

CORAL-REEF RISK ASSESSMENT FROM SATELLITES TO MOLECULES: A MULTI-SCALE APPROACH TO ENVIRONMENTAL MONITORING AND RISK ASSESSMENT OF CORAL REEFS:

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Coral reefs are in decline worldwide. Resource managers require indicators of the biological condition of reef environments that can relate data acquired through remote-sensing, water-quality and benthic-community monitoring to stress responses in reef organisms. A multi-level approach to monitoring and risk assessment of coral-reef resources is proposed that can provide managers with affordable procedures upon which to base management decisions. Advantages of implementing and further developing the proposed approach can be immediate, as well as long term and economic. Most of the technology and procedures are currently available. The ongoing global alert system, based on satellite data, should be expanded to include both estimates of radiant energy reaching the sea surface and water transparency anomalies (i.e., 'photooxidative hotspots' and 'plume' alerts). Satellite-derived alerts should be verified locally by *in situ* physical and chemical measurements, community assessments and bioindicators of stress that can be affordably incorporated into any management plan. Risk assessments based on *in situ* measurements and low cost bioindicators will provide resource managers with essential information to decide when more costly chemical or molecular procedures are needed to determine local sources of stress, which can then focus management actions to protect or restore reef resources.

Key words: coral reef, risk assessment, bioindicator, water quality

INTRODUCTION

Coral reefs, along with rainforests, are among the most ecologically diverse ecosystems on Earth. Like tropical rainforests, coral reefs have evolved complex, interdependent community structures (Southwood, 1996; Hill and Hill, 2001) despite, or more likely, because of the paucity of nutrient resources in their environments (Hallock, 2001a,b). A critical factor enhancing diversity in both systems is the abundance of symbiotic and other interdependent relationships that enable tight recycling and efficient use of limited nutrient resources. Another important factor is the vertical structure provided by the most prominent members of these communities (i.e., corals and giant trees), which provides an almost infinite variety of microhabitats for associated organisms (e.g., Southwood, 1996; Hill and Hill, 2001).

In the last three decades of the 20th century, scientists witnessed the decline of coral reefs, first locally, then over whole reef tracts and regions. By

the late 1990's, most scientists recognized that coral reefs had declined worldwide (e.g., Dight and Scherl, 1997; Eakin and others, 1997; Risk, 1999; Wilkinson, 1998). Bryant and others (1998) estimated that 58% of the world's coral reefs were threatened by human activities such as sewage and industrial pollution, deforestation, and overfishing. The Bryant report was being released as the 1997-98 ENSO event was triggering mass bleaching of corals unprecedented in its global scale and intensity (Hoegh-Guldberg, 1999). Wilkinson (2000) subsequently estimated that more than a quarter of the world's coral reefs have been lost, more than half of those to 1997-98 mass bleaching events and their consequences.

Factors contributing to the decline of coral reefs include truly local impacts, local impacts that are so widespread as to be essentially global in scale, regional-scale impacts, and human activities that are altering the Earth's atmosphere, climate, and ocean chemistry. The underlying problem is the ever-growing number of humans on the planet, combined with modern technology. One small piece of the goal to develop sustainable cultures is to reduce anthropogenic impacts on invaluable natural resources such as coral reefs.

THREATS TO CORAL REEFS

The evidence for reef decline ranges from the obvious, such as burial under fluvial sediments or mine tailings, to the gradual replacement of reef-building corals by macroalgae and sponges (e.g., Brown, 1988, 1997; Hallock and others, 1993; Harborne and others, 2001). The causes of burial are generally straightforward and the consequences are typically permanent loss of reef-constructive capability (Hatcher and others, 1989). On the other hand, the causes of gradual changes in community structure continue to be controversial and arguments rage as to whether reef decline is "anthropogenic" or "natural" (e.g., Szmant and Forrester, 1996; Lapointe, 1997; Risk, 1999; Szmant, 2002).

Among the most obvious natural events that impact coral reefs (Table 1) are storms that temporarily reduce salinity while increasing sediment and nutrient loads, i.e., episodic alteration of water quality (Larcombe and others, 2001). As a consequence, the emergence of even small freshwater streams naturally produce breaks in fringing reefs (Torres and others, 2001). Hurricanes, typhoons and cyclones can induce massive physical damage to coral colonies, especially branching species (e.g., Done, 1992; Rogers, 1993; Blanchon and Jones, 1997). Those same branching species are typically fast-growing and recruit from fragments, allowing rapid recovery from even extensive physical impact if environmental conditions are otherwise amenable (Connell, 1997). Biotic interactions such as parasitism, bioerosion, overgrowth, grazing and predation are also natural processes in any ecosystem (e.g., Carpenter, 1997; Glynn, 1997; Peters, 1997). However, these interactions can become serious problems when the ecosystem is subjected to anthropogenic stresses such as eutrophication and overfishing (e.g., Hallock and others, 1993; Brown, 1997; Connell, 1997).

Human activities both amplify natural stresses and produce "new" stresses (Table 1) such as human-synthesized chemical pollutants (Peters and others, 1997). Increased flux of nutrients and sediments into reef environments as a consequence of coastal and upland deforestation for agriculture and urbanization (e.g., Harborne and others, 2001) provide examples of anthropogenic amplification of natural stresses, while many of the accompanying chemical pollutants are new stresses. Global climate change can amplify temperature and photic stress, as well as frequency and intensity of tropical and

extratropical storms and ENSO events (e.g., Glynn, 1996; Shick and others, 1996; Hallock, 2001a).

Table 1 Examples of natural stresses, natural stresses that are amplified by human activities, and new anthropogenic stresses that affect coral-reef ecosystems.

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- A. Natural processes that can induce stress (examples):
- Storms (physical damage, sedimentation, reduced salinity, nutrients)
 - Currents (tidal currents or upwelling changing water temperatures)
 - Terrestrial runoff (sedimentation, reduced salinity, nutrients)
 - Oxygen (oxidative stress, insufficient oxygen)
 - Temperature (heat or cold stress)
 - Sunlight (photoinhibition, shading)
 - Biotic interactions (predation, competition, bioerosion, parasitism, disease)
 - Chemical stress (naturally occurring trace metals and hydrocarbons)
- B. Natural stresses that can be amplified by anthropogenic activities (examples):
- Physical damage (ship grounding, anchor damage, dredging, fishing with explosives)
 - Currents (engineered water flow, changes in salinity or water quality)
 - Terrestrial runoff (deforestation, agriculture, urbanization)
 - Temperature (thermal effluent, circulation changes, climate change)
 - Sunlight (ozone depletion/increased UV-B, reduced water transparency)
 - Biotic interactions (invasive species, overfishing, phase shifts)
 - Chemical stresses (pollution by trace metals, hydrocarbons)
- C. New stresses resulting from anthropogenic activities
- Chemical pollutants (e.g., fungicides, herbicides, pesticides, antifouling agents, antibiotics, synthetic hydrocarbons)
 - Synthetic radionuclides
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THE CHALLENGES

Major challenges for coastal management include recognizing and managing activities that result in new stresses and locally amplified natural stresses, while distinguishing damage from stresses that are being amplified by regional and global changes. Although local managers can do little to reduce either natural stresses or regional and global amplifiers, they can work to minimize local amplification and to ensure that local water quality supports ecosystem resilience.

Risk (1999) assailed the marine-science community for failing the world's coral reefs. He argued that neither lack of monitoring strategies nor lack of monitoring effort were factors in reef decline. Jameson and other (2002) described reef monitoring programs as "ubiquitous". Risk's (1999) primary recommendation was to develop "community-based bioindicator" programs designed to link monitored parameters to management efforts to preserve remaining reef resources.

Jackson and others (2000, p. vii) defined an ecological indicator as “a measure, an index of measures, or a model that characterizes an ecosystem or one of its critical components”. They further noted “the primary uses of an indicator are to characterize current status and to track or predict significant change. With a foundation of diagnostic research, an ecological indicator may also be used to identify major ecosystem stress.” McCarty and others (2002) noted that the established definition of biomarker (NAS/NRC, 1989) emphasized measurement rather than establishing cause-effect linkages. McCarty and others’ (2002) definition of bioindicator emphasizes that it must be “quantifiable” and either “correlated or causally linked to effects at one or more of the organism, population, community or ecosystem levels of organization” (p. 161). McCarty and others (2002) concluded that “Success in adoption of the bioindicator concept in environmental risk assessment will depend largely on improved understanding of the fundamentals of ecosystem structure and function” (p. 163).

To address the challenges of risk assessment in reef systems, reef managers need to choose a toolbox of reliable and economically feasible indicators, including physical, chemical, geological, and biological parameters, which can be used both to assess and monitor coral reef ecosystem structure and function, and to diagnose specific problems. Jameson and others (1998, 2001) made a giant step toward this goal by summarizing attributes and metrics currently available for reef-ecosystem assessment. Risk and others (2001) recommended a bi-level approach to assessment: a) utilizing simple, low technology techniques to detect evidence for stress on reefs anywhere in the world, regardless of local technological resources, and b) the use of physical and geochemical measurements to identify sources of stress. The challenge with any individual or suite of bioindicators is to successfully and economically adapt the procedures to reef assessment, and to understand both their sensitivity and specificity. The U.S. Environmental Protection Agency has developed useful guidelines for evaluating ecological indicators that include conceptual relevance to ecosystem function, feasibility of implementation, response variability in time and space, and ability to convey information on ecological condition that is meaningful to resource management (Jackson and others, 2000).

Since corals are the defining members of coral-reef communities, the obvious question is “Why not use corals as bioindicators?” Coral communities have been monitored for more than two decades (Risk, 1999; Gardner and others, 2003) and strategies for monitoring are numerous. Jameson and others (1998) listed 9 categories and approximately 40 references for potential scleractinian coral-based bioindicators. Nevertheless, interpretable parameters remain elusive (Brown, 1988; Risk, 1999). Jameson and others (1998) found no formal interpretive framework for any categories except coral fecundity and recruitment, although Ginsburg and others (2001) and Risk and others (2001) have since recommended coral mortality indices as potentially the most useful direct coral-based procedures. Such an index was developed by Gomez and others (1994) and effectively used in Indonesia to distinguish between polluted and non-polluted reefs by Edinger and others (1998). Other parameters can be problematic, in part because corals are long lived and coral community responses to chronic stress may be relatively slow. For example, coral cover can be deceptive because long-lived adults may remain healthy where juveniles can no longer recruit, and conversely, larvae may recruit into areas that cannot sustain reef growth over years to decades (Kinsey, 1988). Diversity in western Atlantic reefs can remain high despite precipitous decline in coral cover (e.g., Porter and others, 2002). Even coral growth rates can be rapid where corals are dying, as

described by Edinger and others (2000) and explained physiologically by Hallock (2001b).

A strong case can be made for non-coral bioindicators if they can be demonstrated to effectively reflect coral community function and responses to stress. Ideally the bioindicators should respond rapidly and measurably to specific stresses and not require consumptive sampling of corals or other protected species. Consequently, micro- and meiobiota have great potential as indicators in reef ecosystems (Jameson and others, 1998). Techniques for microbiota, which are already developed for public health applications, should be adapted for use in coral reef ecosystems (e.g., Griffin, 2001; Griffin and others, 2001; Lipp and others, 2002). Like other biota, microbiota are characteristically diverse in reef ecosystems, so lack of taxonomic information can be an impediment. Exceptions are the Foraminifera, for which generic-level taxonomy has been well known for decades (e.g., Loblich and Tappan, 1987).

The goals of this paper are: a) to examine some of the tools that are currently available for risk assessment and monitoring of coral reefs, b) to propose a suite of tools, including micro- and meiobiota, that can be tailored to local resources, c) to discuss how these tools can assist local resource managers in recognizing types and sources of stress, thereby providing evidence to support sound and defensible management decisions; and d) to recommend other promising approaches, including some which utilize micro- and meiobiota, that should be further developed and evaluated to strengthen diagnostic capability.

TOOLS TO ADDRESS SPECIFIC QUESTIONS

Are Coral Reefs Stressed?

The first issue to be addressed when assessing a reef ecosystem is the simple question “are the corals stressed?” A number of researchers have suggested that corallivores can provide evidence for stress in corals by either their presence or absence. Outbreaks of Crown-of-Thorns starfish (*Acanthaster planci*) or *Drupella* snails on coral reefs might indicate that corals are stressed and therefore unable to fend off predators (Jameson and others, 1998). On the other hand, Reese (1981, 1994) proposed using obligate corallivorous butterflyfish to indicate coral stress because they decrease in abundance and diversity with decreasing food quality and quantity. Crosby and Reese (1996) proposed a monitoring protocol to census butterflyfish abundance as an early warning of stress, which was applied by Ohman and others (1998) on reefs in Sri Lanka. Major limitations of this approach are the low sensitivity of response, since both corals and butterfly fish are relatively long lived, and that corallivorous butterflyfish are not ubiquitous in reef ecosystems.

Molecular biomarker technology is a promising approach for identifying and quantifying stress responses in corals and other reef organisms (e.g., Downs and others, 2000; Anderson and others, 2001; Morgan and others, 2001). For example, the Molecular Biomarker System (MBS), introduced by Downs and others (2000), uses the diagnostic tools of biomedical research to assay specific parameters of molecular and cellular physiology that govern organismal health and fitness. The MBS quantifies cellular and physiological condition of the organism, identifies types of stress (e.g., heat stress, oxidative stress, xenobiotic stress), and indicates if defenses have been mounted against a particular stress (e.g., pesticide, heavy metal, PAH) (Downs and others, 2000, 2001). While research aimed at understanding how molecular biomarkers in specific reef

organisms are responding to the environment only recently began, measurement of suites of molecular biomarkers on single samples holds promise for determining sources of stress in corals and other organisms (Downs and others, 2000, 2002). Potential limitations include expense and necessity for consumptive sampling, which resource managers may be reluctant to permit without strong justification. In addition, transport of coral samples is restricted by international treaties, which increases the difficulty in applying these tools in developing countries. Nevertheless, molecular biomarkers are potentially the best tools for identifying sources of stress that more affordable techniques detect.

Indicators of coral stress can also be found within the coral surface microlayer (CSM), which includes both coral mucus and microbiotic assemblages. Frias-Lopez and others (2002) observed distinct differences among bacterial assemblages in seawater, and healthy, diseased and dead coral surfaces. Lyons and others (1998) proposed that DNA damage in microorganisms associated with the CSM could be used as an indicator of stress from biologically damaging ultraviolet radiation (UV-B). Technology is available to develop more extensive microbial assays of CSM. An important advantage of this approach is the direct assessment of coral colonies without destructive sampling. Promising methods include molecular screening procedures to characterize microbial assemblages (e.g., Cooney and others, 2002; Frias-Lopez and others, 2002), assessing for specific indicators of stresses such as UV-B damage (Lyons and others, 1998) and toxic metals (Webster and others, 2002), or detecting lysogenic viral activity in response to pollutants (Cochran and others, 1998). McDaniel and others (2001) are developing a prophage induction assay for use in detecting carcinogenic compounds in marine environments that should be adaptable to reef assessment.

Does Terrestrial Input Reach Remote Reef Environments?

The decline of Western Atlantic and Caribbean reefs has been particularly dramatic over the past 30 years (Gartner and others, 2003), in part because reefs have failed to recover from mortality events (Hughes, 1994; Connell, 1997; Ostrander and others, 2000). Oceanic reefs of the Indo-Pacific appear to be much more resilient (Connell, 1997; Wilkinson, 2002). While fringing and other nearshore coral reefs can obviously be directly impacted by terrestrial sediments, pollutants and pathogens (e.g., Paul and others, 1995a,b, 1997, 2000; Szmant and Forrester, 1996; Torres and others, 2001), the potential for anthropogenic pollutants to affect offshore reef systems such as barrier reefs and atolls has remained controversial, as have the causes of decline of offshore reefs (e.g., McClanahan and Muthiga, 1998; McClanahan and others, 2001; Szmant, 2002).

Hallock and others (1993) used remote-sensing images to demonstrate that no Caribbean reef was beyond fluvial influence. They illustrated the Mississippi River plume reaching across the Gulf of Mexico to the Dry Tortugas, the Orinoco River plume extending north across the Caribbean to Hispaniola and Puerto Rico, and the Magdalena plume reaching the Nicaraguan Rise (see also Hallock and Elrod, 1988). Hallock and others (1993) further postulated that anthropogenic amplification of fluvial influence has played a role in the decline in Caribbean reefs. Andrefouet and others (2002) subsequently demonstrated that hurricane-enhanced river plumes from Honduran rivers reached the atolls of

Belize, which McClanahan and Muthiga (1998) had assumed were too remote for anthropogenic influence.

Interpretations from satellite imagery have been validated by *in situ* data. Jaap (1984) noted that low salinity water occasionally reaches the Florida reef tract following Mississippi River flooding. King and others (2001) found that flood plumes from the Burdekin River of northeastern Australia reach the middle reefs of the Great Barrier Reef tract, which are 60 km or more offshore. Furthermore, most river runoff is profoundly influenced by human activities; Walsh and others (1984) estimated that the nutrient load of a river is directly proportional to the human population in the watershed feeding the river. Furnas and Mitchell (2001) stated “Terrestrial runoff is the largest source of nutrients directly influenced by human activities” (p. 38).

Paul and others (1995, 1997, 2000) and Griffin and others (1999) used bacteriophage tracers to demonstrate that injection of partially treated wastewater into highly porous limestone results in contamination of Florida Keys coastal waters. Viral tracers were recovered from marine surface waters in as little as 10 hr after seeding of waste disposal systems. These studies provided direct evidence for both pathogen and nutrient contamination of coastal environments. Szmant and Forrester (1996), however, argued that nutrient pollution from the coastal zone of the Florida Keys is quickly taken up by nearshore biotic communities and does not reach the reef tract.

Several researchers have detected terrestrial pathogens in coral mucus. Frias-Lopez and others (2002) found bacterial sequences indicating terrestrial origin in coral surfaces infected by black-band disease. Lipp and others (2002) noted that coral mucus may accumulate enteric microorganisms and suggested analyzing mucus to assess risk to humans as well as to coral populations. Basic protocols for assessing sewage influence on reefs can be adapted from techniques used to assess human health risks in coastal waters (e.g., Griffin, 2001; Griffin and others, 2001).

Risk and others (2001) recommended using a geochemical approach to detecting sewage input to reef ecosystems. Changes in ratios of the stable isotopes of nitrogen ($^{15}\text{N}/^{14}\text{N}$) in the tissues of reef organisms can provide evidence for sewage influence (Risk and others, 1993; Sammarco and others, 1999; Heikoop and others, 2000; Yamamuro and others, 2003). However, as Hallock and others (1993) noted, sewage is only one of many sources of nutrients to a reef that can be increased by human activities.

Hallock and others (1993) also postulated that dust, blown by windstorms originating from areas of Africa that have experienced desertification, could also be influencing Caribbean reefs by further contributing to nutrient loading. Satellite imagery has dramatically demonstrated the transport of African dust (e.g., Shinn and others, 2000; Hayes and others, 2001; Griffin and others, 2002). Geochemical analyses of Pleistocene paleosols of islands in the western Atlantic have revealed that Saharan dust is the principal source of clay minerals in otherwise carbonate provinces (Muhs and others, 1990; Herwitz and others, 1996). However, Hayes and others (2001) have shown that climate change in the past 30 years has significantly increased dust transport to the Caribbean basin. The dust is rich in iron, a key growth-limiting micronutrient (e.g., Duce and Tindale, 1991; Martin and others, 1991; Coale and others, 1996). The dust also transports pathogens including *Aspergerillus sydowii*, a terrestrial fungus that infests seafans (*Gorgonia ventalina*) (Smith and others, 1996, 1998; Shinn and others, 2000). Comprehensive studies of microbes both in dust samples and in the reef environment (Smith and others, 1996, 1998; Griffin and others, 2002) have provided the key to interpreting this mechanism of influence.

Hayes and others (2001) also suggested that biologically-available iron in the dust promotes outbreaks of otherwise iron-limited marine pathogens and opportunistic toxic algae.

Does Water Quality Support Reef Growth?

Coral reefs worldwide have exhibited profound changes in benthic community structure and function over the past 30 years. In many areas, coral or mixed coral-algal communities have been replaced by fleshy and filamentous algae (Dustan and Halas, 1987; Hughes, 1994; Bell and Elmerti, 1995; LaPointe and others, 2002; many others). Phytoplankton concentrations in the vicinity of the Low Isles in the Great Barrier Reef lagoon have significantly increased over the past 65 years, while plankton assemblages have changed from mixed centric and pennate diatoms to predominantly pennate diatoms and small flagellates, indicating community response to anthropogenic nutrients (Bell and Elmerti, 1995).

As nutrient supplies and plankton populations increase, heterotrophic macroinvertebrates, including sponges, tunicates, and barnacles increase in abundance (e.g., Rose and Risk, 1975; Tomascik and Sander, 1987; Kinsey, 1988). Percent cover or abundances of selected heterotrophic macroinvertebrates are included in several reef-monitoring protocols (Hodgson, 1999; McManus and others, 1997; Porter and others, 2002). Jameson and others (2001) considered assessment of both sessile epibenthos and benthic macroinvertebrates to be essential for creating an effective Index of Biological Integrity for coral reefs. Studies in both the Caribbean and Pacific (e.g., Rose and Risk, 1985; Sammarco and Risk, 1990; Holmes, 1997) have demonstrated that internal bioerosion of reef rubble and live coral colonies increases with increasing nutrient flux. Risk and others (2001) proposed a bioerosion index based on work by Holmes (1997) and Holmes and others (2000) on bioerosion in coral rubble.

Erdmann and Caldwell (1997) recommended using stomatopod crustaceans (mantis shrimp) as bioindicators of pollution stress because they are sensitive to a variety of pollutants, they are abundant and ubiquitous on reefs worldwide, they are easily sampled, and non-specialists can readily be taught to identify them. Jackson and others (1989) and Steger and Caldwell (1993) found that these invertebrates are highly sensitive to oil pollution. Risk and others (2001) concluded that these crustaceans are among the most promising first-level indicators of pollution stress.

The effects of environmental and anthropogenic stresses on amphipod crustaceans have been studied since the 1970's, revealing that these organisms are both behaviorally and physiologically sensitive to a variety of pollutants and toxicants (Thomas, 1993). The U.S. Environmental Protection Agency already uses these crustaceans in marine biomonitoring protocols (e.g., Engle and Summers, 1999). The exceptional sensitivity exhibited by these invertebrates enhances their potential for use as indicators of early stages of stress, permitting time for mitigation or other management decisions. Realization of the potential of this group requires either taxonomic inventories in the regions where they are to be used (Thomas, 1993) or development of more economical molecular-genetic screening protocols (e.g., Rohwer and others, 2001, 2002; Peterson and others, 2002).

MacKenzie (1999) suggested that parasites in marine organisms are a promising indicator of the effects of pollution. Since the free-living transmission stages of such internal parasites as helminthes are highly sensitive to

environmental conditions, the incidence of helminth endoparasites tends to decrease as water quality declines. On the other hand, marine ectoparasites tend to be highly tolerant of pollutants, thereby tending to increase in abundance as water quality declines. Jameson and others (1998) noted that timed, visual censuses of fish ectoparasites within areas of concern have been proposed as an assessment tool, though no formal interpretive framework has been developed. Diamant and others (1999) combined parasitological investigations with physiological assessments of rabbitfish in the Red Sea and the Mediterranean, to further demonstrate the potential for using fish parasites as bioindicators.

There is little argument that benthic community structure has changed throughout the Caribbean region over the last several decades. Hallock and others (1993), Lapointe (1997), and Lapointe and others (2002) postulated that anthropogenic amplification of nutrient flux has played a major role in the phase shift to algal dominance of former coral reefs. Numerous other researchers, including Hughes (1994), Miller and others (1999), McClanahan and others (2001) and Szmant (2002) have argued that reduction in herbivory, associated with both overfishing and the 1983 mass dieoff of *Diadema* urchins (Lessios and others, 1984; Lessios, 1988), has enabled macroalgae to take over benthic habitat when corals are killed by other mechanisms. Most monitoring and assessment programs reviewed by Jameson and others (1998, 2001) included sessile epibenthos, macrophytes, benthic macroinvertebrates, and fish in their procedures, most of which Jameson and others (2001) considered to be “non-diagnostic”.

Cockey and others (1996) postulated that foraminiferal assemblages in sediments should be useful in determining if changes in water quality have played a role in the decline of Florida Keys reefs. Foraminiferal assemblages include both mixotrophic taxa that are dependent upon algal endosymbionts for enhanced growth and calcification, and purely heterotrophic taxa that are herbivorous and detritivorous. These investigators argued that, if the decline in coral cover in the Florida Keys was the consequence of physical damage, decline in grazing following the *Diadema* dieoff and coral-specific diseases, the foraminiferal assemblages should show little or no change. Cockey and others (1996) demonstrated that foraminiferal assemblages in the sediments profoundly changed in Florida reef-tract sediments, from domination by algal-symbiont bearing larger taxa in the 1960s to domination by the smaller heterotrophic taxa in the 1990s. This shift in the foraminiferal assemblage is ecologically consistent with the decline in reef-building corals and their replacement by benthic algae and sponges over the same period. Cockey and others (1996) concluded that increasing nutrient flux to the reef tract was likely impacting community structure.

Based on data sets collected by Cockey and others (1996) in the Florida Keys, Donnelly (1993) in Puerto Rico, and Hammond and others (2000) in Antigua, Hallock and others (2003) proposed and evaluated a single-metric index based on foraminiferal assemblages from sediment samples using guidelines proposed by Jackson and others (2000). The resulting Foraminifera in Reef Assessment and Monitoring (FORAM) Index provides a tool for assessing whether water quality in the environment is adequate to support mixotrophy (i.e., algal symbiosis) as the dominant nutritional mode in an ecosystem. The FORAM Index can also be used for risk assessment of proposed restoration projects, as well as assessment of management outcomes. A key advantage of this index is that it is based on functional groups of Foraminifera, which are recognizable worldwide, thus minimizing the need for local taxonomic specialists.

Lidz and others (1985) and Hallock (1988) suggested that the composition of reef sediments as a whole also reflects benthic community structure and function. The basic premise is that mixotrophic organisms, particularly those that calcify, are more successful competitors in low nutrient environments. As a consequence, reef sediments can be dominated by readily identifiable skeletal components, particularly larger foraminifers (McKee and others, 1956). As nutrient supplies increase, skeletal fragments of calcareous algae and herbivorous gastropods increase in abundance, as do bioeroded coral fragments. When nutrient supplies increase even further, such that fleshy algae, boring sponges and bivalves become increasingly prevalent, recognizable coral and larger foraminiferous constituents become increasingly rare, while unrecognizable fragments, along with the shells and fragments of heterotrophic foraminifers and bivalves, become dominant. Because carbonate sediment composition reflects community structure and function, this technique also can be applied as an assessment tool worldwide. Lidz and Hallock (2000) used this model to interpret changes in sediment constituents along the Florida reef tract. Further evaluation is needed to produce a standardized protocol.

IDENTIFYING SPECIFIC STRESSES

Specific stresses that human activities are amplifying include extremes of temperature, irradiance, salinity, oxygen, nutrients, trace elements, and sedimentation. New anthropogenic stresses include synthetic hydrocarbons such as pesticides and herbicides, artificial radionuclides, antibiotics and others. In the continuum between natural, amplified-natural, and new stresses are the nearly infinite variety of hydrocarbons ranging from crude oil to combustion products. Unfortunately, stresses are seldom singular. The most common impact on water quality, increased nutrient flux associated with agricultural runoff and sewage, is typically associated with increased sediment flux, along with a brew of pesticides, herbicides, fungicides, antibiotics, and hydrocarbons.

Specific effects of most chemicals on marine organisms are poorly known. Richmond (1993) demonstrated that selected organophosphates can inhibit settlement of coral larvae and McKenna and others (2001) developed procedures for utilizing this information in assessment. However, since corals generally spawn annually, availability of larvae restricts the widespread applicability of such methods.

Several efforts to detect gene expression to specific pollutants have been reported. Ammar and others (2000) assayed expression of the metabolic enzymes fructose-1,6-bisphosphatase and succinate-dehydrogenase in an azooxanthellate octocoral by isolating the cDNA for both. The levels of expression of these enzymes were correlated with stress from sedimentation. Morgan and others (2001) presented a method for utilizing gene probes as toxicant-specific biomarkers of coral stress responses. Morgan and Snell (2002) used this technology to determine exposure of the coral *Acropora cervicornis* to dibrom, a pesticide used in mosquito control in south Florida.

The MBS (Downs and other, 2000) assays for general indicators of cell integrity including glutathione and ubiquitin, molecular chaperones that indicate the integrity of enzyme pathways, small heat-shock proteins, and antioxidant enzymes indicative of oxidative stress. Examining an array of stress responses provides the potential to differentiate between a global stressor (e.g., high sea-surface temperature) and a local stressor (e.g., fungicide, an organometalloid, endosulfan) (Downs and others, 2000; 2001; 2002). The MBS also is able to

differentiate between bleaching due to oxidative stress and bleaching due to other stresses because cells respond differently (Downs and others, 2002).

Peters and others (1997) examined three classes of pollutants on reefs, heavy metals, petroleum, and synthetic organics including herbicides and pesticides. They concluded that heavy metal pollution has caused physiological stress, reduced reproductive success and mortality in fish and invertebrates, though risk assessment criteria are needed. Haynes and others (2002) reported that Great Barrier Reef management is focusing on heavy metals, persistent chlorohydrocarbons, PCBs and petroleum-related compounds originating from agricultural runoff. Owen and others (2002) demonstrated experimentally that measured concentrations of the antifouling *s*-triazine herbicide in coastal waters of Bermuda, Florida Keys and St. Croix in the U.S. Virgin Islands were sometimes sufficiently high to suppress photosynthesis by coral zooxanthellae. Scott and others (2002) found evidence of pesticide runoff, particularly of endosulfan, in south Florida estuaries. The most widely applied biocriterion for pollution is the global "Mussel Watch" program, which monitors incorporation by bivalves of trace metals, artificial radionuclides, petroleum compounds and halogenated hydrocarbons (e.g., Goldberg and others, 1978; Hungspreugs and Yuangthong, 1984).

The antifouling agent, tributyltin, induces imposex, the imposition of male sexual organs on female, in marine snails. Ellis and Pattisina (1990) recognized the potential of this phenomenon as a bioindicator, proposing the "relative penis size index" (RPS Index) to measure the proportional responses of gastropod imposex to tributyltin exposure. Foale (1993) evaluated the index, which is now widely applied and intercalibrated among laboratories (e.g., Davies and Minchin, 2002) worldwide. In tropical and subtropical waters, the RPS Index has been used in monitoring (e.g., Bech, 2002a,b), in assessing the effectiveness of regulations (e.g., Gibson and Wilson, 2003), and in risk assessment using transplant experiments (e.g., Bech and others, 2002).

Yanko and others (1994, 1998), Samir (2000), and Samir and El-Din (2001) studied the responses of benthic foraminifers to a variety of pollutants in the eastern Mediterranean, including hydrocarbons and heavy metals. These researchers detected not only assemblage responses, but also noted that shells in polluted environments were commonly stunted, pyritized or deformed. The FORAM Index (Hallock and others, 2003) is based upon relative abundances of three functional groups of foraminifers: larger taxa that host algal symbionts, smaller taxa that bloom when nutrification increases food resources, and stress-tolerant taxa that dominate when water quality is seriously impacted by eutrophication, heavy metals, or other pollutants.

Jackson and others (1989) and Steger and Caldwell (1993) studied the effects of a major oil spill on coral reef and associated mangrove communities in Panama, and reported that mantis shrimp were particularly sensitive to oil pollution. Gesteira and Dauvin (2000) concluded that amphipods are useful indicators of the impact of oil spills on soft-bottom communities, noting that polychaete/amphipod ratios can indicate impact or recovery of the soft-bottom macrobenthos. Similarly, assessment of nematod/copepod ratios (e.g., Ansari and Ingole, 2002; Montagna and others, 2002) can indicate impact or recovery of meiobenthos following a spill.

Solar radiation reaching the sea surface and the seafloor are also natural parameters that are being anthropogenically modified. A major factor in coral bleaching is photooxidative stress; elevated water temperatures lower the threshold for photoinhibitory damage to Photosystem II in coral zooxanthellae (Lesser, 1996, 1997; Warner and others, 1999; Downs and others, 2002; others).

Stratospheric ozone depletion has progressed to the degree that biologically damaging UV-B intensities previously experienced only near the summer solstice on mid-latitude reefs are now experienced from April through August (Shick and others, 1996). Thus, at the same time that global warming is producing longer, warmer summers, particularly during ENSO years, UV-B reaching the sea surface has also increased. Furthermore, mass bleaching events in 1983, 1987-88 and 1991 (Hoegh-Guldberg, 1999) followed major volcanic eruptions that depleted stratospheric ozone globally by as much as 4% (Randell and others, 1995; Roscoe, 2001). UV-B penetrates highly transparent oceanic waters sufficiently to have both direct and indirect effects. Gleason and Wellington (1995) found UV-B damage in corals and coral larvae. Lyons and others (1998) documented UV-B induced DNA damage in microorganisms found in the coral-surface microlayer.

Bleaching in both zooxanthellate corals (Fitt and Warner, 1995) and reef-dwelling foraminifers (Hallock and others, 1995; Williams, 2002) has been shown experimentally to be accelerated by exposure to shorter, higher energy wavelengths of solar radiation, including the violet to blue (400-500nm), UV-A (320-400 nm), and UV-B (280-320 nm). Anderson and others (2001) showed that warm, calm, stratified sea surface waters promotes photobleaching of chromophoric dissolved organic matter (CDOM), further enhancing penetration of shorter wavelengths of solar radiation. Williams (2002) documented that bleaching intensity and prevalence in the foraminifer *Amphistegina gibbosa* in the Florida Keys peaks with the summer solstice, rather than with the late summer thermal maximum. She also demonstrated, using SEAWIFS satellite imagery, that prevalence and intensity of bleaching in *A. gibbosa* was inversely related to CDOM concentrations in overlying waters. Toler (2002) found that shell damage by predation or infestation in these foraminifers was most prevalent at intermediate bleaching incidences, indicating that the stress that induces partial bleaching can reduce the foraminifers' ability to avoid predation and infestation.

A BASIC TOOLBOX

Tools that are currently available to assess and monitor stresses and impacts on coral reefs are numerous (e.g., Jameson and others 1998, 2001) and range from satellite imagery to microbial and molecular assays. Potential tools based on state-of-the-art technology are also plentiful. This section examines several basic classes of tools and suggests ways to expand their usefulness.

Remote Sensing

Satellite-derived sea surface temperatures (SSTs) are currently being used to predict coral mass-bleaching events (Gleason and Strong, 1995; Carriquiry and others, 2001; Aronson and others, 2002). Because coral bleaching events have been noted in areas where the SST exceeds the climatological maximum for that region by 1° C or more, charts that illustrate anomalies are updated at least weekly (NOAA/NESDIS Coral Reef Team, 2003).

Satellite technology is in place that could support similar monitoring of total ultraviolet and visible solar radiation reaching the sea surface and the sea floor at shallow-reef depths. For example, Mumby and others (2001) concluded that dense cloud cover in the Society Islands during the 1998 ENSO event

limited bleaching damage despite high water temperatures. A photic-stress index could be developed that is based on UV-B, UV-A, and visible (400-700 nm) radiation reaching the sea surface, which can be estimated based on stratospheric ozone, cloud cover, and aerosols. Algorithms that remove the effects of clouds and aerosols and that distinguish the effects of water turbidity, chlorophyll pigments and CDOM (e.g., Lee and others, 1998, 2002; D'Sa and others, 2002; Lee and Carder, 2002) can provide estimates of irradiance reaching depth (e.g., 10-20 m).

Irradiance estimates, combined with temperature, would allow prediction of the total photooxidative stress to the benthic community. This same technology would permit tracking ocean color or turbidity plumes that indicate major terrestrial runoff events (e.g., Andrefouet and others, 2002), offbank transport of sediments, dense water plumes (Acker and others, 2002), or other anomalous events (e.g., Hu and others, submitted), which could have detrimental impacts on reefs. Ongoing "coral reef hotspot" monitoring now can direct researchers to where corals can be expected to bleach. Similarly, more comprehensive monitoring that included solar irradiance and ocean color would enable researchers to more effectively determine the interplay between temperature, sunlight and water transparency in their effects on reef communities.

In Situ Monitoring of Physical and Chemical Parameters

A basic, low cost, low technology monitoring program based at marine laboratories around the Caribbean regularly records selected physical and biological parameters and monitors for change, with the goal of distinguishing natural from anthropogenic disturbance (e.g., Alcolado and others, 2001). A more comprehensive program of reef-based monitoring buoys and regular *in situ* sampling of physical and chemical parameters including temperature, conductivity, wind speed, turbidity, solar radiation, nutrients (ammonium, nitrate, orthophosphate, total N and total P), and chlorophyll concentrations, as has been initiated for the Florida reef tract (e.g., Ogden and others, 1994; Szmant and Forrester, 1996; Boyer and Jones, 2001). Data from this network can provide measurements needed to calibrate satellite-derived data and establish local stress alerts (e.g., Hendee and others, 2001). Underwater irradiance, turbidity, and chemical measurements tend to be compromised by fouling of the sensors (Nittis and others, 2001), which can take place within hours to days of emplacement. Technology to protect sensors has been recently developed (Williams, 2002). Nevertheless, how to relate the physical and chemical environment to individual, population and community responses remains a major question (Risk, 1999).

Community Structure and Function

Taxonomic groups fundamental to reef condition are algae, corals and fish. A variety of survey protocols have been developed, based on objectives and available resources (reviewed by Jameson and others, 1998, 2001). There are volunteer-focused programs such as ReefBase Aquanaut (McManus and others, 1997) and ReefCheck (Hodgson, 1999). The Florida Keys National Marine Sanctuary Coral Reef Monitoring Project monitors coral cover and other parameters at fixed sites (e.g., Porter and others, 2002). Aronson and others

(1994) proposed a quick and inexpensive procedure for reef resource assessment. The Atlantic and Gulf Rapid Reef Assessment AGRRA protocol (Ginsburg and others, 2000) has been adapted Caribbean-wide in recent years, and includes coral cover, coral mortality and algal and fish biomass. Miller and others (2002) developed a procedure to assess how benthic habitats respond to protection from fishing. Hudson and Goodwin (2001) provided protocols for assessment following ship groundings on coral reefs. Thus, as Risk (1999) argued, the issue is not how to monitor what is happening to reefs, but rather how to determine causes of change.

Whether a reef community has the resilience to recover from impact may not be reflected by coral measures such as percent cover, diversity or growth rates, but rather by bioerosion rates and whether local water quality supports nutrition based on algal symbiosis. Thus, in areas where coral communities are largely intact, assessments based on stomatopod abundance and diversity (Erdmann and Caldwell, 1997), the FORAM Index (Hallock and others, 2003), sediment constituents (Lidz and Hallock, 2000) or bioerosion (Risk and others, 2001) can indicate if the system is likely to recover if a mass mortality event occurs. Sediments dominated by larger foraminifers and easily recognizable coral constituents indicate that the environment supports algal symbiosis as the dominant modes of nutrition and carbonate production. On the other hand, sediments dominated by algae and molluscan fragments, or difficult to recognize bioeroded and coated grains, indicate that the environment favors autotrophy and bioerosion as dominant processes. In the latter case, when corals are lost to disease or a mortality event, the reef community has a low probability for recovery, particularly if herbivores are scarce as a result of disease or chronic overfishing.

In the western Atlantic and Caribbean, coral cover has been lost to hurricanes (Hughes, 1994), disease (Gladfelter, 1982; Goreau and others, 1998; Ostrander and others, 2000), sedimentation, (e.g., Nemeth and Nowlis, 2001, Torres and others, 2001), oil spills (Jackson and others, 1989), ship groundings (e.g., Gittings and others, 1988), and other events or chronic stresses. Coral cover, which was often in excess of 50% in the 1960s and 1970s, is now commonly below 10% (e.g., Dustan and Halas, 1987; Hughes, 1994; Ostrander and others, 2000; Gardner and others, 2003). In such cases, the FORAM Index and sediment constituents should be useful in determining if water quality will still support reef growth, even though regional or global stresses (e.g., bleaching, dust-borne diseases, chronic photooxidative stress increasing susceptibility to disease, bleaching) have reduced coral cover. Accompanying fish and urchin surveys can provide evidence as to whether reduced grazing has contributed to algal overgrowth of the substrate (e.g., Miller and others, 1999).

Local versus Global Stress: A Low Cost Bioindicator

Where decline has occurred as visually indicated by degrading reef structure or historic records, or is occurring as indicated by coral mortality indices (Gomez and others, 1994; Edinger and others, 2000) or a disease outbreak (e.g., Santavy and others, 2001), the challenge is to determine whether causes are local or global. A promising low cost tool is the response of *Amphistegina* populations (Hallock-Muller, 1996; Hallock, 2000; Williams, 2002).

Amphistegina are commonly the most abundant and one of the most widely distributed, symbiont-bearing foraminifers on reefs and carbonate shelves

worldwide (Langer and Hottinger, 2000). Two closely related species, *A. gibbosa* in the western Atlantic/Caribbean and *A. lessonii* in the Indo-Pacific, are known to be sensitive to water quality and to bleach in response to excess radiant energy (Hallock and others, 1986, 1995; Talge, 2002; Williams, 2002). Low-level bleaching stress increases the susceptibility of these protists to predation and infestation (Hallock and Talge, 1994; Toler and Hallock, 1998; Toler, 2002). Richardson and others (2001) recommended assessing bleaching and disease in coral populations in late spring, when disease activity starts to increase, and late summer, when bleaching and disease tend to be highest. These are also the optimum times to assess *Amphistegina* populations to determine if water quality is favorable for mixotrophic nutrition, based on abundance (Hallock-Muller, 1996; Hallock, 2000), or if photic stress is likely to induce susceptibility to disease or bleaching in mixotrophs, based on size distribution, bleaching prevalence and intensity (Williams, 2002), or breakage prevalence (Toler, 2002).

Responses of *Amphistegina* populations can provide managers worldwide with a relatively quick, low cost method to make informed decisions about when to employ more expensive technology, such as the $\delta^{15}\text{N}$ assessment (Risk and others, 2001), the MBS (Downs and others, 2000, 2002) or other specific detection protocols to determine sources of stress, including DNA damage (Anderson and others, 2001), heat stress (Weins and others, 2000), or pesticides (e.g., Morgan and Snell, 2002). A key advantage of regular monitoring of *Amphistegina* is their visually quantifiable and distinct responses to water quality and chronic photooxidative stress.

Other Potential Tools in the Micro- and Meiobiota

Other micro- and meiobiota provide the potential for development of additional tools for reef monitoring and risk assessment. In particular, changes in the microbiota in the CSM is the logical place to look for clues to disease susceptibility. Richardson and others (2001), Ritchie and others (2001), and Santavy and others (2001) have proposed strategies and methodologies for assessing and investigating disease incidences and outbreaks. As noted previously, several researchers have detected terrestrial pathogens in corals or their mucus (Lipp and others, 2002; Patterson and others, 2002). Promising methods include molecular screening procedures to characterize microbial assemblages (e.g., Cooney and others, 2002; Frias-Lopez and others, 2002), assessing for specific indicators of stress (e.g., Lyons and others, 1998; Webster and others, 2002), or detecting lysogenic viral activity in response to pollutants (Cochran and others, 1998; McDaniel and others, 2001). The challenge, as with any bioindicator, is to successfully and economically adapt the procedures to reef research and to understand both their sensitivity and specificity.

Any procedure that has been developed for use in monitoring or risk assessment in other marine environments has the potential to be adapted for use in marine environments. Methods based on taxonomy of micro- or meiobiotic assemblages must confront problems of diverse, poorly known assemblages. However, just as technology is available to assay coral surface microlayer assemblages, similarly assays of sediment microbiota and meiobiota can be used to test for change in assemblages or for specific gene expression in response to pollutants or other stresses.

Amphistegina spp. have other potential applications in risk assessment: laboratory bioassays of water quality and field transplant experiments to determine how they respond to sites or environmental conditions of concern.

These foraminifers are large enough to be individually assessed and manipulated, yet they are easily and inexpensively maintained in culture (Hallock, 2000). Research aimed at developing protocols to determine gene expression associated with specific stress responses are critical to expanding application of these protists in reef research.

RECOMMENDATIONS

Potential tools for coral reef monitoring and risk assessment are numerous, though relatively few have been formally evaluated for conceptual relevance, feasibility of implementation, response variability, sensitivity and limitations (see Jackson and others, 2000). The following approach, which includes currently available procedures and technology, as well as recommendations for further development and evaluation, can provide reef managers with solid information upon which to make management decisions (Figure 1):

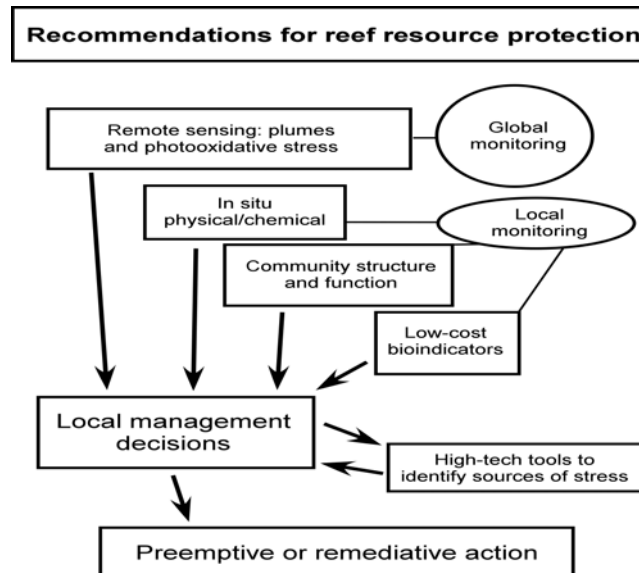


Figure 1 Proposed strategy for reef resource protection

a) Remote sensing: Currently, global temperature “hot spot” anomalies are posted on a website at least once per week (NOAA/NESDIS Coral Reef Team, 2003). The effort has successfully predicted episodes of mass bleaching in corals (e.g., Carriquiry and others, 2001). Expanding the products of this website to include estimates of radiant energy (visible and UV) reaching the sea surface, as well as seawater absorption coefficients (e.g., Lee and others, 1998; D’Sa and others, 2002; Lee and Carder, 2002), would enable identification of both “photooxidative hotspots” and anomalous absorption or reflection features such as the “blackwater” event off Florida in 2002 (Hu and others, 2002) or river flood plumes (Andrefouet and others, 2002). b) *In situ* physical and chemical data: *In*

situ monitoring should be carried out at whatever level is economically feasible, i.e., ranging from daily measurements by a marine lab staff (e.g., Alcolado and others, 2001) to continuous or hourly data streams from well maintained moored buoys that can support local alert systems (e.g., Hendee and others, 2001). Including UV-A and UV-B sensors, as well as collection of temperature and all irradiance parameters at standard depths (e.g., 10 m and 20 m), would substantially enhance modeling of photooxidative stress at the sea floor.

c) Assessment of community structure and function: A variety of procedures have been developed to assess coral and fish communities, ranging from the volunteer-focused efforts (e.g., McManus and others, 1997; Hodgson, 1999), to protocols that require professional field and laboratory teams (e.g., Ginsburg and others, 2000; Porter and others, 2001; Miller and others, 2002). Sampling of reef rubble and sediments, either during other monitoring efforts or as an independent effort, for assessment of bioerosion (Risk and others, 2001), the FORAM Index (Hallock and others, 2003), or sediment constituents (Lidz and Hallock, 2000), can provide evidence on benthic community function relative to algal symbiosis, calcium carbonate production and bioerosion. These assessments should be carried out every three to five years to assess resources, identify reefs in decline, and focus protection efforts.

d) Low-cost bioindicators of stress: Stomatopod diversity and abundance (Erdmann and Caldwell, 1997; Risk and others, 2001) can be monitored annually for pollution stress. Bleaching, disease and mortality in coral (e.g., Santavy and others, 2001; Risk and others, 2002) and *Amphistegina* populations (Hallock 2000; Williams 2002) should be monitored at sites of interest twice yearly, in late spring and late summer. In addition, if remote sensing, *in situ* physical/chemical monitoring, or community assessment detect evidence or potential for stress, or when a disease outbreak or mortality event is reported, *Amphistegina* populations can provide evidence to distinguish between water quality and acute or chronic photooxidative stress as possible sources.

e) Determining causes of stress: When coral, stomatopod or *Amphistegina* populations indicate stress responses, resource managers are equipped to request or defend decisions to expend resources to identify sources of stress. If sewage pollution is suspected, nitrogen isotope analysis (Risk and others, 1993, 2001; Heikoop and others, 1998, 2001), analysis of CSM biota for sewage-derived microbes (e.g., Frias-Lopez and others, 2002), and viral tracers (e.g., Griffin and others, 1999; Donaldson and others, 2002) can be employed to verify sources and pathways of dispersal. If photooxidative stress is suspected, with or without temperature stress, species should be assessed using the MBS (Downs and others, 2000, 2002) or for DNA damage (Lyons 1998; Anderson and others, 2001). If trace metals or pesticides are suspected, the MBS or gene expression technology (e.g., Morgan and others, 2001) can be employed. While a suite of techniques is currently available for the study of coral diseases (e.g., Richardson and others, 2001; Ritchie and others, 2001; Santavy and others, 2001), additional research and development are needed to optimally utilize the CSM microbiota to diagnose specific stresses and disease outbreaks.

Advantages of implementing and further developing this multi-level approach for risk assessment and monitoring of coral reefs can be immediate, as well as long term and economic. While a suite of tools are currently available to managers to focus actions to protect valuable reef resources, many of the proposed bioindicators and biomarkers require further development and assessment (e.g., Jackson and others, 2000). Marine protected areas, at least in some cases, can provide reference sites for comparison with impacted sites and to aid in distinguishing local impacts from regional or global stresses. The larger

challenges are to find the political will and monetary resources to fundamentally improve agricultural, industrial, urban and coastal-development practices in ways that reduce harmful emissions, runoff, and release of toxic chemicals to the atmosphere and biosphere.

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