# Mapping coral reef resilience

Structure:

1. Definitions of resilience, specifically in the coral reef context.
2. Overview of what RS can map directly on coral reefs, depth limits of passive optical data, some mention of the effects of spatial and spectral characteristics on accuracy.
3. Spatial predictive modeling basics – to go beyond what can be mapped directly.
4. Case studies
   1. Knudby Fiji
   2. Rowlands Saudi
   3. Pittman UVI
   4. Wooldridge water quality
   5. Maina Global exposure
5. Pulling everything together to map resilience (rather than indices): Suggest integration through simple mechanical models (Mumby) or more elaborate models that include dynamic human disturbances (Melbourne-Thomas).
6. Management applications – what would a manger do with a resilience map?

## Introduction

Global climate change is now recognized as one of the most important threats to coral reefs, primarily due to the increased frequency of mass coral bleaching events and severe storms expected as a result of our warming climate (Hoegh-Guldberg 1999, Hughes et al. 2003). A new paradigm has therefore emerged for coral reef management: Restoring and maintaining the natural ability of coral reef ecosystems to rebound to a desired state after exposure to climate-driven disturbances (Bellwood et al. 2004). That ability is most commonly termed “resilience” (Holling 1973). This chapter presents a review of how remote sensing and distribution modeling can be used to map coral reef resilience and thus help inform coral reef management in the 21st century.

## Coral reef ecosystem resilience

There are two broad notions of resilience, both focused on a system in dynamic equilibrium with deviations from a stable (climax) state caused by periodic disturbances (Holling 1996, Gunderson 2000). “Engineering resilience” assumes that the system in question has a single stable state that it will return to in the absence of disturbance; engineering resilience is typically quantified as the magnitude of deviation from, and speed of return to, the stable state following a disturbance. “Ecological resilience”, on the other hand, assumes that multiple stable states exist, each bounded by a domain of attraction; ecological resilience is thus considered the amount of disturbance the system can be exposed to without moving beyond its current domain of attraction and transitioning to another stable state (Holling 1996). Although these represent separate definitions of resilience, in the context of frequent disturbances the two definitions are linked because the magnitude of deviation from, and speed of return to, the stable state following one disturbance (i.e. engineering resilience) will determine the system state at the time of the following disturbance, and thus influence the amount of disturbance the system can be exposed to without moving beyond its current domain of attraction (i.e. ecological resilience) (Gunderson 2000).

The determination of both the state and internal dynamics of a system depends on the specific definition of the system as opposed to the external environment that influences its dynamics. In addition, the spatial and temporal scales at which the system is considered influence notions of stability and resilience. For example, the local functional extinction of a keystone species may result in dramatic state change that is irreversible in the short term, but with resettlement of this species from nearby source reefs, and subsequent recovery of the population, the resulting fluctuations in state may part of a long-term dynamic equilibrium. For example, coral reef ecosystems considered stable within the last ~50 years of monitoring may already have been pushed into non-climax unstable states by pre-historic and historic fishing pressure (Jackson 2001, Pauly), from which recovery is not possible on human time scales. It is thus typically not clear from observation alone what the stable state is, and thus what a coral reef ecosystem will rebound to after exposure to climate-driven disturbances.

Although the relevant spatial and temporal scales are rarely specifically defined, it is broadly accepted that multiple stable states exist for coral reef ecosystems (Knowlton 1992, Mumby et al. 2013 in Oikos, but see Dudgeon et al. 2010). Typically these include a desired coral-dominated state that, given a combination of press and pulse disturbances, can be replaced by an undesired macroalgae-dominated state. Stable states dominated by other organisms have also been documented (Davis 1982, Aronson et al. 2002, Loya 2004). Ecological resilience has thus been broadly adopted as the relevant resilience concept by the coral reef community, but it is difficult or impossible to measure in the absence of observed transitions between stable states, and thus not practical as a basis for resilience assessment or management. Engineering resilience, on the other hand, can be assessed by focusing on its two aspects, often termed “resistance” (to disturbance) and “recovery” (from disturbance), which can be quantified through natural experiments by monitoring relevant system state variables before, during and after a disturbance. In order to manage for resilience, factors that influence resistance and recovery can thus be identified and protected. Given the link between engineering and ecological resilience described above, assessment of engineering resilience is sufficient to inform management decisions.

A recent survey of expert opinion and scientific evidence (McClanahan et al. 2012) identified 11 principal factors that influence the resistance or recovery of coral reef ecosystems to climate-driven disturbances and are also feasible to assess from field observations. These include aspects of the coral fauna (presence of stress-resistant species, a diversity of coral species, high levels of coral recruitment, and absence of coral disease) and its competition for space (low presence of macroalgae) as well as moderators of that competition (herbivore biomass), the physical environment (high annual temperature variability, low nutrient and sediment levels), and direct human impacts (physical impacts and fishing pressure). To map coral reef resilience using remote sensing and modeling, these factors represent a list of mapping targets that, combined, have the potential for characterize the resilience of a coral reef ecosystem to climate-driven disturbance.

## What remote sensing can map on coral reefs

Passive optical remote sensing can provide a synoptic view of coral reef ecosystems and can, in theory, provide pixel-by-pixel estimates of a wide range of biological and physical variables (Mumby et al. 2004 review, Phinn et al. 2013 book chapter). However, this capability is limited to optically shallow areas where the seafloor can be discerned from adjacent deep water areas. In the clear tropical waters where most coral reefs are found this limits passive optical remote sensing to depths below 20-30 m; beyond this depth active optical (lidar) or acoustic (sonar) instruments, which provide different information, are necessary.

In addition, derivation of biological or physical information from the reflected solar radiation is complicated because the spectral radiance recorded by an airborne or space-based sensor depends on a combination of seafloor reflectance, water depth and water optical quality, and is additionally influenced by environmental (mainly atmospheric) and sensor noise (Brando and Dekker 2003). This limits the accuracy with which the biological and physical environment of relatively deep reef areas can be mapped.

In addition to shallow-water bathymetry (Lyzenga, Lee, Hedley), remote sensing of coral reefs has typically been used to map geomorphologic zones (Smith 1975, Andréfouët and Guzman 2005, Purkis et al. 2010), and benthic habitat categories (Ahmad and Neil 1994, Green et al. 1996, Mumby et al. 1997). Notable methodological developments have included object-based (Roelfsema et al. 2013) and semi-automated (Suzuki 2001) delineation of geomorphology, as well as a shift from per-pixel to object-based classification of benthic habitat (Leon and Woodroffe 2011, Phinn et al. 2012, Roelfsema et al. 2013). Due to the widespread use of fractional live coral cover as an indicator of reef health, methods have also been developed to map live coral cover (Hochberg, Goodman, Joyce, Hedley, Dekker). These methods typically rely on spectral unmixing approaches applied to airborne hyperspectral data, and are most successful in clear shallow water with few spectrally similar non-coral benthic cover types (Mumby and Hedley 2004, Dekker et al. 2011).

Individual coral species cannot be distinguished with remote sensing (Hochberg and Atkinson 2000, Hochberg et al. 2003), except in rare circumstances (Purkis et al. 2006), and as a result coral diversity or the presence of stress-resistant corals can also not be directly inferred from the remote sensing data. Although it has been demonstrated that corals affected by disease have a distinct spectral response measured in-situ (Anderson et al. PLOS ONE 2013), coral disease is unlikely to be detectable with existing remote sensing instruments, as is coral recruitment. As such, remote sensing is not capable of directly mapping any of the four aspects of the coral fauna identified as important resilience factors. Macroalgal cover, on the other hand, can be derived from the same spectral unmixing approaches used to map live coral cover (Goodman and Ustin 2007, Lee, Hedley book chapter), again with most success in clear shallow water. Herbivore biomass, an important factor moderating the competition for space between corals and macroalgae, cannot be mapped directly with remote sensing.

Remote sensing is better able to characterize the physical environment surrounding the reef ecosystem, with demonstrated applications of remote sensing for mapping concentrations of chlorophyll-a and coloured dissolved organic matter (Morel and Prieur 1977, Moses et al. 2009, more), development towards an operational algorithm for mapping sedimentation in coral reef environments (Ouillon et al. 2008), and operational systems used to characterize past (and present) temperature variability of the sea surface (Maina et al. 2008, McClanahan et al. 2007, NOAA 2013 http://www.ospo.noaa.gov/Products/ocean/sst.html).

Finally, there have been no demonstrated applications of remote sensing to map direct human physical impacts on coral reefs, and although the spatial distribution of fishing pressure is unfeasible to map directly, an indirect method based on local fishing fleet quantification has been developed (Rowlands et al. 2012).

## Spatial distribution modeling

|  |  |
| --- | --- |
| **Resilience indicator** | **Can be mapped with remote sensing** |
| Stress-resistant coral species |  |
| Coral species diversity |  |
| Coral disease |  |
| Coral recruitment |  |
| Macroalgal cover | √ |
| Herbivore biomass |  |
| Annual temperature variability | √ |
| Nutrient levels | √ |
| Sediment levels | √ |
| Direct physical impact |  |
| Fishing pressure | (√) |

## Case studies

This section needs a good introduction that explains why we are presenting all these different case studies and how they form a coherent whole. The basic idea is to provide a broad picture of the work that has been done so far on mapping all things relevant to coral reef resilience, including both resistance and recovery indicators and exposure.

## Integration through mechanistic modeling

Again, my basic idea here is that if we can populate a model like Mumby’s or the CORSET model, and introduce spatial dynamics, we get a way to map not just indicators, but an actual quantification of resilience (e.g. Mumby’s model can produce “probability to end in coral-dominated state after XX years).

## Management applications

The great problem for current coral reef conservation is to operationalize our understanding of ecosystem resilience, and apply it for management. This section should basically answer the question of “so you have a map of reef resilience, now what do you do?”

## Conclusion

Summary…