, but sometimes due to practicality or convenience the sampled population is a restricted part of the target population. It is important to note that if a difference among these populations exists, conclusions drawn from the sample only apply to the sampled population (Cochran 1977).

Population and community assessments correspond only to fish populations within the survey domain, thus the selection and accurate demarcation of the domain is essential for meaningful fish management decisions. Although management efforts in the SFCN are directed towards the areas within park boundaries, the reef fish assemblages in these parks depend on and interact with surrounding areas over a much larger spatial-scale. To comprehensively monitor and manage the reef fish inside the parks and assess effectiveness of NPS management strategies, the survey domain should incorporate areas outside park boundaries as well.

3.2 Selection of the Sample

There are many ways to select a sample, but the more information available about a population, the easier it is to devise a selection method which provides accurate and precise survey estimates. Simple random sampling (SRS) is the simplest and most fundamental probability-based survey design allowing inferences to be made from sample units to the sampled population. In the previous section, we defined the sample unit for a given sampling method to have a known constant area. The complete list of all non-overlapping, independent sample units comprises the sampling frame. The SRS design considers all sample units in the sampling frame equal (i.e. all sample units have the same probability of being selected) and thus is appropriate for situations where there is no spatial structure in the variance of investigated metrics. Fish populations and communities are rarely homogenous in nature. More often, the principal metrics of reef fish show strong association with benthic habitats, depths, salinity, and other environmental covariates (Ault *et al.* 1999; Kendall *et al.* 2003, 2004) and thus are heterogeneous.

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A survey design will sample a population more effectively if the corresponding variations in the spatial structure of variance within the survey domain is known and taken advantage of. A stratified random sampling design (StRS) uses the variance structure to select sample units more efficiently from the sampling frame. A StRS design may divide the survey domain into regions of relatively homogenous variance called strata and by sampling more intensively in highly-variable strata, a StRS can achieve better results than a SRS using the same sample size.

For any given metric the best criteria to use when constructing strata is the variance structure of the metric itself. As this knowledge is unknown (or else sampling would not be need), the next best criteria are variables that are highly correlated with the metric of interest. Maps of environmental covariates at the appropriate spatial scales and spatial extent are ideal, since the sampling frame is situated in a spatial framework.

The benthic habitat maps of shallow-water (depth 0 - 20 m) areas by FMRI (1988), Kendall *et al.* (2001, 2005) and Franklin *et al.* (2003) are exemplary maps of a covariate for BISC, BUIS, DRTO, SARI and VIIS. These maps classify benthic habitats that are strongly correlated with principal reef fish population and community metrics. Ault *et al.* (1999, 2005a) and case studies A and C have shown parsing the survey domain according to benthic habitat types will dramatically improve sampling efficiency compared to a SRS.

To effectively parse the survey domain into strata requires an analysis of covariance among reef fish metrics and environmental variables (e.g. benthic habitat, salinity, depth). A broad assortment of variance analysis techniques such as plots of stratum standard deviations against stratum means, Analysis of Variance (ANOVA), generalized linear models, and resampling methods can be used to investigate the variance structure of metrics. **Figure 3** is a plot of the sample standard deviation of snapper (Family: Lutjanidae) against the corresponding average sample density among 10 distinct benthic habitat types around BUIS. The graph shows that snappers were not homogeneously distributed throughout the survey domain, and that stratification according to benthic habitat can be effective in partitioning the domain into areas with differing variances. **Figure 3** also indicates that a stratification scheme employing benthic habitat type can be simplified by merging relatively similar benthic habitat types (e.g. linear reef and aggregated patch reef) into a single stratum.

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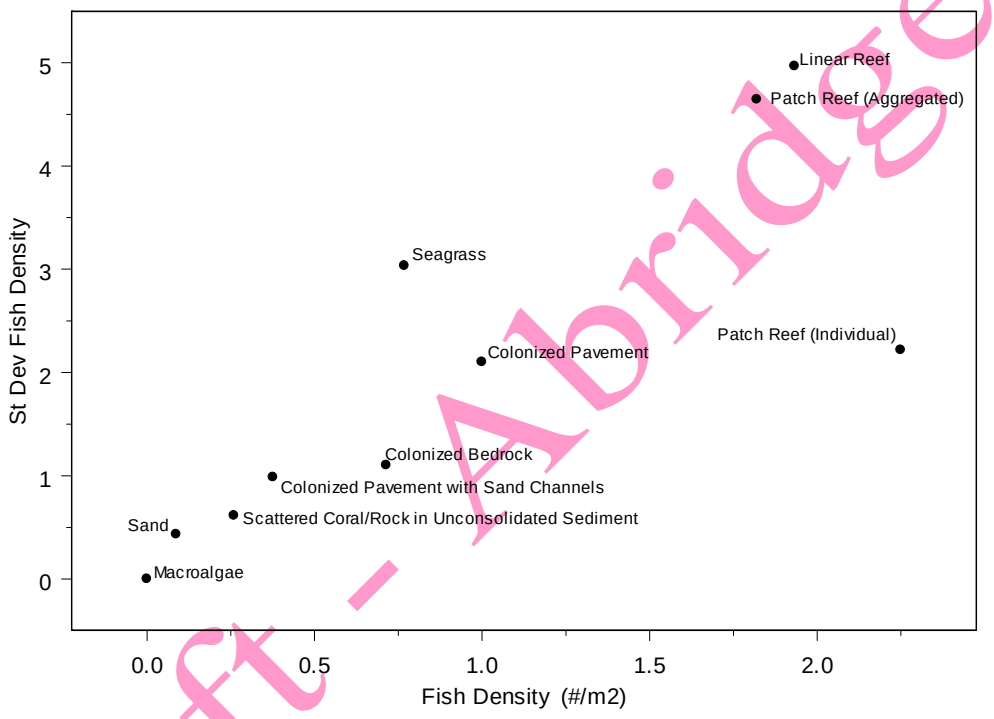

Figure 3: A plot of the sample standard deviation of snapper (Family: Lutjanidae) against the corresponding average sample density among 10 distinct benthic habitat types around BUIS. In conjunction with a map of benthic habitat types, the information in this plot may be used to develop an efficient survey design.

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Another feature of **Figure 3** is the strong relationship between stratum mean density and standard deviation. This is a common phenomenon in surveys of marine animal populations (e.g., Ault *et al.* 1999, 2003). Thus, a stratification scheme that effectively partitions the domain with respect to variance of animal density may also effectively partition the domain into areas of differing mean densities.

Commonly, a monitoring program will initially use a SRS design because of a scarcity of fish and covariate data needed to ascertain spatial relationships. As data are gathered and covariance analyses are performed, more efficient survey designs such as a StRS can be adopted. The exploration of covariance after sampling using strata differing from those actually implemented requires poststratification analysis on domains of study. Ault *et al.* (1999) use poststratification as a comparative stratification scheme analysis tool for pink shrimp in Biscayne Bay. Cochran (1977) describes the process of poststratification and corresponding computations for both SRS and StRS designs.

Although statistical techniques such as ANOVA can reveal trends and regions of relatively homogenous variance in the survey domain, they cannot be used for hypothesis tests, unless the underlying population data structure in each stratum is known and the data conforms to test-specific assumptions (e.g., homogeneity of variance, normality, independence, etc.). Goodness-of-fit tests (D'Agostino and Stephens 1986) identify suitable distributions and consequently the most appropriate tests to use if hypothesis tests are required. In some cases, applying a transformation modifies the data structure to one assumed by a particular statistical technique. Commonly used transformations are listed in Box and Cox (1964), Sokal and Rohlff (1995), and Zar (1999).

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4 Candidate Sampling Design Analysis

The main goal of statistical sampling surveys is to obtain accurate, high-precision estimates of population and community metrics at relatively low cost. The statistical estimation methods of survey design presume that the population of interest is finite and inhabits a finite spatial domain; consequently, these methods are well suited for application to reef fish populations and communities at ecosystem scales appropriate for resource management. Principles of statistical survey design are outlined in Kish (1967), Cochran (1977, 1983), Williams (1978), Yates (1981), Kalton (1983), Kalton and Anderson (1986), Thompson and Seber (1996), and Lohr (1999).

The objective of sample design analysis is to determine the appropriate number of samples to be taken to achieve a certain level of precision for detecting change in population and community metrics (e.g., species numbers-at-size, species composition) used to understand ecological processes and to make management decisions. The specification of a degree of precision desired is an important step in sample surveys and is the responsibility of the park managers and researchers who use monitoring data.

4.1 Basic Concepts of Sampling Theory and Designs

As discussed in section 3.2, populations of coral reef fishes within an ecosystem-scale sampling domain are usually heterogeneously distributed in space rather than homogeneously distributed. In this situation, a StRS design that effectively partitions the domain into distinct strata which are internally homogenous will usually outperform other types of sampling designs (e.g., simple random, systematic, etc.). Basic concepts of StRS designs are illustrated using two population metrics, fish density $\overline{Y}$ (number of individuals per unit area) and fish abundance Y (total number of individuals). The concepts can be applied to SRS designs as well by taking the number of strata (L) equal to one. Observations of density y_i for a given species are the number of individuals observed or captured in a standard sample unit i, (e.g., a belt transect of 100 m^2). An estimate of the mean density in stratum h ($\overline{Y}_h$) is given by

$$\overline{y}_h = \frac{\sum_{i=1}^{n_h} y_{hi}}{n_h} \tag{4.1}$$

where n_h is the number of units sampled in stratum h and y_{hi} is the density in stratum h and sample unit i. An estimate of the stratum variance (S_h^2) is given by

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$$s_h^2 = \frac{\sum_{i=1}^{n_h}\left(y_{hi} - \bar{y}_h\right)^2}{n_h - 1} \tag{4.2}$$

The estimated stratum variance is used to estimate the variance of mean density in stratum h,

$$\mathrm{var}\left[\bar{y}_h\right] = \left(1 - \frac{n_h}{N_h}\right)\frac{s_h^2}{n_h} \tag{4.3}$$

where N_h is the total possible sample units in stratum h. The quantity $\left(1 - \frac{n_h}{N_h}\right)$ in equation (4.3) is termed the finite population correction (FPC), where $\frac{n_h}{N_h}$ is the sampling fraction, or the proportion of the domain of stratum h that is actually sampled. Note in equation (4.3) that increasing sample size n_h reduces the variance of the estimate of mean density in two ways, first by reducing the quantity $\frac{s_h^2}{n_h}$ and second by reducing the FPC. In practice the FPC can be ignored whenever the sampling fraction is less than 5% (Cochran 1977). The resulting equations are simpler, but variance estimates are higher.

Given that $\bar{y}_h$ represents the stratum mean number of animals per sample unit, it follows that stratum abundance is estimated by multiplying mean density by the total number of sampling units,

$$y_h = N_h \bar{y}_h \tag{4.4}$$

Variance of y_h is estimated in a similar manner,

$$\mathrm{var}\left[y_h\right] = N_h^2\,\mathrm{var}\left[\bar{y}_h\right] \tag{4.5}$$

Note that controlling the variance of stratum mean density (equation 4.3) in turn controls the variance of stratum abundance (equation 4.5).

A beneficial property of sampling design theory is that stratum estimates of population means (equation 4.1), totals (equation 4.4), and their associated variances (equations 4.3 and 4.5) are unbiased (i.e., accurate) provided that sampling is done in a random manner (Cochran 1977). The randomization procedures employed in case studies A, B, and C provide practical approaches. Sample units within a stratum were uniquely identified with respect to geographical location in a GIS. The units were then assigned a number from 1 to N_h. Specific units to be sampled within a

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stratum (totaling n_h) were selected from the complete list of N_h units using a random number procedure based on the discrete uniform probability distribution, which assigns equal selection probability to each sample unit. The procedure was repeated for each stratum in the sampling domain.

Domain-wide estimates of population means and totals are computed from the individual stratum estimates taken from samples. Mean density for the stratified survey domain is obtained by summing the weighted averages of sample strata means,

$$\bar{y}_{st} = \sum_{h=1}^{L} W_h \bar{y}_h \tag{4.6}$$

where L is the number of strata, and strata weighting factors (W_h) are given by

$$W_h = \frac{N_h}{\sum_{h=1}^{L} N_h} = \frac{N_h}{N} \tag{4.7}$$

where N is the total number of possible sample units in all strata. The weighting factor W_h represents the proportion of the overall survey domain (or sampling frame) contained within stratum h. In a SRS design $W_h = 1$.

The variance of $\bar{y}_{st}$ is estiamted as

$$\mathrm{var}\left[\bar{y}_{st}\right] = \sum_{h=1}^{L} W_h^2 \,\mathrm{var}\left[\bar{y}_h\right] \tag{4.8}$$

Domain-wide population abundance y_{st} and associated variance $\mathrm{var}\left[y_{st}\right]$ are obtained by summing equations (4.4) and (4.5), respectively, over all strata,

$$y_{st} = \sum_{h=1}^{L} y_h \tag{4.9}$$

and

$$\mathrm{var}\left[y_{st}\right] = \sum_{h=1}^{L} \mathrm{var}\left[y_h\right] \tag{4.10}$$

An important point to remember about a StRS design is that the variance of the domain-wide mean or total depends on the estimates of stratum variance. If a heterogeneously distributed population were divided into strata such that all strata were homogenous (i.e. $\sum_{h=1}^{L} \mathrm{var}\left[\bar{y}_h\right] = 0$), then population estimates would be made without error. Consequently, the basic objective of stratification is to

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partition the sampling domain into sectors of homogenous variance for population metrics such as animal density. Section 3.2 describes stratification techniques.

Developing a StRS design in practice requires both a scheme for stratifying the sampling domain and a scheme for allocating sample units among strata. There are two allocation schemes commonly used for StRS designs. The first is proportional allocation, in which sample units are allocated among strata according to stratum size,

$$n_h = n \cdot W_h \tag{4.11}$$

where n is the total sample size for the survey. The second scheme is Neyman or optimal allocation in which sample units are allocated according to both stratum size and the strata standard deviations of a considered population metric (e.g. density),

$$n_h = n \cdot \left(\frac{W_h s_h}{\sum_h W_h s_h} \right) \tag{4.12}$$

Under this strategy, larger and more variable strata will receive more sampling effort, and vice versa for smaller, less variable strata.

Neyman allocation, in concert with an effective stratification scheme, can substantially reduce the variance of domain-wide population estimates (e.g., equation 4.8) compared to a simple random sampling (SRS) design of similar sample size (Cochran 1977). In contrast, reductions in estimate variance (i.e., increases in the precision of estimates) may not be achieved for a proportional allocation scheme. In theory, a SRS design with a sufficiently large sample size will be equivalent to a StRS design employing proportional allocation with respect to domain-wide estimates of population means (e.g., equation 4.6) and variances (e.g., equation 4.8). However, when sampling heterogeneous populations such as reef fishes in practice, a StRS design with proportional allocation will at least ensure that all strata will be sampled and thus provide a guard against bias in domain-wide estimates of population means and totals as discussed above. This will especially be true for surveys with relatively modest sample sizes.

4.2 Sampling Design Performance Measures

Performance of sampling designs involves the trade-offs between survey costs (usually measured by sample sizes) and the precision of population estimates. Several performance measures can be computed to evaluate the efficacy of sampling designs. The most basic and perhaps most familiar performance measure is the standard error (SE) of an estimate, computed by taking the

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square root of the variance of an estimate. For the case of mean density, the standard error is given by

$$SE\left[\bar{y}_{st}\right] = \sqrt{\mathrm{var}\left[\bar{y}_{st}\right]} \tag{4.13}$$

The standard error can be used to compare the performance among design types (eq. 3.1). A relative measure of precision is the coefficient of variation (CV) of mean density,

$$CV\left[\bar{y}_{st}\right] = \frac{SE\left[\bar{y}_{st}\right]}{\bar{y}_{st}} \tag{4.14}$$

in which the standard error is expressed as a proportion (or percentage) of the mean. A key performance measure is n^*, the estimated sample size required to achieve a specified variance in a future survey. Computation of n^* (presumed optimal allocation) is carried out for mean density using

$$n^* = \frac{\left(\sum_h w_h s_h\right)^2}{V + \frac{1}{N}\sum_h w_h s_h^2} \tag{4.15}$$

where N is the total sample units in the domain and V is the desired variance. A convenient way to express the desired variance is

$$V = \left(CV\left[\bar{y}_{st}\right] \cdot \bar{y}_{st}\right)^2 \tag{4.16}$$

using a target CV of domain-wide mean density.

Alternatively, if the performance measure is a margin of error (d) as used in confidence intervals then

$$V = \left(\frac{d}{t}\right)^2 \tag{4.17}$$

where t is the normal deviate corresponding to the probability that the error will exceed d. This error is commonly referred to as Type I error. Case study A uses a performance measure which subsumes both Type I and Type II error rates in computations of n*.

Aspects of design performance are illustrated in **Figure 4**, which shows performance data for estimates of black grouper density from StRS surveys in the Florida Keys coral reef ecosystem, including Biscayne National Park, during 1994-2002. Habitat-based stratification and visual sampling methods for these surveys were similar to those described in Case Study C (Ault *et al.*

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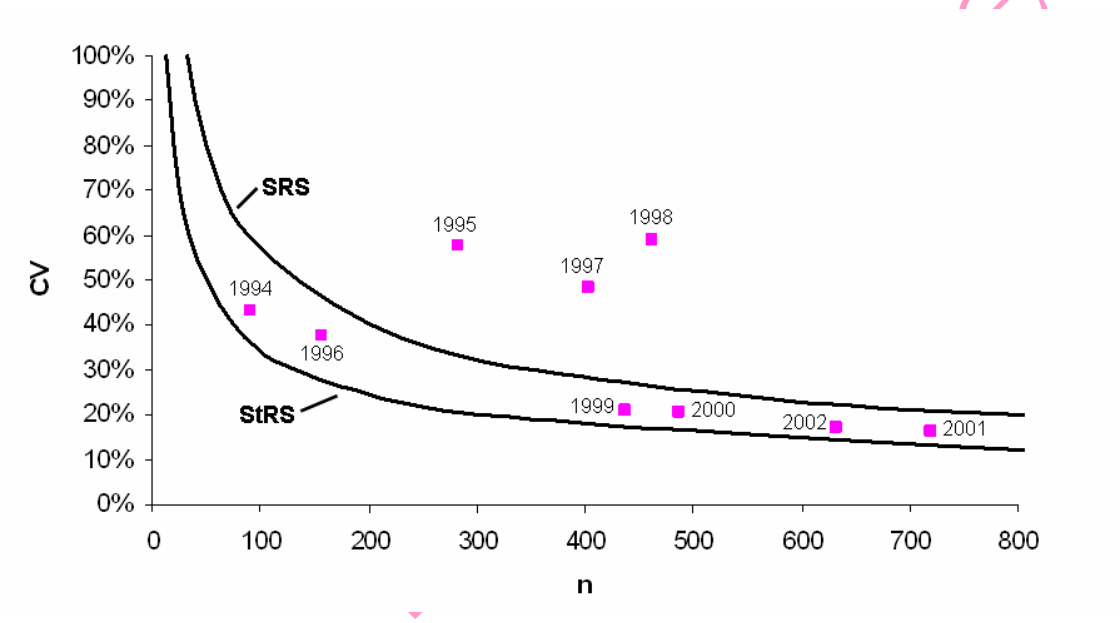

Figure 4: Survey design performance data for annual estimates of black grouper density in the Florida Keys coral reef ecosystem, including Biscayne National Park, during 1994-2002. Annual estimates are compared to the relationship of CV versus *n* for a stratified by benthic habitat-based survey design (StRS) and a simple random survey design (SRS). Habitat-based stratification and visual sampling methods for surveys were similar to those described in Ault *et al.* (2001).

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2001). The CV-n relationship (solid line) for the StRS design was estimated using equations (4.15) and (4.16). The CV-n curve shows that gains in precision (i.e., decreases in CV) occur as *n* increases (i.e., as the sampling budget increases), but the gains are not limitless. For the case of black grouper, increasing *n* from 50 to 200 would be expected to result in a substantial decrease in CV, but increasing *n* from 700 to 800 would be expected to result in almost no appreciable decrease in CV.

A standard benchmark for performance of StRS designs is to compare these results with those obtained for a simple random sampling (SRS) design. The difference between a SRS design and alternate sampling design is known as the design effect. It is typically described as the ratio of the variance from the more complex design to the variance from a SRS design with the same sample units. The CV-n curve for a SRS design for black grouper was estimated by considering the whole survey domain as a single stratum. Comparing the CV-n curves for the SRS and StRS designs highlights the potential for achieving gains in precision through stratification of the domain by variables that account for spatial heterogeneity in density. Estimates of n*, the value for n in CV-n curves, presume that sample units are allocated among strata according to a Neyman scheme. The CV-n curve for the StRS design thus represents a kind of minimum bound of CV that could be achieved in practice for a given n, because it presumes that samples are allocated on the basis of stratum size as well as stratum variance. The vertical distance between the actual CVs for black grouper density (point values by survey year) and the corresponding potential CV (CV-n curve) represents the gain in precision that could be achieved by more effective allocation of samples among strata. For the Florida Keys surveys, formal procedures of stratification, allocation, and randomization were instituted in 1999. The example of Figure 4 thus shows that achieving high precision is not simply a matter of cost, but rather the combination of effective stratification and allocation along with total sample size.

4.3 Composite Sampling Designs

Surveys for reef fishes will usually entail multiple target species, multiple species life stages (e.g., juvenile, adult, exploited), and multiple metrics (e.g., population abundance, community diversity). It is likely that a sampling design that performs well for one case may not perform well for other cases, requiring some sort of compromise. Obtaining a compromise from a constrained set of metrics will prove less challenging than for numerous metrics. A sensible initial step is to reduce the number of metrics to a set deemed most important and representative of other metrics.

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A practical strategy for design development in this situation is illustrated in Ault *et al.* (1999) where analysis of density-habitat relationships showed that the spatial distribution of pink shrimp differed with respect to life stage (juvenile and subadult). The iterative design analysis process outlined above was applied to each life stage, and then a 'composite' stratification and allocation scheme was formulated that performed reasonably well for both life stages, although a more efficient StRS design could have been implemented for either juvenile or subadult pink shrimp. Cochran (1977) describes a process to determine a composite allocation scheme for correlated metrics when a single stratification scheme is used. The compromise is taken from the average of optimum stratum allocations among metrics. Alternative, but computationally-intensive optimization techniques are given by Chatterjee (1967), Kokan and Khan (1967), Bethel (1989) and Rahim and Currie (1993).

4.4 Iterative Learning

Development of efficient (high precision, low cost) sampling designs for marine animal populations in practice usually occurs through an iterative learning process of design formulation, sampling, and performance analysis that leads to improved design formulation, sampling, and so on. The study by Ault *et al.* (1999) provides a detailed application of this iterative process to develop an efficient StRS design for a roller-frame trawl survey targeting pink shrimp in BISC. The main steps of the iterative learning process were as follows:

1) Pilot surveys were conducted in different seasons to obtain information on the temporal and spatial dynamics of the pink shrimp population in Biscayne Bay, and also to refine field sampling methods (e.g., optimal tow distance, etc.).

2) A variety of statistical methods, including some of the modeling tools described in Chapter 3, were used to identify key habitat variables influencing the spatial distribution of pink shrimp within and among seasons.

3) Alternative stratification schemes were developed based on different combinations of influential habitat variables. The design performance of these alternative schemes was evaluated using the technique of post-stratification to identify the most efficient StRS design for a future survey.

4) The refined StRS design was used to conduct a new survey, and steps 2 and 3 were repeated to further improve the sampling design.

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5 Population and Community Assessments

The range and types of statistical analyses that will be performed to assess the status and dynamics of reef fish populations and communities in National Parks depends on the specific management questions and resource goals to be addressed. These analyses, by and large, utilize the range of fundamental survey data (abundance, size, taxonomic identification) outlined and recommended for collection in survey monitoring programs in Section 2. These survey data are then used to generate metrics for individual species and assemblages to assess status and trends of reef fish communities and populations over time and in relation to specific sustainability metrics.

5.1 Species and Community Metrics

5.1.1 Frequency of Occurrence

Survey data relating species frequency of occurrence, i.e., the proportion of sampled sites that given species are seen, constitutes a primary index of fish community dynamics. Frequency of occurrence makes no specific reference to the actual numbers of a species at sites, but rather that they were simply observed or not. The measure can be used to assess changes in species spatial distributions over time.

5.1.2 Diversity Indices

Diversity indices are measures of species composition. A large number of indices have been proposed to compute species diversity and these are outlined in the seminal works by Pielou (1969, 1977), Hurlburt (1971), Margalef (1974), Peet (1974), Legendre and Legendre (1998) and Magurran (1988).

Species richness is the simplest index and is purely the number of distinct species at sites or that are observed at all sites during a particular monitoring survey. As a fish community index, this statistic is a general measure of fish biodiversity. More complex diversity indices, such as the Shannon index (Shannon 1948), Simpson index (Simpson 1949), or Pielou's J (Pielou 1966) integrate both the number of species and the proportion of individuals in each species. A survey with many species equally represented by the same number of individuals will have a higher diversity index than a survey with fewer species or with an unequal distribution of individuals amongst species.

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5.1.3 Relative and Absolute Population Abundance or Biomass

Relative abundance (i.e., density $\equiv \frac{N_i}{a_i}$, or numbers of animals observed or captured per unit sample area) in a sample unit (*i*) is computed as the total number of individuals (N_s) of species *s* within a given sample area (a_i). This quantity can be configured to represent either a relative index, or it can be related to the absolute average population size, $\overline{N}_{abs}$, by

$$\overline{N}_{abs} = \frac{C}{f} = \frac{N_i}{a_i} = q\overline{N} \tag{5.1}$$

where C is the number of fish observed (or captured) within the unit sample area, f is the nominal unit of effort (here it equals the area searched for 1 unit sample), q is the fraction of the population seen per unit sample, and $\overline{N}$ is the average population size at the time of sampling. In this simple example, it is assumed that the design is proportional to the population (i.e., simple random sample) where all sample units have equal sampling probabilities. In a stratified survey, each of the various strata must be computed individually and weighted as discussed in section 4.1. Relative abundance has been used extensively in fisheries to characterize changes in fish population sizes for status and trends in stock assessments (Quinn and Deriso 1999, Haddon 2001, Gulland 1983) and in reef ecosystems (e.g., Bell 1983; Alcala 1988; Cole *et al.* 1990; Polunin and Roberts 1993; Dufour *et al.* 1995; Russ and Acala 1996; Friedlander and Parrish 1998 and Nagelkerken *et al.* 2000).

The total average population size (e.g., mid-year average) consisting of ages *a* at time *t* would be

$$\overline{N}(t) = \int_{ac}^{a\gamma} N(a,t)da \tag{5.2}$$

where a_c = minimum age at first observation (or capture) and a_γ = oldest age in the population.

Population biomass is an integrated measure of the total mass (W, weight) of living biotic matter (both somatic and reproductive) for given ages at a given time. The most common procedure to estimate population biomass is to determine the relative density or abundance at a sample site, and then use species-specific allometric growth relationships to convert observations of length-at-age to weight-at-age for each individual fish (Bohnsack and Harper 1988; Ault *et al.* 1998, 2005b; Froese and Pauly, 2005). The allometric relationship between weight and length is

$$W(a,t) = \alpha L(a,t)^{\beta} \tag{5.3}$$

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where *W(a,t)* is the weight of a fish at age *a* and time *t*, *L(a,t)* is the length of a fish at age *a* and time *t* and α and β are coefficients of the allometric relationship. Consequently, population biomass, $\overline{B}$, can be calculated for a given species with

$$\overline{B}(t) = \int_{ac}^{a\gamma} N(a,t)W(a,t)da = \int_{ac}^{a\gamma} N(a,t)\left[\alpha L(a,t)^{\beta}\right]da \qquad (5.4)$$

where parameters are as defined before. This can be done for *s* species in the reef fish community and added together for an assemblage estimate.

5.1.4 Population Size-Structure

Size-structure, as derived from the sampled population, is a distributional statistic that reflects the interactions of the population-dynamic processes of individual growth, mortality and recruitment among all sizes and ages of fish in a population (Quinn and Deriso, 1999; Haddon 2001). Park managers should be interested in the status and trends of population size-structure because it provides an integrated metric of what has happened and what will happen to a fish assemblage.

Size-structure, in and of itself, is a complex measure to quantify without the aide of some summary statistic that characterizes the distribution. Ault *et al.* (1998, 2005b) have shown average length of the exploited part of a population ($\overline{L}$) can be a robust indicator of community response to exploitation. This statistic is the principal stock assessment indicator variable to quantify population status (Beverton and Holt 1957; Ricker 1963; Pauly and Morgan 1987; Ault and Ehrhardt 1991; Ehrhardt and Ault 1992; Kerr and Dickie 2001). The statistic $\overline{L}$ is a metabolic based indicator that reflects fishing mortality, because exploitation removes large individuals and species from the community (Ault et al 2002; Gislason and Rice 1998; Pauly *et al.* 1998; Kerr and Dickie 2001). Theoretically, $\overline{L}$ at a given instant is expressed as

$$\overline{L}(t) = \frac{F(t)\int_{ac}^{a\gamma} N(a,t)L(a,t)da}{F(t)\int_{ac}^{a\gamma} N(a,t)da} \qquad (5.5)$$

where a_c = minimum age at first capture or observation, a_γ = oldest age in the stock or population, *N(a,t)* = abundance for age class *a*, *L(a,t)* = length-at-age, and *F(t)* = instantaneous fishing mortality rate at time *t*. In practice, because age is unknown, $\overline{L}$ is calculated between lengths corresponding

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to length at first capture and largest fish observed in the population. Case Study C describes the methods involved in calculating $\bar{L}$ and estimating fishing mortality rates as input into calculations of sustainability benchmarks for the reef fish community in DRTO.

5.2 Assessing Changes to Reef Fish Metrics

An advantageous property of statistical sampling theory is that survey design estimates, such as stratum density (equation 4.1) and the variance of stratum density (equation 4.3), do not require knowledge of the underlying probability distribution (e.g., normal, gamma, etc.) of the respective population metric, (e.g., observations of stratum animal density y_{hj}) (Cochran 1977). A second property, based on large sample theory, is that survey design estimates (e.g., population means and totals) are normally distributed due to the central limit theorem if sample size is large. These properties facilitate the analysis of survey design estimates among times or areas (e.g. monitoring density over time, assessing MPA effectiveness).

A simple, straightforward approach to performing statistical tests for differences among survey estimates for a particular time or area is via inspection of confidence intervals (CI). If the sample is relatively large (n > 100) and has a Normal distribution, the survey mean will lie within a CI bounded by

$$\bar{y}_{st} \pm t_{\alpha/k,df} SE\left(\bar{y}_{st}\right) \tag{5.6}$$

with a probability of α, the Type I error rate, and where t is the critical value of Student's t-distribution, and degrees of freedom $df = n_h$-1. Most commonly used reef metrics (see section 5.1) do not posses a Normal distribution which means the Type I error rate will not equal α. Cochran (1977) states α will be very close to what is expected if

$$n > 25 \times G_1^2 \tag{5.7}$$

where G_1^2 is Fisher's measure of skewness. If a sample is too small and the population is heavily skewed, transforming the data (e.g. $\log\left[y_{st}+1\right], \left[y_{st}\right]^2$) may help.

CIs can be used to test the hypothesis that samples were drawn from the same population, as is done to assess temporal change or determine MPA effectiveness. Cochran (1977), Sokal and Rohlff (1995), and Zar (1999) describe methods using CIs to test for differences among means. A comparison of multiple CIs (e.g. a time series) requires a Bonferoni adjustment to α. The Bonferoni adjustment is necessary because the true Type I error rate of simultaneous multiple tests is not α, as it should be for a single test.

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A useful relationship stemming from equation (5.6) is that the 95% CI for a population metric is approximately twice the CV, because the value of t for $\alpha = 0.05$ and $df > 20$ is approximately 2. Thus, for example, a StRS survey that provides a domain-wide estimate of abundance with a CV of 15% would be able to statistically detect a minimum change of 30% in population abundance between survey time periods (with a Type I error rate = 0.05).

5.3 Population Mortality Rate Assessments Using Size-Structure

Exploitation (or other) effects from fishing mortality could be specifically assessed by bounding the integral for Equation (5.5) to reflect the ages/sizes affected. For example, a minimum size limit L_c would constrain the solution to consider the average size ($\overline{L}$) between L_c and L_γ, the minimum size limit and the maximum size observed in the catch (or seen in the visual samples) or population. In a population where fishing mortality is strictly proportional to the stock, $\overline{L}$ of fish on the dock would be exactly equal to the $\overline{L}$ of those fish remaining in the sea, assuming that recruitment remained constant within a finite range of population sizes.

Average size has been used by several analytical studies to assess the impacts of exploitation on reef fish populations and communities, and thus guides management decisions regarding policies to achieve sustainability of reef fish resources (Williamson *et al.* 2004; Ault *et al.* 1997, 2005a, 2005b, 2006; Nemeth 2005) [see Case Studies].

For the case where no fishing occurs, equation (5.5) in combination with fishery-independent data, could be used to compute the natural mortality rate M, or the life-time expectation of survivorship (i.e., average maximum age in the population).

5.4 Population Biomass Assessments in Relation to Sustainability Benchmarks

An important measure of stock reproductive potential is population spawning biomass. One which is used more frequently in fishery management is spawning stock biomass (SSB). SSB is expressed as

$$\overline{SSB}(t) = \int_{ac}^{a\gamma} N(a,t)W(a,t)da \qquad (5.10)$$

where a_m is the minimum age (or size) of sexual maturity. Case study C provides an example of using the SSB and the derived spawning potential (SPR) ratio to assess fishery management in DRTO. SPR is a management benchmark that measures the stock's current reproductive potential to

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produce optimal yields on a sustainable basis. It is simply the ratio of spawning stock biomass from exploited and unexploited populations. Estimated SPRs can be compared to U.S. Federal standards which define 30% SPR as the overfishing threshold at which the stock is no longer sustainable at the current exploitation levels.

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6 Case Study Foreword

The preceding five sections provide a framework for generating a standardized monitoring protocol for use in SFCN park units. The framework outlines useful methods for a monitoring program, but is not a single standardized monitoring protocol for all park units. The variability in ecological condition, size, management capability, expertise, and available data sets among park units implies different parks will have distinct management objectives and logistical constraints. Monitoring programs must be crafted for individual park units considering specific monitoring needs and abilities. One size of monitoring program does not fit all park units.

Three reef fish monitoring case studies are presented which build upon the presented monitoring framework using park-specific data sets, management concerns, and local partnerships. The case studies are offered to provide persons implementing a monitoring program with the information required to understand the pertinent: 1) management issues, 2) sampling methods, and 3) analytical methods used in monitoring reef fish in SFCN managed areas. The case studies employ distinct methodologies because they reflect differences among park needs and abilities. The case studies are similar, but utlitize different measurement methods, sampling designs, and analyses. These differences among case studies are summarized in **Table 1**.

Case study A implements a stratified random sampling design in BUIS and VIIS. Regional benthic habitat maps are used to increase survey design performance. Field work is undertaken by the NOAA Biogeography Team in cooperation with NPS. Design performance, temporal changes, and MPA effectiveness for several fish assemblages are investigated using survey data from 2001-2005. This case study is a good example of effectively utilizing a moderate amount of resources (e.g. multiple boats, dive teams) to obtain precise metrics of the community and several assemblages of special management concern.

Case study B uses the stationary visual census technique to sample at multiple, permanent, high-diversity coral reef reference sites around and in VIIS. This strategy effectively makes use of few resources to monitor constrained areas with high precision. Field work is conducted by the University of Hawaii in Hilo and NOAA Biogeography Team. Data collceted from 1988-2000 are analyzed for MPA effectiveness and trends.

Case study C employs a two-stage stratified random sampling design to sample over hard bottoms in DRTO. Surveys are conducted by the University of Miami in cooperation with the NOAA Southeast Fisheries Science Center. A regional benthic habitat map and distinct management zones are used to stratify the large survey domain. Analyses of survey design

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performance, sustainability status of exploited fish species, and MPA performance using 1999-2004 data are provided. This case study shows the effective use of a live-aboard dive vessel, multiple dive teams, and cluster sampling to efficiently survey a very large area (320 km^2) and obtain precise metrics for fishery assessments.

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Table 1: Summary of differences in (A) measurement methods, (B) sampling designs and (C) data analyses among case studies.

(A) Method of Measurement

Approach	Reason(s)	Case Study - Section
Belt transect	(1) Increase sighting frequency of small fish, and (2) Some sites characterized by moderate visibility (>2 m), rugose benthic structure, or adjacent to mangrove prop roots	A-3.3
Stationary visual census	(1) Decrease measurement bias due to diver movement, and (2) Majority of sites have good visibility (>7.5 m)	B-3.4, C-2.3

(B) Sampling Design

Approach	Reason(s)	Case Study - Section
Stratified random sampling design. Survey domain encompasses whole park and surrounding areas. Strata classified according to a covariate benthic habitat map.	(1) Increase survey estimate precision and reduce sampling cost compared to SRS (2) Obtain representative samples (3) Survey/Make inferences to whole fish community in park (4) Obtain estimates of specific areas in survey domain (e.g. MPA)	A-3.2
Stratified random sampling design. Survey Domain encompasses only reference sites.	Concentrate few resources into understanding references sites very well	B-3.2
Multi-stage stratified random sampling design. Strata classified according to benthic habitat map	(1) Same as first, and (2) Cluster sample units according to mapped covariate (3) Sample over a large area (4) Increase precision by applying finite population correction on first stage of sampling	C-2.2

(C) Analysis of Data

Approach	Reason(s)	Case Study - Section
Analysis of differences in survey estimates among years and areas using confidence intervals.	Assessment of change in survey estimates among years or among areas	A-5.3, B-4.5, C-3, C-5
Analysis of size structure, average size, and spawning stock biomass	Assessment of population mortality rates and biomass sustainability benchmarks	C-4
Analysis of trend in survey estimates using Generalized Linear Model	Assessment of long-term linear trends in survey estimates among years	B-4.5, B-4.6

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