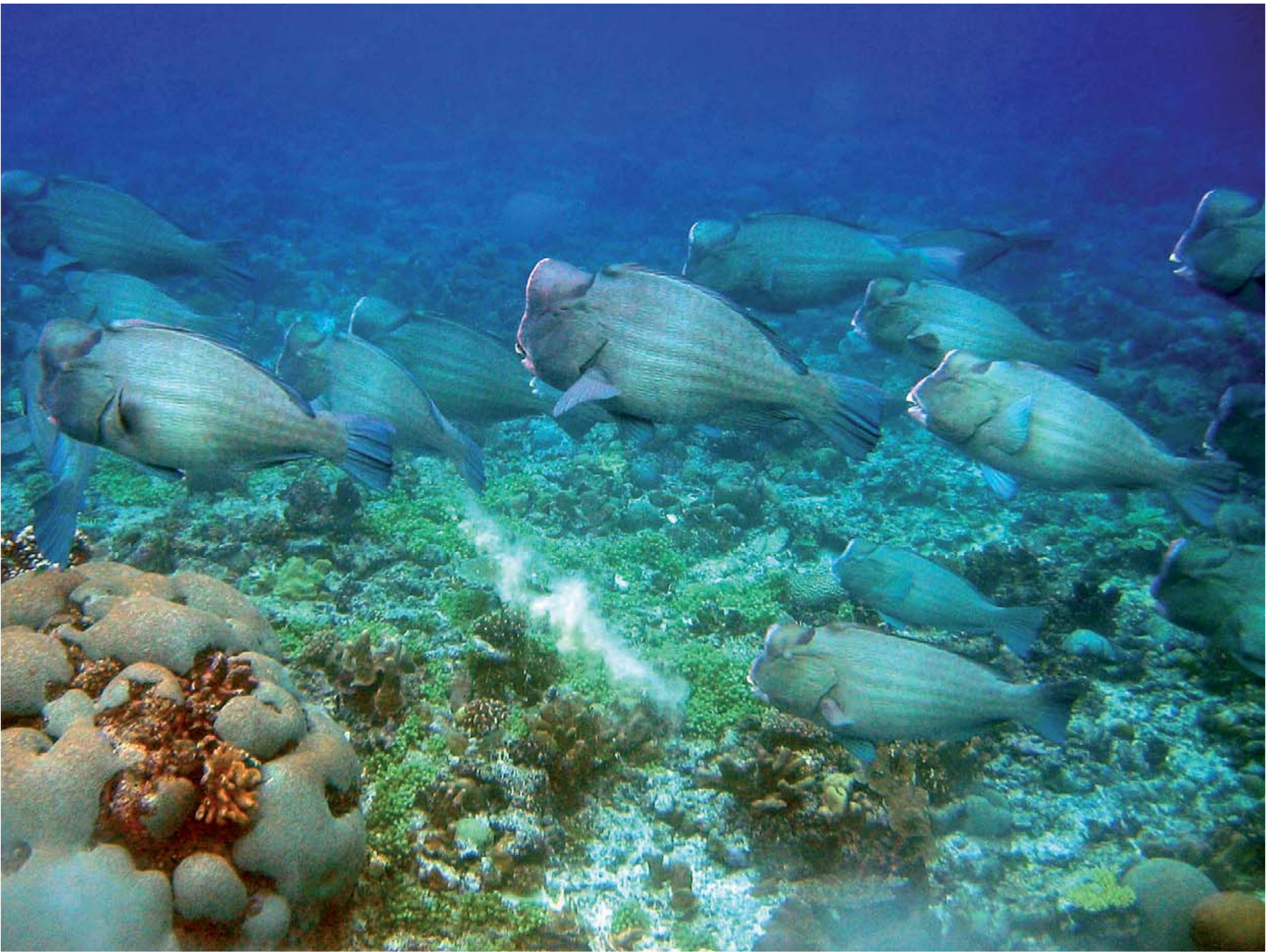
Monitoring Functional Groups of Herbivorous Reef Fishes as Indicators of Coral Reef Resilience

A practical guide for coral reef managers in the Asia Pacific Region

Alison L. Green and David R. Bellwood



IUCN RESILIENCE SCIENCE GROUP WORKING PAPER SERIES - NO 7

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Alison L. Green and David R. Bellwood

Cover photography:

Front cover: Humphead parrotfish (*Bolbometopon muricatum*). Image by R. Hamilton.

Back cover: Mixed school of herbivorous reef fishes (parrotfishes and surgeonfishes) on the reef crest. Image by A. Lewis.

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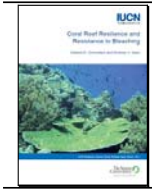
IUCN Resilience Science Group Working Paper Series – No 7

About the IUCN Climate Change and Coral Reefs Marine Working Group

The IUCN Climate Change and Coral Reefs Marine Working Group (formerly the IUCN Resilience Science Working Group), focused on coral bleaching, resilience and climate change, was established in 2006 by the Global Marine Programme of IUCN on a 3-year grant from the John D. and Catherine T. MacArthur Foundation. The goal of the working group is to draw on leading practitioners in coral reef science and management to streamline the identification and testing of management interventions to mitigate the impacts of climate change on coral reefs. The working group consults and engages with experts in three key areas: climate change and coral bleaching research to incorporate the latest knowledge; management to identify key needs and capabilities on the ground; and ecological resilience to promote and develop the framework provided by resilience theory as a bridge between bleaching research and management implementation.

One of the outputs of this group was the setting up of a website that provides links to projects, events, partners and publications. For more information, see http://www.iucn.org/cccr/publications/ This publication is the 7th in a series of publications on management tools to promote resilience in marine ecosystems. The other five available from IUCN's Global Marine Programme are listed below:

	**Coral Reef Resilience and Resistance to Bleaching** IUCN Resilience Science Group Working Paper Series – No 1 Gabirel D. Grimsditch and Rodney V. Salm © IUCN/TNC, October 2006
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INTRODUCTION

Coral reefs are the most structurally complex and taxonomically diverse marine ecosystems on earth, providing ecosystem goods and services for millions of people worldwide. These reefs are seriously threatened by a variety of anthropogenic threats, particularly overexploitation of marine resources, destructive fishing practices and runoff from poor land use practices. Over half of the world's reefs have already been lost or are under threat from these activities, with serious and widespread declines in reef health reported from around the world.

Climate change also represents a new and increasing threat to coral reefs and associated ecosystems. Major threats include rising sea temperatures, rising sea levels and changes in ocean chemistry. Urgent action is now required to halt or reverse these threats and declines in coral reef health. One approach to the protection of reefs during this period of change is to manage for resilience.

CORAL REEF RESILIENCE

Resilience is the ability of an ecosystem to absorb shocks, resist phase shifts and regenerate after natural and human-induced disturbances. For coral reefs, it is the ability of reefs to absorb recurrent disturbances, and rebuild coral dominated systems rather than shifting to algal dominated systems. Coral reef resilience will be increasingly important in future as disturbances become more frequent and severe with climate change.

Several key factors are critical for maintaining coral reef resilience. They are predominantly factors that facilitate coral recruitment and survivorship, including the availability of coral larvae, good water quality, conditioning by biological agents and a stable, consolidated substratum. In contrast, factors that negatively affect coral recruitment and survivorship include a lack of larval supply, loose rubble or unconsolidated substratum, thick algal mats or large stands of macroalgae, some sessile invertebrates (e.g. soft corals), and poor water quality (particularly runoff of sediments and nutrients from poor land use practices).

ROLE OF FUNCTIONAL GROUPS OF HERBIVOROUS REEF FISHES

Herbivores play a critical role in coral reef resilience by limiting the establishment and growth of algal communities that impede coral recruitment. On coral reefs in the Indo-Pacific Region, fishes are the dominant group of herbivores, while both echinoids and fishes are both important in the Caribbean. Major families include surgeonfishes, parrotfishes, rabbitfishes and rudderfishes.

Herbivorous reef fishes are diverse and do not constitute an ecologically uniform group. They comprise several functional groups that differ in terms of how they feed, what they consume, and their impact on the underlying substratum. This study focuses on four functional groups of herbivorous reef fishes that each play different and complimentary roles in coral reef resilience: scrapers/small excavators, large excavators/bioeroders, grazers/detritivores, and browsers.

Scrapers/small excavators: There are two groups of parrotfishes that show major differences in their jaw morphology and feeding behaviour. Both feed on epilithic algal turf, and remove some component of the reef substratum as they feed. They differ in the amount of the substratum they remove while feeding, and their contribution to ecosystem processes such as bioerosion. The majority of parrotfishes (*Hipposcarus* and *Scarus* species) are scrapers. They take non-excavating bites and remove algae, sediment and other material by closely cropping or scraping the reef surface, leaving shallow scrape marks on the reef substratum. Excavating species (*Bolbometopon muricatum, Cetoscarus bicolor* and all *Chlorurus* species) differ from scrapers by taking deeper excavating bites and removing greater quantities of substrata with each bite. Scrapers and small excavators (individuals <35cm standard length) play similar roles in coral reef resilience by limiting the establishment and growth of macroalgae while intensely grazing epilithic algal turf, and providing areas of clean substratum for coral recruitment.

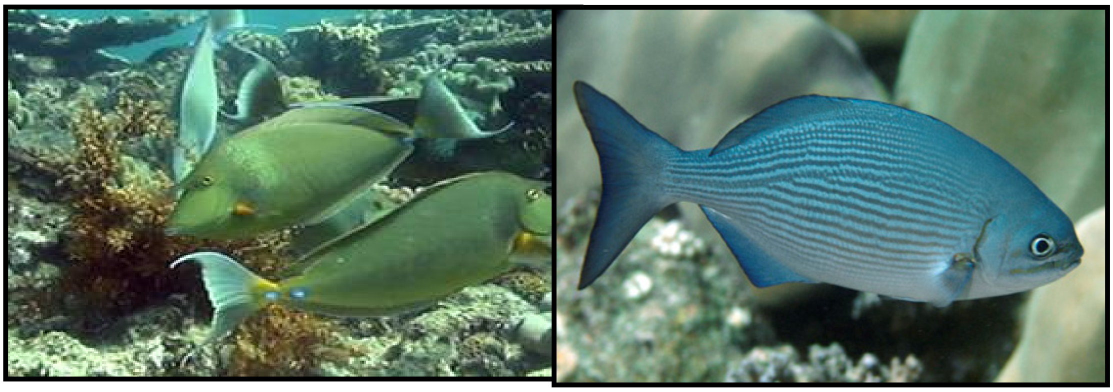
Large excavators/bioeroders play a similar role in coral reef resilience to scrapers and small excavators. However, they are also major agents of bioerosion on reefs, removing dead coral and exposing hard, reef matrix for coral recruitment. They include all large individuals of excavating species (see above: individuals > 35cm standard length). Five species have also been observed grazing on live corals on Indo Pacific reefs, although coral only accounts for a substantial proportion of the diet of one species (*B. muricatum*). Since these species have a greater affect on the underlying substratum than scrapers and small excavators, they play a different role in coral reef resilience by opening up new sites for colonization by coralline algae and corals.

Two groups of parrotfishes: a scraper (*Scarus flavipectoralis*: left) and a school of large excavators/bioeroders (*Bolbometopon muricatum*: right). Images by G. Allen and R. Hamilton respectively.

Grazers/detritivores play an important role in coral reef resilience by intensely grazing epilithic algal turfs, which can limit the establishment and growth of macroalgae. Unlike parrotfishes, grazers do not scrape or excavate the reef substratum as they feed. Grazers include most rabbitfishes, small angelfishes (all *Centropyge* species), and many species of surgeonfishes (all *Zebrasoma* and *Acanthurus* species except those that feed on exclusively on plankton or are grazers/detritivores). Grazers/detritivores include *Acanthurus* species that feed on a combination of epilithic algal turf, sediment and some animal material. Although only a small proportion of their diet is algae, grazers/detritivores are combined with grazers because many are schooling species that can be abundant and consume significant amounts of algal turf.

Grazers (*Acanthurus triostegus*: left) and a grazer/detritivore (*Acanthurus nigricauda*: right): both surgeonfishes. Images by G. Allen.

Browsers consistently feed on macroalgae. They select individual algal components and remove only algae and associated epiphytic material. Browsers play an important role in reducing coral overgrowth and shading by macroalgae, and can play a critical role in reversing coral-algal phase shifts. They include some unicornfishes, rudderfishes, batfishes, a rabbitfish and parrotfishes of the genus *Calotomus* and *Leptoscarus.*



Browsers: unicornfish (*Naso unicornis*: left) and a rudderfish (*Kyphosus vaigiensis*: right). Images by A. Hoey and G. Allen respectively.

MONITORING FUNCTIONAL GROUPS OF HERBIVOROUS REEF FISHES

Coral reef monitoring has traditionally focused on monitoring the status of coral communities and populations of conspicuous species, particularly fisheries species (fish and invertebrates). While these measures provide useful information on the current status of coral reef communities and associated fisheries, they do not provide information on the status of key ecological processes that are essential for maintaining coral reef resilience.

Developing new metrics for monitoring coral reef resilience that are process oriented is an urgent priority for the improved management of coral reefs. Monitoring coral reef resilience will require a combined approach to monitoring key ecological processes, and functional groups that contribute to these processes including:

- Coral population dynamics (size structure and patterns of recruitment).
- Factors that influence coral recruitment and survivorship, particularly water quality, substratum consolidation, and benthic communities (particularly macroalgae).
- Factors that influence the establishment and growth of macroalgal communities, particularly functional groups of herbivorous reef fishes.

Methods for assessing and monitoring coral reef resilience have been developed by the International Union for Conservation and Nature (IUCN) Working Group on Climate Change and Coral Reefs. The following is a protocol for monitoring functional groups of herbivorous reef fishes, which was developed as part of that process.

These protocols were developed using standard monitoring methods for coral reef fishes to facilitate their integration into new or existing programs, where field practitioners are interested in monitoring for multiple objectives (e.g. status of key fisheries species, coral reef resilience). They are not intended to represent the only method that should be used to monitor coral reef fishes, and in most locations, they are expected to form part of broader monitoring programs based on multiple objectives.

This document provides practical advice to field practitioners based on an example from the Asia Pacific Region. Key functional groups of herbivores are identified, species are assigned to each functional group, and methods are provided for monitoring their abundance, biomass and size structure.

Key components include:

- A hierarchical or stratified sampling design.
- Site selection of representative areas with adequate space to conduct the monitoring program.
- Standardising coral reef exposures and zones (reef slopes and crests on exposed linear reef fronts).
- Survey timing (between 9am and 4pm) and frequency for long term monitoring and rapid assessments of coral reef resilience: Long term monitoring requires a baseline survey, repeated every three years or more frequently if required to monitor the success of management actions or impacts of large scale disturbances. Rapid assessments usually comprise a single survey on one occasion.
- Underwater visual census methods for rapid assessments (20 minute timed swims) and long term monitoring (a combination of five 50m belt transects and 20 minute long swims).

Advice is also provided on data analysis, interpreting results, simplifying methods, and modifying the protocol for other biogeographic and geographic regions.

This document represents the first attempt to develop a monitoring program that is specifically designed to monitor key functional groups of herbivorous reef fishes as indicators of coral reef resilience. Even though it is based on the best available information, it is important to remember that the science underpinning these methods is still new and developing. Further research is now required to address knowledge gaps and refine monitoring methods.

BACKGROUND

CORAL REEFS AND THEIR ECOSYSTEM SERVICES

Coral reefs are the most structurally complex and taxonomically diverse marine ecosystems on earth, providing habitat for tens of thousands of associated fishes and invertebrates (Knowlton 2001, Jackson et al 2001). Despite occupying less than 0.1% of the world's marine environment (Spalding et al 2001), they support almost one third of the world's marine fish species (Reaka-Kudla 1996).

Coral reefs are critically important for the maintenance of biodiversity and the ecosystem goods and services they provide (Moberg and Folke 1999, Hughes et al 2003). Globally, more than 100 countries have coastlines with coral reefs (Moberg and Folke 1999), and almost 500 million people, or 8% of the total population, live within 100kms of a coral reef (Bryant et al 1998).

In most of these countries, many people live subsistence lifestyles and millions of people depend on coral reefs for all or part of their protein intake (Salvat 1992, Moberg and Folke 1999). The average global value of annual ecosystem services of corals reefs has been estimated to be US$375 billion per year (Constanza et al 1997).

THREATS

Threats to coral reefs and the ecosystem goods and services they provide are both natural and anthropogenic (see *Glossary*) in origin.

Natural Disturbances

Coral reefs and associated ecosystems have a high degree of natural variability due to large scale episodic disturbances such as major storms (hurricanes, typhoons and cyclones: Brown 1997, Green et al 1999). While these disturbances can result in large scale fluctuations in coral communities (Brown 1997, Green et al 1999, Bruno and Selig 2007), they are part of the natural disturbance regime under which coral reefs have evolved (Moberg and Folke 1999). In the absence of anthropogenic threats, reefs are generally able to recover from these disturbances in a few years to decades (Brown 1997, Green et al 1999).

Anthropogenic Threats

Coral reefs are seriously threatened both locally and globally by a variety of direct and indirect anthropogenic threats (Brown 1997, Bryant et al 1998, Jackson et al 2001, Halpern et al 2008). Of primary concern are threats to ecosystem health from overexploitation of marine resources, destructive fishing practices, runoff of sediments and nutrients from poor land use practices, coastal development and uncontrolled tourism activities (Bryant et al 1998, Jackson et al 2001, Fabricius 2004, Wilkinson 2008, Halpern et al 2008).

Another issue of concern is the degree to which human activities have influenced the proliferation of other threats, particularly outbreaks of corallivorous crown of thorns starfish. In the last few decades, starfish outbreaks have substantially reduced coral cover on many reefs in the Indo-Pacific Region (Jackson et al 2001, Bruno and Selig 2007). While the causes of these outbreaks remain controversial, their increasing incidence and severity is increasingly linked to human activities, particularly increased runoff of nutrients from land and overfishing of starfish predators (Jackson et al 2001, Fabricius 2004, DeVantier and Done 2007. Sweatman 2008).

Climate change also represents a new and increasing threat to coral reefs and associated ecosystems worldwide (Hoegh-Guldberg 1999, McLeod and Salm 2006, Hoegh-Guldberg et al 2007, Veron 2008). Major threats include rising sea temperatures leading to mass coral bleaching, rising sea levels that threaten coastal ecosystems (e.g. mangrove forests), and changes in ocean chemistry that affect the ability of calcifying organisms (including corals) to deposit their calcium carbonate skeletons (Hoegh-Guldberg 1999, McLeod and Salm 2006, Hoegh-Guldberg et al 2007, IPCC 2007, Veron 2008). Another issue of concern is the degree to which global climate change may influence the frequency and intensity of tropical storms (IPCC 2007), leading to shorter recovery times between recurrences (Hughes et al 2003). Climate change has also been linked to the recent proliferation of other threats, particularly coral diseases (Bruno et al 2007, Harvell et al 1999, 2007).

STATUS AND TRENDS

The diversity, frequency, and scale of these threats have now increased to the extent that many coral reefs have suffered severe, long-term declines in abundance, diversity and habitat structure, and are threatened globally (Pandolfi et al 2003, 2005, Hughes et al 2003, Wilkinson 2008).

A recent assessment found that the world has effectively lost 19% of its coral reefs, with 35% under threat in the next 10 to 40 years (Wilkinson 2008). Fortunately, 46% of the world's reefs are regarded as being relatively healthy, and not under immediate threat of destruction (except from climate change).

Declines in coral reef status vary around the world (reviewed by Wilkinson 2008), with a higher proportion of reefs effectively lost in the western Atlantic and the Indian Ocean than in the Red Sea, Australia and the Pacific Islands. The most serious declines have been recorded in Asia and the Arabian Gulfs.

Within the Asia Pacific Region, the reefs of Southeast Asia are most seriously threatened, with 40% of reefs effectively lost, 45% under threat, and 15% at low threat. In contrast more reefs in Australia and the Pacific Islands are in better condition, with 2 to 8% effectively lost, 2 to 35% under threat, and 44 to 90% at low threat. Nonetheless, Bruno and Selig (2007) showed that average coral cover on Indo-Pacific reefs has declined from approximately 50% to 22% in the last four decades, due to major storms and a variety of direct and indirect human impacts.

These studies demonstrate that there are few, if any, reefs that have not been affected by human activities (Jackson et al 2001, Pandolfi et al 2003, Bruno and Selig 2007, Halpern et al 2008), and urgent action is required to halt or reverse these declines in coral reef health. One approach is to manage for coral reef resilience.

CORAL REEF RESILIENCE

WHAT IS IT, AND WHY IS IT IMPORTANT?

Resilience is the ability of an ecosystem to absorb shocks, resist phase shifts and regenerate after natural and human-induced disturbances (Nyström et al 2000). A core component of resilience is the ability of an ecosystem to maintain its key ecological functions and processes after disturbance by either resisting or adapting to change (Gunderson 2000, Carpenter et al 2001, Nyström and Folke 2001). For coral reefs, it is the ability of reefs to absorb recurrent disturbances (such as coral bleaching events), and rebuild coral-dominated systems rather than shifting to algal dominated systems (Marshall and Schuttenberg 2006, Hughes et al 2007). Coral reef resilience will be increasingly important in future as disturbances such as coral bleaching become more frequent and severe with climate change (see *Threats*).

Several key factors are critical for maintaining coral reef resilience. They are predominantly factors that facilitate coral recruitment (see *Glossary*) and survivorship, including the availability of coral larvae (Pearson 1981, Roberts 1997, Birrell et al 2008), good water quality (Hunte and Wittenberg 1992, Fabricius 2004), and stable, consolidated substratum (Fox 2002, 2004, Fox et al 2003, Fox and Caldwell 2006). In some situations conditioning by biological agents such calcareous red algae, particularly crustose coralline algae, may also enhance coral recruitment (Harrison and Wallace 1990, Harrington et al 2004, Birrell et al 2008).

In contrast, factors that negatively affect coral recruitment and survivorship, and therefore coral reef resilience, include a lack of larval supply (Hughes and Tanner 2000, Ayre and Hughes 2004, Ledlie et al 2007), loose rubble or unconsolidated substratum (Fox 2002, 2004, Fox et al 2003, Fox and Caldwell 2006), thick algal mats or large stands of macroalgae (Birrell et al 2008), some sessile invertebrates such as soft corals (Maida et al., 1995), and poor water quality (particularly runoff of sediments and nutrients from poor land use practices: Hunte and Wittenberg 1992, Fabricius 2004).

Dense stands of macroalgae inhibit coral reef resilience by overgrowing and killing coral colonies, and impeding coral recruitment by lowering coral fecundity, settlement rates, and post settlement survival (reviewed in Birrell et al 2008). Therefore factors that are important in controlling algal communities, and preventing dense stands of macroalgae from becoming established, play an important role in coral reef resilience.

Factors that play an important role in controlling algal communities include herbivory, nutrients, physical disturbance, substratum type, available space and algal recruitment dynamics (Steneck 1988, McClanahan et al 2001, 2003, McCook et al 2001, Fabricius 2004, Hughes et al 2007). Herbivory is particularly important, and in many situations, is the primary driver controlling algal communities (Steneck 1988, Lirman 2001, Williams and Polunin 2001, Mumby et al 2006, Hughes et al 2007).

On coral reefs of the Indo-Pacific Region (see *Glossary*), herbivorous reef fishes are the dominant group of herbivores, while both fishes and echinoids are important in the Caribbean (Klumpp et al 1987, Choat 1991). These fishes play a critical role in coral reef resilience by controlling algal communities and influencing competitive interactions between corals and macroalgae (Lirman 2001, Williams and Polunin 2001, Bellwood et al 2004, Mumby et al 2006, 2007, Hughes et al 2007).

Resilient reef (left) and a dense stand of macroalgae overgrowing a coral colony (right). Images by E. Turak and A. Hoey respectively.

ROLE OF FUNCTIONAL GROUPS OF HERBIVOROUS REEF FISHES

Herbivorous reef fishes are diverse, and do not constitute an ecologically uniform group (Choat 1991). Instead, they comprise several functional groups that each play important roles in coral reef resilience by preventing coral-algal phase shifts.

Functional groups are defined as a collection of species that perform a similar function, irrespective of their taxonomic affinities (Steneck and Dethier 1994). For reef fishes, functional groups are generally synonymous with guilds of species from different trophic levels within a food chain (e.g. predators and herbivores), reflecting their roles as major conduits for energy flow on reefs (Bellwood et al 2004). Functional groups can also be identified by their roles in ecosystem processes (Bellwood et al 2004).

Several studies have demonstrated the critical role herbivorous reef fishes play in influencing competitive interactions between corals and macroalgae (Lewis 1986, Lirman 2001, Williams and Polunin 2001, Mumby et al 2006, 2007, Hughes et al 2007), and therefore coral reef resilience (Nyström and Folke 2001, Bellwood et al 2004). Many of these studies have demonstrated a strong negative relationship between fish grazing intensity (particularly biomass) and macroalgal cover (Lewis 1986, Williams and Polunin 2001, Mumby et al 2006, 2007, Fox and Bellwood 2007; but see Wismer et al 2009). However, while grazing is generally considered to provide a net benefit to coral reef resilience, some herbivores (notably some parrotfish species) may also have detrimental effects on coral reef resilience in some instances by consuming live coral in addition to algae (Bellwood and Choat 1990, McClanahan et al 2005, Rotjan and Lewis 2005, Rotjan et al 2006).

Herbivorous reef fishes are also important agents of bioerosion on reefs (i.e. the removal of material from the reef matrix by biological processes: Choat 1991). Large excavating parrotfishes play a key role in bioerosion by excavating the surface of the reef matrix or living coral as they feed (Bellwood and Choat 1990, Choat 1991, Bellwood 1995, Bellwood et al 2003). This material is processed by their specialized jaws, reduced to sediment and expelled back onto the reef or surrounding areas (Choat 1991, Wainwright and Bellwood 2002).

Bioerosion plays a critical role in coral reef resilience by removing dead coral and cleaning areas of substratum for colonization by benthic organisms, facilitating the settlement, growth and survival of coralline algae and corals (Steneck 1988, Hoey and Bellwood 2008). Many herbivorous reef fishes also recycle sediment by taking it up directly from reef aprons, lagoon floors or algal turfs where it can accumulate (Choat 1991).

Herbivorous reef fishes can also play an important role in reversing coral-algal phase shifts, although this process is not well understood (Bellwood et al 2006, Fox and Bellwood 2008). When phase shifts occur, reversals back to coral dominated reefs are uncommon, and it is difficult to predict which species may be important in facilitating this change, since they are likely to be different to those that are important in avoiding phase shifts in the first place (Bellwood et al 2006).

Herbivorous reef fishes can also play an important role in avoiding or reversing other alternate states on coral reefs (see below: Bellwood and Fulton 2008).

Avoiding Coral-Algal Phase Shifts

Several studies have demonstrated how the loss of herbivorous fishes can trigger coral-algal phase shifts on coral reefs (Lewis 1986, Hughes 1994, Hughes et al 2007, McClanahan et al 2001, Bellwood et al 2006). For example in Jamaica, overfishing of herbivorous fishes played a critical role in a dramatic coral-algal phase shift (Hughes 1994, Gardner et al 2003). However, the consequences of the declining populations of herbivorous fishes did not become apparent until after a hurricane caused widespread damage to coral communities, and a disease outbreak caused mass mortality of the other dominant herbivore (the echinoid *Diadema antillarum*: Hughes 1994). After the hurricane and *Diadema* die-off, coral communities were unable to become re-established because the substratum became dominated by macroalgae due to the lack of herbivorous reef fishes. Adult coral colonies that survived the hurricane were also killed by algal overgrowth.

Perhaps the clearest example of the key role that herbivorous reef fishes can play in avoiding coral-algal phase shifts is from a large scale manipulative experiment on the Great Barrier Reef (Bellwood et al 2006, Hughes et al 2007). This experiment simulated the effects of depleting large herbivorous fishes by chronic overfishing, and investigated the role of these herbivorous in the regeneration of reefs after a mass bleaching event and associated coral mortality.

The experiment demonstrated that the exclusion of larger fishes profoundly eroded coral reef resilience, and the ability of the coral community to regenerate after bleaching (Hughes et al 2007). In control areas where herbivorous fishes were abundant, algal abundance remained low, and coral cover almost doubled over three years due to coral recruitment. In contrast, the exclusion of large herbivorous fishes triggered a dramatic phase shift from a system dominated by epilithic algal turf (see *Glossary*) and corals to one overgrown by macroalgae, and demonstrated that even in the absence of other pressures the exclusion of large herbivorous fishes profoundly eroded coral reef resilience (Hughes et al 2007). Once herbivorous fishes were allowed access to the area, a reversal to a coral dominated state occurred (see *Reversing Coral-Algal Phase Shifts* below).

While populations of herbivorous reef fishes play a key role in avoiding coral-algal phase shifts, other factors that influence the population dynamics of macroalgal communities can also be important, including nutrients, physical disturbance, substratum type, and algal recruitment dynamics (Steneck 1988, McClanahan et al 2001, 2003, McCook et al 2001, Fabricius 2004). For example, McClanahan et al (2001) reported that a lack of herbivory by reef fishes and sea urchins was only one of the factors influencing macroalgal cover on Kenya reefs following a severe coral bleaching event. Even though herbivory was the best predictor of macroalgal cover on these reefs, physical disturbances (by waves and currents) and nutrients were also important.

Furthermore, while healthy populations of herbivorous fishes are a key component of coral reef resilience, they are unable to prevent coral-algal phase shifts in the absence of other key factors. For example, Ledlie et al (2007) showed that a marine reserve in the Seychelles underwent a dramatic coral-algal phase shift following extensive coral mortality associated with a mass coral bleaching event, despite the presence of apparently healthy herbivorous fish populations. In that case, the coral-algal phase shift was probably due to a lack of coral larvae, following the mass mortality of corals throughout the Seychelles as a result of the bleaching event, although overfishing of critical herbivores (particularly browsers) outside the marine reserve may have also been a contributing factor underpinning the increase in macroalgae (Fox and Bellwood 2008).

The role of herbivorous reef fishes in promoting reef recovery also depends upon their feeding preferences (see *Key Families and Their Feeding Modes*), and their numerical abundance and biomass relative to benthic cover (Ledlie et al 2007). For example, coral cover was implicated as an important factor in determining the impact of herbivorous fishes on algal communities in the Seychelles, where their impact was likely to be greater where coral cover was high and there was stronger competition for limited algal resources (Ledlie et al 2007).

Similarly, several studies in the Caribbean have demonstrated that while herbivorous reef fishes play important roles in maintaining algal assemblages in a cropped state, they are unable to maintain all areas of reef substratum in that condition, particularly in areas of low coral cover, high nutrients and where the urchin *Diadema antillarum* remains scarce (Williams et al 2001, Mumby 2006, Mumby et al 2005, Paddack et al 2006).

In summary, herbivorous reef fishes play an important role in influencing the dynamics of macroalgal communities, and play a critical role in avoiding coral-algae phase shifts. However, they are not the only factor that influences macroalgal dynamics, and therefore must be considered in the broader context of coral reef resilience.

Reversing Coral-Algal Phase Shifts

While phase shifts from coral to macroalgal dominated communities are not uncommon following disturbances to coral reefs such as storms or mass coral bleaching events (Hughes 1994, McClanahan et al 2001, Graham et al 2006, Ledlie et al 2007), there are very few documented cases of reversals in coral-algal phase shifts (Bellwood et al 2006). That is because the macroalgal communities that become dominant are often characterised by species with physical and/or chemical deterrents that render them less palatable or digestible for herbivores (reviewed in Hay 1991, Steneck and Dethier 1994), such as cyanobacteria, red and brown algae (Hatcher 1984, Ledlie et al 2007, Schroeder et al 2008). In some situations these macroalgae communities become increasingly resistant to perturbations (McManus and Polsenberg 2004) and can become stable (i.e. alternate stable states) unless removed by physical disturbances such as major storms (Hatcher 1984).

The best recorded case of a phase shift reversal was from a large scale experimentally induced phase shift on the Great Barrier Reef, where areas of reef were caged to exclude herbivores, resulting in a coral-algal phase shift where the reef became dominated by dense stands of brown algae (*Sargassum*: Bellwood et al 2006, Hughes et al 2007). When the cages were removed, a phase shift reversal occurred from a macroalgal-dominated community to a coral- and epilithic algal-dominated community (Bellwood et al 2006). Surprisingly, the herbivores that are known to be important in preventing coral-algal phase shifts were not responsible for the reversal. Instead the phase-shift reversal was primarily driven by a batfish species (*Platax pinnatus* in *Sargassum*, left; Image by D. Bellwood), which was previously regarded as an invertebrate feeder (Bellwood et al 2006)! This species was consistently observed removing and ingesting large pieces of *Sargassum*, and may have contributed to algal removal by dislodging the algae while feeding. Bellwood et al (2006) coined the term “sleeping functional group” for species (or groups of species) like this, which may be capable of performing a particular functional role but which do so only under exceptional circumstances.

It is possible that other reef fish species may play similar roles to batfishes in reversing coral-algal phase shifts, although these species are difficult to predict and are likely to vary along many spatial and temporal scales (Bellwood et al 2006). Furthermore while some species may be capable of reversing coral-algal phase shifts, they may not be present in sufficient numbers to reverse phase shifts when they occur (Ledlie et al 2007, Fox and Bellwood 2008).

The extent to which herbivorous reef fishes can facilitate phase shift reversals will therefore depend on their functional role, abundance and the type of algae they consume. In some cases, they may include more traditional herbivorous species.

Families likely to play significant roles in coral-algal phase shifts reversals based on their diet, behaviour and feeding mode include rabbitfishes, rudderfishes and unicornfishes in the Indo-Pacific and sparisomatine parrotfishes in the Caribbean (Bellwood et al 2006). One example is an Indo-Pacific rabbitfish species (*Siganus canaliculatus*), which is known to feed on *Sargassum* (Mantyka and

Bellwood 2007, Fox and Bellwood 2008). However since this species is not amenable to underwater visual census techniques, its role in phase shift reversals may be difficult to predict and monitor.

Therefore, while herbivorous reef fishes may play an important role in reversing coral algal phase shifts, it is not possible to predict with certainty which species may be important at a particular location. Nor is it feasible to monitor these species using underwater visual census methods, since many appear to avoid divers (Bellwood et al 2006). Consequently, this protocol will focus on monitoring species that are important in preventing coral-algal phase shifts.

Reversing/Avoiding Other Alternate States on Coral Reefs

While coral-macroalgal phase shifts are the most widely discussed phase shifts on coral reefs, it must be noted that there are a range of other alternate states on reefs, and that the shift to macroalgae may not be the commonest phase or regime shift (Bruno et al 2009), especially on Indo-Pacific reefs and in areas away from the coast or reef flat. These alternate states were highlighted by Knowlton (1992) and Bellwood et al (2004) and have been recently reviewed by Norström et al (2009). In most cases, the alternate state is dominated by some type of algae although anemones or soft corals may be involved. In terms of alternate states, one of the most common on Indo-Pacific reefs may be algal turfs. Although a common feature of most reefs, turfs vary widely in their composition and those with heavy cyanobacterial loads may represent an undesirable state. Likewise, turfs with heavy sediment loads may represent a stable (i.e. highly resilient) but undesirable condition (Bellwood and Fulton 2008). While these alternate states are poorly understood, some are likely to be influenced by the size and composition of local herbivore communities, particularly parrotfishes (Bellwood and Fulton 2008). The monitoring of herbivores therefore will provide an indication of the relative contribution of herbivores to any changes in the composition of benthic algal cover.

MANAGING CORAL REEF RESILIENCE

Global climate change represents a serious threat to the long term future of coral reefs, and strong policy decisions and management actions are urgently required to reduce global greenhouse gas emissions to slow or reverse the rate of ocean warming and acidification (Hughes et al 2003, Pandolfi et al 2005, Hoegh-Guldberg et al 2007). There is also an urgent need to focus management on maintaining coral reef resilience by addressing direct and indirect anthropogenic threats to coral reefs.

Resilience-based management represents a novel and timely approach to coral reef management, and a fundamental change of focus from reactive to proactive management (Hughes et al 2005, 2007). It recognises that while large scale disturbances such as coral bleaching cannot be managed directly by local managers, local management efforts that support coral reef resilience by addressing other threats can afford significant protection against these large scale disturbances (Hughes et al 2003, 2007, Pandolfi et al 2005).

Managing for coral reef resilience will therefore require a two pronged approach to coral reef conservation by reducing threats to coral reef resilience, and applying ecosystem-based adaptation strategies (reviewed in Marshall and Schuttenberg 2006, McCook et al 2008, TNC 2009).

Reducing Threats to Coral Reef Resilience

Given the critical role that coral reef resilience plays in sustaining coral reefs, there is an urgent need for local management to focus on managing for and protecting coral reef resilience (Nyström and Folke 2001; McClanahan et al 2002, Hughes et al 2003, Bellwood et al 2004). This will require identifying key ecological processes that contribute to coral reef resilience, and managing human activities that threaten these processes (McClanahan et al 2002, Bellwood et al 2003, 2004, Hughes et al 2003).

Coral reef resilience is eroded by chronic human impacts that cause persistently elevated rates of mortality and reduced rates of recruitment of coral reef organisms, particularly corals (reviewed in Hughes et al 2003). Reduced herbivory from overfishing, and runoff of sediments, nutrients and other pollutants from poor land use practices, can impair resilience and prevent recovery following acute-disturbance events like cyclones or coral bleaching (Hughes et al 2003, Fabricius 2004). The key to successful management will be maintaining coral reef resilience and thereby reducing the likelihood of undesirable phase shifts (Hughes et al 2007), since restoring marine ecosystems after they have degraded is much more difficult than maintaining them in good condition (Hughes et al 2005). In particular, it is essential to secure the future of functional groups that play a critical role in the ecological processes that support coral reef resilience (McClanahan et al 2002, Bellwood et al 2004, Hughes et al 2005). This may require a two pronged approach to protecting functional groups of herbivores, by protecting species that prevent phase shifts and those that facilitate reversal and recovery (Bellwood et al 2006).

Key strategies for addressing threats to coral reef resilience include:

- Establishing no-take marine reserves that provide the most effective protection from extractive activities such as fishing (Lubchenco et al 2003, Russ 2002). If adequately enforced, no-take areas allow critical functional groups to persist and contribute to ecosystem resilience (Bellwood et al 2004, Hughes et al 2005, 2006, Mumby et al 2006, 2007) by affording a spatial refuge for a portion of the stock from which larvae and adults can disperse to adjoining exploited areas (Lubchenco et al 2003, Russ 2002). However at present, less than 1.5% of the world's coral reefs are located within no-take areas (Mora et al 2006), and there is a pressing need to greatly increase the rate at which these areas are established (Hughes et al 2003, Bellwood et al 2004).
- Since the vast majority of coral reefs are not located in no-take areas (Mora et al 2006), an ecosystem approach to fisheries management is also required outside of no-take areas to achieve regional-scale management of critical functional groups to support coral reef resilience (Hughes et al 2003, 2006, Bellwood et al 2004). This is important because most no-take areas are unlikely to be self sustaining, because they are small relative to the scale of the disturbances and dispersal distances of many larvae and migrating adults (Bellwood et al 2004). Therefore, the success of no-take areas may be dependant on an influx of adults or larvae from surrounding areas, and they must be managed in the context of the entire seascape (Bellwood et al 2004).
- While no-take areas and an ecosystem approach to fisheries management can be effective tools for managing threats that arise from within the marine environment, they cannot address threats to coral reef resilience that originate from outside the marine environment particularly runoff from poor land use practices (Lubchenco et al 2003). Improved management of land based threats is therefore required to address all human activities that affect marine life (Kelleher 1999, Lubchenco et al 2003, Hilborn et al 2004), and therefore coral reef resilience (Hughes et al 2003).

Ecosystem-based Adaptation Strategies

Ecosystem-based adaptation strategies (see *Glossary*) provide responses to the impacts of climate change, and empower managers and policy makers to build resilience to climate change into their conservation programs.

In recent years, principles for designing and managing marine protected area (MPA) networks that are resilient to the threat of climate change have been developed (West and Salm 2003, Grimsditch and Salm 2006, McLeod and Salm 2006, McLeod et al 2009, TNC 2009). They include: addressing uncertainty by spreading the risk through representation and replication of major habitats; protecting critical habitats, particularly those demonstrating strong resilience; understanding and incorporating patterns of biological connectivity; and reducing other threats (particularly unsustainable fishing practices and runoff from poor land use practices). These principles are now being applied to MPA design throughout the Asia Pacific Region (e.g. Green et al 2007, in press) and elsewhere around the world.

Ecosystem-based adaptation strategies also offer sustainable solutions by fostering healthy ecosystems that are better able to withstand the negative affects of environmental change. For example climate change, and specifically sea level rise, is threatening coastal populations and habitats with erosion and inundation. Protecting, restoring, and managing key coastal ecosystems (e.g., wetlands, mangroves, coral reefs, and barrier beaches) provides cost-effective and natural shoreline protection against storms and flooding (CBD 2009).

MONITORING CORAL REEF RESILIENCE

Coral reef monitoring has traditionally focused on monitoring the status of coral communities (e.g. coral cover) and populations of conspicuous species, particularly fisheries species (fish and invertebrates: see review in Hill and Wilkinson 2004). While these measures provide useful information on the current status of these communities, they do not provide information on the status of key ecological processes that are essential for maintaining coral reef resilience (Bellwood et al 2004, Hughes et al 2005). Therefore, they do not provide an indication of whether coral reefs are likely to recover after disturbance.

Developing new metrics for monitoring coral reef resilience that are process oriented is an urgent priority for the improved management of coral reefs (Hughes et al 2005). Monitoring coral reef resilience will require a combined approach to monitoring key ecological processes, and functional groups that contribute to these processes, including:

- Coral population dynamics (size structure and patterns of recruitment).
- Factors that influence coral recruitment and post-settlement survivorship, particularly water quality, substratum consolidation and benthic communities (particularly macroalgal communities).
- Factors that influence the establishment and growth of macroalgal communities, including functional groups of herbivores, nutrients and algal recruitment dynamics.

SCOPE AND OBJECTIVES OF THIS STUDY

Methods for assessing and monitoring coral reef resilience have recently been developed by the International Union for Conservation and Nature (IUCN) Working Group on Climate Change and Coral Reefs (http://www.iucn.org/cccr/), which is an alliance between scientists and managers from governments, non-government organisations and scientific institutions.

The objective of this study was to contribute to that process by developing protocols for monitoring functional groups of herbivorous reef fishes that play a critical role in coral reef resilience by controlling macroalgal communities and preventing coral-algal phase shifts (Bellwood et al 2004, Hughes et al 2005).

These protocols were developed using standard monitoring methods for coral reef fishes to facilitate their integration into new or existing programs, where field practitioners are interested in monitoring for multiple objectives (e.g. status of key fisheries species, coral reef resilience). They are not intended to represent the only method that should be used to monitor coral reef fishes, and in most locations, are expected to form part of broader monitoring programs based on multiple objectives.

This document provides practical advice to field practitioners based on an example from the Asia Pacific Region (the coral reefs of Southeast Asia and the Pacific Islands: see *Glossary*). Key functional groups of herbivores are identified, species are assigned to each functional group, and methods are provided for monitoring their abundance, biomass and size structure.

While this document focuses on the Asia Pacific Region, these methods have global application and can be easily modified for other regions. Practical advice is also provided on data analysis, interpreting results, simplifying methods, and modifying the protocol for other biogeographic and geographic regions.

HERBIVOROUS CORAL REEF FISHES

KEY FAMILIES AND THEIR FEEDING MODES

Herbivorous reef fishes are taxonomically diverse, comprising at least nine families (Choat 1991). The most important families are the Acanthuridae (surgeonfishes and unicornfishes), Pomacentridae (damselfishes), Labridae (Tribe Scarini[1]: parrotfishes), Siganidae (rabbitfishes) and Kyphosidae (rudderfishes: Choat 1991). Some of these families are quite speciose, particularly surgeonfishes and parrotfishes (Choat 1991).

Another potentially important family is the Blenniidae (blennies), which is usually overlooked because of their cryptic habits. Other families comprising some herbivorous representatives, include the Pomacanthidae (angelfishes), Gobiidae (gobies) and Ephippidae (batfishes: Choat 1991, Bellwood et al 2006). Some Monacanthidae (filefishes), Balistidae (triggerfishes) and other Labridae (wrasses) may also be herbivores, but their trophic status has not been confirmed (Myers 1999, Randall et al 1996, G.P Jones *unpubl. data*).

Herbivorous reef fishes exhibit a range of feeding modes and ingest a variety of plant material, including macroalgae, epilithic algal turf, detrital material, algal mats and associated organisms (including bacteria: reviewed by Choat 1991). There is considerable variation among and within families, and they do not all perform the same role nor do they have similar impacts on coral reef ecosystems (Choat 1991).

There are two groups of parrotfishes, which show major differences in their jaw morphology and feeding behaviour: excavators and scrapers (Bellwood and Choat 1990). The majority of parrotfishes (scarines[1]) are scrapers (most *Scarus* and *Hipposcarus* species) that feed on epilithic algal turf growing on hard substrata (Bellwood and Choat 1990), although some Indo-Pacific species graze mainly over sand (Choat 1991). They take non-excavating bites and scrape material from the surface of the substratum. Excavators differ from scrapers by taking deeper, excavating bites and removing more of the substratum as they feed (Bellwood and Choat 1990). They include *Bolbometopon muricatum, Cetoscarus bicolor* and species of the genus *Chlorurus*. Members of the genus *Calotomus* are browsers, while some members of the Caribbean subfamily Sparisomatinae feed on marine angiosperms (Choat 1991). *B. muricatum* also grazes substantial amounts of live coral (Choat 1991, Bellwood et al 2003). Four other Indo-Pacific parrotfish species have been observed grazing on live corals (*Chlorurus microrhinos*, *Cetoscarus bicolor, S. frenatus* and *S. rivulatus*: Bellwood and Choat 1990, Bellwood 1995), although corals only account for a small proportion of their diet (Hoey and Bellwood 2008).

There is a greater diversity of feeding habits within the Acanthuridae (Choat 1991). Most species of the genera *Acanthurus* and *Zebrasoma* graze on epilithic algal turf over reef substrata (Choat 1991, Choat et al 2002), although some *Acanthurus* species graze over sand and feed on a combination of algae and detritus (ringtail surgeonfishes including *A. blochii*, *A. dussumieri*, *A. leucocheilus*, *A. maculiceps*, *A. nigricauda*, *A. olivaceus*, *A. pyroferus*, *A. tristis* and *A. xanthopterus*) while others are planktivores (*A. albipectoralis*, *A. mata*, *A. nubilus* and *A. thompsoni*: Choat 1991, Choat et al 2002). All members of the genus *Ctenochaetus* are primarily sediment and detritus feeders (Choat 1991), while the single species of the monotypic genus *Paracanthurus* is a planktivore (Choat 1991, Choat et al 2002). The majority of unicornfishes are also planktivores (Choat 1991, Choat and Clements 1998, Choat et al 2002), with the largest individuals (*N. annulatus*, *N. hexacanthus* and *Naso lopezi*) feeding on macroplankton in open water (Choat 1991). A second group of unicornfishes, including *N. unicornis* and *N. tuberosus*, feed on large macroscopic algal species (Choat and Clements 1998).

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[1] Recent phylogenetic analyses have confirmed that parrotfishes, previously Family Scaridae, belong to the Tribe Scarini within the Family Labridae (Westneat and Alfaro 2005, Cowman et al in press). Therefore parrotfishes, previously known as scarids, are now known as scarines.

Herbivorous reef fishes (clockwise from top left): parrotfish (*Scarus frenatus*), surgeonfish (*Acanthurus nigricauda*), rudderfish (*Kyphosus vaigiensis*), angelfish (*Centropyge bicolor*), damselfish (*Dischistodus chrysopoecilus*) and rabbitfish (*Siganus virgatus*). Images by G. Allen.

Siganids and kyphosids are less speciose than scarines and acanthurids, and are more conservative in their feeding habitats (Choat 1991, Choat et al 2002). Both families tend to bite or ‘crop’ algae leaving the basal portions intact (Choat et al 2002). While most siganids are grazers that feed on epilithic algal turfs, some species browse on macroalgae (*Siganus canaliculatus*: Choat 1991, Mantyka and Bellwood 2007, Fox and Bellwood 2008) or are grazers/detritivores (*S. lineatus*: Fox et al in press)**.** Kyphosids feed on large macroscopic algae, and are generally considered browsers (Choat et al 2002).

Pomacentrids comprise a lot more species than the other families (reviewed in Choat 1991). They are also more diverse in their feeding habits although almost all species appear to take some algal material. The most speciose genera are either aggregating planktivores (*Chromis* and *Dascyllus*) or omnivores (*Pomacentrus* and *Chrysiptera*). However within the omnivores, a number of species have pronounced tendencies toward herbivory and territorial behaviour (e. g. *Chrysiptera. biocellata*, *Pomacentrus grammorhynchus* and *P. wardi*). The most explicitly herbivorous members of the Pomacentridae are the large territorial species of the genera *Dischistodus, Hemiglyphidodon, Plectroglyphidodon* and *Stegastes,* (Polunin and Klumpp 1989, Choat 1991), although some of these species may feed primarily on detritus within their territories (Wilson and Bellwood 1997). The extent to which territorial pomacentrids rely on an invertebrate component in their diet is also unclear (Choat 1991).

ONTOGENETIC CHANGES IN FEEDING MODES

Most reef fishes go through a planktonic larval stage (Leis and Rennis 1983), and little is known about their feeding modes while they are in the plankton (Leis 1991). However, most are presumed to feed on other plankton (Choat 1991).

Once they settle onto the reef, the majority of herbivorous species become herbivores and stay that way throughout their lives, including most acanthurids, blenniids and siganids (reviewed by Bellwood 1988 and Choat 1991). Similarly, the pomacanthid *Centropyge bicolor* appears to be an omnivore throughout its benthic life (Bellwood 1988).

Others change feeding modes at different stages of their life history (ontogeny) after they have settled onto the reef. Some species of unicornfishes commence benthic life as herbivores then feed on open water plankton as adults (*Naso annulatus, N. brevirostris, N. maculatus, N. mcdadei* and *N. vlamingii*: Choat 1991, Choat and Clements 1998, Choat et al 2002). For these species, ontogenetic change in diet tends to occur by at least 20cm standard length (SL).

In contrast, parrotfishes pass through a period of carnivory during their early post-settlement phase (Bellwood 1988), before progressively changing to herbivores within the first few weeks of benthic life (by 32mm SL: Bellwood 1988). Similarly, herbivorous pomacentrids undergo ontogenetic changes in diet from an omnivorous to a predominantly herbivorous diet (reviewed in Bellwood 1988).

These differences may be explained by phylogeny (Choat 1991). Acanthuroid fishes (including acanthurids and siganids) settle at relatively large sizes with well-developed sensory, locomotor, and alimentary systems (Leis and Rennis 1983), and can function as herbivores (Choat 1991). Labroids generally, and scarines in particular, settle at smaller sizes (Leis and Rennis 1983). Herbivory in scarines is based on the development of specialized mouthparts (the pharyngeal mill), associated musculature and the alimentary tract (reviewed in Choat 1991). The functioning of the mill may be size dependent, working efficiently only when individuals reach a certain size or mass. Therefore, scarines may feed on copepods while they are small (Bellwood 1988), because their particular mode of feeding has a size threshold. Similarly, Lassuy (1984) suggested that the inclusion of crustaceans in the diet of small juveniles of the herbivorous pomacentrid *Stegastes lividus* may be a result of limited digestive capabilities of these individuals.

TERRITORIAL BEHAVIOUR

Herbivorous coral reef fishes show a variety of feeding modes that are reflected in their movement patterns and behavioral traits (Choat and Bellwood 1985). Some species within the families Acanthuridae, Labridae (Tribe Scarini) and Siganidae feed over a relatively wide area, frequently forming mixed feeding schools and rarely displaying aggression towards other species. In contrast, some species of herbivorous pomacentrids and acanthurids are territorial, and fiercely defend defined areas of reef substratum (usually 2-15m^2 in area: Russ 1987) against the feeding activities of other herbivores (Choat and Bellwood 1985, Choat 1991, Craig 1996). Aggressive defense is focused on those species with similar feeding habitats (i.e. that graze epilithic algal turf and associated biota: Choat and Bellwood 1985, Choat 1991). This behaviour often results in an increase in the standing crop of algae within their territories, as well as differences in species composition, algal diversity and rates of nitrogen fixation by algae compared to other areas (reviewed in Russ 1987 and Klumpp et al 1987). These species can be very abundant, and their territories and algal turfs can cover a high proportion of the reef substratum (up to 77% on reef flats: Klumpp et al 1987).

In the context of coral reef resilience, these territories and their defenders play an important role in coral-algal dynamics on reefs. However, no generality has emerged regarding the relationship between territorial herbivores and coral recruitment, growth and survivorship, and therefore coral reef resilience. On one hand, territorial damselfishes have been considered to have a negative influence on coral recruitment by allowing algae and other fouling organisms to grow and out-compete coral recruits for space (Sammarco and Williams 1982, Wellington 1982, Russ 1987). Conversely, coral recruits that do settle within damselfish territories may benefit from a refuge from other grazers (particularly parrotfishes) that may inhibit their growth and survivorship by consuming live coral in addition to other resources (Wellington 1982, Bellwood and Choat 1990, Rotjan and Lewis 2005, Rotjan et al 2006).

Highly territorial damselfishes can also damage adult coral colonies (Kaufman 1977, Potts 1977, Wellington 1982) and influence coral community structure by causing the mortality of adult corals susceptible to overgrowth by algae, while allowing others that are better competitors to survive (Sammarco and Williams 1982). The exclusion of other herbivores, particularly parrotfishes, may also reduce the rate of bioerosion within damselfish territories (Sammarco et al 1986).

Territorial damselfish (*Dischistodus prosopotaenia*: left) and surgeonfish (*Acanthurus lineatus*: right). Images by G. Allen and A. Lewis respectively.

BIOGEOGRAPHIC AND REGIONAL DIFFERENCES

The tropical oceans of the world comprise four major biogeographic regions: the Indo-West Pacific, Eastern Pacific, Western Atlantic and Eastern Atlantic (Paulay 1997). The Indo-West Pacific and Western Atlantic (including the Caribbean) both have diverse and abundant reef communities, although diversity is much higher in the Indo-West Pacific (Paulay 1997, Veron 2000, Bellwood et al 2003). In contrast, the Eastern Pacific and Eastern Atlantic have only limited reef development (Paulay 1997). These four regions display considerable variation in species composition and diversity, resulting from differences in their evolutionary history and oceanographic conditions (Veron 1995, 2000, Birkeland 1997, Paulay 1997; Johnson et al 1995, Bellwood and Wainwright 2002).

The relative importance of different groups of herbivores in tropical waters is ocean dependent (reviewed in Klumpp et al 1987 and Choat 1991). For example fishes are the dominant group of herbivores in the Indo-Pacific region, while both echinoids and fishes are important in the Western Atlantic (Caribbean).

There are also striking differences in the composition of functional groups of herbivorous reef fishes on both biogeographic and geographic scales (reviewed in Bellwood and Choat 1990, Bellwood et al 2003, 2004). For example, while the Caribbean and the Indo-West Pacific Region share the same suite of functional groups (in broad terms), species richness and taxonomic composition within functional groups is markedly different between and within these regions. This is largely a biogeographic legacy of the evolutionary history of isolation and loss of taxa in the Caribbean basin (Johnson et al 1995, Bellwood and Wainwright 2002).

These differences are demonstrated by the taxonomic composition of functional groups of parrotfishes in different biogeographic regions. For example, both excavators (*Bolbometopon*, *Cetoscarus* and *Chlorurus* species) and scrapers (*Scarus* and *Hipposcarus* species) are present on reefs throughout the Indo-West Pacific, although only scrapers (*Scarus* species) are present in the Eastern Pacific (Bellwood and Choat 1990). While in the Caribbean, most species are scrapers (*Scarus* species), although some larger species appear to be excavators (particularly *Sparisoma viride:* Gygi 1975*).*

PATTERNS OF DISTRIBUTION AND ABUNDANCE WITHIN GEOGRAPHIC AREAS

Within geographic areas, the distribution and abundance of herbivorous reef fishes varies among and within reefs. For example, several studies have described how herbivorous reef fishes vary among reefs at different locations on the continental shelf (Russ 1984a, Williams 1991) and among zones on individual reefs (Russ 1984b).

Variation Among Reefs

Russ (1984a,b) conducted a detailed study of the distribution and abundance of herbivorous reef fishes (parrotfishes, surgeonfishes and rabbitfishes) on the Great Barrier Reef, and found a high degree of variation across the continental shelf. He found that assemblages of herbivorous fishes on inshore reefs were distinct from those on mid and outer shelf reefs, with significantly less species and individuals on inshore reefs. However, patterns of distribution and abundance varied among families. Surgeonfishes increased in abundance and diversity with distance from shore, while rabbitfishes were most abundant and diverse on mid shelf reefs. In contrast, there was no significant difference in diversity or abundance of parrotfishes between outer and mid-shelf reefs, although both diversity and abundance were much lower on inshore reefs. Most species also showed significant cross shelf changes in their abundance, with most of the variability associated with distance from shore (with some species absent from or in low abundance on inshore reefs). On the Great Barrier Reef, this cross shelf variation is greater than the variation associated with latitude (north to south: Williams 1991).

Hoey and Bellwood (2008) also found differences in functional groups of parrotfishes on reefs across the continental shelf on the Great Barrier Reef. They found that inner shelf reefs supported a high density but low biomass of parrotfishes (particularly *S. rivulatus*), which resulted in high rates of scraping and sediment reworking. In contrast outer shelf reefs were characterized by low densities and a high biomass of parrotfishes (particularly *B. muricatum*), so rates of bioerosion and coral predation were high. Mid shelf reefs were characterized by moderate levels of both scrapers and bioeroders. This marked variation in the roles of parrotfishes across the continental shelf suggests that inner, mid and outer shelf reefs are shaped by fundamentally different processes (Hoey and Bellwood 2008), which may be reflected in the composition of benthic communities (Wismer et al 2009).

Variation Within Reefs

A major component of the variability in herbivorous reef fishes is among zones on individual reefs. This variation is largely associated with depth (Russ 1984b), and can be greater than the variability among reefs at different locations on the continental shelf (Russ 1984b).

Herbivorous fishes inhabit most coral reef zones, including reef slopes, crests, reef flats, lagoons and back reefs (Russ 1984b, Hoey and Bellwood 2008). These habitats are defined as (modified from Bellwood and Wainwright 2001):

- Reef slopes are the steeply inclined area on the seaward side of reefs immediately below the reef crest, with depth ranges of 5 to 50m or more.
- Reef crests are the area that marks the transition between the shallow upper areas of the reef flat, and the steeply inclined reef slope (depth 1-4m, 0-1m above mean spring low tide).
- Reef flats are the area behind the reef crest, which can comprise both inner (sheltered) and outer (wave exposed) reef zones (depth 0-1m above mean spring low tide).
- Lagoons are bodies of comparatively shallow water separated from the deeper ocean by a shallow or exposed reef flat. Depth generally ranges from 1-80ms.
- Back reefs, where present, lay at the rear of the reef flat where the reef begins to fall away again into the deeper water of the lagoon.

Russ (1984a) found that herbivorous reef fishes (parrotfishes, surgeonfishes and rabbitfishes combined) occurred in assemblages that were characteristic of each zone. For example, assemblages in deep, outer slope zones were different from those in shallow zones. There were also distinctive assemblages of herbivorous fishes among shallow zones (reef flats, reef crests, lagoons and back reefs).

Each family also showed distinct patterns of zonation within reefs (Russ 1984b). Surgeonfishes and parrotfishes generally had higher numbers of species and individuals on reef crests and lagoons than on reef flats or reef slopes, while siganids had higher numbers of species and individuals in lagoons and back reefs than in other zones.

Functional groups of herbivores also vary among zones on reefs, with bioeroding species most abundant on reef crests and outer reef slopes on Indo-Pacific reefs (Bellwood and Choat 1990, Choat 1991, Fox and Bellwood 2007, Hoey and Bellwood 2008). This suggests that most of the important processes attributable to bioerosion by reef fishes occur on the growing crests of reefs and that the sediment produced contributes to the apron surrounding the reef base (Choat 1991). Some species also appear to feed on living corals and contribute to the sedimentation process by breaking down living coral skeletons (Bellwood 1996). The most important species in this context is *B. muricatum* which is also characteristic of reef crests and outer reef slopes (Hoey and Bellwood 2008).

Russ (1984b) also found that different functional groups of herbivores were distributed differently among zones on the Great Barrier Reef. Grazers were most abundant near windward and leeward edges of reefs, while grazers/detritivores that feed over sand were most abundant in back reefs and lagoons. Scrapers and browsers were more abundant in shallow (reef crests, reef flats and lagoons) than deeper zones (reef slopes and back reefs).

BIODIVERSITY AND FUNCTIONAL REDUNDANCY

Biodiversity is often associated with functional redundancy (Bellwood et al 2003), where richer biotas are considered more likely to have higher levels of functional redundancy than depauperate biotas, and therefore greater resilience (Bellwood et al 2003, 2004, Steneck et al 2002, 2004, Hughes et al 2005). For example, the Caribbean and the Great Barrier Reef share the same suite of functional groups (in broad terms), but species richness and the number of species in each functional group are much higher on the Great Barrier Reef than in the Caribbean (reviewed in Bellwood et al 2004). This suggests that functional redundancy within groups should be higher on the Great Barrier Reef, where the loss of any one species may be compensated for by the actions of another (Bellwood et al 2004). In contrast, functional redundancy within groups should be lower in the Caribbean, and these reefs may be more vulnerable to minor changes in biodiversity (Bellwood et al 2004).

However the evidence for this is equivocal (reviewed in Bellwood et al 2003, 2004). While high diversity does provide the potential for functional redundancy, redundancy in some functional groups can be limited. For example, despite high diversity on Indo-Pacific reefs, the potential for replacement of parrotfishes by other functional groups is limited because they are the primary group of bioeroders on reefs that consume structural reef carbonates. Echinoids are functional replacements only in that they remove carbonate, because erosion by echinoids is very different to that of parrotfishes (echinoids are not restricted to convexities as are parrotfishes: Bellwood et al 2003). Echinoids are also far more destructive bioeroders than parrotfishes, and large numbers can be highly detrimental to reefs (Eakin 1996). This demonstrates that the loss of functional redundancy can come at a cost, even when some members of a group can compensate for others.

Another important factor to consider is response diversity (reviewed in Bellwood et al 2004). If all species within a functional group respond similarly to pressures such as overfishing, then higher biodiversity will not afford additional protection (Elmqvist et al 2003, Hughes et al 2005). In such cases, the insurance value of high species richness and functional redundancy may be negligible.

SIZE AND ROLE IN ECOSYSTEM PROCESSES

When considering the role of herbivores in coral reef resilience, it is important to consider size, since a number of studies on the feeding ecology of parrotfishes on both Pacific and Atlantic reefs have found significant differences in the impact of different size classes on the reef substratum (Bruggemann et al 1994, Bonaldo and Bellwood 2008).

For example, Bonaldo and Bellwood (2008) investigated the effect of size on the functional role of the parrotfish *Scarus rivulatus* on the Great Barrier Reef. They found that small parrotfishes scrape a greater substratum area per unit biomass than larger parrotfishes, while larger parrotfishes take a greater volume of material per unit biomass. Furthermore smaller parrotfishes usually only crop the algal surface and have little or no visible effect on the consolidated substratum, while large parrotfishes seem to affect both algal cover and the underlying substratum, and are responsible for the effective removal of algae and opening new colonization sites on reefs. Consequently large individuals appear to play a more significant role in coral reef resilience than small individuals (Bonaldo and Bellwood 2009).

School of large scrapers (parrotfishes: *Scarus ghobban*) feeding on epilithic algal turf and scraping the substratum clean. Image by G. Allen.

MONITORING FUNCTIONAL GROUPS OF HERBIVOROUS REEF FISHES

WHAT CAN BE ACHIEVED?

It is important to remember that monitoring coral reef resilience is a new approach, and the underlying science is still developing. At present, we have a good understanding of the role that functional groups of herbivores play in avoiding coral-algal phase shifts, although their role in reversing phase shifts is less clear (see *Reversing Coral-Algal Phase Shifts*).

Therefore, this protocol will focus on monitoring species that play an important role in avoiding coral-algal phase shifts. While some of these species may also be important in reversing phase shifts, other key species may be missing. Further studies are now required to ensure that other key species are included in future. Other refinements may also be required as more information becomes available regarding the diet and behaviour of herbivorous reef fishes (see *Refining the Protocol*).

MONITORING FUNCTIONAL GROUPS OF HERBIVOROUS REEF FISHES THAT PREVENT CORAL-ALGAL PHASE SHIFTS

The aim of this section is to provide field practitioners with a practical protocol for monitoring functional groups of herbivores that play important roles in preventing coral-algae phase shifts. Functional groups were selected that are central to the capacity of reefs to resist coral-algal phase shifts, and regenerate and retain critical functions after disturbance (see *Role of Functional Groups of Herbivorous Reef Fishes*). For the Asia Pacific Region, these are predominantly herbivorous reef fishes (see *Biogeographic and Regional Differences*).

Herbivorous reef fishes do not constitute an ecologically uniform group (Choat 1991), and there are four functional groups that each play an important role in preventing coral-algal phase shifts (see *Functional Groups*): scrapers/small excavators, large excavators/bioeroders, grazers/detritivores, and browsers. Species of each of six major families of herbivorous reef fishes [Acanthuridae, Labridae (Tribe Scarini), Siganidae, Kyphosidae, Ephippidae and Pomacanthidae] were assigned to each of these functional groups based on a case study for the Asia Pacific Region (see *Assigning Species to Functional Groups*). Species were assigned based on best available information from the scientific literature and expert opinion.

Survey methods are provided for monitoring these species (see *Monitoring Methods*), including specific advice regarding sampling design, site selection, standardising coral reef exposures and zones, survey timing and frequency, and underwater visual census methods for both rapid assessments and long term monitoring. Advice is also provided on how to design a new program to specifically monitor functional groups of herbivores, how to modify existing programs to include functional groups of herbivores, and how to integrate monitoring for resilience with other monitoring objectives.

While this protocol is based on the Asia Pacific Region, it can be adapted easily for other regions by modifying the species monitored in each functional group (see *Adaptations for Other Biogeographic and Geographic Regions*). Methods can also be modified for field practitioners with lower levels of expertise (see *Simplifying the Protocol*).

Functional Groups

Four functional groups of herbivorous reef fishes were identified that play a critical role in coral reef resilience: scrapers/small excavators, large excavators/bioeroders, grazers/detritivores and browsers. Their roles differ in terms of how they feed, what they consume, and their impact on the underlying substratum. Each play different and complementary roles in coral reef resilience.

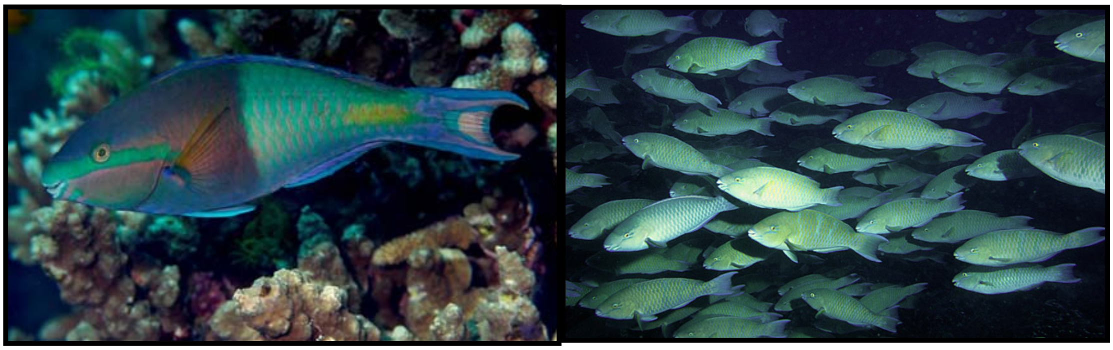
Scrapers/small excavators

There are two functional groups of parrotfishes that show major differences in their jaw morphology and feeding behaviour (Bellwood and Choat 1990): scrapers and excavators. Both feed on epilithic algal turf, and remove some component of the reef substratum as they feed (Bellwood and Choat 1990). They differ in the amount of the substratum they remove while feeding (Bellwood and Choat 1990), and their contribution to ecosystem processes such as bioerosion.

The majority of parrotfishes are scrapers, including most *Hipposcarus* and *Scarus* species (Bellwood and Choat 1990). They take non-excavating bites and remove algae, sediment and other material by closely cropping or scraping the reef surface, leaving shallow scrape marks on the reef substratum (Bellwood et al 2004, Hoey and Bellwood 2008).

Excavating species include *Bolbometopon muricatum, Cetoscarus bicolor* and all species of the genus *Chlorurus (*Bellwood and Choat 1990). They differ from scrapers by taking deeper excavating bites and removing greater quantities of substrata with each bite (Bellwood and Choat 1990, Hoey and Bellwood 2008).

Scrapers and small excavators (individuals <35cm SL) play similar roles in coral reef resilience by limiting the establishment and growth of macroalgae while intensely grazing epilithic algal turfs (Paddock et al 2006, Hughes et al 2007). They also provide areas of clean substratum that facilitates the settlement, growth and survival of coralline algae and corals (Steneck 1988, Hoey and Bellwood 2008).



Scrapers: *Scarus flavipectoralis* (left) and *S. ghobban* (right). Images by G. Allen.

Large excavators/bioeroders

Large excavators play a similar role in coral reef resilience to scrapers/small excavators. However, they are also major agents of bioerosion on reefs (Bellwood and Choat 1990, Choat 1991, Bellwood et al 2003), removing dead coral and exposing hard, reef matrix for settlement by coralline algae and corals (Bellwood and Choat 1990, Bellwood et al 2003, 2004).

Large excavators/bioeroders include all large individuals (≥35cm SL) of excavating species: *Bolbometopon muricatum, Cetoscarus bicolor* and all species of the genus *Chlorurus* (Bellwood and Choat 1990). *B. muricatum* is the largest species of excavating parrotfish on coral reefs (up to 120cm total length: Bellwood et al 2003), and each individual ingests 5 tonnes of reef carbonate per year, almost half of which is live coral skeletons (Bellwood et al 2003). Due to the large volume of live and dead coral this species consumes, it is considered one of most important bioeroders on coral reefs (Bellwood et al 2003). Four other species of Indo-Pacific parrotfish have also been observed grazing on live coral skeletons (including *Chlorurus microrhinos* and *C. bicolor*), although live corals only account for a small proportion of their diet (see *Key Families and Their Feeding Modes*). Other reef fishes also feed on live corals and their skeletons, including puffers (*Arothron* species: Myers 1999), and occasionally porcupinefishes (*Diodon* species) and triggerfishes (Cole et al 2008). However, they are not included here because they are generally uncommon in the Asia Pacific Region, and are unlikely to play significant roles in coral reef resilience.

Because *B. muricatum* is a coral predator, its role in coral reef resilience is complex. However, this species is a natural component of coral reefs in the Asia Pacific Region, and in the absence of other impacts, it is not a threat to coral reef resilience. Unlike other coral predators like crown-of-thorns starfish and the snail *Drupella*, *B. muricatum* does not undergo population outbreaks and cause serious degradation to coral communities. In fact on relatively unexploited oceanic reefs, total ingestion rates balance estimated rates of reef growth (Bellwood et al 2003). This species also tends to target fast growing corals such as *Acropora* and *Pocillopora* (Bellwood et al 2003), and can act as an important agent of intermediate disturbance (Connell 1978) on reefs, contributing to the maintenance of high biodiversity. While chronic predation by coral-eating fishes (including parrotfishes) may exacerbate the effects of climate induced bleaching on coral communities (Cole et al

2008), *B. muricatum* is unlikely to cause serious damage because it is extremely vulnerable to overexploitation and is now rare or uncommon throughout much of its range (Dulvy and Polunin 2004, Chan et al 2007).

Despite being coral predators, large excavators/bioeroders play a critical role in coral reef resilience by opening up new sites for colonization by coralline algae and corals, and their absence may have serious consequences for coral reef resilience.

Excavators: *Bolbometopon muricatum* (left) and *Chlorurus microrhinos* (right). Images by R. Hamilton and G. Allen respectively**.**

<u>Grazers/detritivores</u>

Grazers play an important role in coral reef resilience by intensely grazing epilithic algal turfs, which can also limit the establishment and growth of macroalgae (Paddock et al 2006, Hughes et al 2007). Unlike scrapers and excavators, they do not scrape or excavate the reef substratum as they feed.

Grazers include most rabbitfishes (except *Siganus canaliculatus* and *S. lineatus*), small angelfishes (all *Centropyge* species), and many species of surgeonfish including all *Zebrasoma* and *Acanthurus* species except those that feed on exclusively on plankton (see *Key Families and their Feeding Modes*) or are grazers/detritivores (see below). *Acanthurus lineatus* is included in this group because while they are predominately territorial (Polunin and Klumpp 1989, Craig 1996) they also move around and consume algal turf outside their territories (Craig 1996).

Grazers: surgeonfish (left: *Acanthurus triostegus*) and angelfish (right: *Centropyge loricula*). Images by G. Allen.

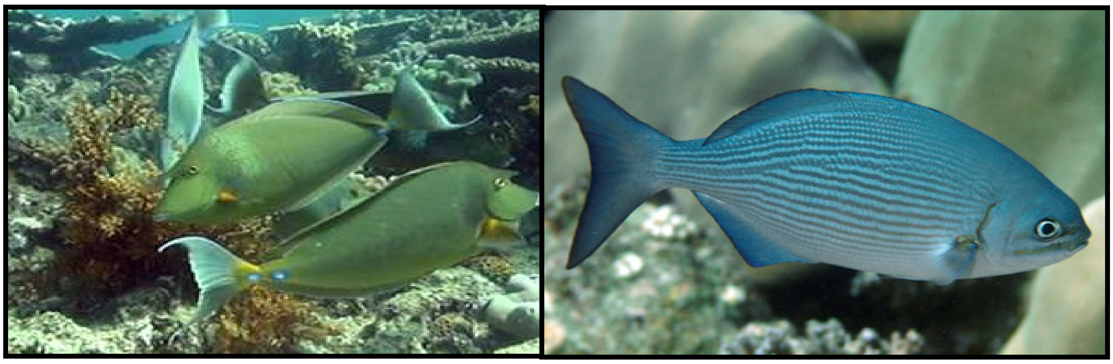
Ringtail surgeonfishes of the genus *Acanthurus* (see *Key Families and their Feeding Modes*) are grazers/detritivores that feed on a combination of epilithic algal turf, sediment and some animal material (Choat 1991). Similarly, the rabbitfish *S. lineatus* appears to be a grazer/detritivore, feeding primarily on off-reef detrital aggregates (Fox et al in press). Although only a small proportion of their diet is algal turf, grazers/detritivores are combined with grazers because many are schooling species that can be abundant and consume significant amounts of epilithic algal turf. They also play similar roles in coral reef resilience by removing macroalgae before it becomes established (see above).

Grazers/detritivores: rabbitfish (left: *Siganus lineatus*) and ringtail surgeonfish (right: *Acanthurus nigricauda*). Images by G. Allen.

Browsers

Browsers consistently feed on macroalgae. They select individual algal components and remove only algae and associated epiphytic material (Hoey and Bellwood 2008). Browsers play an important role in reducing coral overgrowth and shading by macroalgae, and may play an important role in reversing coral-algal phase shifts (see *Reversing Coral-Algal Phase Shifts*).

Browsers include some unicornfishes that feed on algae throughout their lives (e.g. *N. unicornis*), and small individuals of other species that feed on algae when they are small (<20cm SL) and plankton as adults (≥20cm SL e.g. *N. annulatus*: see *Ontogenetic Changes in Feeding Modes*).



Browsers: unicornfish (left: *Naso unicornis*) and rudderfish (right: *Kyphosus vaigiensis*). Images by A. Hoey and G. Allen respectively.

Also included are rudderfishes, batfishes (which will eat both plant and animal material: Bellwood et al 2006), a rabbitfish (*S. canaliculatus*: Fox and Bellwood 2008) and parrotfishes of the genus *Calotomus* (Choat 1991, Choat et al 2002) and *Leptoscarus*.

Browsers: batfish (left: *Platax teira*).
Image by G. Allen.

Assigning Species to Functional Groups

The practical application of monitoring key functional groups of herbivores requires assigning each species to a functional group in each geographic area. The following is an example of how to assign species to functional groups, based on a case study for the Asia Pacific Region.

This study focuses on six families of herbivorous reef fishes that each play important roles in coral reef resilience: Acanthuridae, Ephippidae, Kyphosidae, Pomacanthidae, Labridae (Tribe Scarini) and Siganidae (see *Key Families and Their Feeding Modes*). Other herbivorous reef fishes are not included because:

- Small, cryptic families (blennies and gobies) are not amenable to visual census techniques, and are unlikely to make a significant contribution in terms of ecosystem resilience.
- Damselfishes are small, and hard to identify. They also comprise a wide variety of diets (herbivores, detritivores, algae grazers and planktivores), and it is a complex and difficult task to assign them to functional groups. Species that are herbivorous also tend to be small and contribute less in terms of ecosystem resilience, because they are territorial and farm algae for their own consumption (see *Territorial Behaviour*).
- While some filefishes, triggerfishes and a wrasse (*Pseudodax mollucanus*: Cowman et al in press) may be herbivores, their trophic status has not been confirmed (Myers 1999, Randall et al 1996, G.P Jones *unpubl. data*).

A species list for each family in the Asia Pacific Region was provided by Dr. Gerry Allen (Allen et al 2003). Each species was assigned to a functional group (see *Functional Groups* above) based on the best available literature, and expert opinion (Table 1). Species that did not belong to one of these functional groups were deleted, including species that feed exclusively on detritus (*Ctenochaetus* species) or plankton (some *Acanthurus* and *Naso* species: see *Key Families and Their Feeding Modes*).

Ontogenetic changes in diet were taken into account for some species (see *Ontogenetic Changes in Feeding Modes*). For example, only juveniles of some species of unicornfishes were included, because they undergo an ontogenetic change from browsers to planktivores at around 20cm SL. However, ontogenetic changes in parrotfishes from carnivory to herbivory were not considered, because this change occurs at such a small size (32mm SL: Bellwood 1988) that juvenile parrotfishes that are still carnivores would not be counted using underwater visual census methods (see *Monitoring Methods* below).

These species will provide the basis of a monitoring program for key functional groups of herbivores on reefs in the Asia Pacific Region.

Table 1. Functional group of each species of herbivorous reef fish in the Asia Pacific Region to be used for resilience monitoring.

Family (Tribe)	**Species**	**Functional Group**	**Source**
Acanthuridae	*Acanthurus achilles*	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus auranticavus	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus barine	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus blochii	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus dussumieri	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus fowleri	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus guttatus	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus japonicus	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus leucocheilus	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus leucopareius	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus leucosternon	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs
	Acanthurus lineatus	Grazers/detritivores	Choat et al 2002
	Acanthurus maculiceps	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus nigricans	Grazers/detritivores	Choat et al 2002
	Acanthurus nigricauda	Grazers/detritivores	Choat et al 2002
	Acanthurus nigrofuscus	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus nigroris	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus olivaceus	Grazers/detritivores	Choat et al 2002
	Acanthurus pyroferus	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus tennentii	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus triostegus	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus tristis	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Acanthurus xanthopterus	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
	Naso annulatus	Browsers (<20cm SL only)	Choat 1991, Choat & Clements 1998; Choat et al 2002
	Naso brachycentron	Browsers	J.H. Choat pers. obs.
	Naso brevirostris	Browsers (<20cm SL only)	Choat & Clements 1998; Choat et al 2002
	Naso elegans	Browsers	J.H. Choat pers. obs.
	Naso lituratus	Browsers	Choat & Clements 1998
	Naso maculates	Browsers (<20cm SL only)	J.H. Choat pers. obs.
	Naso mcdadei	Browsers (<20cm SL only)	J.H. Choat pers. obs.
	Naso tonganus	Browsers	Choat 1991, Choat et al 2002
	Naso unicornis	Browsers	Choat 1991, Choat & Clements 1998; Choat et al 2002

Family (Tribe)	**Species**	**Functional Group**	**Source**
Acanthuridae cont.	*Zebrasoma desjardinii*	Grazers/detritivores	Choat 1991
	Zebrasoma flavescens	Grazers/detritivores	Choat 1991
	Zebrasoma rostratum	Grazers/detritivores	Choat 1991
	Zebrasoma scopas	Grazers/detritivores	Choat 1991, Choat et al 2002
	Zebrasoma veliferum	Grazers/detritivores	Choat 1991
Ephippidae	*Platax batavianus*	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
	Platax boersi	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
	Platax orbicularis	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
	Platax pinnatus	Browsers	Bellwood et al 2006
	Platax orbicularis	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
	Platax teira	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
	Zabidius novemaculeatus	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
Kyphosidae	all species	Browsers	Clements & Choat 1997; Choat et al 2002
Pomacanthidae	all *Centropyge* species	Grazers/detritivores	J.H. Choat & D.R. Bellwood pers. obs.
Labridae (Scarini)	*Bolbometopon muricatum*	Scrapers/small excavators (<35cm SL);	Bellwood & Choat 1990; Bellwood et al 2003, J.H. Choat & D.R. Bellwood pers. obs..
		Large excavators/bioeroders (≥35cm SL)	
	Calotomus carolinus	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
	Calotomus japonicus	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
	Calotomus zonarchus	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
	Calotomus spinidens	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
	Cetoscarus bicolor	Scrapers/small excavators (<35cm SL);	Bellwood & Choat 1990, J.H. Choat & D.R. Bellwood pers. obs.
		Large excavators/bioeroders (≥35cm SL)	
	Chlorurus bleekeri	Scrapers/small excavators	Bellwood & Choat 1990
	Chlorurus bowersi	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Chlorurus capistratoides	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Chlorurus enneacanthus	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Chlorurus frontalis	Scrapers/small excavators (<35cm SL);	Bellwood & Choat 1990, J.H. Choat & D.R. Bellwood pers. obs.
		Large excavators/bioeroders (≥35cm SL)	
	Chlorurus japanensis	Scrapers/small excavators	Bellwood & Choat 1990
	Chlorurus microrhinos	Scrapers/small excavators (<35cm SL);	Bellwood & Choat 1990; Bellwood et al 2003, J.H. Choat & D.R. Bellwood pers. obs.
		Large excavators/bioeroders (≥35cm SL)	
	Chlorurus oedema	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Chlorurus perspicillatus	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.

Family (Tribe)	**Species**	**Functional Group**	**Source**
Labridae (Scarini) cont.	*Chlorurus sordidus*	Scrapers/small excavators	Bellwood & Choat 1990
	Chlorurus strongylocephalus	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Chlorurus troschelii	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Hipposcarus hairid	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Hipposcarus longiceps	Scrapers/small excavators	Bellwood & Choat 1990
	Leptoscarus vaigiensis	Browsers	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus altipinnis	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus chameleon	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus dimidatus	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus dubius	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus festivus	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus flavipectoralis	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus forsteni	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus frenatus	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus fuscocaudalis	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus ghobban	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus globiceps	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus hypselopterus	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus longipinnis	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus niger	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus oviceps	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus prasiognathos	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus psittacus	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus quoyi	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus rivulatus	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus rubroviolaceus	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus scaber	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus schlegeli	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus spinus	Scrapers/small excavators	Bellwood & Choat 1990
	Scarus tricolor	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus virdifucatus	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus xanthopleura	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
	Scarus sp.	Scrapers/small excavators	J.H. Choat & D.R. Bellwood pers. obs.
Siganidae	*Siganus canaliculatus*	Browsers	Mantyka and Bellwood 2007, Fox and Bellwood 2008
	All other species	Grazers/detritivores	Choat 1991, Choat et al 2002, Fox et al in press, J.H. Choat & D.R. Bellwood pers. obs.

Monitoring Methods

The aim of this section is to provide practical advice to field practitioners on how to design and implement a monitoring program (see *Glossary*) for coral reef resilience based on functional groups of herbivores. Methods are described that focus on sampling design, site selection, standardising coral reef exposures and zones, survey timing and frequency, and underwater visual census methods (summarized in Table 2). Survey methods are also identified for a range of situations.

Sampling Design

Ideally, as for all monitoring programs, a hierarchical or stratified sampling design should be used to address variation at multiple scales: coral reef type (taking into account geomorphology, distance to shore and varying degrees of exposure to wave energy), individual reefs within each coral reef type, replicate sites within each reef, and replicate censuses within each zone (e.g. reef slopes, crests, flats and back reefs) at each site.

Detailed advice on how to design a hierarchical sampling design is provided by English et al (1997), and an excellent example for assessing functional groups of herbivores is provided by Hoey and Bellwood (2008). However in most situations time is limited and it will be necessary to standardize on one exposure and one or two zones (see *Standardising Coral Reef Exposures and Zones* below).

Site Selection

Sites should be selected after a general survey of the area to ensure that they are representative of that reef (English et al 1997), and there is adequate space to conduct the monitoring program (at least 400m for timed swims, and 650m for belt transects and long swims combined: see *Underwater Visual Census Methods* below). All sites should be similar with respect to physical characteristics, slope and coral cover. Where possible, sites should be separated from each other by a reasonable distance (at least several hundred meters, preferably 500m), and the exact location of each site should be recorded with a Global Positioning System **(**GPS) so it can be relocated in future. Manta tows can provide a useful method for site selection. For more information see English et al (1997).

Standardising Coral Reef Exposures and Zones

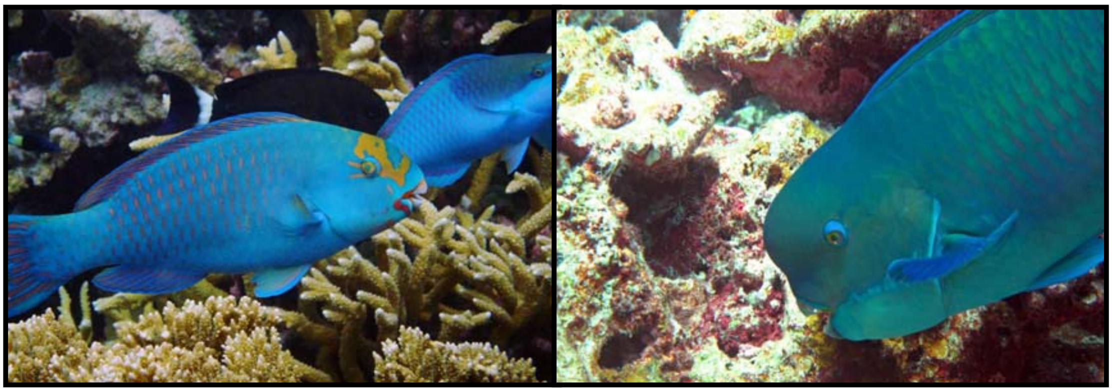
Key functional groups of herbivores inhabit most coral reef areas. Within geographic areas, much of the variation in the size and structure of herbivore populations is associated with reef exposure (exposed versus sheltered reefs) and zone, with different species favouring different exposures and zones (see *Patterns of Distribution and Abundance within Geographic Areas).*

Since there is rarely enough time to monitor all exposures and zones, monitoring programs should standardise on exposed linear reef fronts. Exposed linear reef fronts are recommended because they are consistently available in most coral reef areas, and they generally comprise a high diversity and abundance of key functional groups of herbivorous reef fishes (see *Patterns of Distribution and Abundance within Geographic Areas*). They also comprise the growing crests of reefs, where bioeroding species are most abundant and bioerosion is highest (Russ 1984b, Choat 1991, Fox and Bellwood 2007, Hoey and Bellwood 2008).

Within exposed reef fronts, monitoring should focus on two zones: outer reef slopes and crests (see *Variation Within Reefs*). Reef slopes are recommended because they are consistently available on most reefs, and they support a high diversity and abundance of herbivores. If reef crests are present, they should be surveyed because they generally have the highest numbers of acanthurids and scarines (Russ 1984b). They also comprise a different fauna to reef slopes (Russ 1984b, Williams 1991), and some key species tend to be more abundant on crests (e.g. *B. muricatum*, *C. frontalis*, *C. microrhinus, C. sordidus* and some *Nasos*: Russ 1984b, Choat 1991, Hoey and Bellwood 2008). However distinctive reef crests are not present in all areas. In those situations, reef slopes should be surveyed at two depths: 8-10m and 2-4m.

Surveys should be conducted along the reef slope first, followed by the reef crest (see *Diver Safety* below). This will provide a good appreciation of not only these habitats, but also the outer reef flat which will be visible during the survey of the reef crest. While lagoons and back reefs are important in some areas, they are not recommended for this protocol because they are not uniformly available on all reefs and are less common than reef crests. If time permits, it may be interesting to survey these habitats, although the results will only be comparable with similar zones in other areas. If censuses are conducted in lagoons and back reefs, they should cover a shallow depth range (1-10m) where there are coral communities rather than bare sand.

Similarly, if reef channels, passes or distinctive reef promontories (points) are present in an area, they can be surveyed also (since large reef fishes are often abundant in these areas), but since they represent different habitat types they should be analysed separately.



Two parrotfish species that tend to be more abundant on reef crests: *Chlorurus frontalis* (left front) and *C. microrhinos* (right feeding). Images by K. Pollock and R. Salm respectively.

Survey Timing and Frequency

Where possible, surveys should be conducted during daylight hours from 9am to 4pm to avoid peak spawning times for some species (reviewed in Clayton 2004). However since spawning in other species can occur at other times of the day, counts should be suspended if a spawning aggregation (see *Glossary*) is encountered.

Long term monitoring will provide the most useful information for assessing functional groups of herbivores. This usually involves an initial baseline survey, followed by surveys on a regular basis – usually every one to three years (e.g. Sweatman et al 2005). If the coral reefs in the study area are in good condition with healthy herbivore populations and no obvious threats, surveys every three years may be adequate for monitoring long term trends in herbivore populations. However, if herbivore populations are low, threats are high, or if management actions have been taken to protect species targeted in the monitoring program, more regular monitoring (yearly) may be required to monitor management success for adaptive management (see *Adaptive Management*). Additional surveys may also be required after large scale disturbances, such as tropical storms, crown-of-thorns starfish outbreaks or mass coral bleaching events.

Rapid assessments usually comprise single surveys undertaken on one occasion (e.g. Green et al 2006a, Hoey and Bellwood 2008).

Underwater Visual Census Methods

Underwater visual census methods are the most effective way to monitor herbivorous reef fishes, particularly in remote locations (Choat and Pears 2003). Monitoring methods need to be carefully selected to suit the aim of the monitoring program, and the biology and behavior of the species and coral reef habitats. For a detailed description of the range of underwater visual census methods, and the costs and benefits of each method, see Hill and Wilkinson (2004).

In this document, recommendations are made regarding developing new monitoring programs, and modifying existing monitoring programs, to census functional groups of herbivorous reef fishes. Whatever methods are used, both abundance and size (total length, TL[2] in cms) of fishes should be monitored, since there is a strong negative relationship between biomass of herbivorous fishes and macroalgal cover (Williams and Polunin 2001, Mumby et al 2006), and therefore coral reef resilience.

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[2] TL is used because it is easier to estimate underwater than SL. Therefore, TL will be used instead of SL for size cutoffs for species that change feeding modes throughout their lives (see Table 1). Since the greatest error in this method is underwater size estimation, TL can be considered equal to SL for this purpose.

The way in which biomass is proportioned (i.e. the size structure of the populations - lots of small individuals versus a few large ones) is also important, because different size structures will have different impacts on coral reef resilience (see *Size and Role in Ecosystem Processes*). Large and small individuals of some species also have different feeding modes and need to be assigned to different functional groups (see Table 1).

If a new monitoring program is developed to monitor key functional groups of herbivores, then one of two methods may be appropriate depending on program objectives, time constraints, and the degree of precision required: timed swims and a combination of belt transects and long swims.

Timed swims are much faster than transect methods that require measuring tapes to be deployed (Hill and Wilkinson 2004). Therefore, they allow for larger areas and more sites to be surveyed in less time, and are recommended for the rapid assessment of functional groups of herbivores. However, timed swims are less precise than belt transects for most species (Hill and Wilkinson 2004), and transects should be used if a higher degree of precision is required (e.g. for long term monitoring: Wilkinson et al 2003, Hill and Wilkinson 2004), there are less time constraints, and if conditions are suitable for laying tapes (i.e. low current and wave exposure).

Timed swims are also less amenable to monitoring for multiple objectives than are belt transects. For example, many field practitioners are interested in developing monitoring programs that will simultaneously assess coral reef health (coral and reef fish communities, or indicator species), the status of key fisheries species, and coral reef resilience. Monitoring for multiple objectives requires monitoring a wide range of species, and there is a limit to how many species can be censused effectively during a timed swim, which generally comprises only one pass of an area (English et al 1997, Hill and Wilkinson 2004). Belt transects are more amenable to monitoring for multiple objectives, because they allow for more species to be quantitatively assessed by multiple passes of the transects (English et al 1997, Sweatman et al 2005). However, if belt transects are used, they need to be combined with a long swim method that provides the most effective method for quantifying the abundance, biomass and size structure of populations of large, vulnerable reef fishes, including large excavators/bioeroders (Choat and Pears 2003).

Stationary plot methods are another method that is often used to census reef fishes. These methods are not recommended for new monitoring programs for herbivorous fishes, because they are not suitable for censusing small species (e.g. *Centropyge*: see Hill and Wilkinson 2004) or large species where it is necessary to cover a large census area (Choat and Pears 2003). This method is also not recommended because it maximises diver affects (because the observer is in the middle of the census area and therefore maximises the chances of attracting some species and repelling others), and maximises error by estimating every boundary. Stationary plot methods are also less suitable than belt transects for field practitioners who require a higher degree of precision (Hill and Wilkinson 2004), and who are interested in developing monitoring programs to assess multiple objectives (because they are less effective for monitoring a wide range of species: Hill and Wilkinson 2004).

In many situations, monitoring programs already exist, most of which are focused on monitoring key fisheries species or indicators of coral reef health (see review in Hill and Wilkinson 2004). They are generally based on one of three census methods: timed swims (roving diver techniques), belt transects, and stationary plots (see Hill and Wilkinson 2004). These monitoring programs can be easily modified to include functional groups of herbivores by ensuring herbivores are counted along with other species of interest, and that the results for herbivores and other groups are analysed separately (see *Data Analysis*).

The following is a description of the two preferred methods for monitoring functional groups of herbivores: timed swims and a combination of belt transects and long swims (summarized in Table 2). Examples of modified stationary plot methods for censusing herbivorous reef fishes are provided in Williams and Polunin (2001) and Ledlie et al (2007).

For both methods, observers should count all individuals of the study species in the census area from the reef substratum to the water surface, and be careful not to re-census fish that have left and subsequently re-entered the census area (Hoey and Bellwood 2008). Care should also be taken not to count the same fish twice in different areas (Hoey and Bellwood 2008). All data should be recorded directly onto pre-prepared data sheets on underwater paper (Appendix 1), and individuals should be recorded based on taxonomic group during the survey and assigned to functional groups later during data analysis.

It is also important to minimise disturbance to the fish communities being counted by not driving the boat over the area prior to the census, the observers being the first people to swim through the census area, swimming very quietly while censusing, and waiting for at least 5mins after getting in the water before starting the census (English et al 1997).

Timed Swims

Timed swims are the preferred method for rapidly assessing functional groups of herbivorous reef fishes (see above). The method described here is a combination of two timed swim methods developed specifically to survey functional groups of parrotfishes (Hoey and Bellwood 2008) and large, vulnerable reef fishes (including large parrotfishes: Choat and Pears 2003). Where possible, these methods should be combined to simultaneously monitor all functional groups of herbivorous as follows.

One census should be conducted at each site, which includes two zones (reef slopes and crests). Each census consists of two observers swimming side by side along the reef slope parallel to the reef crest at a consistent depth of 8-10m for 20mins, and then along the reef crest for 20mins (just below the reef crest so it is possible to simultaneously monitor the reef crest, outer reef flat and reef slope). If the zones are separated by more than 20m (Hoey and Bellwood 2008) and there is not a strong current, they can be surveyed in different directions (i.e. surveying the reef slope in one direction, and then surveying the reef crest back in the other direction). However, if the zones are less than 20m apart or there is a strong current, they should be surveyed consecutively in the same direction.

Each observer censuses different components of the fish fauna, counting different sized individuals using different transect widths. One observer swims 1-2m above the substratum, counting and estimating the size of all small (<20cm TL and more than the minimum size for inclusion: see below) and medium (20 to <35cm TL) sized individuals of the species listed in Table 1 using a transect width of 5m (2.5m either side of the observer). The second observer swims 2-3m above the substratum and next to or slightly behind the other observer (so as not to disturb the small and medium sized fishes), counting and estimating the size of large (≥35cm TL) individuals of the same species on 20m wide transects (10m either side of the observer). If only one observer is available, two timed swims should be conducted through the area, where medium and large sized individuals are counted on the first pass, and small individuals are counted on the second pass.

A minimum size of 10cm TL should be used to exclude juveniles that are not amenable to rapid visual census methods (Hoey and Bellwood 2008), except for small species (*Acanthurus nigrofuscus, A. triostegus, Zebrasoma scopas* and *Centropye* species) where a minimum size of 5cm should be used (or most individuals will not be recorded). Fishes should also be assigned to size categories, where 2.5cm size categories are used for fishes less than 10cm in size, and 5cm size categories are used for fishes 10cm or greater in size (i.e. 5 to <7.5cms, 7.5 to <10cms, 10 to <15cms, 15 to <20cms etc).

Prior to commencing each survey, observers should calibrate their swimming speeds (see below), the accuracy of their transect width estimates (using tape measures), and their size estimation of fishes underwater (see English et al 1997).

In order to calculate fish density and biomass (see *Data Analysis*), it will be necessary to calculate the area covered by each timed swim (distance covered by each timed swim multiplied by transect width). The width of each transect is 5m for small and medium sized individuals, and 20m for large individuals. Where possible, the length of each timed swim should be measured by marking the beginning and end of each swim with a differential GPS, and measuring the distance between them using the GPS (Bellwood and Wainwright 2001). Alternatively, a GPS can be attached to a floating buoy that is towed by the divers, which can more accurately record their track. The buoy will also help the boat drivers keep track of the divers while they are swimming (see *Diver Safety* below).

If that is not possible, the average distance covered by both observers in a 20 min period can be estimated by measuring the distance they swim in 5mins in a range of situations (current speeds, high and low density of fishes) and converting this to an average distance covered in 20mins (Choat and

Pears 2003). This method is less precise than using a differential GPS, and the GPS should be used if possible.

The optimal distance covered in each timed swim should be 400m to ensure enough distance is covered to encounter highly mobile species that tend to be rare, patchy or clumped in distribution (Choat and Pears 2003). If observers are consistently covering distances less than 400m in each timed swim (e.g. due to a large number of small and medium sized individuals that one observer has to count), the methods should be modified until an average distance of 400m per timed swim is attained. This can be achieved by either: 1) Increasing the duration of each swim from 20mins to 25 or 30mins; 2) Reducing the transect width for small and medium sized individuals from 5m to 3m (1.5m either side of the observer); or 3) Changing the fish sizes that each observer has to count so that both observers are counting a similar number of individuals (e.g. by one observer counting small individuals only, and the other observer counting medium and large sized individuals). In that situation, transect widths for each size group (small, medium or large) remain the same.

Combination of Belt Transects and Long Swims

Belt transects should be used where a higher degree of precision is required or when monitoring for multiple objectives, and when time and conditions are suitable for laying tapes at each site. This method provides the most effective technique for monitoring most small (<20cm TL and more than the minimum size for inclusion) to medium (20 to <35cm TL) sized reef fishes that are amenable to visual census techniques, including most of the species listed in Table 1. However, to effectively monitor all of the species listed in Table 1, it will be necessary to combine the transects with a long swim method that will provide more precise estimates of the abundance, biomass and size structure of large (≥35cm TL), highly mobile species, that tend to be rare, patchy or clumped in distribution (including large parrotfishes: Choat and Pears 2003). If there is insufficient staff, time or resources to conduct both methods, the survey should comprise just belt transects for all sizes. However if long swims are not conducted, the density, biomass and size structure of large individuals of key species will be underestimated.

Five 50m belt transects should be used to monitor all species listed in Table 1. If there is insufficient time to complete five 50m transects at each site, it is better to do less replicates (a minimum of three) rather than use shorter transects, because longer transects are more appropriate for most species (English et al 1997).

Transects should be laid consecutively along a consistent depth contour of 8-10m parallel to the reef crest, with the start of each transect separated by at least 5m from the end of the previous transect (English et al 1997). Transect tapes should be laid during the first pass of the transects by an assistant following the observer (to minimise disturbance to the fish communities being counted: Green et al 2006b). The tapes should then remain *in situ* until all surveys are completed at that site. Fish counts (i.e. each pass of the transect) should be separated by a waiting period of 5 to 10 minutes between counts.

Different transect widths should be used for different sized individuals as described for the timed swims (5m wide for small and medium sized individuals, and 20m wide for large individuals). If two observers are available, all sizes can be counted on a single pass of the transects as described for the timed swims, where one observer counts small to medium sized individuals and the other counts large individuals. If only one observer is available, different sized individuals should be counted on different passes of the transects, with medium and large fishes counted on the first pass, and small individuals counted on the second pass. Each pass should be separated by a waiting period of 5 to 10 minutes between counts to minimise disturbance to the fish communities.

Diver surveying herbivorous reef fishes using belt transects. Image by D. Wachenfeld.

Once the transects have been completed, a long swim should be conducted starting from the end of the transects and continuing in the same direction, so the observers do not cover the same area as the transects (to minimise disturbance to fish communities). Long swim methods are described by Choat and Pears (2003). They consist of a 20 minute timed swim at a standardised swimming speed (20m per minute) swimming parallel to the reef crest at a consistent depth of approximately 5m on the reef slope (just below the reef crest, so it is possible to simultaneously monitor the reef crest, outer reef flat and reef slope). All large individuals ($\geq$35cm TL) of species listed in Table 1 should be counted and their size estimated[3]. Appropriate transect dimensions are 400m x 20m.

Prior to commencing each survey, observers should calibrate their swimming speeds (see *Timed Swims* above), the accuracy of their transect width estimates (using tape measures), and their size estimation of fishes underwater (see English et al 1997). A minimum size for inclusion and size categories should also be used (see *Timed Swims* above).

In order to calculate fish density and biomass per unit area (see *Data Analysis*), it is necessary to calculate the area covered by each method. The area of each belt transect is 250m^2 for small and medium sized individuals, and 1000 m^2 for large individuals. The area surveyed during the long swims will need to be calculated as described above for the timed swims.

Diver Safety

In recognition of safe diving practices, coral reef habitats should be surveyed in decreasing order of depth (e.g. reef slopes followed by reef crests). Maintaining a maximum depth of less than 10m will also maximise dive time, and minimise the risk of decompression sickness.

When conducting timed or long swims, observers should tow a surface buoy so the boat drivers can monitor their location at all times.

Equipment

Equipment required includes underwater slates and pencils, pre-prepared datasheets (Appendix 1) on underwater paper, a GPS, five transect tapes (for belt transects), an underwater depth gauge and stopwatch, and a surface buoy with a tow rope (see *Diver Safety* above).

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[3] If both transects and long swims are used to count large individuals, the data for the long swims should be used in preference to the transect data for those individuals, since long swims provide better estimates of their abundance than transects.

Data Analysis

After the survey has been completed, each species should be assigned to a functional group as defined in Table 1, and density, biomass and size structure calculated for all herbivorous reef fishes and each functional group.

Density

For each census, the number of individuals counted should be converted to a total density per unit area (hectare or ha[4]) for timed swims, and mean density (per ha) for belt transects. Where: density (per ha) = (number of individuals per sampling unit ÷ area of the sampling unit in m^2) x10,000.

Biomass

Size estimates should be converted to biomass using known length-weight relationships for each species using the formulae $W = aL^b$ as described in Kulbicki et al (2005). Where: W = weight of the fish in grams (g); L = fork length (FL) of the fish in cms; and a and b are constants calculated for each species or genus. Biomass constants (a and b) for each species are provided in Appendix 2 based on the best available information for the Asia Pacific Region.

For each census, biomass should be converted to a total biomass per unit area (kg per ha) for timed swims and mean biomass (per ha) for belt transects. Where: biomass (kg[5] per ha) = [(total biomass per sampling unit in g ÷ 1000) ÷ area of the sampling unit in m^2] x10,000.

Please note that underwater visual estimates of size are generally based on TL, which is easier to estimate than FL for many species. However, length-weight relationships for biomass are generally based on FL. For species with rounded or square tails, FL and TL are the same. However for species with forked tails, TL should be converted to FL to use for biomass estimates. Where detailed conversion ratios are not available for local species, a good rule of thumb is that FL is approximately 90% of TL for most species with forked tails (Kulbicki *pers. comm.*). Also, if size categories are used, fish lengths used for biomass estimates should be the mid value for each size category (e.g. use 12.5cms for size category 10 to <15cms).

While there may be an error associated with this approach, it is generally considered less than the error associated with underwater size estimation, which is the greatest source of error in this method (see review in Kulbicki et al 2005). Since underwater size estimation is highly dependant on diver training, observers should ensure that they are well trained prior to each census period (see English et al 1997).

Size Structure

Size structure of herbivores should be analysed also, because different size structures will have different impacts on coral reef resilience (see *Size and Role in Ecosystem Processes*). In particular, the biomass of small (<1kg) and large ($\geq$1kg) herbivores, particularly functional groups of parrotfishes (scrapers/small excavators and large excavators/bioeroders), should be analysed separately since they consume different quantities of algae and have different effects on the underlying substratum.

Analysis of Coral Reef Resilience

Coral reef resilience can be assessed using one of two methods: a low tech approach (a visual assessment of results) or a high tech approach (multivariate assessment using principal component analysis). Either method can provide a useful assessment of coral reef resilience, and the method that should be used will depend on the technical expertise of the survey team. If possible, analytical methods (principal component analysis) should be used, since they provide a more rigorous analysis of the data. However, if that is not possible, a visual assessment of results may be useful.

Both methods should be based on three key characteristics that are likely to be the most important indicators of coral reef resilience in terms of herbivorous reef fish populations: total biomass of herbivores; number of functional groups represented; and biomass of large parrotfishes ($\geq$1kg: see *Size and Role in Ecosystem Processes*).

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[4] One hectare is equal to 10,000m^2.

[5] 1kg = 1000g

To conduct a visual assessment of results, each site should be assigned a resilience rank based on these three characteristics. If information is available on what is "natural" for the area, results should be compared to what would be expected under natural conditions for that area (see *Interpreting Results*). However if that information is not available, sites can be ranked in terms of their relative resilience (i.e. relative to each other) to identify sites that are likely to have high, moderate or low resilience in terms of herbivorous reef fishes. Where sites that comprise a high total biomass of herbivorous reef fishes, all functional groups, and a high biomass of large parrotfishes are likely to have high resilience; sites that comprise a low total biomass of herbivorous reef fishes, only one or two functional groups of herbivores, and a low biomass of large parrotfishes, are likely to have low resilience; and sites that comprise moderate levels of all three characteristics are likely to have moderate resilience.

Coral reef resilience can also be assessed using principle component analysis based on the same three key characteristics of herbivorous reef fish populations to see which sites clump together, and whether these groupings seem representative of the potential resilience at these sites.

A case study demonstrating the use of both of these methods for analysing coral reef resilience is provided in Green and Muljadi (2009).

Table 2. Summary of recommended methods for long term monitoring and rapid assessments of herbivorous reef fishes as indicators of coral reef resilience.

Methods	Details	Modifications (if required)
Sampling Design	Use a hierarchical or stratified sampling design.	
Site Selection	Select representative areas with adequate space to conduct the monitoring program (400m for rapid assessments, and 650m for long term monitoring: see *Underwater Visual Census Methods* below). Sites should be separated by several hundred meters, preferably 500m, and each site location recorded using a GPS.	
Standardising Coral Reef Exposure and Zones	Standardise for exposure (exposed linear reef fronts) and zones: outer reef slopes (8-10m) and crests (just below the reef crest, so it is possible to simultaneously monitor the reef crest, outer reef flat and upper reef slope).	If distinctive reef crests are not present at some sites, reef slopes should be surveyed at two depths: 8-10m and 2-4m. Other exposures or zones (lagoons, back reefs, reef channels, passes or promontories) can also be surveyed, but must be analysed separately.
Survey Timing	Daylight hours from 9am to 4pm to avoid peak spawning times for some species.	Counts should be suspended if a spawning aggregation is encountered.
Survey Frequency	Long term monitoring should comprise an initial baseline survey followed by surveys every one to three years, depending on the status of coral communities, herbivorous reef fish populations, threats, and management actions. Rapid assessments usually comprise a single survey on one occasion.	

Methods	Details	Modifications (if required)
Underwater Visual Census Methods	Long term monitoring methods should comprise a combination of belt transects and a long swim. Five 50m belt transects should be used to count small (<20cm TL and more than the minimum size for inclusion: see below) and medium (20 to <35 cm TL) sized individuals, using a transect width of 5m. Transects should be laid consecutively along a depth contour of 8-10m parallel to the reef crest, with transects separated by at least 5m. Once the transects have been completed, large individuals (≥35cm TL) should be counted using a 20 min long swim, a transect width of 20m and a standardised swimming speed of 20m per minute. The long swim should start at the end of the transects and continue in the same direction just below the reef crest (see *Standardising Coral Reef Exposure and Zones*). Appropriate transect dimensions for the long swim are 400m x 20m.	If there is insufficient time to complete five 50m transects at each site, use three 50m transects rather than using shorter transects, because longer transects are more appropriate for most species. If there is insufficient time to conduct a long swim, large individuals can be counted on the belt transects using a transect width of 20m. However this should be avoided if possible, since long swims provide more precise methods for censusing large, highly mobile species. If both transects and long swims are used to count large individuals, the data for the long swims should be used in preference to the transect data for those individuals, since long swims provide better estimates of their abundance than transects.
	Rapid assessments: Timed swims should be used to census all herbivorous reef fishes at each site in each of two zones (reef slopes and crests). Each census consists of two observers swimming side by side along the reef slope parallel to the reef crest at a consistent depth of 8-10m for 20mins, then along the reef crest for 20 mins. Appropriate transect dimensions for timed swims are 400m x 20m.	For timed and long swims: If observers are consistently covering distances less than 400m in each timed or long swim, methods should be modified until an average distance of 400m per swim is attained by either: 1) Increasing the duration of each swim from 20 to 25 or 30mins; 2) Reducing the transect width for small and medium sized individuals from 5m to 3m; or 3) Changing the fish sizes that each observer has to count (see *Number of Observers and What They Count* below) so one observer counts small individuals, and the other observer counts medium and large sized individuals.
Number of Observers and What They Count	If possible, use two observers where each observer censuses different components of the fish fauna, counting different sized individuals using different transect widths. One observer swims 1-2m above the substratum, counting and estimating the size of small and medium sized individuals of the species listed in Table 1 using a transect width of 5m. The second observer swims 2-3m above the substratum, and next to or slightly behind the other observer, counting and estimating the size of large individuals of the same species on 20m wide transects.	If only one observer is available, he/she should conduct two passes through the survey area, where medium and large sized individuals are counted on the first pass, and small individuals are counted on the second pass.

Methods	**Details**	**Modifications (if required)**
Data to Record During the Survey	Record both the abundance (number of individuals) and size categories of each species listed in Table 1, by counting all individuals of the study species in the census area from the reef substratum to the water surface. Record data directly onto pre-prepared data sheets (Appendix 1) printed on underwater paper. Data should be recorded at the species level during the survey, and assigned to functional groups during data analysis (see *Data Analysis* below).	If observers cannot identify all herbivores to the species level, the method can be modified to a higher taxonomic level by lumping species at the family and genus level as summarized in Tables 3 and 4 (see *Simplifying the Protocol*).
Minimum Size for Inclusion	Use a minimum size of 10cm TL to exclude juveniles that are not amenable to rapid visual census methods, except for small species (*Acanthurus nigrofuscus, A. triostegus, Zebrasoma scopas* and *Centropye* species) where a minimum size of 5cm should be used (or most individuals will not be recorded).	If observers cannot identify these small species, a minimum size of 10cm can be used for all species, although it is not recommended (see *Simplifying the Protocol*).
Size Categories	Use 2.5cm size categories for fishes less than 10cm in size, and 5cm size categories for fishes 10cm or greater in size (i.e. 5 to <7.5cms, 7.5 to <10cms, 10 to <15cms, 15 to <20cms etc).	If this is too complicated, use 5cm size categories for all sizes (see *Simplifying the Protocol*).
Minimising Disturbance to Fish Populations	Observers should be the first people to swim through the census area, swimming very quietly, and waiting at least 5 to 10 mins after getting in the water before starting the census or between passes of the census area. If transect tapes are used, they should be laid during the first pass of the transects by an assistant following the observers. Field teams should also avoid driving the boat over the survey area prior to the census.	
Training Prior to Surveys	Observers should calibrate their swimming speeds, the accuracy of their transect width estimates (using tape measures), and their size estimation of fishes underwater prior to commencing each survey.	
Diver Safety	Coral reef habitats should be surveyed in decreasing order of depth (e.g. reef slopes followed by reef crests). Divers should also maintain a maximum depth of less than 10m so they can maximise their dive time, and minimise the risk of decompression sickness. When conducting timed or long swims, observers should tow a surface buoy so boat drivers can monitor their location at all times.	
Equipment Required	Underwater slates and pencils, pre-prepared datasheets (Appendix 1) printed on underwater paper, a GPS, five transect tapes (for belt transects), an underwater depth gauge and stopwatch, and a surface buoy with a tow rope (see *Diver Safety* above).	

Methods	**Details**	**Modifications (if required)**
Data Analysis	Assign each species to a functional group (see Table 1).	
	Calculate the area of each sampling unit: For timed or long swims the area equals the distance covered by each timed/long swim multiplied by the transect width. The width of each transect is 5m for small and medium sized individuals, and 20m for large individuals. The length of each timed swim should be measured by marking the beginning and end of each swim with a differential GPS, and measuring the distance between them using the GPS (or attaching the GPS to a floating buoy that is towed by the divers to track the distance travelled). For belt transects: the area of each transect is 250m^2 for small and medium sized individuals (50x5m transects), and 1000m^2 for large individuals (50x20m transects).	For timed or long swims: The average distance covered by both observers in a 20 min period can also be estimated by measuring the distance they swim in 5mins while censusing in a range of situations (current speeds, high and low density of reef fishes) and converting this to an average distance covered in 20mins. This method is less precise than using a GPS, and the GPS should be used if possible.
	Calculate the total density (per ha) and total biomass (kg per ha) for timed or long swims, and the mean density (per ha) and mean biomass (kg per ha) for belt transects, for all herbivorous reef fishes and each functional group.	
	Analyse the size structure of all herbivorous reef fishes and each functional group, particularly the biomass of small (<1kg) and large (≥1kg) functional groups of parrotfishes (scrapers/small excavators and large excavators/bioeroders).	
	Analyse coral reef resilience using principal component analysis based on three key characteristics of the herbivorous reef fish populations: total biomass of herbivores; number of functional groups represented; and biomass of large parrotfishes.	If it is not possible to use principal component analysis, a visual assessment of results may be useful based on the same three key characteristics of the herbivorous reef fish populations.
Interpreting Results	Results should be interpreted in terms of what is “natural” for an area, and what the results mean in the broader context of coral reef resilience.	

Interpreting Results

Interpreting the results of a monitoring program for functional groups of herbivorous reef fishes requires understanding what is “natural” for an area, and what the results mean in a broader context.

What is “Natural” for an Area?

Patterns of distribution and abundance of herbivorous reef fishes vary on both biogeographic and geographic scales (see *Biographic and Regional Differences* and *Patterns of Distribution and Abundance within Geographic Areas*). So in order to interpret the results of a monitoring program, it is important to understand what is natural for that area. For example on the Great Barrier Reef, *B. muricatum* is naturally more abundant on mid and outer shelf reefs, and less abundant on inner shelf reefs (Hoey and Bellwood 2008). So if the results of a monitoring program showed that this species was not abundant on inner shelf reefs, there may not be cause for concern. However, if a monitoring program found that this species was absent or only present in very low abundances on outer shelf reefs, there may be cause for concern.

Furthermore, while extremes may be easy to interpret (e.g. lots of large *B. muricatum* or none in areas where they were previously abundant), intermediate results will be more difficult to understand (e.g. what does one or two medium sized individuals mean?). In that situation, information regarding natural patterns of distribution and abundance of herbivorous reef fishes will be required to interpret results at each location.

Therefore where possible, monitoring results should be compared to those of existing long term monitoring programs, which may provide an insight into the “natural” density, biomass and size structure of herbivorous reef fishes in that area. In doing so, it will be important to guard against shifting baselines (Jackson et al 2001), where current conditions may be considered “natural” because they have been that way for a long time (years or decades), even though there may have been dramatic changes in the longer term (Pandolfi et al 2003, Bellwood et al 2004). In that situation, local or traditional knowledge may provide valuable information over a longer time frame. For example, elders in the community may remember large numbers of species such as *B. muricatum* that may not have been seen in the area for years.

If long term data is not available, interpreting the results of a single survey at one site may be problematic. If that is the case, broad scale surveys of surrounding areas may provide a broader context for interpreting the results. Of particular value would be surveys of similar habitats in effectively enforced no-take MPAs, which may provide insight into what may be “natural” for that area.

Information regarding the biology of the species may also be useful. For example, Hamilton (2004) and Aswani and Hamilton (2004) reported that the preferred recruitment habitat for *B. muricatum* is lagoons. Therefore, areas without lagoons are unlikely to support large populations of that species.

What Do Results Mean in a Broader Context?

When interpreting results, it is important to remember that functional groups of herbivorous reef fishes are only one factor that plays a critical role in maintaining coral reef resilience (see *Coral Reef Resilience*). Therefore, the results of herbivore monitoring programs must be interpreted in a broader context where all factors that contribute to coral reef resilience are considered (see *Monitoring Coral Reef Resilience*).

Recently, the IUCN Working Group on Climate Change and Coral Reefs developed an integrated monitoring program aimed at assessing all components of coral reef resilience (Obura and Grimsditch 2009, see also Maynard et al ms). These protocols provide a broader context for interpreting the results of herbivore monitoring programs.

The results of herbivore monitoring programs will also need to be interpreted in terms of the role that each functional group plays in coral reef resilience (see *Functional Groups*). Scrapers/small excavators and grazers/detritivores play a critical role in preventing coral algal phase shifts by intensely grazing epilithic algal turfs, and limiting the establishment and growth of macroalgae. Scrapers/small excavators also provide areas of clean substratum that facilitates the settlement, growth and survival of coralline algae and corals. Therefore, the minimum requirement for preventing coral-algal phase shifts may be an army of small parrotfishes, surgeonfishes and rabbitfishes mowing down the algae and scraping surfaces clean for coral recruitment.

An army of parrotfishes, surgeonfishes and rabbitfishes mowing down the algae and scraping surfaces clean for coral recruitment**.** Image by A. Lewis.

Fortunately these small herbivores tend to be less susceptible to overfishing than large herbivores and other trophic groups such as carnivores (Craig et al 1997, Jennings and Polunin 1996, Russ and Alcala 2003, 2004; Sandin et al 2008). Populations of herbivorous reef fishes, particularly parrotfishes, also tend to recover quickly after protection in no-take MPAs (McClanahan et al 2007). Consequently, fish biomass on intensely fished reefs is often dominated by small herbivores, while lightly fished reefs tend to have a higher proportion of large herbivores and carnivores (Jennings and Polunin 1996, Sandin et al 2008). This is relatively good news in the context of coral reef resilience, since fishing pressure must be intense before all functional groups of herbivores are overfished, reducing coral reef resilience.

The bad news is that browsers are more susceptible to overfishing than scrapers/small excavators and grazers/detritivores. Browsers (rudderfishes, batfishes, a rabbitfish and some unicornfishes) consume macroalgae, and play critical roles in preventing and reversing coral-algal phase shifts. Therefore, it is important that these species are well represented in the fish fauna.

Large excavators/bioeroders are also extremely vulnerable to overfishing, particularly *Bolbometopon muricatum* (left; Image by G. Allen). This species has recently been listed as Vulnerable on the IUCN Red list (Chan et al 2007) in recognition of its slow growth and late sexual maturity (Hamilton et al 2008), and the ease with which stocks can be wiped out by spearfishers (Dulvy and Polunin 2004, Hamilton 2003, Gillet and Moy 2006). Consequently, populations of large excavators/bioeroders such as *B. muricatum* have been reduced across much of their range, and are now common only in remote areas or areas where fishing is prohibited (reviewed in Dulvy and Polunin 2004, Chan et al 2007). These large excavators/bioeroders play a critical role in ecosystem processes, and their absence may have serious consequences for processes such as bioerosion and coral reef resilience (see *Functional Groups*).

Furthermore while other trophic groups (such as carnivores) may not play a key role in preventing coral-algal phase shifts, there is evidence to suggest that coral reefs with low fishing pressure and relatively intact food webs (including large carnivores that are also vulnerable to overfishing) provide greater overall resilience to threats such as climate change (Sandin et al 2008).

Adaptive Management

Long term monitoring programs must be designed to provide useful information for coral reef conservation and management. In the context of coral reef resilience, monitoring programs should provide information on long term trends in herbivorous reef fish populations and other key factors that are critical for maintaining coral reef resilience, to identify the need for management intervention (Adaptive Management: see *Glossary*). They should also be used to monitor the success of management actions (Wilkinson et al 2003), such as using no-take areas to protect herbivorous reef fish populations.

Unfortunately, the results of long term monitoring programs are not always available, and managers need to interpret the results of single surveys for management purposes. This is problematic because natural patterns of distribution and abundance can be quite variable (see *Interpreting Results* above). However, some general advice can be provided. For example, if fishing pressure is low and the density and biomass of herbivorous reef fishes is high relative to that recorded in similar areas, adaptive management to protect these species may not be required. However, if fishing pressure is moderate to high and the density and biomass of herbivorous reef fishes is low relative to that recorded in similar areas, it may be advisable to take a precautionary approach and improve the management of these species to determine if herbivore populations respond with increased density and biomass. If they do respond, it may be important to continue to manage fisheries for these species in order to manage for coral reef resilience. However if they do not respond, they may have never been abundant in that area (see *Interpreting Results* above). Alternatively, overfishing may have been so severe that these species are now recruitment limited and may take a long time to recover (Russ 1991, Russ 2002). In that situation, effective management and long term monitoring will be required.

SIMPLIFYING THE PROTOCOL

This document represents the first attempt to put theory into practice, by designing a protocol to monitor functional groups of herbivores as indicators of coral reef resilience. These methods require a moderate level of technical expertise (e.g. in species identification). If technical expertise is limited, the following modifications can be used to provide a simplified protocol (summarized in Table 2).

Using a Lower Taxonomic Level

In this protocol, herbivorous reef fishes were assigned to functional groups at the species level for some families (see Table 1). If possible, the best approach is for observers to learn to identify the species that occur in their area (not all species occur in all areas), because monitoring at the species level will provide the highest degree of resolution for monitoring coral reef resilience.

If that is not possible, the method can be modified to a lower taxonomic level by lumping species as described below and summarized in Tables 3 and 4. However, it is important to recognise that this may result in less reliable results.

If a lower level of taxonomic resolution is required, herbivores can be assigned to functional groups as follows:

- All rudderfishes and batfishes are browsers
- All angelfishes of the genus *Centropyge* are grazers (all other genera are excluded).
- All rabbitfishes are grazers/detritivores except *S. canaliculatus,* which is a browser.
- All grazers belong to the functional group of grazers/detritivores.

For parrotfishes, the best way to approach this is to learn the genera and assign them to functional groups as follows:

- All humpheaded parrotfishes of the genera *Bolbometopon, Cetoscarus* and *Chlorurus* are excavators. All individuals ≥35cm SL should be considered large excavators/bioeroders, while those <35cm SL should be considered small excavators. [Please note that it is important not to include *Scarus rubroviolaceus* in this group. Even though it has a humphead, *S. rubroviolaceus* is not an excavator, but a scraper. This species can be easily recognised by its distinctive colour pattern (see local fish guides or FishBase).]
- All individuals of the genera *Scarus* and *Hipposcarus* are scrapers.
- All individuals of the genera *Calotomus* and *Leptoscarus* are browsers.
- All scrapers and small excavators belong to the functional group scrapers/small excavators.

Surgeonfishes are more complicated, because some genera and species comprise different functional groups. If possible, the best approach is to learn enough genera and species to assign them to functional groups as follows:

1. Recognise and exclude all planktivorous species (*A. albipectoralis*, *A. mata*, *A. nubilus, A. thompsoni* and *Paracanthurus hepaticus*: see Allen et al 2003). They can be recognised by a combination of colour, shape, and behaviour (they tend to hang out off the reef front and feed on plankton in the water column).
2. Recognise and exclude all *Ctenochaetus* species (see Allen et al 2003), because they are exclusively detritivores.
3. All other species are grazers/detritivores, including all species of *Zebrasoma* and all other *Acanthurus* species.

Unicornfishes are also complex. The only species that are included are those that are either browsers throughout their lives (N. *brachycentron, N. elegans, N. lituratus, N. tonganus* and *N. unicornis*) or as juveniles (<20cm SL: *N. annulatus, N. brevirostris, N. maculatus, N. mcdadei*, and *N. vlamingii*). The only way to include these species is to identify them at the species level, and assign them to functional groups as appropriate (by size for some species). All other species are planktivores and should be excluded. If this is not practical, unicornfishes can be excluded from the protocol. These species have different feeding rates and nutritional ecology than other herbivores (i.e. what they feed on, and how they process it), and they don't make major contributions to herbivory in most places compared to other herbivores such as parrotfishes. However, they should be included if possible, because they are among the few herbivores that feed on macroalgae and may be capable of playing a significant role in reversing coral-macroalgal phase shifts (Bellwood et al 2006).

The angelfish genus *Centropyge* can also be excluded if necessary (if they are difficult to identify). However, they should be included if possible because they can be numerous and important herbivores in some locations.

Minimum Size for Inclusion

A minimum size for inclusion is necessary, because many small individuals are cryptic in behaviour and colouration (Bellwood and Choat 1989), and specialized methods are required to census them (Green 1992, 1998). Observers are also likely to miss large individuals if they are searching for small ones (and vice versa).

A minimum size of 10cm TL is often used for underwater visual census of herbivorous reef fishes, such as scarines (e.g. Hoey and Bellwood 2008). In this protocol we recommended using the same minimum size, but recognised that this would effectively exclude some of the smaller species of grazers/detritivores (*Acanthurus nigrofuscus, A. triostegus, Zebrasoma scopas* and *Centropye* species). Therefore, a minimum size of 5cm was recommended for those species. If observers cannot identify these small species, a minimum size of 10cm could be used for all species, although it is not recommended.

Size Categories

In this protocol, we recommended that fishes be assigned to size categories, given that most observers do not have the level of skill required to estimate size underwater at a high degree of accuracy (see Kulbicki et al 2005). We recommended using 2.5cm size categories for fishes less than 10cm in size, and 5cm size categories for fishes 10cm or larger in size (i.e. 5 to <7.5, 7.5 to <10, 10 to <15, 15 to <20 etc). If that is too complicated, 5cm size categories could be used for all sizes. While that may affect the accuracy of biomass estimates for smaller fishes, it is unlikely to have a significant effect on the overall results (unless most of the individuals are small). Also, if a minimum size of 10cm is used for all species (see above) only 5cm size categories will be required.

Table 3. Functional groups within each family, genera and species of herbivorous reef fishes

Family (Tribe)	**Common Name**	**Functional Group**	**Genera and Species**
Labridae (Scarini)	Parrotfishes	Scrapers/small excavators	All *Scarus* and *Hipposcarus* All *Bolbometopon*, *Cetoscarus* and *Chlorurus* <35cm SL
		Large excavators/bioeroders	All *Bolbometopon*, *Cetoscarus* and *Chlorurus* ≥35cm SL
		Browsers	All *Calotomus* and *Leptoscarus* species
Acanthuridae	Surgeonfishes	Grazers/detritivores	All species except those that are planktivores (*A. albipectoralis, A. mata, A. nubilus, A. thompsoni* and *P. hepaticus*) or detritivores (*Ctenochaetus* spp.)
	Unicornfishes*	Browsers	All *N. brachycentron, N. elegans, N. lituratus, N. tonganus* and *N. unicornis*
			Juveniles (<20cm) of *N. annulatus, N. brevirostris, N. maculatus, N. mcdadei*, and *N. vlamingii*
Siganidae	Rabbitfishes	Browsers	*S. canaliculatus*
		Grazers/detritivores	All other species
Kyphosidae	Rudderfishes	Browsers	All
Ephippidae	Batfishes	Browsers	All
Pomacanthidae	Angelfishes*	Grazers/detritivores	All *Centropyge* species

*Can be excluded if too difficult to identify.

Table 4. Families, genera and species within each functional group of herbivorous reef fishes, and their role in coral reef resilience.

Functional Group	Family (Tribe)	Common Name	Genera and Species	Role in Coral Reef Resilience
Scrapers/ small excavators	Labridae (Scarini)	Parrotfishes	All *Scarus* and *Hipposcarus* All *Bolbometopon*, *Cetoscarus* and *Chlorurus* <35cm SL	Limiting the establishment and growth of macroalgae while intensely grazing epilithic algal turf, and scraping the substratum as they feed (providing areas of clean substratum for coral recruitment).
Large excavators/ bioeroders	Labridae (Scarini)	Parrotfishes	All *Bolbometopon*, *Cetoscarus* and *Chlorurus* ≥35cm SL	Limiting the establishment and growth of macroalgae while intensely grazing epilithic algal turf, and excavating the substratum as they feed (providing areas of clean substratum for coral recruitment). They are also major agents of bioerosion on reefs.
Grazers/ detritivores	Acanthuridae	Surgeonfishes	All species except those that are planktivores (*A. albipectoralis, A. mata, A. nubilus, A. thompsoni* and *P. hepaticus*) or detritivores (*Ctenochaetus* spp.)	Limiting the establishment and growth of macroalgae while intensely grazing epilithic algal turfs. Unlike scrapers/small excavators or large excavators/bioeroders, they do not scrape or excavate the reef substratum as they feed.
	Siganidae	Rabbitfishes	All species except *S. canaliculatus*	
	Pomacanthidae	Angelfishes*	All *Centropyge* species	
Browsers	Kyphosidae	Rudderfishes	All	Reducing coral overgrowth and shading by selectively feeding on macroalgae.
	Siganidae	Rabbitfishes	*S. canaliculatus*	
	Acanthuridae	Unicornfishes*	All *N. brachycentron, N. elegans, N. lituratus, N. tonganus* and *N. unicornis*	
			Juveniles (<20cm) of *N. annulatus, N. brevirostris, N. maculatus, N. mcdadei*, and *N. vlamingii*	
	Ephippidae	Batfishes	All	
	Labridae (Scarini)	Parrotfishes	All *Calotomus* and *Leptoscarus* species	

*Can be excluded if too difficult to identify.

ADAPTATIONS FOR OTHER BIOGEOGRAPHIC AND GEOGRAPHIC REGIONS

This protocol was developed using the Asia Pacific Region as a case study. However, the monitoring methods and functional groups used here can be applied anywhere in the world by modifying the species used (and biomass constants) for different biogeographic and geographic regions.

This method can be easily adapted for other areas of the Indo-West Pacific Region (e.g. the West Indian Ocean) by expanding the species list in Table 1 to include additional species as required, and assigning them to functional groups based on the scientific literature and expert opinion for those areas.

A similar approach can be taken for other biogeographic regions (i.e. the Eastern Pacific, Western Atlantic and Eastern Atlantic), by modifying the list to include local species of herbivorous reef fishes. However, for other biogeographic regions, particularly the Caribbean, consideration should also be given to including other functional groups of herbivores (particularly echinoids) that play an important role in herbivory and coral reef resilience in those areas (see *Biogeographic and Regional Differences*). If other functional groups are included, other survey methods may also be required for those species (e.g. belt transects for *Diadema*: see Hill and Wilkinson 2004).

Field practitioners working in other geographic areas of the Indo-West Pacific Region will also need to decide what biomass constants to use. If they are using the same species, they can use the constants provided in this document (Appendix 2), since the variation associated with geographic area is less than the error associated with underwater visual size estimates (see *Data Analysis*). Alternatively, they can find biomass constants for their geographic areas in FishBase (2007). Biomass constants for species in other biogeographic regions are also available in FishBase (2007).

REFINING THE PROTOCOL

This document represents the first attempt to develop a monitoring program that is specifically designed to monitor key functional groups of herbivorous reef fishes as indicators of coral reef resilience. Even though it is based on the best available information, it is important to remember that the science underpinning these methods is still new and developing. Further research is now required to address knowledge gaps and refine monitoring methods.

Areas where some uncertainty still exists that will require further research and refinement include:

- Identifying key functional groups to monitor: This is the first attempt to identify functional groups of herbivores for monitoring coral reef resilience, and these categories may require refinement over time. For example, as the science develops, it may become apparent that other groups also play a critical role in coral reef resilience, and need to be included in this protocol. In particular, it will be important to identify and include other groups or species that may play an important role in reversing coral-algal phase shifts.
- Assigning species to each functional group: Detailed dietary information is not available for many species of herbivores. In this protocol, species were assigned to functional groups based on best available information in the scientific literature and expert opinion. However, this may need to be refined as more dietary information becomes available.
- Research at multiples scales to aid in the interpretation of results (see *Interpreting Results*). In particular, more information is required regarding natural patterns of distribution and abundance of herbivores in a range of biogeographic and geographic areas.

While these methods have been field tested in a range of situations, refinements may also be required at each site based on local conditions, fish abundances, the objectives of the monitoring program, and the technical expertise of the monitoring team. Where possible, modifications have been suggested to accommodate these changes (see Table 2, *Monitoring Methods* and *Simplifying the Protocol*).

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GLOSSARY

Terms

Adaptive management is a systematic process for continually improving management policies and practices by learning from the outcomes of programs in action.

Algal turf is epilithic algal turf.

Anthropogenic threats, processes or materials are those that are derived from human activities, as opposed to those occurring in natural environments without human influences.

Asia Pacific Region comprises the coral reefs of Southeast Asia and the Pacific Islands (Melanesia, Micronesia and Polynesia). It is part of the *Indo-Pacific Region* (see below).

Bioerosion is the removal of material from the reef matrix by biological processes (Choat 1991).

Climate change refers to any change in climate over time, whether due to natural variability or as a result of human activity (IPCC 2007).

Coral recruitment includes larval supply, settlement, and post-settlement growth and survival of juvenile corals (modified from Birrell et al 2008). Coral recruits are visible *in situ*, generally more than six months after settlement (with diameters of 0.5cm or more).

Coral reef resilience is the ability of reefs to absorb recurrent disturbances (such coral bleaching events), and rebuild coral-dominated systems rather than shifting to macroalgal dominated systems (Marshall and Schuttenberg 2006, Hughes et al 2007). [See also *Resilience* below]

Coral settlement involves the attachment of coral larvae to the substratum (modified from Birrell et al 2008). Newly settled corals are generally not visible *in situ* with the naked eye (generally less that six months after settlement, and with diameters of less than 0.5 cm).

Ecosystem-based adaptation identifies and implements a range of strategies for the management, conservation and restoration of ecosystems to ensure that they continue to provide the services that enable people to adapt to the impacts of climate change (IUCN 2009).

Epilithic means living on rocks or hard reef substratum.

Functional groups are a collection of species that perform a similar function, irrespective of their taxonomic affinities (Steneck and Dethier 1994).

Indo-Pacific Region comprises all coral reefs in the Indian and Pacific Oceans, from the Red Sea to the Eastern Pacific. It includes, but is not limited to, the *Asia Pacific Region* (see above).

The *Indo-West Pacific Region* extends from the Red Sea and eastern African coast through the Indian Ocean to eastern Polynesia (Hawaii, Line Islands, Marquesas, and Easter Islands). It does not include the *Eastern Pacific* (tropical western American coastline and offshore islands).

Monitoring is the gathering of data and information on coral reef ecosystems and its users on a regular basis, preferably for an extended period of time (English et al 1997, Wilkinson et al 2003).

Resilience is the ability of an ecosystem to absorb shocks, resist phase shifts and regenerate after natural and human-induced disturbances (Nyström et al 2000). A core component of resilience is the ability of an ecosystem to maintain its key ecological functions and processes after disturbance, by either resisting or adapting to change (Gunderson 2000, Carpenter et al 2001, Nyström and Folke 2001). [See also *Coral Reef Resilience* above.]

Sleeping functional groups are species (or groups of species) that may be capable of performing a particular functional role but which do so only under exceptional circumstances (Bellwood et al 2006).

Spawning aggregations are temporary aggregations by fishes that have migrated for the specific purpose of spawning (Clayton 2004). They can often be identified as large groups of a given species milling around close to reef points and edges, which are not feeding.

Fish Families and Common Names

Acanthuridae (or acanthurids) are surgeonfishes and unicornfishes.
Balistidae (or balistids) are triggerfishes.
Blenniidae (or bleniids) are blennies.
Ephippidae (or ephippids) are batfishes.
Gobiidae (or gobiids) are gobies.
Kyphosidae (or kyphosids) are rudderfishes.
Labridae (labrids) are wrasses, parrotfishes and odacids (Westneat and Alfaro 2005, Cowman et al in press). Parrotfishes (Tribe Scarini), previously known as scarids, are now known as scarines.
Monacanthidae (or monacanthids) are filefishes.
Pomacanthidae (or pomacanthids) are angelfishes.
Pomacentridae (or Pomacentrids) are damselfishes.
Siganidae (or siganids) are rabbitfishes.

Abbreviations

FL = fork length.
GPS = Global Positioning System
IUCN = International Union for Conservation and Nature
MPA = marine protected area.
SL = standard length.
TL = total length.

APPENDIX 1 Data Sheets

Timed Swims (Small and Medium Sized Individuals)

Site Name:	**Exposure:**	**Zone:**
GPS Coordinates:	Start	End
Date:	**Observer:**	**Visibility:**
Duration:	**Transect Width:**	**Transect Length:**

FAMILY (Tribe)*	**SIZE CATEGORIES**						
Acanthuridae	**5-<7.5**	**7.5-<10**	**10-<15**	**15-<20**	**20-<25**	**25-<30**	**30-<35**
Labridae (Scarini)							
Siganidae							
Ephippidae							
Kyphosidae							
Pomacanthidae							

*** List species if known**

Timed Swims (Large Individuals)

Site Name:	**Exposure:**	**Zone:**
GPS Coordinates:	Start	End
Date:	**Observer:**	**Visibility:**
Duration:	**Transect Width:**	**Transect Length:**

FAMILY (Tribe)*	**SIZE CATEGORIES**						
Labridae (Scarini)	**35-<40**	**40-<45**	**45-<50**	**50-<55**	**55-<60**	**60-<65**	**65-<70**
B. muricatum							
C. bicolor							
C. microrhinus							

Site Name:	**Exposure:**	**Zone:**
GPS Coordinates:	Start	End
Date:	**Observer:**	**Visibility:**
Duration:	**Transect Width:**	**Transect Length:**

FAMILY (Tribe)*	**SIZE CATEGORIES**						
Labridae (Scarini)	**35-<40**	**40-<45**	**45-<50**	**50-<55**	**55-<60**	**60-<65**	**65-<70**
B. muricatum							
C. bicolor							
C. microrhinus							

*** List species if known**

50 m Transects (Small and Medium Sized Individuals)

Site Name:	**Exposure:**	**Zone:**
GPS Coordinates (start):	Latitude	Longitude
Date:	**Observer:**	**Visibility:**
Transect Length:	**Transect Width:**	**Trans #:**

FAMILY (Tribe)*	**SIZE CATEGORIES**						
Acanthuridae	**5-<7.5**	**7.5-<10**	**10-<15**	**15-<20**	**20-<25**	**25-<30**	**30-<35**
Labridae (Scarini)							
Siganidae							
Ephippidae							
Kyphosidae							
Pomacanthidae							

*** List species if known**

50 m Transects (Large Individuals)

Site Name:	**Exposure:**	**Zone:**
GPS Coordinates (start):	Latitude	Longitude
Date:	**Observer:**	**Visibility:**
Transect Length:	**Transect Width:**	**Trans #:**

FAMILY (Tribe)*	**SIZE CATEGORIES**						
Labridae (Scarini)	**35-<40**	**40-<45**	**45-<50**	**50-<55**	**55-<60**	**60-<65**	**65-<70**
B. muricatum							
C. bicolor							
C. microrhinus							

Site Name:	**Exposure:**	**Zone:**
GPS Coordinates (start):	Latitude	Longitude
Date:	**Observer:**	**Visibility:**
Transect Length:	**Transect Width:**	**Trans #:**

FAMILY (Tribe)*	**SIZE CATEGORIES**						
Labridae (Scarini)	**35-<40**	**40-<45**	**45-<50**	**50-<55**	**55-<60**	**60-<65**	**65-<70**
B. muricatum							
C. bicolor							
C. microrhinus							

*** List species if known**

Long Swims (Large Individuals)

Site Name:	**Exposure:**	**Zone:**
GPS Coordinates:	Start	End
Date:	**Observer:**	**Visibility:**
Duration:	**Transect Width:**	**Transect Length:**

FAMILY (Tribe)*	**SIZE CATEGORIES**						
Labridae (Scarini)	**35-<40**	**40-<45**	**45-<50**	**50-<55**	**55-<60**	**60-<65**	**65-<70**
B. muricatum							
C. bicolor							
C. microrhinus							

Site Name:	**Exposure:**	**Zone:**
GPS Coordinates:	Start	End
Date:	**Observer:**	**Visibility:**
Duration:	**Transect Width:**	**Transect Length:**

FAMILY (Tribe)*	**SIZE CATEGORIES**						
Labridae (Scarini)	**35-<40**	**40-<45**	**45-<50**	**50-<55**	**55-<60**	**60-<65**	**65-<70**
B. muricatum							
C. bicolor							
C. microrhinus							

***List species if known**

APPENDIX 2 Biomass Constants

Constants *a* and *b* for calculating biomass of herbivorous reef fish species included in the case study for the Asia Pacific Region.

Family (Tribe)	Species	a	b	Source
Acanthuridae	*Acanthurus achilles*	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus auranticavus	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus barine	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus blochii	0.0251	3.032	*A. blochii* (Kulbicki et al 2005)
	Acanthurus dussumieri	0.0426	2.868	*A. dussumieri* (Kulbicki et al 2005)
	Acanthurus fowleri	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus guttatus	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus japonicus	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus leucocheilus	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus leucopareius	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus leucosternon	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus lineatus	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus maculiceps	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus nigricans	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005)
	Acanthurus nigricauda	0.0168	3.168	*A. nigricauda* (Kulbicki et al 2005)
	Acanthurus nigrofuscus	0.0264	3.028	*A. nigrofuscus* (Kulbicki et al 2005)
	Acanthurus nigroris	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005
	Acanthurus olivaceus	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005
	Acanthurus pyroferus	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005
	Acanthurus tennentii	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005
	Acanthurus triostegus	0.0831	2.570	*A. triostegus* (Kulbicki et al 2005)
	Acanthurus tristis	0.0280	2.983	*Acanthurus* spp. (Kulbicki et al 2005
	Acanthurus xanthopterus	0.0267	2.984	*A. xanthopterus* (Kulbicki et al 2005)
	Naso annulatus	0.0510	2.715	*N. annulatus* (Kulbicki et al 2005)
	Naso brachycentron	0.0085	3.250	*Naso* spp. (Kulbicki et al 2005)
	Naso brevirostris	0.0107	3.243	*N. brevirostris* (Kulbicki et al 2005)
	Naso elegans	0.0085	3.250	*Naso* spp. (Kulbicki et al 2005)
	Naso lituratus	0.0085	3.250	*Naso* spp. (Kulbicki et al 2005)
	Naso maculates	0.0085	3.250	*Naso* spp. (Kulbicki et al 2005)
	Naso mcdadei	0.0085	3.250	*Naso* spp. (Kulbicki et al 2005)
	Naso tonganus	0.0085	3.250	*Naso* spp. (Kulbicki et al 2005)
	Naso unicornis	0.0179	3.035	*N. unicornis* (Kulbicki et al 2005)
	Zebrasoma desjardinii	0.0378	3.857	*Zebrasoma* spp. (Kulbicki et al 2005)
	Zebrasoma flavescens	0.0378	3.857	*Zebrasoma* spp. (Kulbicki et al 2005)
	Zebrasoma rostratum	0.0378	3.857	*Zebrasoma* spp. (Kulbicki et al 2005)
	Zebrasoma scopas	0.0291	2.993	*Z. scopas* (Kulbicki et al 2005)
	Zebrasoma veliferum	0.0343	2.866	*Z. veliferum* (Kulbicki et al 2005)

Family (Tribe)	**Species**	**a**	**b**	**Source**
Ephippidae	*Platax batavianus*	0.0443	2.951	*Platax* spp. (Kulbicki et al 2005)
	Platax boersi	0.0443	2.951	*Platax* spp. (Kulbicki et al 2005)
	Platax orbicularis	0.0443	2.951	*Platax* spp. (Kulbicki et al 2005)
	Platax pinnatus	0.0443	2.951	*Platax* spp. (Kulbicki et al 2005)
	Platax orbicularis	0.0443	2.951	*Platax* spp. (Kulbicki et al 2005)
	Platax teira	0.0443	2.951	*Platax* spp. (Kulbicki et al 2005)
	Zabidius novemaculeatus	0.0443	2.951	*Platax* spp. (Kulbicki et al 2005)
Kyphosidae	all species	0.0129	3.151	*Kyphosus* spp. (Kulbicki et al 2005)
Pomacanthidae	all *Centropyge* species	0.0745	2.577	*Centropyge* spp. (Kulbicki et al 2005)
Labridae (Scarini)	*Bolbometopon muricatum*	0.0098	3.1329	*B. muricatum* (Hamilton 2004)
	Calotomus carolinus	0.0122	3.167	*C. carolinus* (Smith and Dalzell 1993)
	Calotomus japonicus	0.0122	3.167	*C. carolinus* (Smith and Dalzell 1993)
	Calotomus zonarchus	0.0122	3.167	*C. carolinus* (Smith and Dalzell 1993)
	Calotomus spinidens	0.0122	3.167	*C. carolinus* (Smith and Dalzell 1993)
	Cetoscarus bicolor	0.0161	3.0049	*Cetoscarus* (Hoey and Bellwood unpubl. data)
	Chlorurus bleekeri	0.0243	2.969	*C. sordidus* (Kulbicki et al 2005)
	Chlorurus bowersi	0.0243	2.969	*C. sordidus* (Kulbicki et al 2005)
	Chlorurus capistratoides	0.0243	2.969	*C. sordidus* (Kulbicki et al 2005)
	Chlorurus enneacanthus	0.0243	2.969	*C. sordidus* (Kulbicki et al 2005)
	Chlorurus frontalis	0.0925	2.85	*C. microrhinus* (Choat et al 1996**)**
	Chlorurus japanensis	0.0243	2.969	*C. sordidus* (Kulbicki et al 2005)
	Chlorurus microrhinos	0.0925	2.85	*C. microrhinus* (Choat et al 1996)
	Chlorurus oedema	0.0243	2.969	*C. sordidus* (Kulbicki et al 2005)
	Chlorurus perspicillatus	0.0243	2.969	*C. sordidus* (Kulbicki et al 2005)
	Chlorurus sordidus	0.0243	2.969	*C. sordidus* (Kulbicki et al 2005)
	Chlorurus strongylocephalus	0.0243	2.969	*C. sordidus* (Kulbicki et al 2005)
	Chlorurus troschelii	0.0243	2.969	*C. sordidus* (Kulbicki et al 2005)
	Hipposcarus hairid	0.0633	2.6184	*Hipposcarus* (Hoey and Bellwood unpubl. data)
	Hipposcarus longiceps	0.0633	2.6184	*Hipposcarus* (Hoey and Bellwood unpubl. data)
	Leptoscarus vaigiensis	0.0163	2.991	*L. vaigiensis* (Kulbicki unpubl. data)
	Scarus altipinnis	0.0184	3.029	*S. altipinnis* (Kulbicki et al 2005)
	Scarus chameleon	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus dimidatus	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus dubius	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus festivus	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus flavipectoralis	0.0202	2.9811	*S. flavipectoralis* (Hoey and Bellwood unpubl. data)
	Scarus forsteni	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus frenatus	0.0279	3.060	*S. frenatus* (Choat et al 1996)

Family (Tribe)	**Species**	**a**	**b**	**Source**
Labridae (Scarini) cont.	*Scarus fuscocaudalis*	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus ghobban	0.0165	3.041	*S. ghobban* (Kulbicki et al 2005)
	Scarus globiceps	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus hypselopterus	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus longipinnis	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus niger	0.0257	3.09	*S. niger* (Choat et al 1996)
	Scarus oviceps	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus prasiognathos	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus psittacus	0.0608	2.90	*S. psittacu*s (Choat et al 1996)
	Scarus quoyi	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus rivulatus	0.0173	3.14	*S. rivulatus* (Choat et al 1996)
	Scarus rubroviolaceus	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus scaber	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus schlegeli	0.0186	3.12	*S. schlegeli* (Choat et al 1996)
	Scarus spinus	0.0164	3.1086	*S. spinus* (Hoey and Bellwood unpubl. data)
	Scarus tricolor	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus virdifucatus	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus xanthopleura	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
	Scarus sp.	0.0234	2.956	*Scarus* spp. (Kulbicki et al 2005)
Siganidae	all species	0.0145	3.122	*Siganus* spp. (Kulbicki et al 2005)

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