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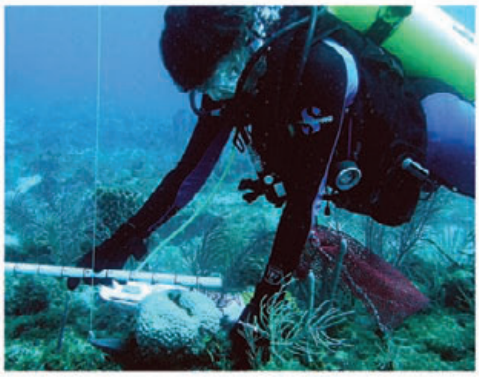
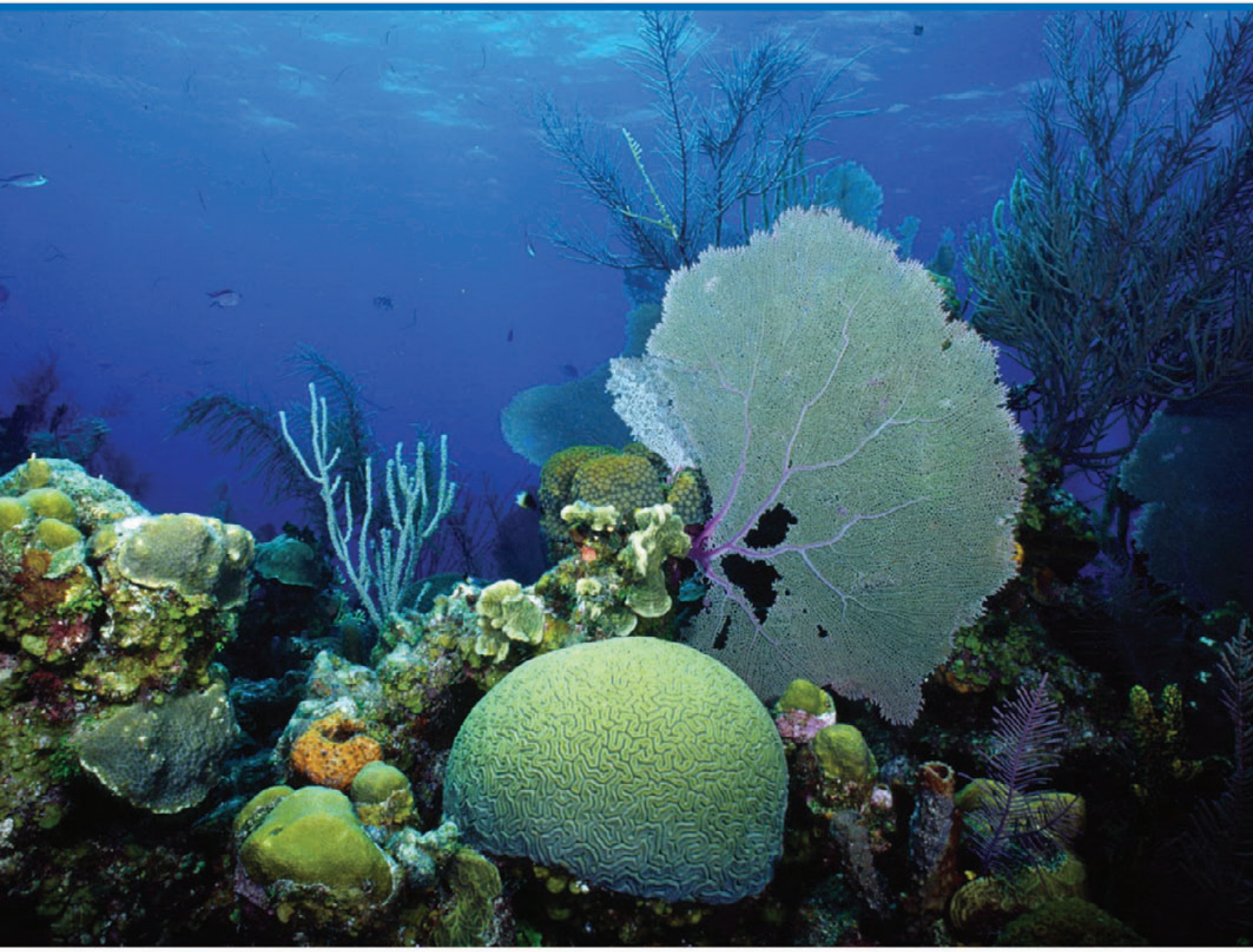



Stony Coral Rapid Bioassessment Protocol



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Stony Coral Rapid Bioassessment Protocol

by

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Executive Summary

At a time when coral reefs worldwide are in the greatest decline of their known existence, and despite the enormous value of coral reef ecosystem services, there are relatively few management tools available to offset the growing impact of human activities. Bioassessments and biocriteria can be used to protect coral reefs in the same way they are used to protect other aquatic resources in the United States. As authorized by the Clean Water Act, U.S. jurisdictions can implement measures of biological integrity (bioassessments) to determine whether a waterbody is meeting resource expectations. When a waterbody is found impaired, jurisdictions have the authority to use those same measures as the basis for implementing corrective action, including changes to human activity in the watershed and waterbody.

The Stony Coral Rapid Bioassessment Protocol (RBP) is an inexpensive, no-contact, nontechnical underwater survey procedure designed for jurisdictions with coral monitoring expertise on staff but with limited time and funding. The protocol focuses on scleractinian (stony) reef-building corals because of their fundamental importance to coral reef ecology and ecosystem value. This focus provides vital information for reef assessment but is not intended to limit development of additional measurements that incorporate other components and processes of the reef community. Only three observations are required—coral identification, size, and proportion of live tissue—all reported for each colony in the sampling transect. These simple underwater observations have been used independently in previous monitoring programs, but when used in combination, they provide a robust array of relevant and informative condition indicators. A unique aspect of the RBP is conversion of colony-size measurements to topographic three-dimensional coral surface area; this augments the number of useful indicators and incorporates both colony and surface area approaches in coral assessment.

A clear benefit of the Stony Coral RBP is the number and relevance of coral condition indicators that can be calculated, indicators that represent numerous biological, physical and ecological aspects of stony corals. For regulatory monitoring, the indicators are screened to determine which respond to human disturbances over natural variability—this is because the Clean Water Act is intended to protect resources against human-induced decline, not decline resulting from natural environmental change. Indicator responses can be influenced by a variety of factors unrelated to human disturbance and will vary for different coral communities at different locations. Because not all indicators will be responsive under all conditions, it is an asset to have many useful candidate metrics to screen.

Biocriteria, or any enforceable regulations derived from bioassessments, require scientifically sound monitoring programs capable of distinguishing impairment. Design of the monitoring program requires a rigorous examination of metric variability, reference conditions, reef classifications, sampling strategies and designated uses and must be sensitive to the limitations of agency resources. Preliminary biological surveys are needed to evaluate these monitoring variables, but once a competent monitoring program is installed, it will serve the jurisdiction for many years and provide valuable, long-term records of coral condition and regulatory compliance.

The principal purpose of the *Stony Coral Rapid Bioassessment Protocol* is to introduce a simple and rapid coral survey method that provides multiple bioindicators to characterize coral condition. This document offers insight on indicator relevance to ecosystem services (societal values), reef condition and sustainability. Information regarding regulatory programs is provided, and a few examples are presented to describe how bioassessment indicators can be incorporated into a regulatory biocriteria program to conserve coral resources.

1. Bioassessment and Regulatory Monitoring

1.1 Role of bioassessment in regulatory monitoring

Biological monitoring is used to detect status and change in the health of living organisms and populations. Health signifies the cumulative and integrated response of organisms to both beneficial and adverse factors in the environment. The health of resident communities (biota) can thus represent the environmental status of a habitat. Measurement of biological attributes to represent environmental status constitutes a biological assessment (or *bioassessment*, Table 1-1). Under the Clean Water Act (CWA), bioassessments can be used to evaluate the condition of a waterbody and to trigger management action if the waterbody fails to comply with biological expectations. Expectations are based largely on *biological integrity*, or the ability of a waterbody to support and maintain a balanced, integrated and adaptive biological system having the full range of elements and processes expected for its region (Karr and Dudley 1981; Karr 1996). Elements and processes that compose biological integrity include species composition, diversity and functional organization comparable to that of natural habitats within the region. In a regulatory context, biological integrity can be used to enforce remediation of a waterbody that does not meet expectations for its designated use.

The environmental status of waterbodies and the health of organisms inhabiting waterbodies are determined by the dynamics of physical, chemical and biological factors in the environment. Natural environmental factors can cause adverse biological effects, but biological monitoring programs are primarily intended to characterize the effects of anthropogenic stressors, which include any "man-made or man-induced alteration of the physical, chemical, biological or radiological integrity of water" (CWA 1988). To apply the authority of the CWA thus requires that indicators be more responsive to human activities than co-occurring natural factors. Such indicators, called *metrics*, exhibit a consistent and logical change along a gradient of human activity (Table 1-1). Natural stressors also influence the condition and sustainability of resources in a waterbody, but natural stresses are regarded as agents of change in an adaptive biological system.

Bioassessments are used to identify impaired waters and to measure the success of remedial actions. Because of this, bioassessments provide a foundation for development of an important regulatory tool, biological criteria (biocriteria). ***Biocriteria** are benchmark, guideline or threshold values that describe the expected (or desired) biological integrity of a waterbody.* The criteria may be either narrative expressions or numeric values adopted into state, territory or tribal water quality standards for assessment thresholds or restoration goals. Section 5 provides some examples of how bioassessment indicators can be used in development of a biocriteria program.

All bioassessments, including those used in biocriteria, are condition measurements that reflect the cumulative and integrative effects of multiple stressors. They are not exposure, stressor or performance measurements. Bioassessments complement the U.S. Environmental Protection Agency's (EPA) traditional chemical-specific water quality standards because they can identify impairment from nonchemical and nonpoint sources of pollution. The combined use of chemical, physical, toxics and biological criteria in water quality standards serves to better protect natural aquatic life and habitats.

Coral reefs occur in waterbodies that provide a wide variety of values for human society. The CWA requires that U.S. jurisdictions develop water quality standards that define designated uses (such as drinking water, recreation and fisheries) for navigable waters and institute criteria for protecting

Table 1-1. Terms used for defining biological condition

Term	Definition
Indicator (sometimes termed *endpoint*)	A measured characteristic that indicates the condition of a biological, chemical or physical system.
Attribute	Measurable part or process of a biological system.
Biological monitoring	Sampling the biota of a place (e.g., corals in a coral reef).
Biological assessment	Sampling the biota to evaluate the environmental condition (status) of a place.
Metric	Attribute empirically shown to change in value along a gradient of human influence. A dose-response context is documented and confirmed.
Multimetric index	An index (expressed as a single numerical value) that integrates several biological metrics to indicate the environmental status of a place.
Biological integrity	Exhibiting a balanced, integrated and adaptive biological system having the full range of elements and processes expected for a particular region. Biological integrity is the product of ecological and evolutionary processes at a place in the relative absence of human influence (Karr 1996).
Biological criteria	Narrative expressions or numerical values that define an expected or desired biological condition for a waterbody and can be used to evaluate the biological integrity of the waterbody. When adopted by U.S. jurisdictions, they become legally enforceable standards.

Source: Karr and Chu (1999)

such uses. EPA has developed a national framework and guidance on tiered aquatic life uses (TALU), which refines designation of aquatic life uses along a biological condition gradient. Use designations stem from political and social considerations as well as insight and appreciation of ecosystem values (services). Prior knowledge of ecosystem services will help to avoid inappropriately high or low waterbody use designations. Thresholds derived for biocriteria are based on conditions at reference sites, historical data, empirical models and regional expert judgment. For example, bioassessment data from reference, or *minimally impaired*, sites might provide reasonable expectations concerning the structure and function of the resident biological community for that particular region.

Pursuant to the purpose of the CWA (section 101[a]), federal and state governments are required to "restore and maintain the chemical, physical and biological integrity of the [n]ation's waters," including coral reefs, within U.S. boundaries and territorial waters. The CWA imparts legal authority to the U.S. Environmental Protection Agency (EPA) to protect and maintain the nation's waters and watersheds and to derive thresholds, such as coral reef biocriteria, for the protection of those habitats. EPA therefore plays a key role in biocriteria development for restoration and maintenance of biological integrity in the nation's waters.

Other sections of the CWA establish various programs and authorities for implementation of its goals and objectives. The following are relevant portions of the CWA that rely on biological monitoring and assessment (CWA 1988):

- Section 303(c)(2)(A) provides statutory authority for states, tribes and territories to develop water quality standards that consist of a designated use that supports aquatic life (e.g., corals) and recreational activities, criteria to protect that use, and an anti-degradation policy to prevent any further loss or degradation in the system. It states, "…[s]tate water quality standards shall protect and enhance the quality of water and serve the purposes of the Act, including protecting and propagation of a balanced indigenous population of fish, shellfish, and wildlife [fishable/swimmable] and recreation in and on the water."
- Section 304(a) provides statutory authority to develop biological criteria: "EPA shall…develop and publish information on methods for establishing and measuring water quality criteria for toxic pollutants on other bases than pollutant-by-pollutant criteria, including ***biological monitoring*** and ***assessment methods***."

- Section 305(b) establishes a process for reporting information about the quality of the nation's water resources. States and tribes are required to assess the general status of waterbodies and identify, in general terms, known or suspected causes of water quality impairments, including ***biological impairments***. This information is compiled into a biennial *National Water Quality Inventory* sent to Congress (i.e., 305(b) Report).
- Section 303(d) requires that states and tribes prepare and submit lists of specific waterbodies that currently violate or have the potential to violate water quality standards, including designated uses and numeric or narrative ***biocriteria***. Those waterbodies *listed* as failing to meet the water quality standards require a total maximum daily load (TMDL) designation. The TMDL process quantifies the loading capacity of a waterbody for a given stressor and ultimately provides a quantitative means to allocate pollutant loads. A TMDL is suitable for chemical as well as nonchemical stressors, such as sediment deposition or physical alteration of habitat.
- Section 319 establishes a voluntary nonpoint source control program by which jurisdictions can control impacts of runoff using guidance and information regarding different types of nonpoint source pollution. ***Bioassessment protocols*** are particularly effective for characterizing cumulative and integrated effects of multiple stressors such as those from nonpoint sources.
- Section 402 makes it illegal to discharge any pollutant to waters of the United States from a *point source* unless authorized by a National Pollutant Discharge Elimination System (NPDES) permit. A permit is required in any case where a discharger could cause a water quality violation, including ***biological impairments***.
- Section 301(h) describes a Waiver Program that allows marine dischargers to defer secondary treatment if they can show that discharge does not adversely affect biological communities. As part of this program, extensive ***biological monitoring*** is required to detect any effects on the biological communities.
- Section 403(c) requires that all ocean dischargers provide an assessment of the ***biological community*** in the area surrounding the discharge.
- Other federal acts that apply to coral reef protection and biocriteria include the Ocean Dumping Act (MPRSA), the Rivers and Harbors Act, and the Coastal Zone Management Act, as well as various programs adopted by states, tribes and territories.

Biomonitoring and bioassessment can be employed in all the above programs, and the Stony Coral RBP can be used when coral reefs are the target resource. Biological monitoring is also an indispensable aspect of problem formulation and effects characterization in ecological risk assessment (USEPA 1992).

Because of this high regulatory relevance, bioassessment procedures and biocriteria programs have been recommended for several aquatic systems. Technical guidance has been prepared for streams and rivers (Plafkin et al. 1989; USEPA 1990; Klemm and Lazorchak 1995; Davis et al. 1996; Barbour et al. 1999; USEPA 2002), estuarine and near coastal waters (USEPA 1997; USEPA 2000a), and lakes and reservoirs (USEPA 1998). A summary of the purpose and history of bioassessment protocols and biocriteria is presented in Barbour et al. (1999). One reason for the success of biocriteria programs is the development of efficient and informative rapid bioassessment protocols (RBP; Table 1-2). However, no bioassessment procedures or regulatory biomonitoring programs (such as biocriteria) have yet been developed or recommended for protection of coral reefs.

Table 1-2. The principal underpinnings of RBPs

Rapid Bioassessment Protocols

- Cost-effective, yet scientifically valid, procedures for biological surveys
- Provisions for multiple site investigations in a field season
- Quick turnaround of results for management decisions
- Scientific reports are easily translated to management and the public
- Environmentally benign procedures

Source: Barbour et al. (1999)

1.2 Rapid Bioassessment Protocol for stony corals

The Stony Coral RBP is designed to provide inexpensive tools to characterize the condition of coral reefs and determine whether waterbodies support biological integrity for a designated aquatic life use. Data from RBPs have been used (Barbour et al. 1999) to accomplish the following:

- Characterize the severity of waterbody impairment
- Help identify sources of impairment
- Evaluate the effectiveness of control actions and restoration activities
- Support use attainability studies and cumulative impact assessments
- Characterize regional biotic attributes of reference conditions

Existing RBPs (e.g., Plafkin et al. 1989; USEPA 1997; USEPA 1998; Barbour et al. 1999) advocate an integrated assessment that compares habitat (e.g., physical structure, flow regime), water quality and biological measures with empirically defined reference conditions (using reference sites, historical data, and models). Reference conditions are established through systematic monitoring of *minimally disturbed* field sites that represent the natural range of variation in water chemistry, habitat, and biological conditions (Gibson et al. 1996). Reference conditions are important for defining expectations (e.g., best-case scenarios) and amending those expectations when conditions are altered by large-scale stressors, such as global climate change or acid rain, that cannot be controlled by local management activities.

Several factors are considered in selecting organisms as biological indicators. Indicator organisms should be reasonably abundant, well-distributed, easily identified to species and not be subject to human exploitation (Jameson et al. 1998). The Stony Coral RBP focuses on a single, phylogenetic group rather than the multiple taxa of other assessment protocols. It is an initial effort that can be expanded to include other taxa as information and procedures are developed. However, stony corals are a dominating influence in the reef ecosystem because they build and maintain the physical infrastructure that supports all other organisms in the community. Consequently, they are considered by many to be primary indicator organisms for coral reef condition (Birkeland 1987; Brown 1988; Jones and Kaly 1996; Done 1997; Kramer 2003; Fisher et al. 2007a). Loya (1972) offered the following justification for a stony coral focus:

> A coral reef constitutes probably the most complex community of the marine environment. It is actually an association of several thousand species of different kinds of animals which occupy various ecological niches. A correspondingly complex community on dry land is, perhaps, the tropical-rain forest. Corals constitute the basic framework and substrate for many other organisms which penetrate the skeletal mass (sponges, polychaetes, sipunculides, bivalves and gastropods). Corals also provide shelter for many fishes as well as various species of polychaetes, crustaceans, mollusks and echinoderms. It is, therefore, of primary interest to obtain an adequate understanding of the coral-community structure as the first step for a better understanding of the complex of interspecific relations between corals and other organisms living in close association with them (Loya 1972, p. 100)

Stony corals, oysters, seagrasses and other habitat-forming biota are unique in that their survival, growth and reproduction dramatically influence the success of the entire community and ecosystem (Figure 1-1).

The Stony Coral RBP provides a quick, reliable and inexpensive means to characterize the biological condition of coral reefs. It relies on three rapid observations (colony identification, colony size and proportion of live tissue) that have been adapted from existing coral reef monitoring programs. When combined, these three measurements generate multiple indicators that characterize the value and sustainability of coral reefs and are likely to be responsive to effects of human disturbance.

Assessment of stony corals using the RBP will not address all issues relevant to resource management. In particular, measurements made only on stony corals, while reflecting reef status, cannot directly address questions related to other taxa (e.g., overfishing). Moreover, the indicators provide an instantaneous reflection of grossly observable coral

Photo: EPA

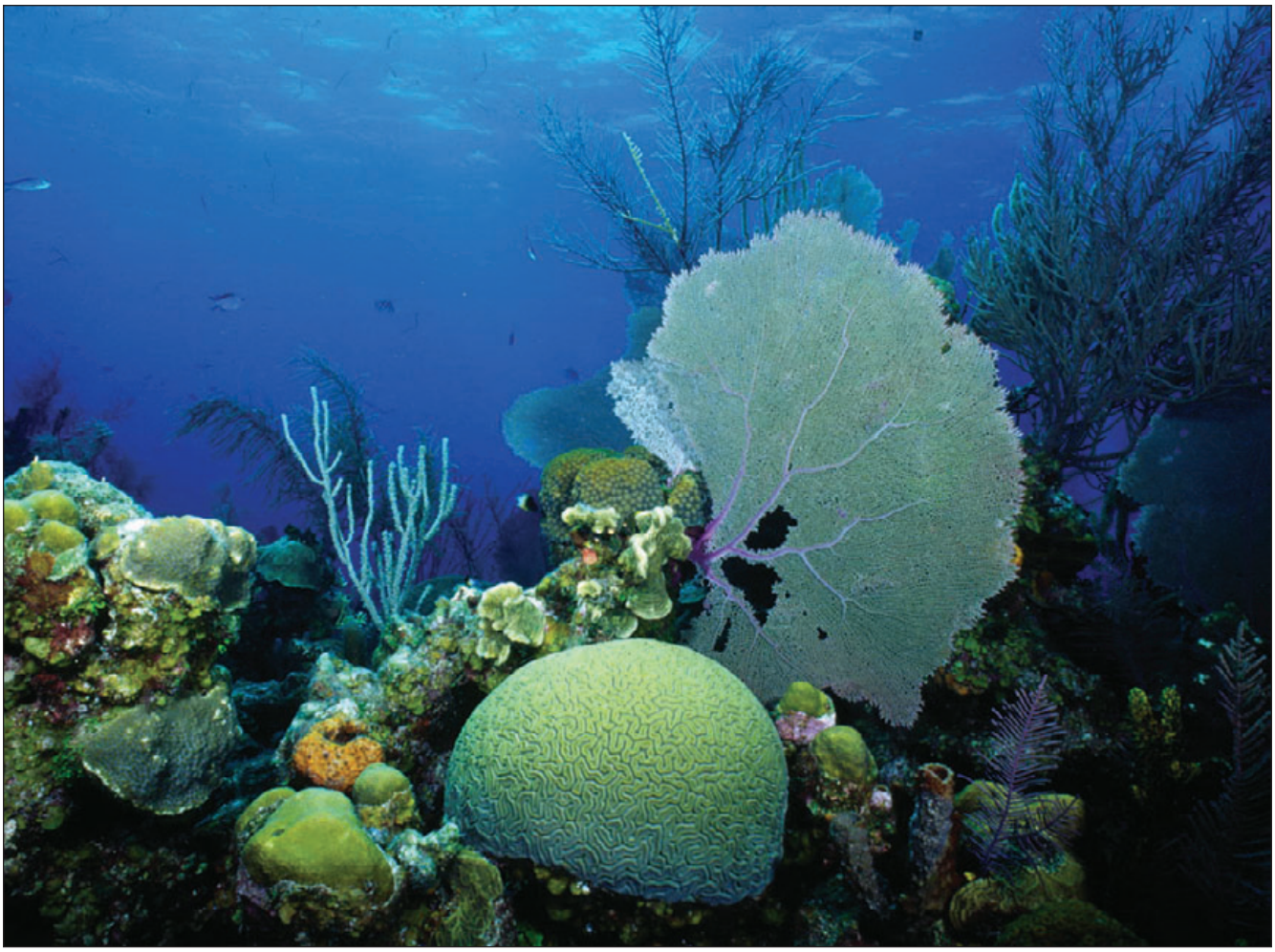


Figure 1-1. Physical structures formed by reef-building stony corals are inhabited by diverse and abundant biota.

characteristics—they do not provide information on physiological function or identify causes of impairment. Additional indicators or direct measurements are needed to identify causes, potential resolutions and avenues for remediation and restoration. It is also unlikely that RBP bioindicators will serve as a conventional early warning system for reef degradation because they might not respond quickly to environmental change. However, the strength of the RBP in regulatory monitoring lies in setting levels of expected conditions that conserve the resource over a long term. It is anticipated that the Stony Coral RBP will eventually be integrated with community, ecosystem function and exposure methods to generate comprehensive multimetric indices, such as an index of biotic integrity (Karr 1991; Jameson et al. 2001). Such indices can also be used as biocriteria and could ultimately fulfill some early warning or even causal objectives.

1.3 Coral reef biological criteria

There is great potential for coral reef biocriteria in U.S. jurisdictions, but implementation requires scientifically defensible assessment protocols and monitoring strategies. Numerous workshops and publications have addressed methods to measure coral reef condition, usually with a focus on development of rapid, reliable, low-cost monitoring approaches (UNESCO 1984; Aronson et al. 1994; Rogers et al. 1994; Crosby et al. 1996; Bruckner and Burrows 2005). The Stony Coral RBP consolidates and integrates some of these approaches for regulatory bioassessment and biocriteria.

Jameson et al. (1998) prepared an overview of potential methods to develop biological criteria for coral reef ecosystems. In particular, they evaluated the existing information, the scientific gaps and underscored the connection among coral

reefs, seagrass beds and mangrove forests. They introduced *assessment tiers* for comprehensive characterization of reef ecosystems; these ranged from desktop screening of data and information (tier 0) to rigorous field surveys repeated over time (tier 3). The importance of biological measurements in marine management programs was emphasized and biocriteria were characterized as scientifically sound, cost-effective tools to protect sensitive biological communities and sustain the chemical, physical and biological integrity of an ecosystem. Karr and Chu (1997) provided a template for development of biocriteria that stressed (1) habitat classification, (2) metric selection, (3) sampling protocols, (4) analytical procedures and (5) communication. The template was expanded by Jameson et al. (1998) into a step-by-step procedure (Table 1-3) that provided a foundation for future development of coral reef biocriteria.

Table 1-3. Process for coral reef biocriteria development

1. Preliminary classification of coral reef ecosystem
2. Biological survey
3. Final classification of coral reef ecosystem
4. Metric evaluation and index development
5. Biocriteria development
6. Implementation of a monitoring and assessment program
7. Protective and remedial management action
8. Continual monitoring and periodic reviews

Source: Jameson et al. 1998

The credibility of water quality standards is highest when criteria are developed within the context of a scientifically sound, long-term monitoring program. Achieving a sound monitoring program requires an initial study, sometimes called a biological survey (Table 1-3), to characterize and optimize the numerous variables that influence a monitoring design. Variables include metric selection, sample numbers and sampling unit size, reef classifications, variability within reef types, management zones, responsiveness of metrics to gradients of human activity and expectations based on reference condition. A comprehensive biological survey will provide the information to generate a competent and efficient monitoring design and is therefore crucial to any bioassessment program.

The *Stony Coral Rapid Bioassessment Protocol* addresses only the sampling methods applicable to development of a scientifically defensible, long-term monitoring program. Different sampling approaches are also being examined for use in biocriteria development (e.g., American Samoa; Houk et al. 2005). While this document does not provide guidance on biocriteria development, there are a few examples of how RBP indicators can be used for that purpose (Section 5). Reviews of coral reef classification systems (Jameson et al. 2003a) and methods to develop reference conditions (Jameson et al. 2003b) are already available, and additional guidance on monitoring designs, waterbody use designations and selection of thresholds (levels of expectation) for biocriteria is anticipated.

2. Coral Reef Attributes and Services

2.1 Biology and distribution of coral reefs

Fossil records indicate that corals appeared on Earth more than 400 million years ago (Allen and Steene 1996). Existing reef-building corals are stony corals of the Order Scleractinia (Phylum Cnidaria, Class Anthozoa, Subclass Hexacorallia; Humann and DeLoach 2002). The primary biological unit of a coral is the sessile polyp, which reproduces by clonal expansion (multiplication of individual polyps) and facilitates deposition of a calcium carbonate skeleton that supports the colony as it grows (Fagerstrom 1987). Colony size and morphology varies among and within species, often dependent on depth and hydrologic factors. Corals of many different species aggregate in communities (Figure 2-1) and establish complex, three-dimensional architectures that form a reef (Stoddart 1969). Although coral reefs vary in size, type and extent, shallow-water reefs are generally classified into (1) fringing reefs that are parallel to and near the shoreline, (2) barrier reefs that run parallel to the shoreline but are deeper and sometimes at the edge of the continental shelf and (3) patch reefs that are small and separated from adjacent land and reef masses (Humann and DeLoach 2002; Turgeon et al. 2002; Bruckner and Burrows 2005). Reef ecosystems include both the skeletal, or geological, component of corals and its diverse community of biological residents.

Many shallow-water corals are hermatypic, or *reef-building*. They flourish through an obligatory symbiosis of animal tissue (polyps) and photosynthetic dinoflagellate algae (zooxanthellae)

Photos: NOAA Photo Library

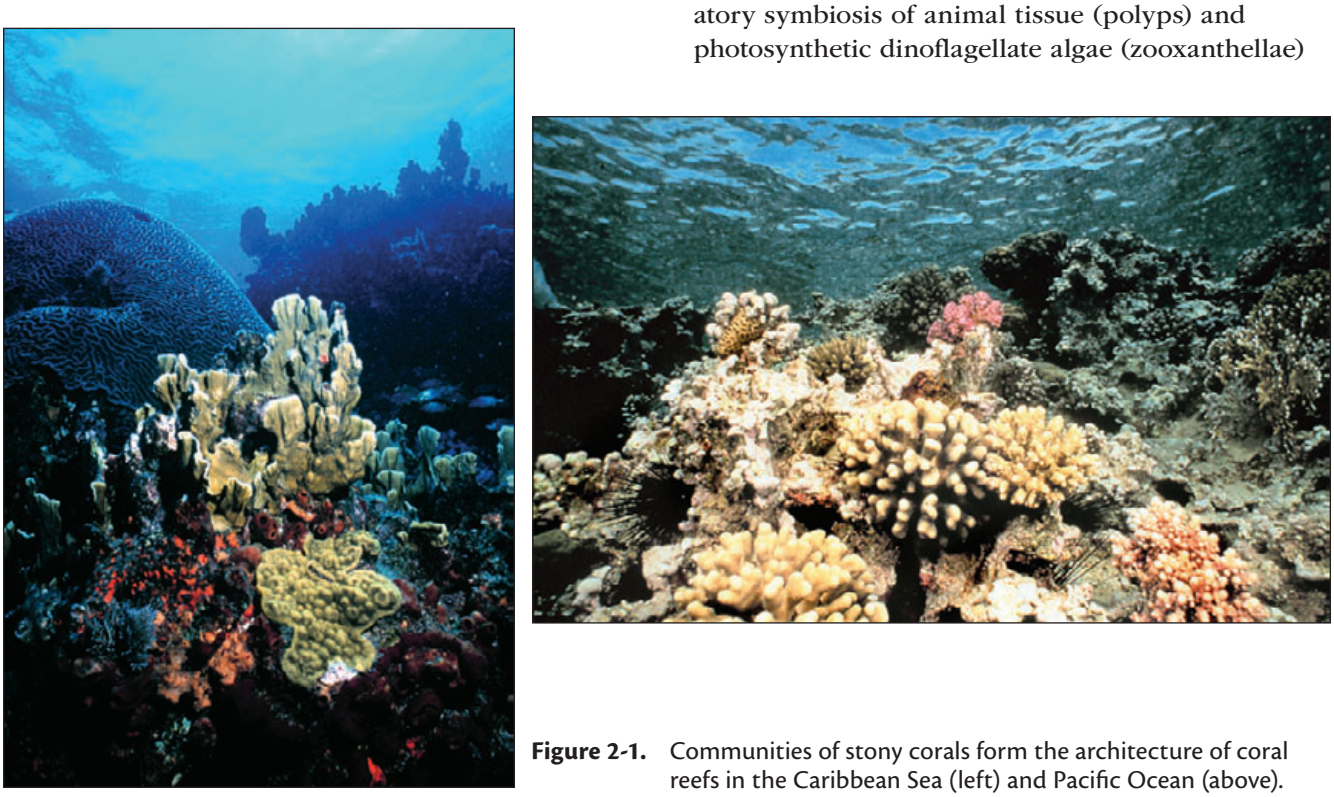
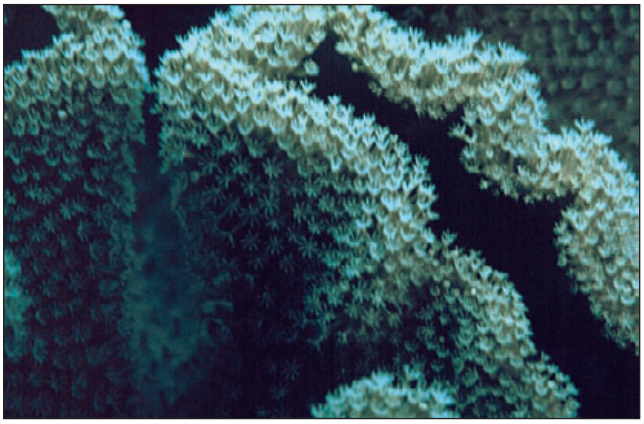


Figure 2-1. Communities of stony corals form the architecture of coral reefs in the Caribbean Sea (left) and Pacific Ocean (above).

belonging to the genus *Symbiodinium* (Figure 2-2; Yonge and Nicholls 1931a, 1931b; Pearse and Muscatine 1971; Muscatine 1973). Reliance on photosynthetic activity of the zooxanthellae limits the distribution of hermatypic coral to shallow depths that are penetrated by light (photic zone). The symbiotic algae provide organic compounds (sugars) to coral polyps, which metabolize them for energy production. This energy is used primarily to facilitate calcification processes required for growth and maintenance of the coral skeleton (Gattuso et al. 1999). The polyps, in return, provide the zooxanthellae with inorganic nitrogen, phosphorus and a secure, well-lit shelter (Goreau and Goreau 1960).

Coral reefs occur predominantly in shallow (50m or less), warm (20 to 30 degrees Celsius) and generally clear waters throughout the tropic and subtropic seas (between 30 °N and 30 °S). They lie adjacent to approximately 100 countries and territories (Wilkinson 2002), and reefs are estimated to cover 284,300 km^2 worldwide (Spalding et al. 2001), or roughly 1 percent of the available area of continental shelf. Coral surface area coverage within U.S. jurisdictional waters has been estimated at 19,702 km^2 (Boesch et al. 2000; Turgeon et al. 2002). Coral reefs included in U.S. jurisdictions are distributed along states, territories, and commonwealths in the Caribbean Sea, Western Atlantic Ocean, Gulf of Mexico and Pacific Ocean. Information regarding the U.S. distribution of corals, their management and regional condition is contained in reports produced by the National Oceanic and Atmospheric Administration (Turgeon et al. 2002; Waddell 2005).

Photo: NOAA Photo Library



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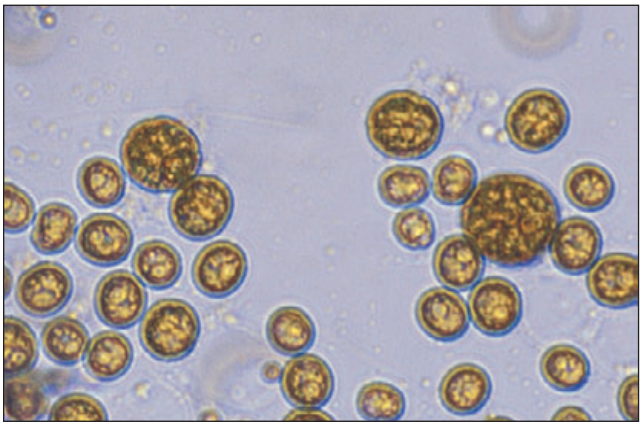


Figure 2-2. Corals grow through a symbiotic relationship of coral polyps (top) that are inhabited by dinoflagellate algae (*Symbiodinium* spp., often called *zooxanthellae*) that can also be free living (bottom).

2.2 Ecosystem services of coral reefs

Enormous value is attributed to coral reefs of the world. Some ecosystem services are linked to economic outcomes (e.g., fishing, tourism, bioprospecting, construction material, shoreline protection) and are estimated to contribute as much as $375 billion annually to the world economy (Costanza et al. 1997; Wilkinson 2002). There are also social and cultural values attributed to coral reefs, especially in island jurisdictions (Copp 1950; Holmes 1974). Other services are related to stability and integrity of the biological community (e.g., biodiversity, trophic complexity, primary production). Proliferation of human populations along coastlines, accompanied by resource extraction and water quality degradation, threatens the sustainability of these services (Wilkinson 1996). Nearly half a billion people, or 8 percent of the total global population, live within 100 km of coral reefs (Bryant et al. 1998). This demographic is not without adverse effect.

Coral reefs in Florida and the Caribbean basin have experienced unprecedented levels of bleaching, disease and mortality during the past three decades (Jaap et al. 2000; Wheaton et al. 2001; Gardner et al. 2003; Kramer 2003). Stressors believed to have led to this decline include elevated water temperature, increased exposure to solar radiation, novel and opportunistic pathogenic microorganisms and degraded water quality, all of which might be related in some manner to global changes in climate, land use or human activity in coastal areas (Atwood et al. 1992; Hoegh-Guldberg 1999). The consequences of continued stress on corals are diminished growth and reproduction, loss of coral tissue, algal overgrowth of denuded skeleton and eventual disintegration of the skeleton through biological and

physical erosion. Loss of coral and coral skeleton limits the capacity of coral reefs to provide the ecosystem services for which they are valued and has led to calls for greater resource protection. Principal benefits and assets of coral reefs are briefly summarized in the following sections.

2.2.1 Subsistence and commercial fishing

Coral reef and adjacent open water fisheries once supplied the major animal protein source for many island populations (Wilkinson 1996). Coral reef fisheries, in spite of declining catch per unit effort, account for about 9 million metric tons of food worldwide, equal to 10 percent of the world's fisheries. For some Pacific island and Caribbean communities, coral reef seafood once provided more than 80 percent of the animal protein consumed (Pernetta and Hill 1982). High abundance and diversity of commercially harvested reef fish are highly dependent on coral structures (Figure 2-3). The three-dimensional coral skeletons that form the reef topography provide habitat for fish protection, predation and breeding (Bruckner and Burrows 2005). Subsistence and recreational fishing, as well as aquarium trade industries, are therefore tightly linked to the structural habitat provided by coral reefs (Luckhurst and Luckhurst 1978; Roberts and Ormond 1987; Done et al. 1996; Lirman 1999; Ferreira et al. 2001; Perkol-Finkel et al. 2006).

2.2.2 Tourism

Coral reef ecosystems are highly attractive to tourists seeking relatively pristine, unique and diverse habitats teeming with colorful and morphologically diverse organisms. Tourism value of coral reefs includes the aesthetic, recreational and economic aspects of fishing, boating, scuba diving and snorkeling at reef locations (Figure 2-4). Socioeconomic conditions worldwide are influenced by income derived from tourism (Reaser et al. 2000). In the Caribbean in 1990, coral reefs provided 2–6 percent of the gross national product for many island states (Dixon 1993). In Florida alone, reef tourism brought one million visitors in 1990 and

Photo: NOAA Photo Library

Figure 2-3. The three-dimensional coral skeletons constructed by stony coral growth provide physical habitat for numerous recreationally and commercially harvested fish species.

Photo: EPA

Figure 2-4. Reef-building stony corals provide physical habitat for diverse and unique biota that have become valuable tourist attractions.

$46.5 million (Dixon 1993); a more recent study estimated annual visitation for 2000–2001 to be 18 million people and an annual use value of $227 million (Johns et al. 2001). In Hawaii, recreation and tourism related to coral reefs bring an estimated $364 million in annual economic benefits (Cesar et al. 2002).

2.2.3 Shoreline protection

The same coral structures that provide habitat to marine communities also protect coastal shorelines from wave and current erosion (Pernetta 1992; Costanza et al. 1997). Ecological value from this natural protection of estuaries, lagoons and productive coastlines (Figure 2-5) is substantial. Often overlooked is the economic value, which could overreach the economic impacts of all other ecosystem services combined—coastal reinforcement and

Photo: NOAA Photo Library

Figure 2-5. Pacific island surrounded by coral reefs that protect the shoreline from wave and current erosion.

protection barriers were once estimated at $10 million per linear kilometer (Costanza et al. 1997).

2.2.4 Future chemical and pharmaceutical products

Untapped chemical and pharmaceutical products exist within the diverse biota of coral reef ecosystems. Extracting novel compounds from biological organisms (bioprospecting, biomining) has shown particular promise for human health applications. Biochemicals produced by many reef species are currently being used for health care products, medical procedures, and pharmaceuticals. About half the potential pharmaceuticals currently under development are from the ocean (Carte 1996; Fenical 1996; Hay and Fenical 1996) and many of these are from coral reef organisms. Because only a small portion of coral reef biota has been described (approximately 10 percent; Reaka-Kudla 1996), there is considerable potential for discovery of novel chemicals (Adey 2000).

2.2.5 Biodiversity

Coral reefs are complex and highly productive biological systems. A reef is more than an aggregation of corals—the complex physical structure created by corals provides habitat for a uniquely diverse and interactive biotic community (Figure 2-6). In the Indo-Pacific alone, there are more than 719 different species of hard corals and 690 species of soft corals. This coral community provides essential habitat to 4,000 different marine fish and thousands of invertebrate species (Spalding et al. 2001). In all, it is estimated that roughly a million species are dependent on, or contribute to, coral reef ecosystems (Reaka-Kudla 1996).

Although coral reefs are sometimes compared to tropical rainforests as major storehouses of

Photo: EPA

Figure 2-6. Coral reefs are highly productive ecosystems and principal contributors to ocean biodiversity.

biodiversity, 32 of the 34 recognized animal phyla are found on coral reefs compared to only 9 phyla in tropical rainforests (Wilkinson 2002). Much of this diversity can be directly attributed to the complex skeletal infrastructure, which provides a high number and heterogeneity of habitat niches (Loya 1972; Sebens 1994; Bruckner and Burrows 2005). Even a single coral head provides habitat for a rich community (Grassle 1973). Reef inhabitants are also relatively unique, possibly a consequence of geographic and genetic isolation. Caribbean and Indo-Pacific reefs have few species in common, and many species are geographically limited in range (Boesch et al. 2000).

2.2.6 Primary and secondary production

Symbiotic algae (zooxanthellae) that inhabit coral polyps provide energy through photosynthesis. Zooxanthellae occur at densities of more than $10^6/cm^2$ on coral surfaces and are among the dominant primary producers in reef communities (Muscatine 1980, 1990). Gross carbon fixation of coral reefs is relatively high (estimated at 700 x 10^{12} g C/yr globally), and most of this is quickly and efficiently recycled to secondary producers within the reefs (Crossland et al. 1991). Corals thus provide not only the habitat, but also a portion of the energy for a diverse and abundant biological community (Lewis 1977, 1981).

2.2.7 Calcium carbonate deposition and degradation

Skeletal growth of stony corals requires biologically mediated precipitation of inorganic carbon. Carbon is available to symbiotic algae from bicarbonate ions dissolved in sea water and from the respiratory activity of polyps (Muscatine 1990). With photosynthesis, carbon is fixed and used for generating new zooxanthellae, respiration and translocation into skeletal structures (Pearse 1970; Pearse and Muscatine 1971). Fixation rates have been estimated at 9 kg $CaCO_3$ m^2/yr (Chave et al. 1972; Stearn et al. 1977). The ability to fix inorganic carbon places corals among those organisms that influence oceanic CO_2 cycling and several related aspects of seawater chemistry (Kinzie and Buddemeier 1996). One potential adverse effect of increased CO_2 in the atmosphere (from anthropogenic activities) is reduced calcification rates in corals (Gattuso et al. 1999; Kleypas et al. 1999).

Degradation of coral skeletons by physical and biological erosion supplies the surrounding sea floor with sand and other particulate sediments (Scoffin et al. 1980; Hutchings 1986). Thus, coral reefs, and stony corals in particular, influence substrate composition throughout the world. Coral sand is mined for a variety of landscaping and recreational purposes.

2.3 Biological attributes of coral reefs

Many different biological measurements and approaches have been used to quantify coral reef attributes (e.g., Kinzie and Snider 1978; Rogers et al. 1994; Risk et al. 2001; Bruckner and Burrows 2005). This variety has necessarily spawned a number of method comparisons (e.g., Weinberg 1981; Dodge et al. 1982; UNESCO 1984; Chiappone and Sullivan 1991; Foster et al. 1991; Rogers and Miller 2001; Brown et al. 2004). The two biological indicators most often reported in coral reef assessments are live coral cover and diversity of benthic cover (Jameson et al. 1998), both of which have been measured using a variety of protocols, survey designs and calculations. Despite the many disparities, all biological monitoring is intended to promote scientific understanding or inform decisions by resource managers.

Assessment monitoring compares the existing condition of a resource with an expected (reference, target) condition and provides a means to detect change over time. Assessment endpoints, the biological indicators, are field measurements or calculations from field measurements that characterize the attributes of a resource or ecosystem for interpretation (Table 1-1). The relative merit of each indicator depends on how well differences in condition can be detected over time or among stations, reefs or regions, and how relevant the indicator is to a management question. If metrics and assessment endpoints reflect common perceptions of the values, management decisions are more easily instituted and enforced (Jackson et al. 2000).

For convenience, existing field measurements of coral are divided into three categories for discussion—coral condition (biological and physical characteristics of corals), ecological condition (reef community characteristics) and environmental stressors (exposure of coral reefs to anthropogenic or natural stresses). The Stony Coral RBP provides indicators of coral condition only (see Sections 3 and 4), but all three categories are summarized in the following paragraphs. A comprehensive biocriteria program, at

least as envisioned by Jameson et al. (1998), would include components from all three categories.

2.3.1 Biological and physical measurements

Biological status of corals can be measured in ways common to most organisms, including metabolic rates (e.g., growth, photosynthesis), health and life stage. Measurements of live tissue and denuded skeleton can also be made; a coral colony is composed of multiple interconnected polyps, and the colony can survive even when large areas of the polyps have died. Skeletal deposition is an ecologically important measurement—it represents additional coral structure and new habitat for the reef community.

The physical status of coral colonies can be captured by measurements of three-dimensional size, shape and structural complexity (e.g., surface area of hollows, ridges, caverns). Size might be related to colony age or at least life-stage (e.g., new recruit) and can be used to generate size-frequency distributions for particular populations or for a coral community. The physical status of coral communities has been depicted through measurements of coral cover (the amount of coral per unit of sea floor), coral density, relief (height of colonies in a reef), topography (rugosity or complexity) and extent. The geographic extent of coral communities is often used to define the perimeter of a reef ecosystem, which is sometimes delineated using side-scan sonar (Kendall et al. 2004).

Physical status of corals has been measured on the basis of independent colonies or their surface area, and sometimes both are simultaneously quantified (Chiappone and Sullivan 1991). There are clear benefits to both approaches—surface area methods provide estimates of skeleton and coral quantity, and colony-based methods characterize genetically distinct organisms, each with varying potential to survive, grow and reproduce. The more versatile and robust programs, including the Stony Coral RBP, will incorporate both approaches.

2.3.2 Ecological and community measurements

Many aspects of the reef community can be measured to characterize ecological well-being. Reef ecological measurements are important because they can represent a greater portion of ecosystem services. Changes in reef communities can reflect upward or downward trends in *sustainability*, which is the retention of reef values over time. Measurements supporting ecological and community indicators are both structural (e.g., benthic cover, diversity) and functional (e.g., productivity, herbivory).

Benthic cover is among the most-reported community measurements. Its relevance is rooted in the concept of competition for space between corals and macroalgae. When coral tissue dies, the skeleton is left bare and available for colonization. Macroalgae can out-compete coral recruits for the substrate if sufficient nutrient is available and herbivorous fish and invertebrates (e.g., sea urchins) are lacking (Hughes 1989; Chazottes et al. 1995; Tanner 1995). This can result in a shift of community composition from coral to algal domination (Naim 1993; Szmant 2002). Such a shift, often linked to anthropogenic activity, is considered adverse because non-coral colonizers contribute to coral bioerosion and eventual destruction of coral skeletons (Hutchings 1986).

Whereas ecological and community measurements are important aspects of coral reef condition, they are subject to interpretations that sometimes require additional investigation. For example, it is generally believed that eutrophication leads to greater algal growth and bioerosion of coral colonies (Hutchings 1986). Yet, sediment runoff, which often accompanies nutrients from the watershed, can inhibit algal colonization by covering available substrate (Hutchings et al. 2005). Similarly, measurement of benthic cover can be misleading as an indicator of coral condition—interpretation of results is confounded by herbivory, and nutrient availability and coral loss can occur for many reasons unrelated to competition with macroalgae. Interpretations of ecological and community measurements might require more supporting evidence than can be easily provided in a rapid bioassessment.

2.3.3 Exposure measurements

There are numerous natural and anthropogenic factors that adversely affect corals and coral reefs (e.g., Richmond 1993; Dubinsky and Stambler 1996; Wilkinson 1996; Hughes and Connell 1999). Stress generated by exposure to these adverse factors can be acute or chronic, and repetitive exposures decrease the likelihood of coral recovery. Consequences of stress include coral bleaching (loss of photosynthetic algae, Figure 2-7), greater

susceptibility to disease, diminished growth and reproduction, and partial or complete mortality.

Anthropogenic coral stressors include efflux of terrestrial material (nutrients, contaminants, sediments and microorganisms), resource extraction (fishing, bio-prospecting), physical damage (divers, boats), habitat alteration (dredging, coastal development) and introduced and invasive species. Stressors could also include natural conditions such as disease and wave energy (Turgeon et al. 2002). Storm wave damage to corals, for example, has been estimated using maximum wave height (Dollar 1982; Storlazzi et al. 2002; Jokiel et al. 2004).

Climate change is often cited as a coral stressor. Elevated oceanic temperatures during the past half-century have been at least partially attributed to increasing concentrations of greenhouse gases from burning of fossil fuels (IPCC 2001; Levitus et al. 2000, 2001). Climate change encompasses a variety of physical and chemical stresses to corals, including temperature, ultraviolet radiation, sea level rise, storm damage and an oceanic carbonate shift that reduces the ability of corals to deposit calcified skeleton. Climate change also influences weather patterns that interact with global changes in land use to create additional stressors from the watershed (Figure 2-8).

Exposure measurements are not required for development of biocriteria and are not explored in this document. However, exposure measurements are needed to determine causality when bioassessments reveal an impaired waterbody (USEPA 2000b). Isolating a single cause of impairment is difficult because human disturbance is multidimensional.

Photo: EPA

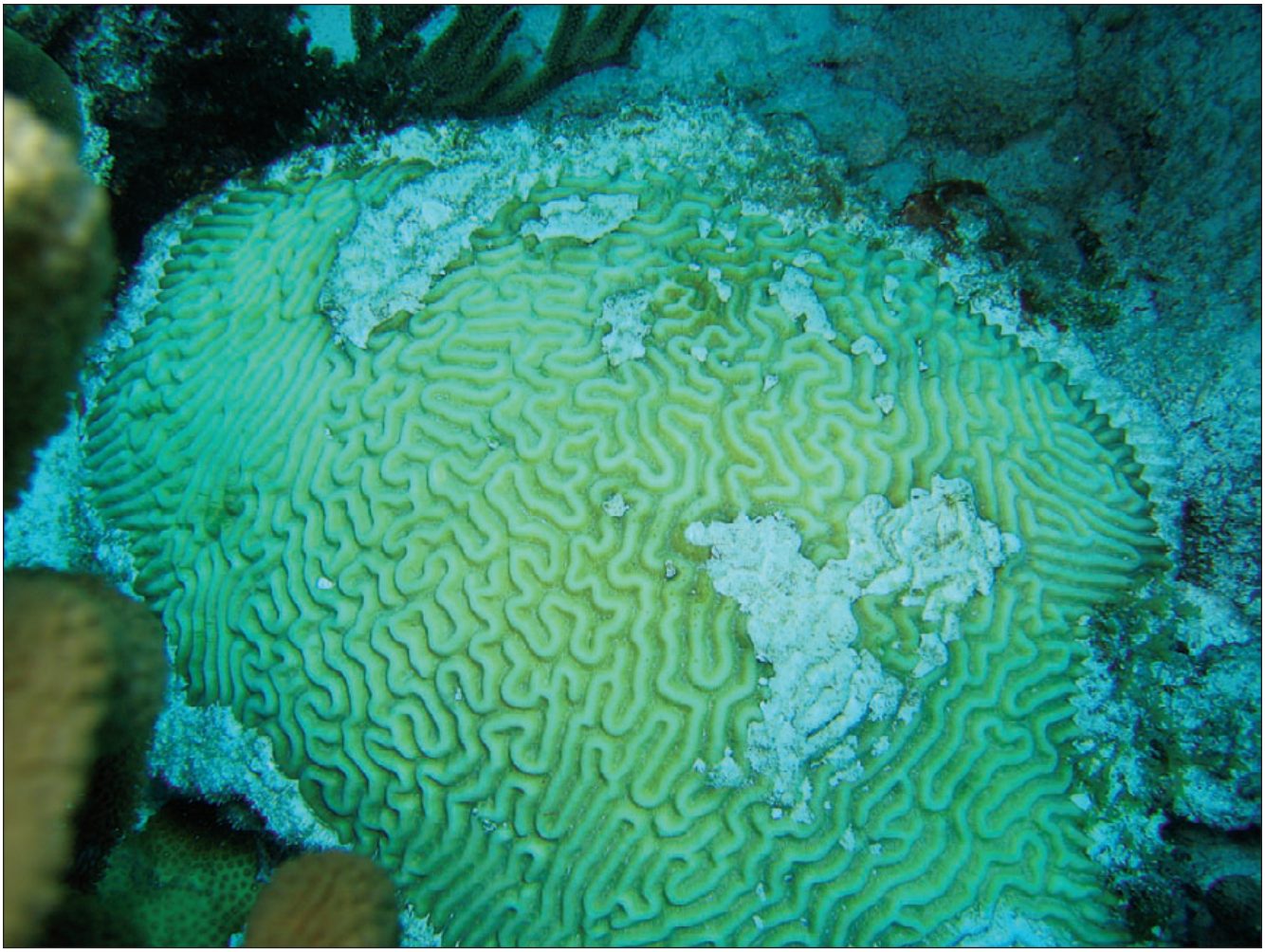


Figure 2-7. Various stresses upset the symbiotic relationship of corals and can cause a loss of symbiotic algae. This leaves a colony, such as the *Diploria strigosa* pictured here, with a *bleached* appearance as the white coral skeleton shows through the translucent polyps.

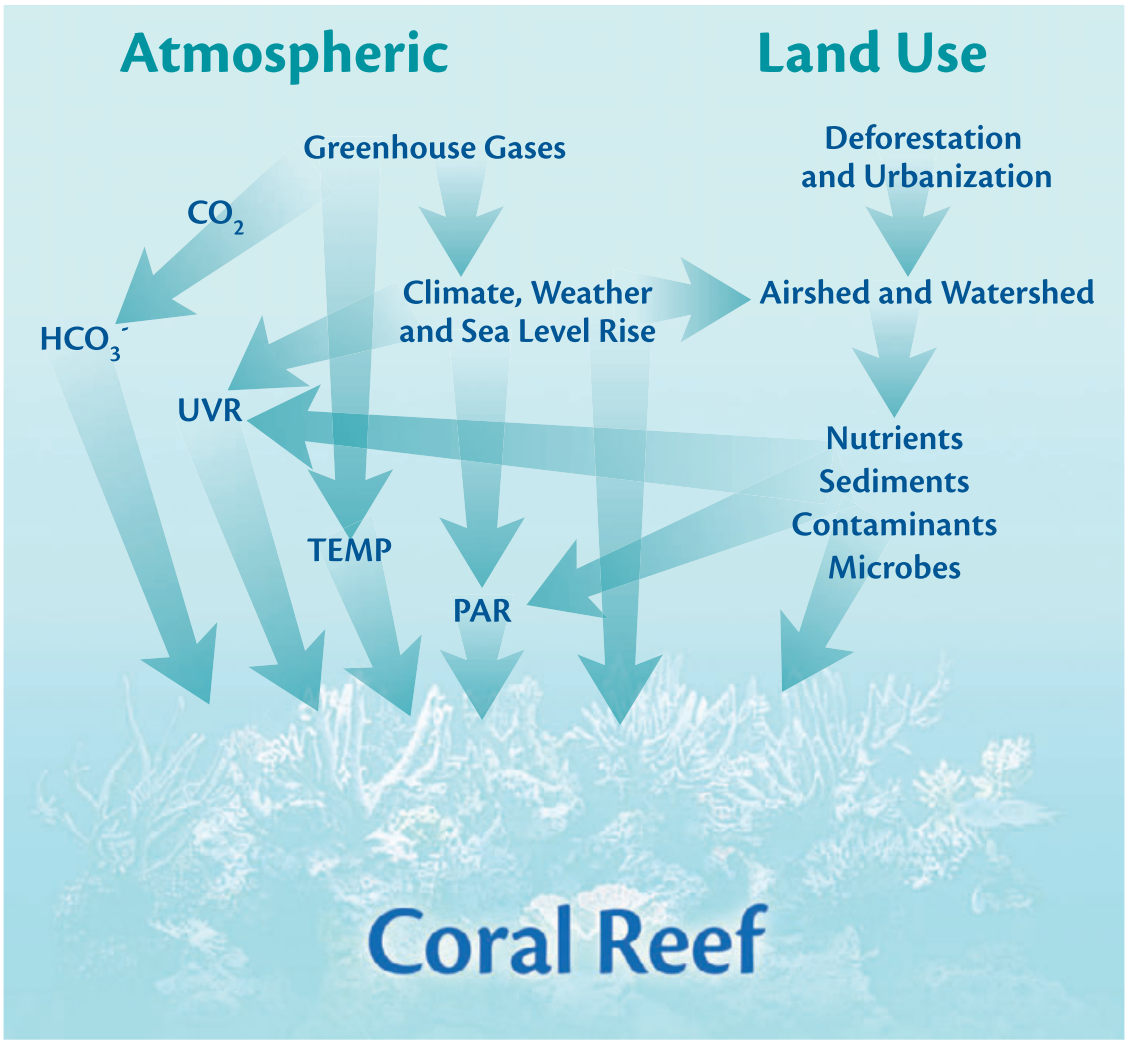


Figure 2-8. Several atmospheric and land use changes are occurring at a global scale, with cumulative and interactive effects on coral reefs. Carbon dioxide (CO_2), carbonate ion in sea water (HCO_3^-), ultraviolet radiation (UVR), temperature (TEMP), photosynthetically active radiation (PAR).

Because controlled experiments are infrequently possible, linking degradation to cause is often correlative. Beyers (1998) has suggested a *weight-of-evidence* approach to evaluate cause-effect relationships, and Jameson and Kelty (2004) have reviewed many potential methods to measure stress exposures.

3. Rapid Bioassessment Protocol for Stony Coral Condition

It is no coincidence that indicators of stony coral condition have been incorporated into nearly all coral reef monitoring programs. The intent of any bioassessment program is to employ practical, affordable measurements that generate ecologically relevant endpoints to support management decisions, enforcement and performance evaluation (Jackson et al. 2000; Jameson et al. 2001). Stony corals are directly responsible for most ecosystem services, so indicators of stony coral condition are very likely to be informative, transparent and authoritative.

A principal objective of the Stony Coral RBP is to provide an efficient, inexpensive, nondestructive method that generates useful indicators for management programs. Three core observations are recommended—species census, colony size and the proportion of live tissue on individual colonies. While additional observations and measurements are not precluded, a variety of useful indicators can be calculated from these three observations alone. The observations have been made in other programs (for example, Lang 2003; TNC 2006; Bruckner and Bruckner 2007), but two aspects—colony-to-surface area conversions and topographic three-dimensional (3D) surface area—are unique to the RBP.

Most existing methods quantify coral abundance by counting colonies or estimating surface area, both of which produce indicators relevant to coral condition. The surface area approach is used to estimate, for example, the proportion of live coral cover, whereas a census (colony approach) provides indicators related to abundance and density. Both approaches are incorporated in the RBP, which converts size measurements made on each colony to surface area.

The potential of the Stony Coral RBP to serve as a regulatory bioassessment protocol has been examined in a pilot study (Fisher et al. 2007a), a modified survey of the Florida Keys (Fore et al. 2006a; Fisher et al. 2007b) and an initial biological survey at St. Croix, U.S. Virgin Islands (Fore et al. 2006b, 2006c). Although the RBP has not been validated in Pacific Ocean reefs, the three core observations should be relatively straightforward. Differences in colony morphology, however, could require unique conversions for colony size measurements and assignment of topographic surface area (see Appendix A).

3.1 Stony coral census

In a coral census, each stony coral colony within the transect perimeter is identified to species or at least genus (e.g., English et al. 1994; Allen and Steene 1996; Veron 2000; Humann and DeLoach 2002). Conventions must be adopted in advance to determine which colonies will be included in the census. For the Stony Coral RBP, common rules are applied (Santavy et al. 2001; Lang 2003; Fisher et al. 2007a):

1. Colonies must be greater than 10 cm (any dimension, including live tissue and denuded skeleton) to be included in the census. The main reason for this convention is that smaller colonies are often difficult to identify and enumerate, which can lead to long dive times and more measurement errors while providing only limited information. Smaller colonies should be included for recruitment assessments, but in such cases, a simple tally and a generic surface area assignment would be more efficient than measuring each colony.

2. Colonies are included in the census if at least 50 percent of the colony lies within the transect perimeter. Any colony large enough to span the

transect perimeter should be included, even if the majority of the colony lies outside the perimeter.

3. Colonies are included in the census even if the living portion of the coral is less than 10 cm and occurs outside the transect perimeter.

4. Corals are included in the census if they can be identified at the genus level. Species-level identification better supports indicators related to community composition. Inability to identify the colony, whether because of size, algal overgrowth or loss of distinguishing characteristics, excludes the colony from the census.

5. Data (colony size and live tissue estimates) are collected from the entire colony, not merely from the portion that lies within the transect perimeter or only from the tops (aerial view) of colonies.

There are relatively few reasons to exclude a stony coral species from census. A recent survey performed in the Florida Keys excluded lesser starlet corals (*Siderastrea radians*) because they were small, difficult to count and provided no vertical relief (Fisher et al. 2007b). Branching fire corals (*Millepora alcicornis*) (hydrocorals) were also excluded because they were more often encrusting than reef-building. In contrast, blade fire corals (*M. complanata*) were included because they supplied relatively permanent vertical structure to the reef. Any exceptions must be clearly documented for comparability among programs, particularly if the species occur regularly in the area. While individual managers might have different objectives, the value of all bioassessment programs is increased when the same methods and approaches are used by many.

In most cases, visual distinction of coral colonies is not difficult. Connell (1973) characterized individuals as any colony growing independent of its neighbors. Sometimes, however, two colonies of the same species grow together and the line of separation is indistinct. If tissue separation is not visible or two separate morphological shapes are not identifiable, this is documented as a single organism (see AGRRA Program, Lang 2003). Some coral colonies break, and the fragments form independent colonies. Although these are genetically identical, they are regarded as distinct organisms because they have varying potential for survival, growth and reproduction. The most difficult challenge is when patches of live tissue, separated by dead areas, occur on a colony skeleton. The patches could be surviving remnants of the colony or could be young recruits. Unless it can be reasonably concluded that the patches belong to the same colony, they are considered independent biological entities (Connell 1973).

3.2 Colony size and 3D surface area

Surprisingly few monitoring programs measure or even estimate the size of coral colonies. Part of the reason for this is that many programs use linear transect methods, rather than a colony-based census, to estimate coral cover. Yet, size is an extremely important coral attribute. Size discriminates the contribution of each colony and species to community habitat, biomass, photosynthetic activity, metabolism and calcium carbonate deposition. Colony size is indispensable when considering growth, reproduction, population dynamics and community interactions.

Various means have been used to quantify colony size (Figure 3-1). Some have estimated the cubic volume of colonies using predetermined size classes (Fisher et al. 2007a, 2007b), and others have measured a colony dimension (Lang 2003; Houk 2005). While measuring is more time-consuming (Figure 3-2), it provides continuous distributions for analysis of population structure. Measurement of three coral dimensions has been applied in disease studies (Bruckner and Bruckner, in press), in pilot surveys by the Florida Reef Resilience Program (TNC 2006) and in biocriteria development surveys in the U.S. Virgin Islands (Fore et al. 2006c). Each of these studies has measured the same three dimensions: height (greatest colony distance perpendicular to the substrate), maximum diameter (planar diameter with greatest aerial projection onto the substrate) and width (diameter orthogonal to the maximum diameter measured at the center of the colony). Some studies have measured maximum width, which does not necessarily occur at the center of the colony. Either is acceptable if consistent.

3.2.1 Estimating 3D surface area

With few exceptions, coral studies have quantified coral surface area in only two dimensions. Coral cover, for example, is estimated as the planar projection of colonies on the underlying substrate as

Source: Lee Courtney, EPA

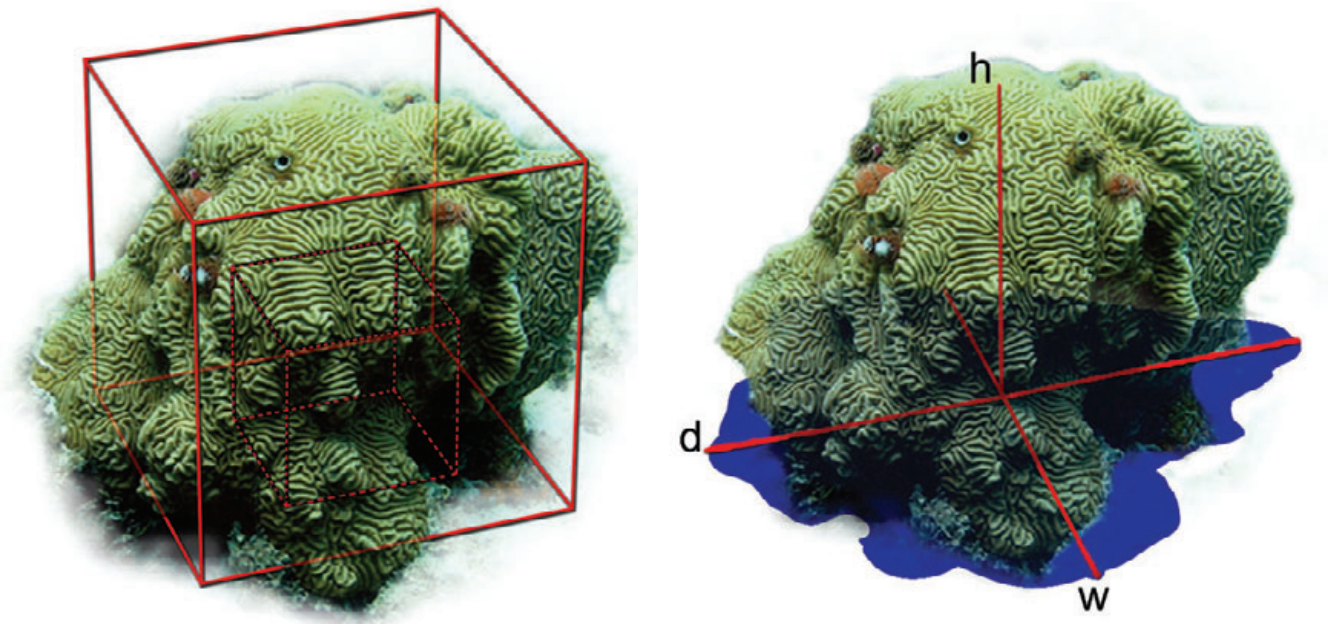
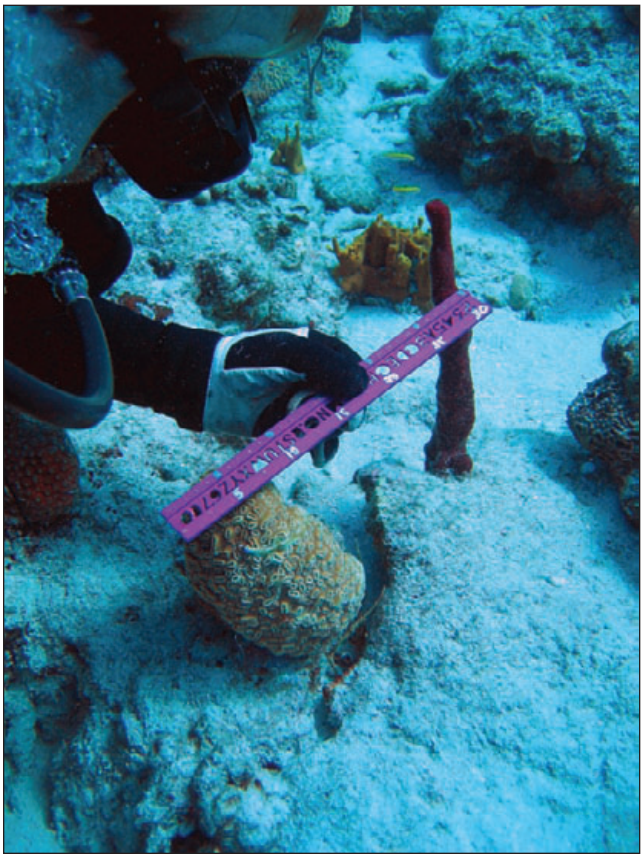


Figure 3-1. Colony size has been quantified by visual grading into volumetric size classes (left). In this example colony (*Diploria strigosa*), volume was better approximated by the larger 10L cube than the 1L cube. The surface area assigned to this colony (five sides of the 10L cube) was 2,311.3 cm^2. Colony size can also be quantified by actual field measurement of height (h), diameter (d), and width (w) from an aerial view (right). When analyzed photographically (Appendix A) this colony measured h = 22.9 cm d = 36.1 cm and w = 29.8 cm, with a surface area of 1,976.9 cm^2.

Photo: EPA



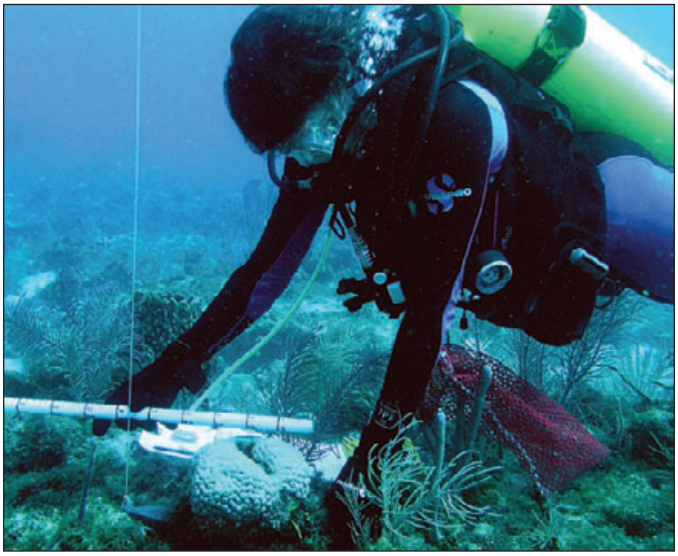


Figure 3-2. Rulers and meter sticks can be used to measure height, maximum diameter and width of individual coral colonies. Each measurement brings greater accuracy to size estimates but requires more underwater time and effort.

viewed from above (aerial view). This approach does not account for height or structural complexity of the colony. Among the several reasons to migrate to 3D quantification of corals is the role of topographic surface area in coral reef ecology. This was emphasized by Dahl (1973):

> The production, occupation, and destruction of surface area are, therefore, basic reef processes, and the balance between them is an essential aspect of the reef ecosystem. The efficient production of surface is a primary function of many reef organisms, and the control of surface by secondary occupants is a basic competitive force and a major determinant of reef communities (p. 240).

Surface area should be measured along all three dimensions because all three support these basic reef processes. A 3D approach provides a more realistic quantification of physical structure (community habitat), live coral (reproduction and growth) and bare skeleton available for recruitment or erosion. Energy transfers occur across the epithelial membranes of coral polyps, so topographic surface area is a rate determinant for photosynthesis, feeding, carbonate deposition, growth, and reproduction (Dahl 1973). These physiological and ecological relationships are fundamental to development of useful coral reef ecosystem and sustainability models, and our coral measurements should reflect this significance.

Although appealing, 3D values for coral surface area are not easy to obtain because corals have different shapes. Substantial morphological variation occurs, even within species and particularly with depth (Goreau 1963; Barnes 1973). Most procedures to measure 3D topographic surface area have been developed for laboratory use (e.g., Marsh 1970; Hughes and Jackson 1985; Meyers and Schultz 1985; Hoegh-Guldberg 1988; Ben-Zion et al. 1991; Stimson and Kinzie 1991; Tanner 1995). All these laboratory methods are time-consuming, destructive and unusable for rapid underwater surveys.

Several investigators have estimated 3D values for surface area using geometric surrogates (Szmant-Froelich 1985; Roberts and Ormond 1987; Babcock 1991; Alcala and Vogt 1997; Bak and Meesters 1998; Fisher et al. 2007a). New photographic techniques employ 3D colony reconstruction to estimate coral surface area with high accuracy (Bythell et al. 2001; Cocito et al. 2003), and this approach has now been successfully applied to four species of field corals (Courtney et al. 2007).

3.2.2 Geometric shapes as colony surrogates

Bioassessment monitoring usually characterizes condition across relatively broad spatial areas. For these programs, reasonable approximations are often more effective than accurate measurements because the time saved by approximation can be used to increase the number of locations sampled. Several studies, noted above, demonstrated the use of geometric shapes to approximate 3D values for surface area. In most cases, morphological dimensions of the colony were simply entered into the surface area formula for a representative geometric shape. For example, the average radius (r) of a hemispheric colony can be used to calculate $2\pi r^2$, the 3D colony surface area (CSA) of a bottomless hemisphere (the bottom is eliminated so that estimates are made for only the above-substrate portion of the coral colony). While many colony shapes are straightforward, some geometric surrogates might require experimentation and validation. Various approaches are reviewed in Appendix A, including a discussion of appropriate scale and level of accuracy. Appendix B addresses the potential conversion of historical two-dimensional coral data to 3D units.

Ultimately, statistical comparison will play a large role in developing methodology for different monitoring programs. For example, three colony dimensions (height, diameter and width) might be measured in the first few years of a monitoring program; then, analysis of the data might indicate that only two measurements are needed to achieve the same programmatic objectives. This was the case when data were examined from a pilot study of the Florida Reef Resilience Program (Appendix C). Monitoring data could also be examined to determine whether measurements are needed for small colonies. Because the influence of small colonies on surface area indicators is comparatively minor, they could all be assigned the same surface area (e.g., an average obtained from a subset of small colonies). Regardless of the strategy, procedures to approximate CSA must be guided by efficiency (optimal accuracy and survey time) over the entire course of the survey. This is true for all aspects of the survey, but it is particularly important for measurements of colony dimension, which are relatively time consuming.

3.3 Percent live coral tissue

The proportion of live coral tissue on a colony reflects the cumulative, integrated effect of both beneficial and adverse environmental factors. Substantial portions of coral tissue can die without lethal consequences to the colony (Figure 3-3), but tissue loss reduces the chance of colony survival and reduces the capacity to augment biomass through growth and reproduction. In most studies the *dead* proportion of a colony, the portion that lacks tissue where tissue once existed, is estimated and reported as *partial mortality* (Sudara and Snidvongs 1984; Brown and Howard 1985; Brown 1988; Dustan 1994; Ginsburg et al. 1996; Bak and Meesters 1998). Estimates of either live or dead (denuded) coral proportions will serve the same purposes because they are converse estimates. The RBP uses a live proportion, percent live tissue (%LT), because the values are used in calculations for live surface area (LSA). Both the proportion and amount of live tissue are useful indicators of colony health.

Several protocols have been used to estimate the proportion of denuded surface on a colony. Ginsburg et al. (1996) and Lewis (1997) graded corals as < 1/3 dead, 1/3–2/3 dead, and > 2/3 dead. Some have used a quartile system (0–25, 26–50, 51–75 and 76–100 percent live or dead), and the value for each colony is reported as the midpoint of the quartile range (12.5, 38, 63 and 88 percent, respectively). Because colonies at the extremes of 0 percent live tissue and 100 percent live tissue can be easily distinguished, an expanded quartile system would provide six categories (0, 1–25, 26–50, 51–75, 76–99 and 100 percent; Fisher et al. 2007b). The Atlantic and Gulf Rapid Reef Assessment Program (Lang 2003) reports partial mortality in 10 percent increments in the mid-ranges, and approaching

Photo: EPA

Figure 3-3. Loss of living tissue on a colony is not necessarily lethal to the colony; it is, however a sign of damage or poor health, and if substantial portions of live tissue are lost, the colony will ultimately succumb.

extremes progressively reports in 5 percent, 2 percent and, finally, 1 percent intervals.

3.4 Recommended monitoring protocol for stony corals

Drawing from the above information, an RBP for characterizing condition of stony corals can be recommended. The protocol is intended for use in a long-term biocriteria monitoring program, which requires exploratory biological surveys to inform and mold the monitoring design and strategy (Section 1.2). Biological surveys provide data to address reef classifications, metric variability, size and number of sampling units and reference conditions. Consequently, these preliminary surveys are indispensable to developing an efficient and defensible, long-term monitoring program.

The following protocol is recommended (Table 3-1). Three core observations are reported for each stony coral colony within the transect perimeter—species identification, size and percent live tissue. Trained and experienced personnel should determine species identification. Initially, size should be determined by measuring three colony dimensions, height, maximum diameter and width. All three measurements should be made until the effect of reducing the number of measurements can be determined for each size-related indicator (Appendix C). Measurements can be eliminated for consistently small species or for all small colonies—these can be tallied and assigned a common, approximate surface area. This should not radically alter results and could save valuable time, especially in reefs with many small colonies. Larger colonies should be measured because they have greater influence on the biological and physical endpoints. Percent live coral can be estimated in 10 percent increments (recommended), using six categories (i.e., 0 percent, 100 percent and quartile ranges of 1–25 percent, 26–50 percent, 51–75 percent and 76–99 percent) or variable intervals such as progressively finer intervals near the extremes (Lang 2003). Statistical comparison of endpoints will illustrate which methods have sufficient accuracy to discriminate differences across reef types and management zones.

It is essential that all three observations—colony identification, size and proportion of live tissue—be made on every colony that occurs within the transect perimeter. For example, measuring a subset of the coral population for size and a different subset for percent live coral tissue dramatically decreases the number and value of indicators that can be calculated. Consistent data collection and calculation for each colony ensures that the characteristics and contributions of different species can be delineated.

Table 3-1. Three recommended core measurements for stony coral surveys

Three Core Observations

1. Identify colonies in transect to species (or at least to genus)
   - Colonies ≥ 10 cm, any dimension
   - More than 50% of the colony is within the transect perimeter or crosses it completely
   - Colonies < 10 cm can be tallied (recruitment data)
2. Measure three colony dimensions; height, maximum diameter and width
   - Fewer measurements possible as warranted
   - Small colonies can be tallied and assigned a common surface area
   - Substitute measurements as needed for specific geometric surface area solutions (e.g., # branches, # columns; Appendix A)
3. Visually estimate the percent live tissue on the colony in 10% increments
   - Estimates based on the total (3D) colony surface area
   - If preferred, a minimum of six percentage intervals can be used
   - Either live or dead proportions can be recorded (converse values)

3.5 Optional reporting

The recommendation for three core observations is not intended to exclude others. Measurements of additional reef attributes should, in fact, be included if dive time permits. For example, evidence of disease or coral bleaching (Figure 3-4) could easily be noted for each colony during the survey. This requires additional expertise and that surveys be performed at the same time each year when disease is at its peak (index period). If the consequence of recent bleaching and disease events is an objective, the survey might differentiate between old and recent tissue mortality (Lang 2003; TNC 2006). The line tender (Section 3.6) could conduct a census of fish, echinoderms, sea stars or soft corals within the transect perimeter or in the surrounding area. These will ultimately be useful for developing community-based and multimetric biocriteria. A potentially valuable addition to the survey is to make a video recording or a series of still pictures of the reef or transect—not necessarily to validate measurements but to visually document changes that occur over time. This complementary evidence could be useful for communicating the extent of degradation or improvement to public stakeholders.

3.6 Radial-belt transect

A radial-belt transect has been used very successfully with the Stony Coral RBP. Other survey approaches, as long as they employ a transect area (not a transect line), should be equally appropriate. In previous studies, radial-belt transects were delineated by two circles 8m and 10m from permanent stakes at each station (Santavy et al. 2001; Fisher et al. 2007a). A 2-m high center pole is placed over a short, permanent stake with a lightweight line attached to a pivot on the upper half of the pole. One diver, the line tender, holds the line above the tops of the colonies and pulls the end of the line away from the stake. Weighted flags or beads are clipped to the line 8m and 10m from the pole to mark the 2-m belt width (Figure 3-5). An underwater buoy is placed at the start point. The line tender maintains a taut line while the surveyor records all corals that fall within the 8–10m belt marked by the beads. Both divers proceed around the circumference of the circles until reaching the underwater (start/stop) buoy.

The survey area of the 8m–10m transect is $(\pi 10^2 - \pi 8^2) = 113.1\ m^2$. However, sampling unit

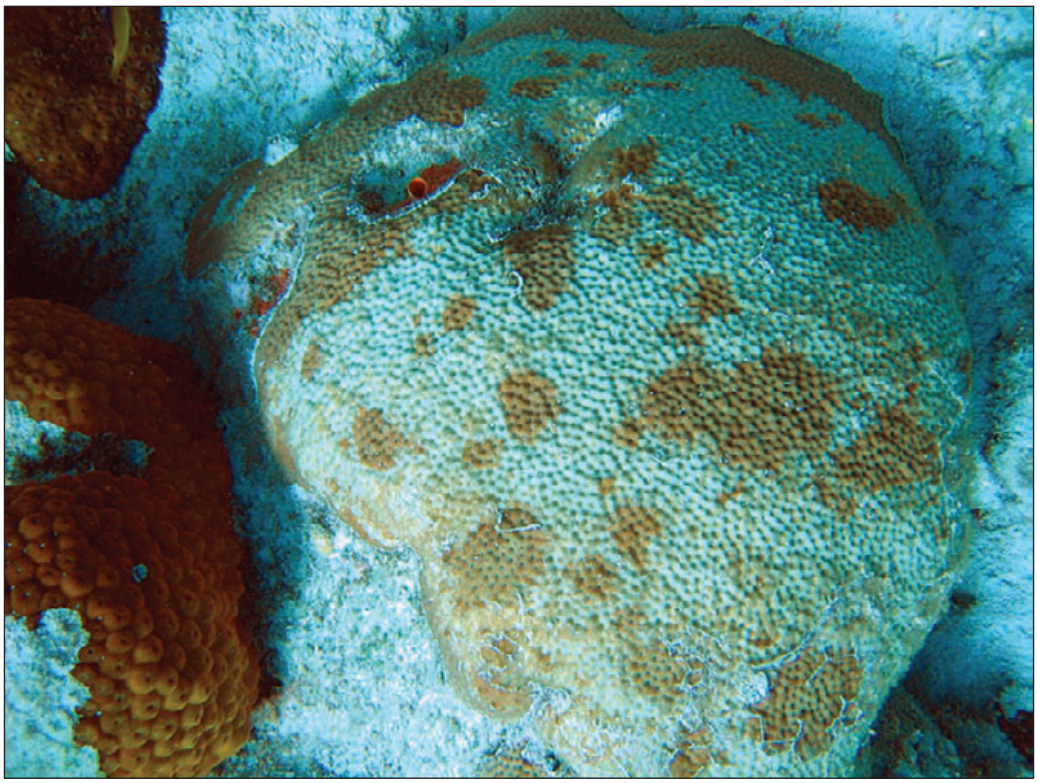


Photo: EPA

Figure 3-4. Observations of bleaching and disease can be made simultaneously with measurements for the Stony Coral RBP but require disease expertise. Here, bleaching from an unknown cause has left a *Siderastrea siderea* colony with a mottled appearance.

Photos: EPA

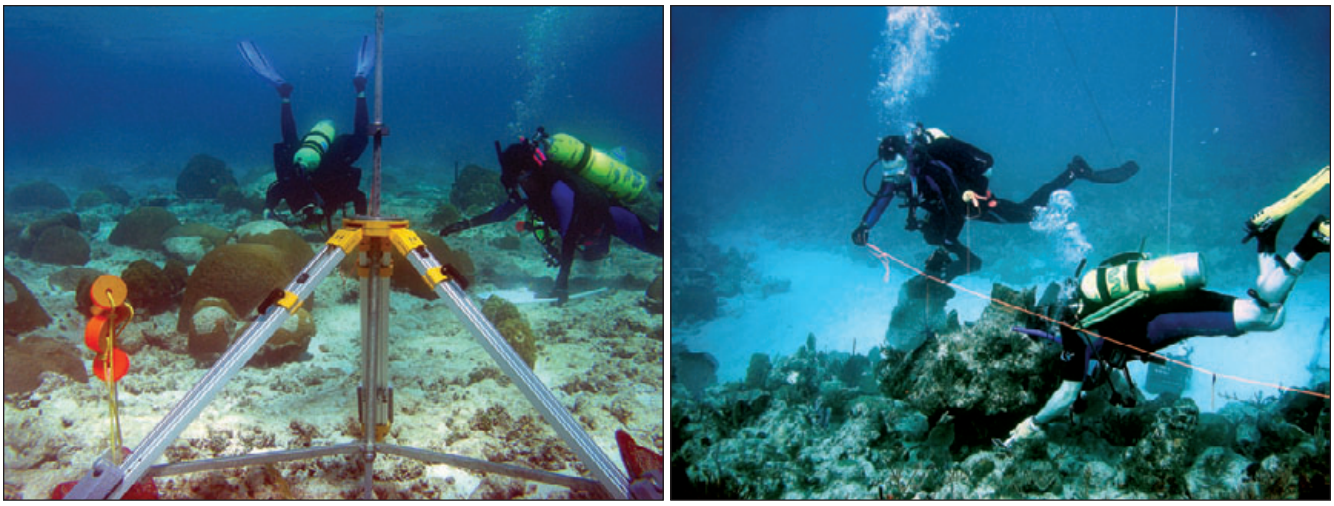


Figure 3-5. A radial-belt transect is established with a fixed stake (permanent station) or temporary tripod (left) at its center. A line is extended from the center and held above the colonies by the line tender (right). Weighted flags or beads are suspended from the line to mark the 2-m width of the belt transect).

size can be unique to each program and should be determined from the biological survey. Previous work indicated that a 50 m^2 transect, which can be obtained from a 3m–5m radial belt, was sufficient to distinguish significant differences among reefs (Fore et al. 2006a, 2006c). It is important that the outer radius of the belt be at least 5m to overcome aggregation of species (Loya 1972, 1978).

A radial-belt transect is suggested for many reasons. (1) It can be established and surveyed quickly. At permanent stations, the center pole is designed to slip over the embedded stake (Santavy et al. 2001) which holds it vertical. For temporary stations (no stake), a weighted tripod is used to hold the central pole vertical. The procedure does not require multiple lines or grids, and lines do not need to be moved or re-set for different measurements. (2) A survey performed using a radial-belt transect reduces the risk of coral tissue damage because the line tender elevates the line above the colonies. (3) The sampling unit size can be altered by simply increasing or decreasing the radii of the belt. This allows the same fixed stations to be used for multiple objectives. More importantly, it provides a consistent procedure for different sampling unit sizes. It is possible that large transects (e.g., 8m–10m belt) are more effective for trend detection (more comparative information), and small transects (e.g., 3m–5m belt) are more effective for status (more stations can be sampled). (4) Finally, the radial-belt transect is a very safe survey approach. Buddy divers are always close together and one diver, the line tender, is able to maintain an awareness of the surroundings while the surveyor focuses on documenting coral condition. The line tender can also take pictures or tally other key reef organisms, such as soft corals, sea urchins and conchs.

3.7 Synopsis

The Stony Coral RBP was specifically designed to support bioassessments preformed by jurisdictions with coral monitoring expertise but with limited resources and personnel. Surveyors are required to establish transects, identify colonies by species, measure colony dimensions and estimate the proportion of live coral on each colony. In nearly all cases, even with high coral densities, stations have been successfully surveyed by two divers (one surveyor and one line tender) on a single dive. Data can be entered directly into a spreadsheet with formulas pre-set to instantaneously calculate indicator values; therefore, feedback on transects and reef characteristics is immediate. The protocol requires no costly technical instrumentation or expertise, so the magnitude of a monitoring program is restricted only by available surveyors and line tenders. Technical transfer of RBP methods is uncomplicated, and data collected by trainees has been found to closely match data from experienced surveyors (Fore et al. 2006c). Indicators that are generated from the RBP and their application to coral reef management are discussed in the remaining sections.

4. Stony Coral Condition Indicators

The three core measurements described in Section 3 are used to calculate a variety of biological condition indicators relevant to coral reef value and sustainability and necessary for regulatory bioassessments. Valuation of the reef and its services, as reflected in several RBP indicators, is a fundamental component of use designation. For example, indicators like LSA and taxa richness can identify rich and healthy reefs that provide abundant ecosystem services. These reefs would reasonably be assigned use designations requiring high levels of resource protection (see Section 1.1). Indicators are also essential for establishing protective standards. Once a waterbody use is designated, criteria are established to ensure that existing ecosystem services are sustained or improved. These protective criteria are based on expected or desired responses of specific indicators (metrics) that are sensitive to human disturbance. Levels of protection are then derived from metric values obtained at reference sites. Indicators are, thus, indispensable elements of regulatory bioassessment and their development an essential component of any monitoring program.

4.1 Multiple indicators from core measurements

Measurements and observations become indicators when they are used to characterize a meaningful aspect of coral condition. Although relatively simple, the three core observations of the RBP can be used to derive several traditional and several unique condition indicators relevant to coral reef value and sustainability. Those described here (Table 4-1) are not exhaustive—others can be derived from the same field data (e.g., Fore et al. 2006c) or with additional data. Not all indicators calculated from the three measurements will be useful for every management purpose. Indicators to be used for regulatory action, for example, must be screened for power of detection and response to human disturbance (Section 5.1). An important feature of the RBP is that it provides multiple indicators for screening. During development of biocriteria for freshwater streams, only 16 acceptable fish and invertebrate metrics were identified out of 178 indicators that were screened (Fore 2003). So, not all indicators will be sufficiently sensitive to anthropogenic factors, and each might be more or less responsive under different environmental, reef and stressor situations. It is a clear advantage then to calculate multiple indicators from the relatively simple RBP field observations.

The indicators listed here are condition indicators. It is re-emphasized that *condition indicators do not identify causes of change*. Diagnostic assessment is different than condition assessment and usually employs physical, biomarker, geochemical and weight-of-evidence approaches to associate coral degradation with nutrients, sediments, contaminants and other anthropogenic stresses (Beyers 1998; Risk et al. 2001). Condition indicators are used to identify impaired waterbodies, and exposure or diagnostic indicators are used to identify the cause of impairment. Exposure and diagnostic indicators have been reviewed (Jameson and Kelty 2004) and are not discussed here.

Many traditional and some novel coral indicators are provided by the RBP. Novel indicators are largely drawn from 3D CSA estimates, which are used to calculate total surface area (TSA), live surface area (LSA) and other indicators that incorporate these values. One such indicator, 3D total coral cover (3DTC), could be used to supplant the conventional chain-transect method for estimating coral topography (i.e., *rugosity*, Appendix B).

For organizational convenience, indicators are introduced in categories of *abundance and composition*, *physical status* and *biological condition* (Table 4-1). All are derived from the three core measurements: Abundance and composition are derived

Table 4-1. RBP coral condition indicators[1]

RBP Coral Condition Indicators
Abundance and Composition
Abundance: *number of colonies*
Density: *number of colonies per m^2 sea floor*
Relative species abundance: *abundance of a particular species per total abundance*
Species (taxa) richness: *number of species occurring in a reef or region*
Species frequency: *proportion of sites where a species occurs*
Species diversity: *index of taxa richness and relative abundance*
Protected species: *richness and abundance of protected coral species*
Community composition: *relative richness or abundance of a species or group of species with some discretionary biological or physical attribute (e.g., tolerance)*
Physical Status
Colony surface area (CSA): *3D skeletal surface area of an entire colony (m^2)*
Total surface area (TSA): *∑ CSA for all colonies at a transect, station or reef*
3D total coral cover (3DTC): *TSA per m^2 sea floor*
Average colony surface area (AvCSA): *TSA / # colonies*
Population structure: *colony size distribution for a species compared to colony number or other attribute*
Community structure: *colony size distribution for all species compared to colony number or other attribute*
Biological Condition
Percent live tissue (%LT): *proportion of live coral tissue on each colony*
Average percent live tissue (Av%LT): *∑ %LT / # colonies*
Colony live surface area: *live tissue on a colony (m^2) = CSA × [%LT / 100])*
Live surface area (LSA): *∑ colony live surface areas at a transect, station or reef (m^2)*
3D live coral cover (3DLC): *LSA per m^2 sea floor*
Percent Live Surface Area (%LSA): *comparative ratio of live and total surface area = ([LSA / TSA] × 100)*

[1] Indicators are derived from three core observations on stony coral colonies and can represent cumulative or average values for transects, stations and reefs or for a particular species or group of species.

from coral identity, physical status from colony size and biological condition from proportion of live tissue (%LT). Indicators are calculated from observations on individual colonies and combined to obtain cumulative or average values for transects, sites, reefs or a particular species.

4.1.1 Coral abundance and species composition

Coral abundance and species composition varies from region to region, reef to reef, and even within a reef. Composition can be influenced by depth, hydrologic patterns and a variety of other environmental conditions. The stony coral community can thus be characterized by the presence and abundance of different species. Because colonies are identified to species in the RBP, values can be aggregated for community characteristics or applied to distinct populations.

Indicators related to abundance and composition are calculated from the transect census. ***Abundance*** is simply the number of distinct, independent coral colonies and does not account for size differences. Abundance can be represented by ***density***, which is abundance normalized by the area of sea floor surveyed (number of corals per m^2 sea floor).

Coral density characterizes the proximity of colonies to one another—a factor that affects disease epizootiology (transmission of infective agents), sexual reproduction (dispersion of gonochoric gametes) and recruitment (settling of planula larvae). Coral abundance can be differentiated by species to obtain ***relative species abundance***, which is the proportion of colonies for one species relative to the abundance of all species combined. This indicator is apparently useful even with patchy distribution and large population shifts (Jameson et al. 2001).

Species identification is used to characterize coral community composition. The total number of species encountered in a reef is reported as ***species richness*** or ***taxa richness***. This attribute is often used to demonstrate the ecological complexity of a community. Species richness is not generally calculated for single transects because it is meaningful only for relatively large spatial areas (Magurran 1988). Because it generally increases with greater sampling effort, species richness is sometimes assigned the asymptote of the rarefaction curve, which depicts the number of new species encountered with additional transects (Peet 1974; Aronson et al. 1994). In some instances, species richness has been reported as the number of species per unit area of sea floor (Chou 1984). However, while providing some spatial context, this endpoint is also influenced by the sampling unit size and number of sites surveyed. ***Species frequency*** is the proportion of sites, relative to the overall number sampled, where a particular species occurred. This indicator is a presence-or-absence observation and does not physically quantify coral abundance or colony size. While infrequently used, species frequency is intended to characterize the geographic distribution and range of a species.

A ***species diversity index*** is often suggested as a means to integrate species richness and relative abundance (or evenness among species); it characterizes the variety and abundance of different types of organisms that inhabit an area. The index incorporates the number of species present and the proportion of individuals in each species. There is greater diversity for reefs with more coral species, and there is greater diversity for an even distribution of individuals among the species. Many diversity, evenness and community similarity indices are available (Simpson 1949; Pielou 1966; Chou 1984), but the one that is most often cited is the Shannon or Shannon-Weaver Diversity Index (Shannon 1949):

$$\text{Diversity Index } (H') = -\left[(P_1 \times lnP_1) + (P_2 \times lnP_2) + (P_3 \times lnP_3) \ldots. (P_n \times lnP_n)\right]$$

where *ln* is natural log, *P* is the proportion of individuals of each species relative to the total number of individuals in species 1 through n. Because *P* is a proportion of the total, diversity does not express the actual abundance of any species—a reef with many colonies could have the same diversity as a reef with few colonies. Stony coral diversity can be calculated using proportions of individuals, proportions of live coral cover (Chou 1984; Aronson et al. 1994), proportions of total skeletal surface area and proportions of live coral surface area (Fisher et al. 2007a), each potentially addressing a different information need. Aronson et al. (1994) recommended that diversity indices should be calculated from coral cover because of the vast size differences among coral colonies and because colony fragmentation and fusion obscure the identification of individuals. Although still widely reported, species diversity indices have several disadvantages (Hurlbert 1971) and are often replaced with species richness and relative species abundance. These indicators provide the same information and are less ambiguous (Karr and Chu 1999).

The number and abundance of ***protected species*** are particularly important in waterbody valuation because listed species carry special protection requirements. Several potential indicators stem from ***community composition*** data, which describe richness or abundance of a species or a group of species on the basis of some discretionary characteristic. As environmental conditions change in a habitat, community composition might shift from intolerant to tolerant, from large to small, brooder to broadcaster or from slow-growing to fast-growing species.

4.1.2 Physical status

Indicators of physical status are derived from estimates of colony size. Colony size is important—larger colonies and larger reefs increase the protection of shorelines from erosion, the habitat available to reef communities, the amount of calcium carbonate sequestered by coral skeletons and, assuming a high proportion of live tissue, the amount of live coral available for photosynthesis, growth and

reproduction. Colony size is also an important analytical tool. For a particular species, size is generally related to colony age, and size-frequency distributions can provide insight to historical and possibly future condition.

Colony surface area (CSA) is the 3D topographic surface area of the entire colony, including both living and denuded portions. Volumetric size classes or measured colony dimensions can be converted into estimates of 3D CSA (m^2) using geometric equations or conversion factors defined by colony morphology (Section 3.2; Appendix A). ***Total surface area*** (TSA) is a summation of CSA for all colonies in a station or reef and reflects the coral surface area available as community habitat. TSA normalized by transect area (TSA per m^2 sea floor) becomes ***3D total coral cover*** (3DTC), which is an indicator of stony coral topography and complexity. It is calculated from stony corals only and does not account for rocks, ridges, mounds, spurs, grooves, crests or other physical structures that also provide sea floor complexity. If overall reef relief is a monitoring objective, the chain transect method (Appendix B), which is sometimes used to measure 2D contours of both coral and non-coral structures, can be employed (Porter 1972; Risk 1972). TSA can also be normalized by the number of colonies at a station or reef to obtain an ***average colony surface area*** (AvCSA; m^2/colony).

Physical data on individual coral colonies provide several avenues to examine ***population structure***. Colony size, at least within a species, is a practical reflection of relative colony age (Connell 1973; Hughes and Jackson 1980; 1985), so analysis of the number of individuals within a particular size class (size-frequency distribution) can reflect changes that have occurred in the population over time. Population structure can also be analyzed as size-surface area (Figure 4-1) because the RBP provides 3D CSA estimates. Size-frequency characterizes the numeric relation between small and large (presumably young and old) colonies and size-surface area characterizes the distribution of surface area between small and large colonies. Similar comparisons can be made for changes in %LT over various size distributions as an indication of size-related condition of colonies (Section 4.1.3). A size-frequency distribution of all colonies in the reef, regardless of species, represents the ***community structure***, which reflects the heterogeneity of coral sizes available for habitation by fish and invertebrate populations.

4.1.3 Biological condition

Several indicators of biological condition are derived from estimating the proportion of live tissue on a coral colony. Percent live tissue (%LT) reflects the cumulative influences of beneficial and adverse environmental conditions. The converse observation, partial mortality (%), has been reported in many studies (Brown and Howard 1985; Brown 1988; Dustan 1994; Gomez et al. 1994; Ginsburg et al. 1996; Lewis 1997; Endean et al. 1988; Lang 2003). It is important to note that, while the two perspectives are converse, estimates of partial mortality are often made from 2D aerial perspectives (tops of colonies) rather than the whole colony.

Estimates of ***percent live tissue*** (%LT) on individual colonies can be averaged for a station or reef (Av%LT). This indicator does not account for colony size but characterizes the average condition of individual colonies across a reef. Just as important, %LT is used to calculate ***colony live surface area*** (= CSA × [%LT / 100]), which reflects the amount of coral tissue on a single colony. Single colony values can be summed to obtain the ***live surface area*** (LSA) on a transect, station or reef that is available for growth, reproduction, and colonization by photosynthetic zooxanthellae. LSA can be normalized per unit of substratum to generate ***3D live coral cover*** (3DLC; m^2 live coral per m^2 sea floor). This indicator can be compared with historical two-dimensional (2D) live coral data using coarse conversion factors or geometric surrogates (Appendix B). Or, if diameter and width measurements are made on each coral colony, the traditional 2D LSA can be directly calculated from RBP measurements. ***Percent live surface area*** can be calculated as a convenient means to compare live and total 3D CSAs (%LSA = [LSA / TSA] × 100) of colonies within transects, reefs or regions. Unlike Av%LT, %LSA incorporates CSA (m^2), so larger colonies in the reef have a greater influence on the indicator value. This calculation is similar to previously reported vitality indices (Brown and Howard 1985; Brown 1988; Dustan 1994; Gomez et al. 1994; Ginsburg et al. 1996) but incorporates 3D, rather than 2D, colony measurements.

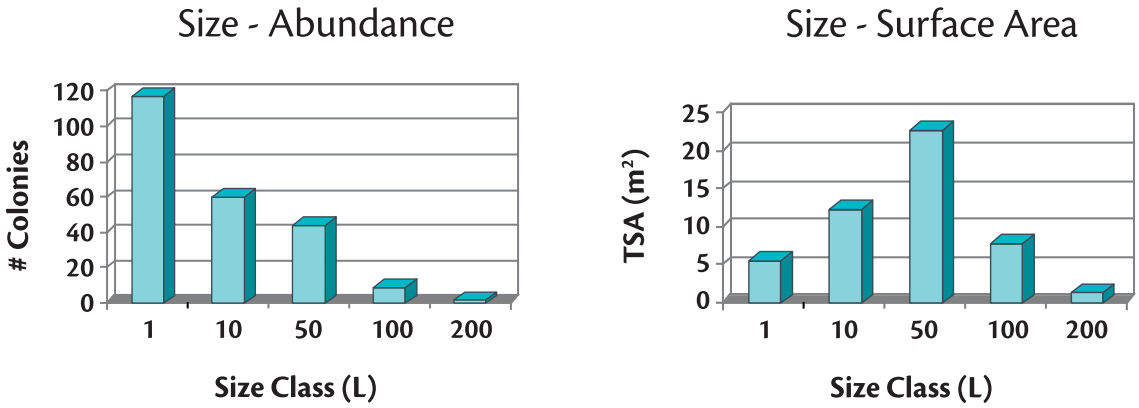


Figure 4-1. Population structure (size-frequency) diagrams of *Diploria clivosa* determined from a pilot study in Dry Tortugas (Fisher et al. 2007a). Population structure analysis can be performed using colony data (size-abundance, left) or surface area data (size-surface area, right). In this study, corals were graded into five different volumetric size classes (L = Liters).

4.2 Linking bioindicators to coral reef value and sustainability

Indicators generated from the Stony Coral RBP are linked to the values (services) and sustainability of coral reefs. Because stony corals are responsible for the physical infrastructure of coral reefs, measurements that characterize their physical and biological condition explain key attributes and functions of the reef ecosystem (Section 1.3). Linkage to resource value supports designation of waterbody uses and facilitates the communication and enforcement of management and regulatory activities. Indicators developed for management applications should have both ecological significance and strong ties to ecosystem values. The latter is a clear asset for public understanding and acceptance of management decisions to protect the resources (Suter 1990; Gentile and Slimak 1992; Fisher et al. 2001). Indicator linkage to sustainability is essential for establishing conservation benchmarks and targets. Stony coral protection is intrinsic to coral reef protection: it is very unlikely that reef values will be sustained if stony corals are not protected (Figure 4-2).

4.2.1 Indicator links to coral reef values

The physical structure provided by corals as community habitat is fundamental to the renowned biological abundance and diversity of coral reef ecosystems (Section 2.2). Coral structure is therefore indirectly responsible for substantive economic benefits, including fishing and tourism. The amount of coral structure available for habitat is represented by two indicators, *TSA* and *3DTC*, both of which are derived from 3D CSA estimates. Recreational and subsistence fisheries benefit from coral *species richness* and *community structure* (McCormick 1994). Tourism benefits from coral communities with high *abundance, density*, *species richness*, *species diversity*, *protected species*, and reefs with high *CSA*—all valued by divers and snorkelers. Several ecosystem functions are represented by the RBP indicators. *LSA* and *3DLC* characterize the quantity of live coral polyps potentially able to support zooxanthellae, grow and reproduce.

All these indicators can be considered when establishing designated uses for waterbodies. For example, if 80 percent of coral species found in an entire region occur at a single reef, if a threatened species is present or if reef topographic complexity is high, the reef might become a high priority for management protection.

4.2.2 Indicator links to coral reef sustainability

The proportion of live tissue on a coral colony (%LT) is a simple indicator of colony health. It does not indicate the physiological health of the remaining live tissue, but it reflects the cumulative loss of tissue relative to the colony's status at its peak. Hughes and Connell (1999) surmised that some level of tissue mortality is expected as corals grow larger; however, large portions of denuded skeleton undoubtedly signal a serious decline from an earlier

Photo: EPA

Figure 4-2. Stony corals form the infrastructure of a coral reef habitat and all biological and physical ecosystem services are obtained directly or indirectly from their sustained existence.

condition. *Percent LSA* also reflects colony health by comparing amounts of live and total coral. Percent LSA is the conceptual converse of a *mortality index* (dead coral cover:total coral cover) proposed by Gomez et al. (1994) and based on 2D planar estimates of tissue loss. Either calculation could be used to illuminate a contention of Ginsburg et al. (1996) that less living than dead coral would be evidence of serious reef decline. In a pilot survey at Key West, Florida, *Acropora palmata* and several sampling stations exhibited %LSA lower than 50 percent (Figure 4-3).

Changes in reef condition might also be revealed by changes in coral *community composition*, which characterizes the distribution, arrangement and abundance of different coral species (Loya 1972, 1976) and could include consideration of size (Rylaarsdam 1983), recruitment (Porter et al. 1981) and mortality (Hughes 1996). Community composition encompasses several possible indicators relevant to different situations. Fewer species, less diversity, a shift to more tolerant species and diminishing abundance of rare species could all indicate degrading environmental conditions (Jameson et al. 2001). It is likely that highly fecund, fast-growing and generally short-lived genera such as *Porites* and *Agaricia* will prosper after an environmental disturbance (Tomascik and Sander 1987). Aspects of coral community structure have been used to characterize spatial zonation of corals (Loya 1972; Odum and Odum 1955), effects of specific stressors (e.g., Odum and Odum 1955; Loya 1976; Porter et al. 1981; Hughes and Jackson 1985; Done 1992; Bak and Nieuwland 1995) and even to propose a zoning strategy for marine reserves (Reigl and Reigl 1996).

LSA is linked to reef sustainability because it indicates the physical potential for coral growth and reproduction. Large corals produce more tissue in absolute quantities than small corals growing at the same rate. More live coral also means that greater biomass is available for gamete production. In fact, a threshold colony size (live tissue) might be required for successful reproduction (Kojis and Quinn 1985; Szmant-Froelich 1985; Soong and Lang 1992; Soong 1993).

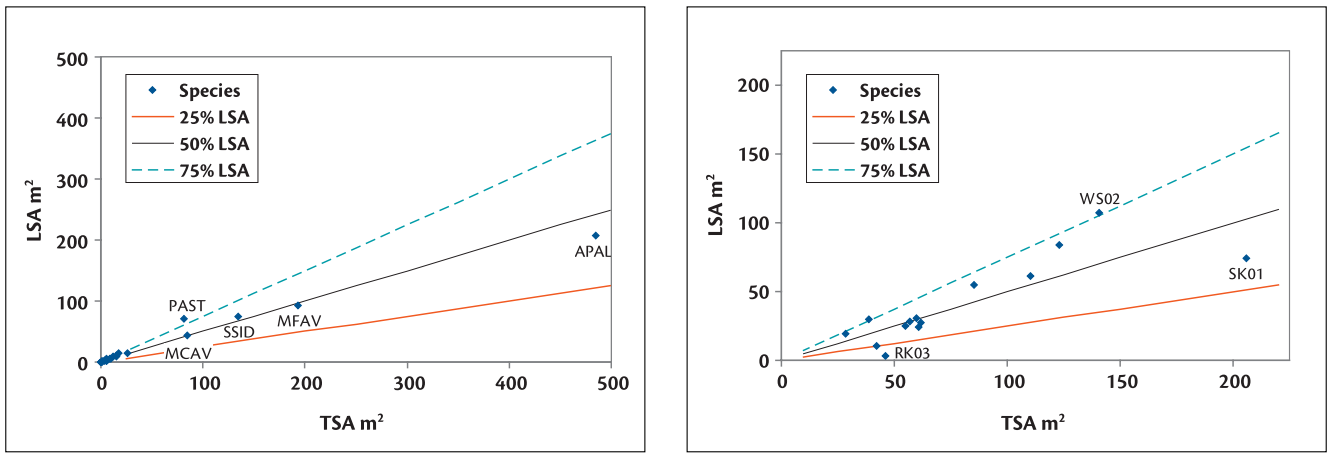


Figure 4-3. Comparison of TSA and LSA (m^2) documented for different species (left) and stations (right) from surveys performed at 14 stations near Key West, Florida. Lines represent 25%, 50% and 75% values for %LSA. Elkhorn coral, *Acropora palmata* (APAL) had the greatest TSA of all species encountered at these stations, but %LSA was less than 50%. Other species, including *Montastraea faveolata* (MFAV), *Siderastrea siderea* (SSID), *M. cavernosa* (MCAV) and *Porites astreoides* (PAST) provided substantial surface area to the reef. The PAST population retained a %LSA greater than 75%. Station SK01 had the highest TSA but less LSA than station WS02. Percent LSA was less than 25% for station RK03 and greater than 75% for WS02. Source: Fisher et al. (2007b).

For reef sustainability, new coral skeleton must be deposited (whether by growth or recruitment) at least as fast as it is degraded (Hutchings 1986). The Stony Coral RBP does not provide growth and degradation rates, but it does estimate the quantity of live coral with potential for growth and, conversely, the quantity of dead coral surfaces vulnerable to erosion. As sustainability models are developed, growth rates (Lewis et al. 1968; Buddemeier and Kinzie 1976) and degradation rates (Hutchings 1986) for particular species, reef types and geographic zones will become critical information.

Insights to reef sustainability can also be gained from examination of *population structure*, or the size-frequency distributions of different species (Babcock 1991; Bak and Meesters 1998). For example, the number of small (young) colonies (e.g., Figure 4-1) could indicate recruitment success and represent the foundation for future growth. Patterns of population structure are likely to be influenced by chronic environmental degradation (Meesters et al. 2001). Another indicator that can be derived from population structure is *size-related condition*, a comparison of %LT with the size of the colony. For example, Lewis (1997) analyzed *Siderastrea siderea* to demonstrate declining proportions of live tissue on larger colonies, a finding that is apparently quite common (Hughes and Connell 1999; Figure 4-4). Bak and Meesters (1998) used the same approach to compare mortality patterns of various species.

High *species diversity* usually implies integrity and stability in a community (Odum 1971). This association has been controversial (Hurlbert 1971; Peet 1974; Karr and Chu 1999) and has not been well-documented or defended for corals (Rogers et al. 1983). The use of *species richness* and *relative species abundance* could capture the same information with less ambiguity (Karr and Chu 1999). High species richness and even distribution of colonies among species is believed to maximize resource acquisition at different trophic levels, retain resources within the ecosystem and reduce the risk of dramatic changes in ecosystem processes in response to directional or stochastic variation in the environment (Chapin et al. 2000).

Indicators calculated from RBP observations could eventually be applied to sustainability models. Protocols that do not incorporate colony number or colony size or provide only 2D CSA estimates will contribute little to dynamic ecosystem models. At a minimum, useful models will require estimates of physical reef structure that serves as habitat, biomass to estimate the potential for coral growth and reproduction, and rates of skeletal accretion and erosion. Ultimately, sustainability models will improve use designation and protection criteria by incorporating projections of future coral condition.

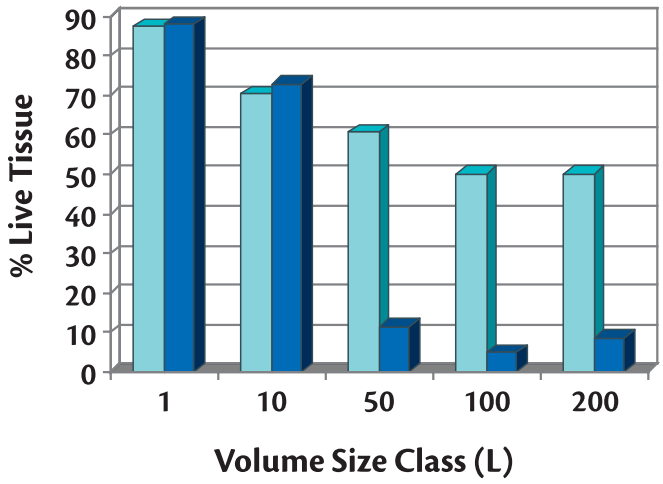


Figure 4-4. Colonies of *Acropora palmata* (dark bars) exhibited much lower %LT for middle and large size classes (≥ 50L volume) than colonies of *Montastraea faveolata* (light bars) on reefs near Key West, Florida. The data were collected in 2003; hurricanes, bleaching and disease before the survey are likely to have caused severe damage to existing colonies. Colonies apparently recruited after the damage (≤ 10L volume) exhibited higher %LT. Source: Fisher et al. (2007a).

4.3 Relation to existing indicators

Most of the Stony Coral RBP indicators are familiar to coral reef researchers and resource managers. For example, indicators derived from species identification and enumeration can be found throughout the literature. These have been measured using quadrats, point-quarter methods (Loya 1978; UNESCO 1984; English et al. 1994), line transects (Loya 1978), belt transects and video transects (Jaap et al. 2000; Wheaton et al. 2001; Brown et al. 2004; Jokiel et al. 2004). The various methods have led to some inconsistencies; for example, reports of species diversity have been calculated from both colony number and live coral cover (Aronson et al. 1994). Indicators related to the proportion of live tissue are less frequent, but all apply some categorical system for estimating percent live tissue and denuded skeleton on individual colonies (Grigg and Dollar 1990; Dustan 1994; Gomez et al. 1994; Ginsburg et al. 1996; Meesters et al. 1996; Lewis 1997; Lang 2003; Fisher et al. 2007a). Largely because of its use in the Atlantic and Gulf Rapid Reef Assessment Program (Lang 2003), most coral researchers are well aware of *partial mortality* and related measurements.

Methods to quantify coral size are less frequently incorporated. One reason for this could be the highly favored, linear-transect method to estimate 2D projected surface area from an aerial view. The linear-transect method does not always account for individual colonies, much less the size of those colonies. Yet, both surface area and colony indicators represent attributes useful for characterizing coral condition. The Stony Coral RBP capitalizes on geometric surrogates to obtain surface area from colony measurements (Appendix A) and thereby takes advantage of both surface area and colony-based indicators. Geometric surrogates provide 3D CSA estimates, which improves traditional indicators of live coral cover and topographic complexity. Historic 2D data, however, must be converted to 3D values for comparisons (Appendix B).

5. Application of RBP Indicators in Regulatory Monitoring

Indicators derived from the Stony Coral RBP are useful for characterizing coral condition, but their ultimate significance will be realized through regulatory monitoring programs that establish a direct link between indicator responses and regulatory action. Section 1 described the many regulatory programs and objectives that can benefit from the RBP. This section describes two examples of how RBP indicators can be evaluated and applied to biocriteria monitoring, a regulatory tool with high potential for protection of coral reefs. The examples include metric screening and developing an effective and defensible monitoring strategy. There are many other aspects related to biocriteria development that are not addressed here but will be the subject of future guidance. A brief summary of monitoring terms used in this section is provided (Table 5.1).

5.1 Indicator responsiveness to human disturbance

The CWA is intended to protect resources against human-induced decline, not decline resulting from natural environmental change. Intrinsic to a biocriteria program then is the need to ascertain and document which indicators are most affected by anthropogenic stressors. No matter how compelling the underlying biology for particular indicators might be, only those that represent human-induced degradation can trigger management action. Biocriteria programs, consequently, are derived from these indicators, which are then called *metrics*.

Evaluating indicators for response to human disturbance can be relatively straightforward (Fore et al. 2006b). An area of human disturbance is located, and sampling sites are selected at and away from the center of the area to represent a stressor gradient (Figure 5-1). If the bioindicator changes in a consistent and logical manner across this gradient, it is very likely responsive to the disturbance. Spacing of the stations depends on the intensity and scale of the disturbance, and stations are selected by judgment, not probability (Section 5.2.3). Repetition of the sampling gradient for each reef type or habitat will increase confidence in the results. For many situations, it might not be necessary to document the exposure (e.g., quantify the sediment, nutrients, or contaminants) as long as a point source or area of human activity is localized and apparent. However, water quality measurements might lead to identification of which stressors are causing the impairment.

Table 5-1. Terms used in the development of monitoring programs

Status: An estimate, or snapshot, of existing resource conditions; e.g., the live coral cover within a region.
Trend: Change in resource condition over time; e.g., the 3D live coral cover declined by x% over 5 years.
Target population: The resource about which information is needed, including a precise definition of the elements of the resource to be measured and a description of their spatial distribution; e.g., stony corals greater than than 10 cm in size living in the near shore environment of St. Croix island at a depth less than 10 meters.
Sampling unit: The individual item or area that will be measured or characterized during sampling; e.g., all the stony corals existing within a belt transect. The sampling unit is also called the population element.
Probability-based survey design: The process of selecting data collection sites where every site or element of the target population has a known, nonzero probability of being selected.

Adapted from EPA Aquatic Resources Monitoring: ***http://www.epa.gov/nheerl/arm/terms.htm***

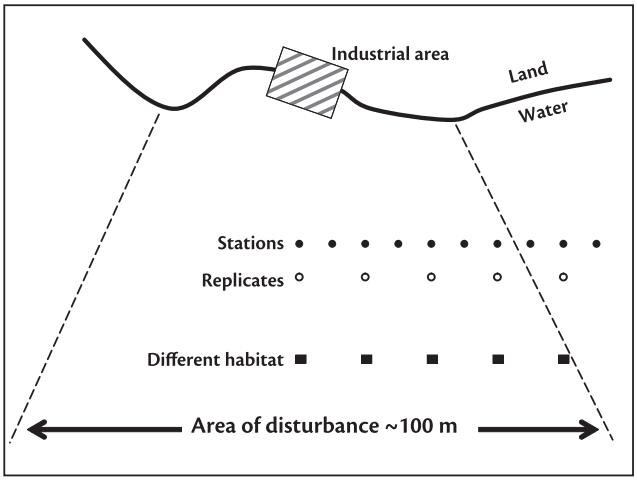


Figure 5-1. To test which indicators can serve as metrics, sampling is targeted across a gradient of human disturbance, such as an industrial area. Metrics will show a consistent improvement in coral condition for stations increasingly distant from the center of disturbance.
Source: Fore et al. (2006b).

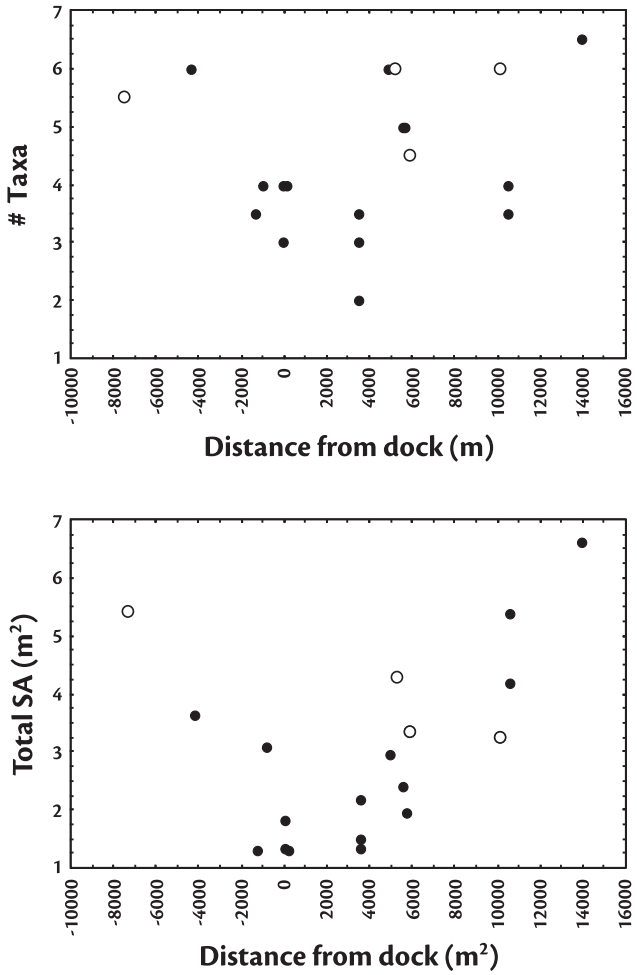


Figure 5-2. Coral condition indicators measured at increasing distances from the center of human disturbance (industrial docks at St. Croix, USVI) showed species richness (top) to have a less consistent response than total 3D CSA (m^2; bottom). Source: Fore et al. (2006c); open (> 6 m) and closed (< 6 m) points represent different depths. Prevailing currents moved west (left on graphs).

According to Karr and Chu (1999), taxa richness has been found to be consistently responsive to human disturbance gradients in freshwater systems. However, a biological survey of corals along the southern shore of St. Croix found TSA more responsive (Fore et al. 2006c; Figure 5-2). It is noted that this approach could selectively identify only those metrics that are responsive to point-source, rather than nonpoint sources. Also, it is difficult to apply this approach to screen mobile indicator organisms (e.g., fish) that could move in, out and within the disturbance gradient.

There is no limit to the number of indicators that can be screened or metrics that can be used to develop biocriteria. Indicator responsiveness is likely to vary with different reefs, coral communities and human stressors. A clear benefit of the RBP is that the same three core measurements provide several indicators, any one of which could serve as a metric in a variety of different circumstances.

5.2 Developing a biocriteria monitoring program

The first step toward developing a defensible biocriteria monitoring program is to evaluate and document the metrics that could be used (above). Next, it must be determined which of these can detect significant change associated with human influence. This requires a monitoring program optimized around the variability of the metrics, the number of management zones and reef types, the objectives of the program and the monitoring capacity of the resource agency. Consequently, the monitoring strategy is generated in iterative steps, starting with preliminary studies (biological surveys) of the target population.

5.2.1 Metric variability

Data collected from individual sampling units (stations) need to be sufficient to characterize condition and detect differences at a level relevant to the monitoring objectives but within the resource constraints of the monitoring agency. Metrics that are highly variable will require larger sampling units or more sampling stations to detect differences. The size of the sampling unit can be determined by comparing variances obtained with different sized survey transects. In the U.S. Virgin Islands, the size of radial-belt transects was examined by comparing variances among transect quadrants, which were established by placing subsurface buoys N, S, E, and W of the transect center (Fore et al. 2006c). Data were examined for variance differences between one-quarter, one-half and three-quarter transects with the full transect. They found that a full transect did not appreciably improve (decrease) variance over a half transect; sampling only half of the radial transect reduced sampling time substantially.

The number of stations to be sampled can be investigated through power analysis. In the above study (Fore et al. 2006c), power analysis was performed to calculate the minimum detectable difference for each candidate metric as it related to the number of stations sampled. In this example (Table 5-2), colony number would have to change by 17 for statistical significance (p = 0.1 for a one-sided t test) if five stations were sampled in a reef type or management zone, but this minimum detectable difference would decrease to 12 and 9 colonies if more stations (10 and 15, respectively) were sampled. Taxa richness would have to differ by three species if five stations were sampled, but only by one if 15 stations were sampled. In all cases, a higher sampling effort lowered the mean detectable difference (increased the sensitivity of the metric). Yet, more sampling increases cost in time and effort, so resource managers must optimize these factors in the monitoring strategy. It is possible, too, that minimum detectable differences for certain metrics are too high to have any functional relevance to reef management.

Table 5-2. Station effects on minimum detectable differences (MDD)[1]

Candidate metric	MDD (5)	MDD (10)	MDD (15)
# Colonies	17	12	9
# Taxa	3	2	1
% Live coral (colonies)	13	9	7
Total SA (m^2)	4	3	2
Living SA (m^2)	4	3	2
% LSA	20	20	10
Average colony SA (m^2)	0.34	0.23	0.19

[1]MDD were calculated from a biological survey in St. Croix, U.S. Virgin Islands; values represent how much a candidate metric would have to change for statistical significance (p = 0.1 for a one-sided t test) if 5, 10, or 15 stations were sampled in each zone. Source: adapted from Fore et al. (2006c).

5.2.2 Management zones and reef types

The number of stations to be sampled is a critical element in developing the monitoring strategy. Stratification by management zone or reef type will increase the required number of sampling stations. Monitoring objectives and metric data should therefore be examined carefully to eliminate any unnecessary stratification. There will be little flexibility in determining the number of management zones, which generally reflect different waterbody use designations and require individual analysis; but the necessity for different reef types (classifications) can be determined through analysis of metric variability.

Classification of a biological resource is used to reduce natural variation in measured attributes (Gibson et al. 1996; Gerritsen et al. 2000). Classification partitions the resource into ecological units for which expectations in structure, function and, most

importantly, measured attributes are similar. Proper classification increases the precision of measured indicators, adding power and value to ecosystem assessments (Gibson et al. 2000). Moreover, classification provides a structure for synthesizing data across regions and jurisdictions (Madden and Grossman 2004). While no unifying scheme has emerged, there are many possible approaches for coral classification (UNESCO 1984; Jameson et al. 1998, 2001, 2003a; Mumby and Harborne 1999).

For all its virtues, classification can be costly if it increases the number of samples necessary for statistically significant results. Despite human tendencies to group similar objects, classification might not be relevant to regulatory monitoring unless it substantially improves data precision (see Kurtz et al. 2006). It is, therefore, counterproductive to automatically classify reefs on the basis of a scheme or perceived differences. Instead, data analysis can determine whether classification is warranted. The many measurable differences in reef geomorphology, hydrodynamics and composition might not necessarily have a substantive effect on metric responses.

One means to reduce metric variability associated with reef type is to exclude unnecessary reef types from the target population (Table 5-2). For example, watershed effects might best be evaluated by reefs close to shore, so offshore habitat types could be excluded. Some reef types might occur in surge zones that are difficult to sample. These can be excluded from the target population and greater focus placed on those reefs that are more easily sampled.

5.2.3 Program objectives

The core measurements of the Stony Coral RBP can be used for a variety of monitoring objectives, and useful data can be collected using most sampling designs, transect types and classification systems. Coral monitoring programs typically include three types of sampling objectives: (1) assessing the current status, (2) detecting trends over time and (3) evaluating conditions at specific locations (targeted or judgment sampling). These approaches differ in the manner in which sampling units (stations) are selected, defined and interpreted (Fore et al. 2006c). Status assessment is best accomplished with random selection of sampling locations every year; trend detection can be accomplished with randomly selected stations that are revisited in subsequent years (at least until temporal variability is characterized); and targeted sampling is accomplished at purposely selected locations to address specific management questions.

Federal and national programs such as EPA's Environmental Monitoring and Assessment Program (now National Coastal Assessment) often assess status to characterize the condition of large regional areas (Larsen 1997). Stations are randomly selected but will not be repeated in subsequent years, so the locations are not permanently marked. The principal advantage of random site selection is that any summary statistics derived from a random sample will be unbiased for the entire population, including all possible sites in the defined region that were not sampled. This means that randomly selected sites can be used to represent the entire region. For status assessment, a larger number of sites at the cost of a smaller transect area might be preferable.

In contrast, jurisdictions could have local mandates to identify sites and sources of degradation and to develop management responses that mitigate local effects. While overall regional condition might be useful for CWA reporting requirements, local resource managers also need to know which waterbodies are degraded (Hall et al. 2000). For targeted sampling, stations are selected to fulfill a particular objective and locations often marked for return visits (permanent or *fixed* stations). Targeted sample monitoring is used, for example, to compare particular reefs, to characterize trends in threatened species, to monitor effects of pollution sources (existing or pending) or to evaluate the success of management activities. Targeted sampling can be used to address management issues raised through status and trend monitoring. Because they are not randomly selected, data from targeted sites are applicable only to those sites and cannot be used to more widely represent a reef, management zone, or region.

Managers at every level are interested in trend assessments to signal whether a resource is improving or declining. The goal is to detect change in condition through time should change occur. Regression is the recommended statistical model for trend detection in which the variable of interest (e.g., total live coral) is regressed against time. The greatest power to detect temporal trend is found by comparison of a particular station with itself (Larsen et al. 1995; Urquhart et al. 1998). This emphasizes temporal variability by eliminating among-site (spatial)

variability, which can be measured separately. Stations for trend monitoring, whether they were targeted or randomly selected, can be permanently marked for repeat sampling. Larsen et al. (2001) have described how to maximize the probability of detecting a trend by balancing sampling effort among sites, replicates and repeat visits. Depending on results from initial surveys, trend detection could be optimal with fewer sites and larger transects.

Regional patterns of reef condition are fundamental to characterizing the extent and severity of decline and developing hypotheses of causality (Ginsburg and Glynn 1994). Yet, disparity among monitoring designs has resulted in duplication of effort and squandered opportunities to integrate local and regional data. In general, existing studies have employed an array of sampling methods focused on local, rather than regional, coral condition. Consequently, statistical comparisons at larger spatial scales are impossible (Kramer 2003). Both local and regional monitoring objectives could be more easily fulfilled if sampling for local monitoring programs were designed to accommodate regional objectives. That is, monitoring programs can be designed so that a subset of randomly selected local stations could serve regional objectives.

All three sampling approaches can be incorporated in a framework that allocates sampling effort proportionately to both regional and local needs (Fore et al. 2006b). In a hypothetical example (Table 5-3) where annual sampling needs outstrip the agency's capacity, the three monitoring designs were allocated across 4 years in a rotating panel (rotating basin). In this example, where the agency can survey only 60 stations a year, different zones are sampled each year, and 60 targeted stations are sampled in the fourth year to investigate hypotheses that are expected to emerge. In this scenario, status for each region is documented every fourth year.

Table 5-3. Hypothetical rotating panel monitoring strategy[1]

Year	1	2	3	4
Zone 1	10 trend 30 status			
Zone 2		10 trend 30 status		
Zone 3			10 trend 30 status	
Targeted	20	20	20	60
Total	**60**	**60**	**60**	**60**

[1]A rotating strategy is depicted for a jurisdiction with three geographic or management zones and a sampling limit of 60 stations per year. The strategy involves re-sampling stations in each geographic region every 4 years and includes status, trend and targeted sampling. The fourth year of the rotating panel provides targeted testing to address specific local questions or hypotheses generated from earlier monitoring. This approach can serve both local and regional objectives. Source: Fore et al. (2006b).

5.2.4 Synopsis

There are several factors that must be considered in developing a regulatory monitoring program. Those factors, briefly introduced here, include screening indicators for metrics (responsive to human disturbance), determining the sensitivity of metrics to changes in coral condition, selecting an appropriate sampling design for local and regional objectives and incorporating the design into a sampling strategy that can be realistically completed and sustained by the responsible resource agency. Other significant aspects of regulatory monitoring include assigning designated uses, prioritizing questions to address and setting levels of expected condition (e.g., biocriteria).

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Appendix A: Estimating 3D Colony Surface Area

Among others, Dahl (1973) stressed the importance of topographic, rather than planar, surface area for quantifying structural and physiological aspects of coral reefs. 3D CSA can be estimated for any coral colony, even if only one morphological measurement is known. How it is estimated depends on requirements of scale and accuracy. The scale must be relevant to project assessment questions, and the level of accuracy must balance statistical significance with overall efficiency.

Relevant scales for 3D CSA. Selecting the appropriate scale for surface area analysis is a significant concern but generally straightforward. Dahl (1973) offered the following:

> There are multiple levels of surface features depending on the scales at which they are considered. The earth, at one scale a smooth sphere, includes mountains which have boulders covered with microscopic ridges, and so on. The scale of surface variation becomes significant when it approaches the scale of the phenomenon being measured. Surface area is, thus, a relative measure depending on the scale considered. The benthic surface area of significance to a large organism will be different from (and far less than) that important to bacteria. (p. 241)

He indicated that for coral reefs, there are at least three scales of functional significance—reef, individual colony and colony surface (polyp scale). Various indicators of topography (surface features of an object or place) are measured at the reef scale and are intended to represent the physical habitat available for reef communities. Measurements of reef-scale topography most often (although not always) include corals plus non-coral geologic deposits such as rock, uncolonized hardbottom, and spur-and-groove or buttress-and-canyon formations. The non-coral components are relatively stable and, except for ship groundings and anchor damage, unaltered by most human stressors. Existing indicators of reef topography employ measures of coral height (1D) or vertical contour (2D *rugosity*), both of which fail to capture the most reliable estimator of physical habitat—the entire CSA measured in three dimensions.

The RBP estimates the physical habitat of a reef by summing CSA measurements of individual colonies. This approach was introduced by Dahl (1973) to estimate quantities of benthic algae and was later applied by Roberts and Ormond (1987) and Alcala and Vogt (1997). Summation is adopted in the RBP because it simultaneously provides reef-scale and colony-scale indicators; no additional lines, methods or transects are needed. Total (live plus dead) and LSA estimated for each colony can be used for colony-scale indicators (e.g., size-frequency distributions) and combined for reef-scale indicators (e.g., 3DTC).

Physiological processes of corals, such as respiration and algal photosynthesis, are best examined at the colony surface (polyp) scale because this is where the measured activities occur. Dahl (1973) suggested that polyp-scale estimates could be made by summing contributions calculated at different scales. In other words, the polyp-scale surface area of a colony can be estimated by adding the surface area of polyps (determined by subsampling) to CSA measured at the colony scale. Finer scales, as noted

earlier, generate larger surface areas but are not necessarily more *correct*. The abundance and diversity of harvestable fish are influenced more by physical habitat provided by reefs and individual colonies than by polyps.

Other scales might be relevant. For example, Roberts and Ormond (1987) selected a 1-cm resolution for fish habitat studies. A slightly larger scale might be used to estimate surface area of *finger* corals. Coral fingers, small branches from the main body of the colony, might not be important as refuge for large, harvestable fish but are important for small prey fish and invertebrates. Finger-scale surface area estimates could be made by combining CSA with surface area of fingers per square meter of colony (fingers are fairly regular across the surface of a coral). A decision to use this approach would depend on the importance of prey species to the assessment questions and the dominance of finger corals in the reef.

Accuracy of CSA estimates. Selecting an appropriate level of accuracy for measurements in a monitoring program is an iterative process. An appropriate level balances the objectives of the program against the amount of time and effort required for greater accuracy. Even if greater accuracy increases statistical power, this advantage is lost if too few stations are sampled. Accuracy is not, in itself, the objective of a monitoring program. Coarse estimates might be sufficient to distinguish significant differences for monitoring objectives. Dahl (1973) noted that for coral surface estimates,

> …absolute accuracy is almost never required, particularly when the areas themselves are so highly variable. What is needed initially is a meaningful basis for comparisons and generalizations, and this can usually be achieved with careful approximation. (p. 241)

The level of accuracy for each monitoring program must be appropriate for the monitoring objectives and feasible within constraints of time and cost. Several methods are available to approximate CSA, including some that provide low (e.g., volumetric surrogates), medium (geometric surrogates) and high (photographic reconstruction) levels of accuracy.

A coarse volumetric surrogate was recently used to estimate CSA (Fisher et al. 2007a). Regardless of species and morphology, coral colonies were visually graded into boxes (cubes) of different predetermined 3D size classes (Figure 3-1). CSA was assigned as the surface area of five sides of the box (omitting the bottom, which is not functionally relevant as community habitat or LSA). Grading colonies according to volumetric size was very rapid, supplied useful information on coral size and LSA, and readily distinguished differences in physical indicators (e.g., TSA and LSA, Figure 4-3) across geographic zones and reef types in the Florida Keys (Fisher et al. 2007b). While coarse colony size estimates might be very effective for most of the indicators included in the Stony Coral RBP, analysis of population structure is limited when using predetermined size classes.

Geometric surrogates have been used by many to estimate CSA (Dahl 1973; Szmant-Froelich 1985; Roberts and Ormond 1987; Alcala and Vogt 1997; Bak and Meesters 1998; Meesters et al. 2001). In this approach, various colony dimensions (usually height, diameter and width, Figure 3-1) are measured and applied to species-specific (or morphologically dependent) geometric solutions for surface area. The simplest example is the hemispheric shape of massive corals. Colony dimensions are used to determine radius (r), which is applied to $2\pi r^2$, the geometric solution for a (bottomless) hemisphere. Solutions for prolate and oblate hemispheres can be used when height is much greater or much less than the projected radius of a colony (Szmant-Froelich 1985). Fore et al. (2006c) used either a hemisphere or a cylinder as a surrogate, depending on the ratio of colony height to average radius. Alcala and Vogt (1997) applied geometric solutions to six different colony growth forms (Table A-1). The surrogates selected will likely vary in different programs and are influenced by the number and type of underwater measurements that will be made.

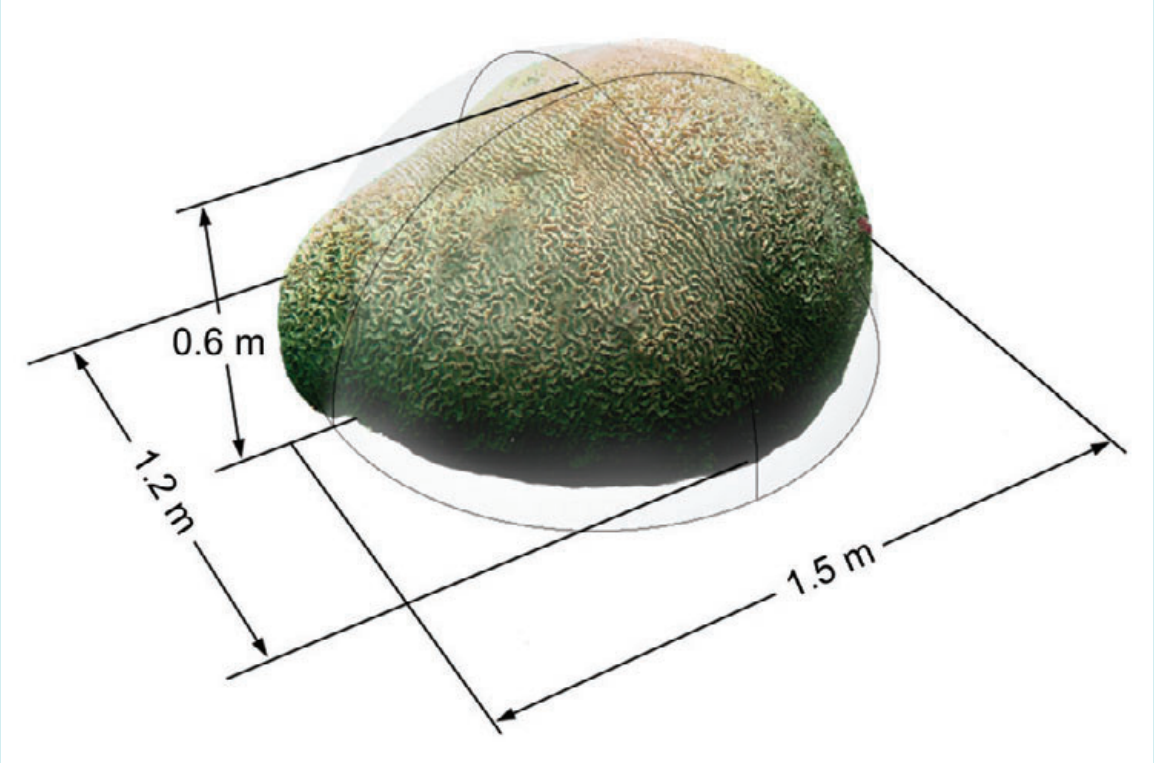
Table A-1. Geometric surrogates and CSA solutions for various colony forms[1]

Morphological form	Geometric surrogate	CSA solution
Massive	Hemisphere	$2\pi r^2$
Free living	Hemisphere	$2\pi r^2$
Branching	Cylinder	$(2\pi rh)$(# branches)
Columnar	Cylinder	$(2\pi rh)$(# columns)
Tabulate	Cylinder	$(2\pi rh)$(# branches)
Foliose	semi-circle; right triangles	$(\pi r^2) / 2 + \frac{1}{2}(bh)$(# plates)

[1] Geometric solutions for various colony forms were described by Alcala and Vogt (1997). Because of the radial growth of coral colonies, planar (overhead view) 2D-SA solutions for all morphological forms were assumed πr^2 (circle), where r=radius, h=height, b=base of triangle, $\pi = 3.14$.

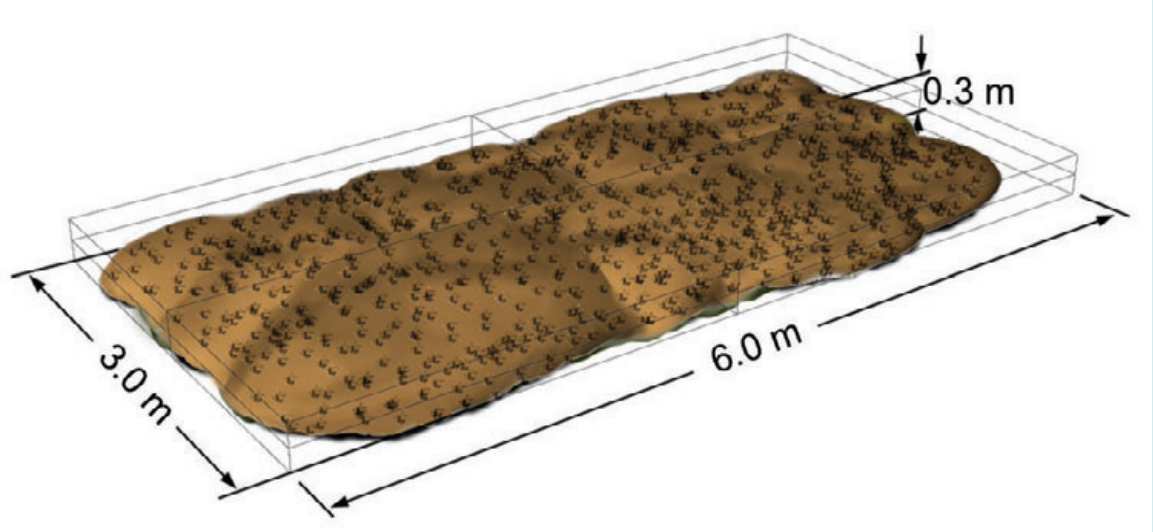
Example 1

A roughly hemispherical colony measures 0.6 m high, 1.5 m diameter and 1.2 m width. Radius is estimated as the average from all three dimensions (r = [0.6 + 0.75 + 0.6] / 3 = 0.65) and surface area calculated as $2\pi r^2$ (hemisphere), or (2 x 3.14 x [0.65^2]) = 2.65 m^2.



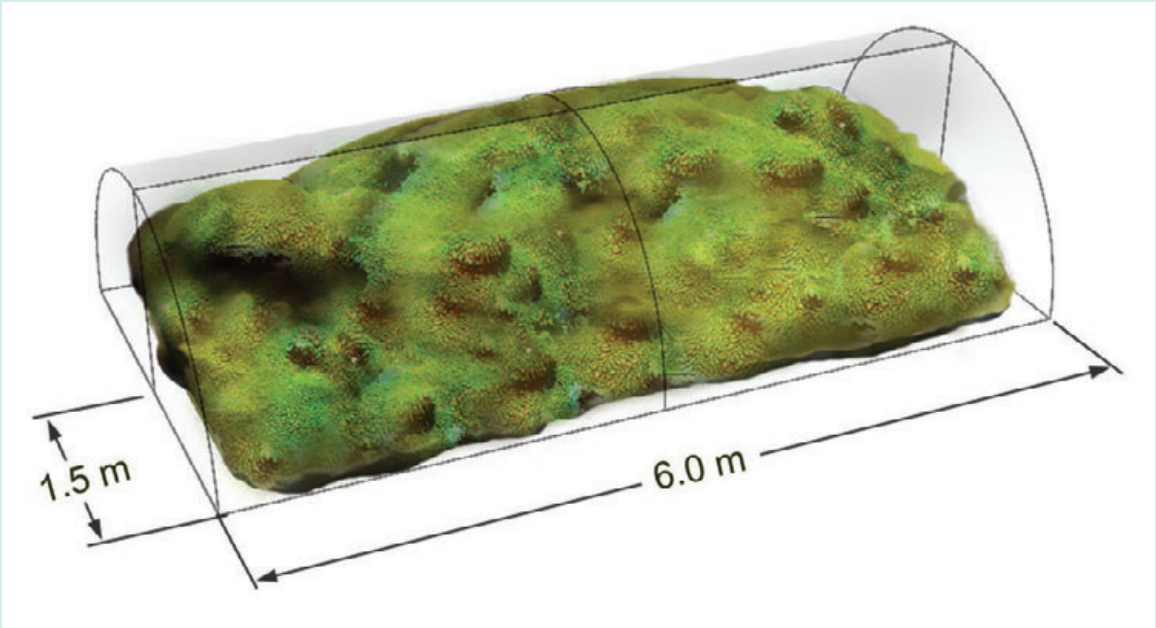
Example 2:

A large, flat colony lies across the transect perimeter. It extends 6 m in diameter, 3 m width and is 30 cm in height. The closest geometric surrogate is a rectangle for which the surface area solution is the sum of surface areas for each of the five sides above the sea floor. SA = (6m x 3m) + 2(6m x 0.3m) + 2(3m x 0.3m) = 23.4m^2.



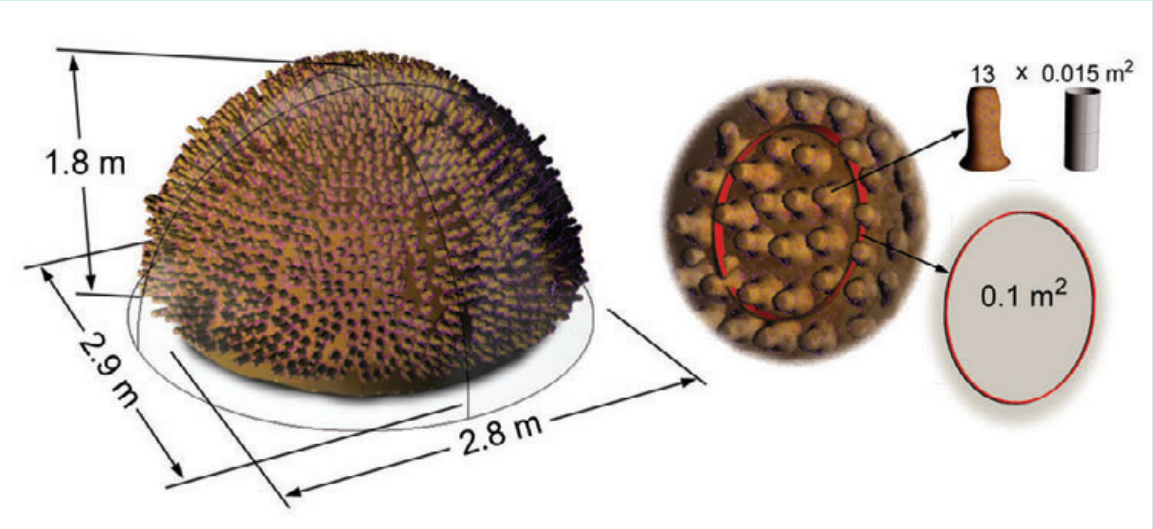
Example 3:

Another large colony, 6m diameter and 3m width, lies across the transect perimeter but, unlike the rectangular colony described above, is shaped more like a half-cylinder lying on its side. The surface area solution for a cylinder is $2\pi r^2 + 2\pi rh$, half of which would represent the above sea floor surface area of the colony ($= \pi r^2 + \pi rh$). In this case, r = 1.5m and h = 6m, so colony SA = (3.14 x [1.5]^2 + 3.14 x 1.5 x 6) = 35.3 m^2.



Example 4:

The surface area of a large hemispherical finger coral was calculated from height, diameter and width dimensions (see first example) as SA = 15.1 m^2. If estimates were needed at the finger scale, the surface area of fingers would be added. The amount to be added can be determined by measurements on a subpopulation of fingers to determine height, radius and density. In this example, height and radius of the fingers are applied to the formula for a cylinder, $2\pi rh$ (the ends of the cylinder are not included), to obtain an average addition of 0.015 m^2 per finger. Density of fingers was found to average 13 per 0.1 m^2 or 130 per m^2. Fingers therefore add 0.015 x 130 = 1.95 m^2 per m^2 coral. The surface area of the colony at the fingers scale is 15.1 + (1.95 x 15.1) = 44.5 m^2, or nearly three times the surface area estimated at the colony scale.



Because very large colonies have such a high influence on TSA for a station, it might be worthwhile to calculate them individually. However, most colonies can be approximated using a common surrogate for the particular species or morphological type. For example, surface areas for Caribbean *Diploria* and *Montastraea* colonies are very closely approximated by a hemispherical surrogate (Courtney et al. 2007). Once surrogates are assigned, geometric solutions for each species can be entered in a spreadsheet and surface areas can be calculated automatically. It is likely that several species will have similar colony morphology (e.g., hemisphere), so only a few solutions might be needed.

The most accurate methods to estimate CSA rely on virtual 3D reconstruction of coral colonies from digital photographs (Bythell et al. 2001; Cocito et al. 2003). Using this approach, multiple photographs are taken from various positions and angles around the coral colony. Images are downloaded into commercial software packages where they are oriented and aligned to form a 3D model of the colony (Figure A-1). Height, diameter, width, surface area and volume of the colony are obtained from the reconstructed model with relatively high accuracy. Because the procedure is time consuming, it is not recommended for field monitoring programs. However, useful conversion factors are generated when a sufficient number of colonies of a particular species or morphological type are analyzed.

Courtney et al. (2007) have applied the photographic method to specimens of Caribbean *Diploria* and *Montastraea* to generate genus-specific conversion factors that allow accurate estimates of CSA from underwater colony measurements (i.e., height, diameter or width). They found the hemisphere to be a very accurate surrogate for these genera, demonstrating high correlation (R^2 = 0.99) when surface area

was calculated ($2\pi r^2$) using an average colony radius derived from all three morphological dimensions. Additional species and morphological types are being examined by the same authors in the same manner. For the more complex morphological forms (e.g., branched colonies, Figure A-1), the reconstructed models provide a means to explore surrogate geometries that can be resolved with the fewest, or easiest, underwater measurements. Although tedious and time consuming, only one comprehensive set of colony reconstructions for a particular species or morphological type is required for most monitoring objectives. As an example, Caribbean programs can now reasonably assign a hemispheric conversion for CSA of *Diploria* and *Monatastraea* colonies without any additional research.

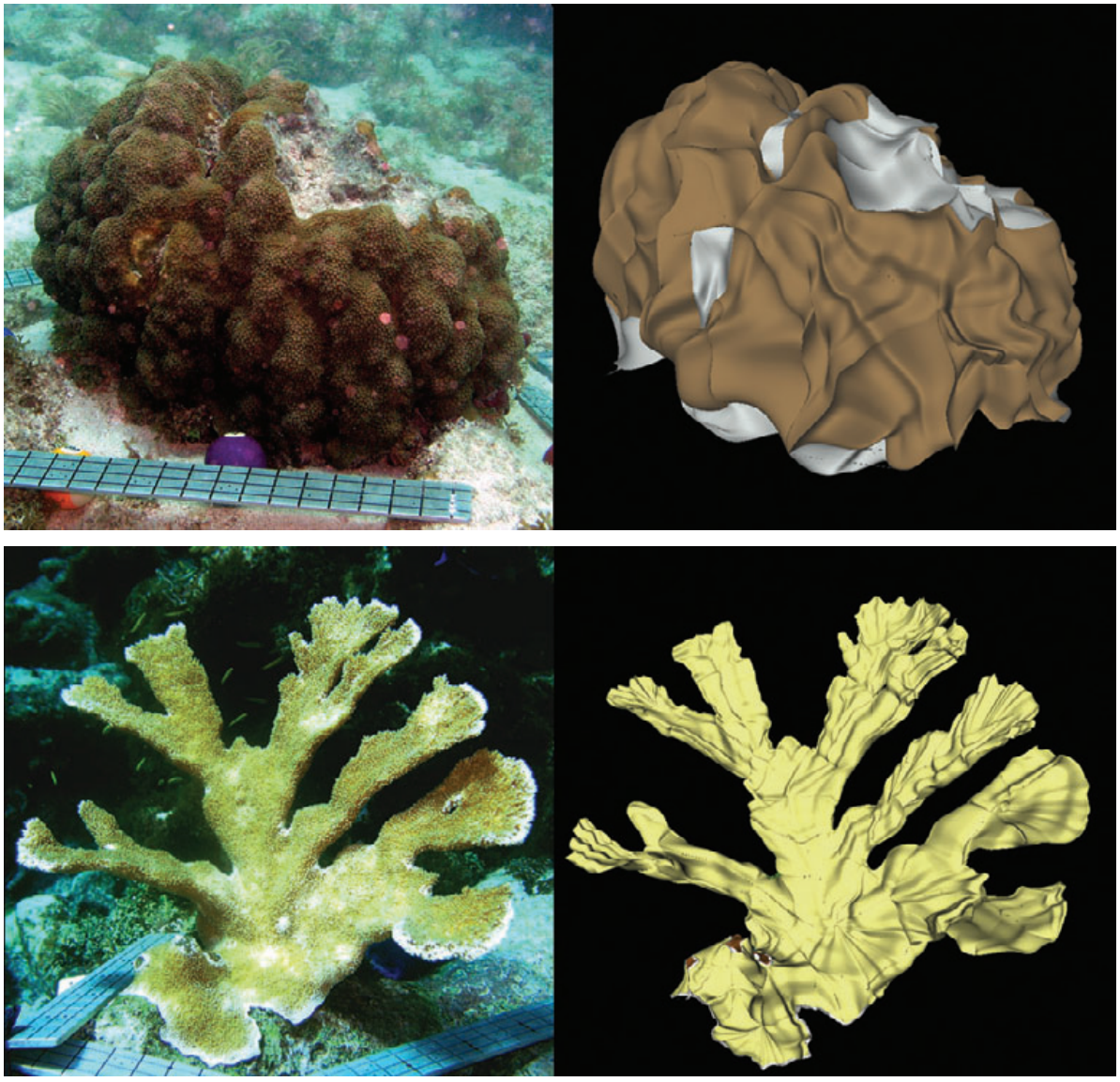


Figure A-1. Photographic methods have been used to measure CSA for a variety of colony shapes (Courtney et al. 2007). Colonies are photographed in the field (left) from multiple positions and angles. Scale bars and reference objects (billiard balls) are placed so that images can be correctly oriented and reconstructed (right) using commercial software. Living and dead surface area (e.g., brown and white in top photographs) can be delineated and quantified. Colony dimensions can be accurately determined from the reconstructed models. Dimensions for the small elkhorn coral shown in the bottom figures are height = 42.2 cm, maximum diameter = 52.5 cm, width = 32.2 cm, CSA = 3445 cm^2, volume = 671 cm^3, and the 2D planar footprint is 954 cm^2 (surface index = 3.6). Source: Lee Courtney, EPA.

Appendix B: Converting Historical Data

Geometric surrogates are used in the Stony Coral RBP to provide 3D rather than traditional 2D estimates of stony coral surface area. Methods are also needed to convert historical indicators, including live coral cover and topographic complexity (rugosity), to 3D units.

Conversion from 2D to 3D. Live coral cover measured in 2D is the indicator most often employed in monitoring programs (Jameson et al. 1998). In fact, there are sufficient live coral cover data to allow meta-analysis and regional documentation of changes in coral condition (Gardner et al. 2003; Kramer 2003). Live coral cover refers to the proportion (%) of live coral relative to the total amount of sea floor surveyed and is a value obtained from either 2D quadrats or 1D linear transects. Even if measured in 1D, live coral cover is intended to represent the 2D planar proportion of coral relative to all other substrata (e.g., sand, vegetation, rock) when viewed from above.

Despite this widespread use and substantial historical data, 2D sampling techniques can lead to deceptive interpretations, including gross underestimation of live coral and its potential to grow and reproduce (Dahl 1973; Alcala and Vogt 1997). Also, Randall (1963) and several others have noted that errors are introduced when comparing fish abundance and diversity to planar 2D rather than topographic surface area. There has always been some uncertainty in measurements of 2D coral cover because of the many different methods employed. In some studies, coral cover is measured as the total area of the quadrat in which any live coral occurs (Gomez and Alcala 1984). In others, the entire colony projection is reported regardless of the proportion of live tissue; in still others, only the live portion of the colony is reported. In the AGRRA program (Lang 2003), live coral cover is estimated by linear transect and partial mortality (percent live coral) is determined from a planar overhead view that does not account for live tissue and bare skeleton on the sides of coral colonies. These inconsistencies in application are often avoided in direct method comparisons (Section 2.3) so that results across methods usually appear to be compatible. In several studies, there is no description of whether live or total coral was reported. Perhaps before recent catastrophic declines, reports of coral cover were presumed to be living coral.

Realistic quantification of coral must incorporate three dimensions (Dahl 1973; Alcala and Vogt 1997), and several means are available to convert historical 2D data. The easiest but least satisfying way is to invoke a generic conversion factor. Odum and Odum (1955) assumed that surfaces of all reef objects were 3X the horizontal area and Risk (1972), perhaps contemplating a highly detailed scale, assumed a 100-fold difference. The best experimental evidence was provided by Alcala and Vogt (1997), who reported that TSA for corals across a 257 m^2 study site averaged 5X the planar surface area. This is a reasonable value at the colony scale, so 2D live coral cover data could be generically multiplied by 5 to obtain 3DLC. It would assume, however, that community composition was similar to that sampled by Alcala and Vogt.

A refinement of this approach is to assign specific conversion factors to particular morphological shapes. Dahl (1973) defined the 3D:2D surface area as the *surface index*, which varies for different morphological types. The surface index (SI) of hemispherical colony, for example, is SI = $2\pi r^2 / \pi r^2 = 2$. Colonies with greater complexity exhibit higher surface indices. For example, Alcala and Vogt (1997) assigned branched colonies SI = 6.88, and free living forms SI = 1.92. Photographic methods noted in Appendix A showed *Diploria* were an average SI = 2.6 (range 1.4 to 4.1), and *Montastraea* were an average SI = 3.0 (range 1.9 to 5.2) (Courtney et al. 2007). Reasonable conversion factors can be obtained or

estimated for most coral morphologies. In contrast to a generic conversion factor, species-specific (or morphology-specific) conversion factors improve the accuracy of conversions across stations and reefs with variable community composition.

It is possible to calculate 2D live coral cover from RBP data. With radial growth, the 2D geometric solution is a circle projected on the planar substrate, or $SA = \pi r^2$, where r is derived from the average of colony diameter and width. Using this approach, there should be reasonable similarities between the RBP and other coral cover methods. As an example, a 2004 RBP survey in the Florida Keys (Fisher et al. 2007b) used a coarse volumetric approach to colony size, which was used to document 3DLC values. However, 2D values could also be calculated from the data using a simple conversion factor. Across the 29 sampling stations, 2D live coral cover was calculated as 6.5%, which is nearly identical to 6.6% found in a 2004 survey of 160 stations using videographic methods (Florida's Coral Reef Monitoring Project; Beaver et al. 2004). Similarly, width and diameter dimensions of colonies in an RBP assessment in southeast Florida were used to calculate 2D live coral cover at 0.7–1.3 percent (Fisher, unpublished), and videographic methods for the same area documented 0.9–1.3 percent live coral cover (SECREMP 2005).

Colony and reef topography. As noted in Appendix A, measures of reef *topography* (topography = surface features) characterize the amount and type of habitat provided by corals and non-coral structures for the greater reef community. Methods that estimate 2D live coral cover are not intended to, and do not, reflect reef topography because coral height and physical complexity are ignored. Yet it is well known that the vertical dimension is essential to flourishing reef communities (Dahl 1973; Luckhurst and Luckhurst 1978; Roberts and Ormond 1987; Ferreira et al. 2001; Scheffers et al. 2003). In fact, some studies have used coral height as an estimator of community habitat (Chou 1984; McCormick 1994; Lang 2003). The term *complexity* is used to describe surface features which, for individual colonies, might be defined as surface area:volume. Like topography, complexity can also be applied at the reef scale, which would be measured as surface area per square meter of sea floor, or 3DTC.

Several studies have applied a *rugosity index* to estimate physical habitat provided by a reef (Porter 1972; Risk 1972; Luckhurst and Luckhurst 1978; Aronson et al. 1994; McCormick 1994; Rogers et al. 1994; Chiappone et al. 2001; Lang 2003; Jokiel et al. 2004). The rugosity index is a 2D measurement applied as a reef-scale indicator of topography and is determined using a chain-transect method that compares the length of a chain draped along the coral colonies of a reef to the length of a taut line across the same linear distance. The procedure is time-consuming, and its application varies depending on how meticulously the line is placed within the nooks and hollows of each rock and colony. In most studies, the chain-transect method is performed independently of a linear transect, but in some cases, the methods have been combined into a single-chain-transect protocol (Rogers et al. 1994). This inconsistency can confound data comparisons, particularly among reefs with high reef topography.

The rugosity index estimates complexity by subsampling a 2D contour of each coral colony (and non-coral substrata) along the draped line. This generates a unitless value that can be used for relative comparisons across stations and reefs. The chain-transect method estimates topography by extrapolation (much as 1D linear transect data are extrapolated to estimate 2D live coral cover). While rugosity accounts for the important vertical dimension, it captures only one horizontal dimension. The chain can lie across any part of a colony but never across the entire colony. It is, therefore, difficult to compare chain-transect data to RBP data for individual colonies. However, comparisons at the reef-scale are possible but have not been performed.

There are clear benefits for migrating to RBP colony-based summations for estimates of reef topography. The estimates are made in 3D rather than in 2D contours or 1D heights; estimates are made on individual colonies, and values can be attributed to different species; estimates focus only on the coral component of reef structure (the component that is managed) and exclude the non-coral components. Measurement of 3D CSA, necessary for calculation of 3DTC, is used in several other indicators and can be applied in value and sustainability models; there are no additional sampling requirements beyond those used to obtain all other RBP indicators (i.e., no additional chain-transect to exclusively collect complexity data).

Appendix C: Evaluating Outcomes of Different Survey Procedures

Different procedures can be used to address varying objectives of a monitoring program. Usually under consideration is the value of spending more time in the field (dive time is a primary concern) for more information or greater accuracy in field measurements. It was proposed (Section 3.4) that three colony dimensions should be measured initially and then analyzed to determine whether all three are actually needed to fulfill the objectives of the program. Such an example is provided here.

In a pilot survey, the Florida Reef Resilience Program (TNC 2006) measured three colony dimensions for each colony encountered in transects from seven different subregions of the Florida Keys reef tract. One objective of the monitoring program was to compare reef TSA among the seven subregions. Hence, the subregions were compared first through analysis of variance (ANOVA) and then Tukey's comparison test. Analyses were performed for seven different calculations of radius on the basis of combinations of one, two and three measurements of colony dimension.

As described for the Stony Coral RBP, the Florida Reef Resilience Program measured colony height (h), maximum diameter (d) and width (w). Estimates of colony radius were derived from height, diameter (d / 2) and width (w / 2). One estimate or an average of combined estimates was applied to a hemispheric surrogate to calculate CSA and reef TSA. Data from all seven combinations were analyzed to identify any differences among regions.

Results from ANOVA (Table C-1) and Tukey's comparison test (Table C-2) indicate that for this monitoring objective, it would be reasonable to measure only two colony dimensions, especially if height was one of the two. It is even possible that a single colony measurement could be used if that single measurement is maximum diameter. Both of these modifications to the monitoring protocol could save valuable dive time. These data and calculations are preliminary and provided only as an example; they are not intended as a recommendation for other monitoring programs.

Table C-1. ANOVA F and p values for different combinations of measurements[1]

# Measurements	Measurements	F-value	p-value
3	h, r_d, r_w	2.71	0.018
2	h, r_d	3.04	0.009
	h, r_w	2.79	0.015
	r_d, r_w	2.38	0.035
1	h	3.94	0.002
	r_d	2.56	0.025
	r_w	2.16	0.054

[1] Different combinations of measurements (height h; radius from diameter r_d; and radius from width r_w) were used to determine colony radius. Radius values were used in a hemispheric model ($2\pi r^2$) to calculate CSA and TSA for the reefs. One-way ANOVA tests revealed significant differences in mean log reef surface areas when $F_{0.05,6,90} > 2.20$.

Table C-2. P-values for Tukey comparisons for different numbers of measurements[1]

Subregions	h, r_d, r_w	h, r_d	h, r_w	r_d, r_w	h	r_d	r_w
Palm Beach v. Lower Keys	**0.029**	**0.022**	**0.027**	**0.050**	**0.026**	**0.040**	0.068
Palm Beach v Middle Keys	**0.046**	**0.039**	**0.049**	0.064	0.051	**0.049**	0.089
Palm Beach v. Upper Keys Transition	0.167	0.136	0.162	0.223	0.156	0.184	0.281
Palm Beach v. Broward County	0.269	0.271	0.295	0.296	0.329	0.278	0.324
Palm Beach v. Northern Transition	0.608	0.802	0.714	0.415	0.993	0.456	0.39
Palm Beach v. Upper Keys	0.326	0.283	0.297	0.442	0.285	0.42	0.479

[1]Tukey comparisons were made between Palm Beach and six other subregions for estimates using 3, 2 and 1 measurements to define radius (r) for calculating the hemispherical surface area. P-values < 0.05 are in **bold**. Additional comparisons among other regions detected no significant differences, and results are not shown.

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