**Temporal effectiveness of biodiversity surrogates in coral reef communities in the British Virgin Islands**

Nicole B. Keefner1

1Department of Natural Resources Science, University of Rhode Island, Kingston, RI 02881, USA.

**Statement of the Problem**

This study aims to investigate the effectiveness of different taxonomic groups (sponges, corals, and reef fishes) as proxies for overall biodiversity in a tropical coral reef system. In addition to testing whether proxies are comparable among sites, I will test if and why they change similarly over time in response to natural and anthropogenic forces. I will compare the effectiveness of surrogates that can be surveyed in the field: recognizable taxonomic units (RTU’s) and functional groups. I will use data collected from reef monitoring in the British Virgin Islands from 1992 to 2017

**Justification for and Significance of the Study**

Biodiversity declines associated with increasing levels of anthropogenic stress disrupt community dynamics and are of great concern (Staudinger et al., 2013; Stork, 2010). However, because biodiversity often cannot be measured directly, proxies are often used. Proxies like diversity indices and species richness are useful when learning about changes in the species composition aspect of biodiversity (Hamilton, 2005). Unfortunately, these proxies that attempt to include all species require large expenditures of time, effort, and taxonomic expertise, and are therefore often prohibitively expensive (Magierowski & Johnson, 2006).

Surrogates are specific abiotic or biotic indicators/indicator groups that provide an estimate of a component of biodiversity (Lambeck, 1997). A good surrogate meets the assumptions that the target-surrogate relationship remains constant over time and space. Many studies have investigated the effectiveness of surrogates, with mixed results. For example, percent canopy cover was found to be a poor surrogate for bird richness in different geographic regions (Pierson, Mortelliti, Barton, Lane, & Lindenmayer, 2016). In contrast, mollusk diversity served as a good surrogate to estimate community diversity on the rocky shores of a marine park in Australia (Smith, 2005) and mycorrhizal fungal diversity was a good surrogate for plant diversity in lab and field experiments (Van Der Heijden et al., 1998).

Most surrogate studies to date have investigated the effectiveness of surrogates at different spatial scales, perhaps because of their widespread use to identify priority conservation areas (Margules, Pressey, & Williams, 2002; Sarkar & Margules, 2002) and sites for designation as marine reserves (T. Ward, Vanderklift, Nicholls, & Kenchington, 1999). For example, hedgerow bird communities act as surrogates for landscape quality at a broad scale and for landscape structure at a local scale, which helps define appropriate indicators for restoration efforts (Padoa-Schioppa, Baietto, Massa, & Bottoni, 2006).

Although the spatial assumption of surrogate effectiveness has been frequently investigated, many authors have noted a lack of studies that investigate the temporal effectiveness of surrogates (Bevilacqua, Mistri, Terlizzi, & Munari, 2018; Lewandowski, Noss, & Parson, 2010; Magierowski & Johnson, 2006; McArthur, Brooke, Przeslawski, Ryan, & Lucieer, 2010; Mellin et al., 2011; Rubal, Veiga, Vieira, & Sousa-Pinto, 2011). The few examples have produced mixed results. For example, percent canopy cover was a poor surrogate for bird population trends over periods greater than ten years (Pierson, Mortelliti, Barton, Lane, & Lindenmayer, 2016). In contrast, a group of 35 biodiversity surrogates defined using a 5-year pilot data set successfully detected changes in the species assemblage structure over a subsequent 5-year test period in a temperate brackish system (Bevilacqua et al., 2018).

In addition to spatial effectiveness, surrogate studies often focus on taxonomic sufficiency (i.e., the taxonomic resolution required to maximize surrogate effectiveness) (Fontaine, Devillers, Peres-Neto, & Johnson, 2015; Musco, Mikac, Tataranni, Giangrande, & Terlizzi, 2011; Noss, 1990; Olsgard & Somerfield, 2000). Few studies have investigated the use of recognizable taxonomic units (i.e. RTU’s) or functional groups when identifying potential surrogates. RTU’s are taxonomic units defined by readily identifiable characteristics in the field (Sebek et al., 2012). Some surrogates identified using functional groups have been consistent with those using taxonomic designations (Rubal et al., 2011). However, functional and taxonomic diversity can provide different information when measured at different scales (Törnroos, Nordström, & Bonsdorff, 2013).

Because they are used as proxies to monitor specific aspects of biodiversity, surrogates are especially relevant when studying high-diversity ecosystems, such as coral reefs. Coral reefs are being progressively degraded by a suite of anthropogenic stressors (Habibi, Setiasih, & Sartin, 2007; Hughes et al., 2017; Stubler, Duckworth, & Peterson, 2015).Reef fishes, hard corals (Scleractinia), and sponges are dominant coral reef organisms that establish and maintain biodiversity by filling multiple functional roles in coral reef systems (Angelini, Altieri, Silliman, & Bertness, 2018). Reef fish richness has been found to be a better surrogate than coral richness for estimating the diversity of corals and fishes when deciding on areas to become marine reserves (Beger, Jones, & Munday, 2003). However, this finding was not investigated over time and sponges were not considered in the study. Understanding whether these groups can be used as surrogates for other taxonomic groups, would provide valuable information to managers with limited monitoring resources.

I will use data collected from the British Virgin Islands (Forrester et al., 2015) to investigate the effectiveness of reef fishes, sponges, and corals as surrogates for biodiversity over space and time. I will test the hypothesis that, of reef fishes, corals, and sponges, reef fishes will act as the most effective surrogate for the others. I also hypothesize that surrogate performance will be consistent over successive years of monitoring. Finally, I hypothesize that using RTU’s to identify surrogates will be consistent with using functional groups and that this relationship will hold for successive years of monitoring.

**Methodology or Procedures**

*Field study design*

There were eight sites around Guana Island in the British Virgin Islands (Fig. 1). All sites were similar in covering 0.6-1.0 hectares of fringing reef adjacent to the island at a depth of 8-10 m. Sites varied in exposure to prevailing weather; sites on the windward north side of the island are more exposed than those on the southern leeward side. Each site was surveyed annually from 1992-2017, except that sponges were not counted in 1993 at Crab Cove or in 2014 at Pelican Ghut. All surveys were done between June and August. Each year, fish densities, coral cover, and sponge abundance relief were measured using 3-12 transects (mean = 4.3) per site. Transects were 30 m long, and placed at selected locations within each site using a haphazard sampling approach.

*Census methods*

Corals, sponges, and fish were sampled using well-established methods. For corals, divers recorded percent cover for 27 recognizable taxonomic units (RTU’s) of hard corals (Forrester et al., 2015). They used the linear point-intercept method and recorded the substrate or coral group every 0.25 m along the 30-m transect. These point observations were converted to surface area estimates of percent cover (Ohlhorst, Liddell, Taylor, & Taylor, 1988). At the same sites, they also recorded counts for 58 RTU’s of sponges using a line intercept method (Forrester et al., 2015). Fishes were counted within a belt transect 30 m long x 1.5 m wide, and a T-shaped bar was used to determine the transect width. Fish counts were restricted to species that are amenable to visual survey, that is, day-active species that are relatively site-attached and reliably visible to divers. Nocturnal species, highly mobile groups such as mackerels (Scombridae) and jacks (Carangidae) that are transient visitors to the sites, and small cryptic groups like gobies (Gobiidae) and blennies (Blennioidei) that often hide in crevices were not surveyed. Newly recruited juvenile fishes (< 1 month on the reef) were also excluded because their abundance is strongly affected by lunar cycles, which complicates the detection of long-term trends.

Differences between observers can influence coral reef survey data, but are unlikely to influence the outcome of this study. Fish and sponge data were both collected by a single expert observer. Coral data were collected by three observers, but new observers’ species identifications and counts were intercalibrated with those of another observer during a training period of at least 15 dives before their data were incorporated into the study

*Recognizable taxonomic units*

Fish, corals and sponges were identified to the most specific taxonomic group possible in the field. All fish were identified to species, while corals and sponges were identified as multi-species RTU’s (D. Ward & Stanley, 2004) rather than species for the following reasons: (1) taxonomists reassigned taxa thought to be different species to the same species after the study began, (2) taxonomists divided a single species into multiple species after the study began, and (3) several species are visually indistinguishable in the field. In all cases, the lowest resolution RTU was used. For example, in 1994 the coral *Montastraea annularis* was recognized to be three separate species (*M. annularis*, *M. faveolata*, and *M. franksi*) (Weil & Knowlton, 1989). Although the species can now be distinguished visually, and were counted separately after 1994, the aggregate was used because the study began in 1992 before the distinction was discovered.

*Functional groups*

Species will also be classified based on their functional role within the ecosystem because the diversity of functional groups has been shown to increase reef resilience (Nyström, 2006). For sponges, the major functional roles consist of erosion, stabilization (accretion), bentho-pelagic coupling, and associations with other organisms such as, settlement substrate for algae, habitat for microorganisms, and protecting bivalves from predation (Bell, 2008). Although not understood as well as the others, bentho-pelagic coupling may have significant impacts on the microhabitats available in the reef because some sponges have pumping rates of two times their own volume of water per hour (Bell, 2008). Coral functional roles will be defined by colony shape and morphology (Bellwood, Hughes, Folke, & Nyström, 2004), as well as life history strategy (Bak & Engel, 1979). Because fish influence the community primarily though their role as consumers, they will be classified by trophic group and maximum body size(Halpern & Floeter, 2008).

**Resources Required**

Coral dataset; Sponge dataset; Reef fish dataset; Program R; R Studio

**Literature Cited**

R Core Team (2017). R: A language and environment for statistical computing.

R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.Angelini, C., Altieri, A. H., Silliman, B. R., & Bertness, M. D. (2018). Interactions among Foundation Species and Their Consequences ­ for Community Organization , Biodiversity , and Conservation, *61*(10). https://doi.org/10.1525/bio.2011.61.10.8

Bak, R. P. M., & Engel, M. S. (1979). Distribution, abundance and survival of juvenile hermatypic corals (Scleractina) and the importance of life history strategies in the parent coral community. *Marine Biology*, *54*, 341–352.

Beger, M., Jones, G., & Munday, P. (2003). Conservation of coral reef biodiversity: a comparison of reserve selection procedures for corals and fishes. *Biol Cons*, *111*.

Bell, J. J. (2008). The functional roles of marine sponges. *Estuarine, Coastal and Shelf Science*, *79*, 341–353. https://doi.org/10.1016/j.ecss.2008.05.002

Bellwood, D. R., Hughes, T. P., Folke, C., & Nyström, M. (2004). Confronting the coral reef crisis. *Nature*, *429*, 827–833. https://doi.org/https://doi.org/10.1038/nature21707

Fontaine, A., Devillers, R., Peres-Neto, P. R., & Johnson, L. E. (2015). Delineating marine ecological units: A novel approach for deciding which taxonomic group to use and which taxonomic resolution to choose. *Diversity and Distributions*. https://doi.org/10.1111/ddi.12361

Forrester, G., Baily, P., Conetta, D., Forrester, L., Kintzing, E., & Jarecki, L. (2015). Comparing monitoring data collected by volunteers and professionals shows that citizen scientists can detect long-term change on coral reefs. *Journal for Nature Conservation*, *24*, 1–9. https://doi.org/10.1016/j.jnc.2015.01.002

Habibi, A., Setiasih, N., & Sartin, J. (2007). A decade of reef check monitoring: Indonesian coral reefs, condition and trends.

Halpern, B. S., & Floeter, S. R. (2008). Functional diversity responses to changing species richness in reef fish communities. *Marine Ecology Progress Series*. https://doi.org/10.3354/meps07553

Hamilton, A. J. (2005). Species diversity or biodiversity? *Journal of Environmental Management*. https://doi.org/10.1016/j.jenvman.2004.11.012

Hughes, T. P., Kerry, J. T., Álvarez-Noriega, M., Álvarez-Romero, J. G., Anderson, K. D., Baird, A. H., … Wilson, S. K. (2017). Global warming and recurrent mass bleaching of corals. *Nature*, *543*(7645), 373–377. https://doi.org/10.1038/nature21707

Lambeck, R. J. (1997). Focal species: A multi-species umbrella for nature conservation. *Conservation Biology*. https://doi.org/10.1046/j.1523-1739.1997.96319.x

Magierowski, R. H., & Johnson, C. R. (2006). *ROBUSTNESS OF SURROGATES OF BIODIVERSITY IN MARINE BENTHIC COMMUNITIES*. *Ecological Applications* (Vol. 16).

Margules, C., Pressey, R., & Williams, P. (2002). Representing biodiversity: data and procedures for identifying priority areas for conservation. *J Biosci*, *27*.

Musco, L., Mikac, B., Tataranni, M., Giangrande, A., & Terlizzi, A. (2011). The use of coarser taxonomy in the detection of long-term changes in polychaete assemblages. *Marine Environmental Research*. https://doi.org/10.1016/j.marenvres.2010.12.004

Noss, R. (1990). Indicators for monitoring biodiversity - A hierarchical approach. *Cons Biol*, *4*.

Nyström, M. (2006). Redundancy and response diversity of functional groups: Implications for the resilience of coral reefs. *AMBIO: A Journal of the Human Environment*, *35*(1), 30–35. https://doi.org/10.1579/0044-7447-35.1.30

Ohlhorst, S. L., Liddell, W. D., Taylor, R. J., & Taylor, J. M. (1988). EVALUATION OF REEF CENSUS TECHNIQUES. *Proceedings of the 6th International Coral Reef Symposium, Australia*, *2*, 319–324.

Olsgard, F., & Somerfield, P. (2000). Surrogates in marine benthic investigations: Which taxonomic unit to target? *J Aquat Ecosyst Stress Recovery*, *7*.

Padoa-Schioppa, E., Baietto, M., Massa, R., & Bottoni, L. (2006). Bird communities as bioindicators: The focal species concept in agricultural landscapes. *Ecological Indicators*. https://doi.org/10.1016/j.ecolind.2005.08.006

Pierson, J. C., Mortelliti, A., Barton, P. S., Lane, P. W., & Lindenmayer, D. B. (2016). Evaluating the effectiveness of overstory cover as a surrogate for bird community diversity and population trends. *Ecological Indicators*. https://doi.org/10.1016/j.ecolind.2015.10.031

Sarkar, S., & Margules, C. (2002). Operationalizing biodiversity for conservation planning. *J Biosci*, *27*.

Sebek, P., Barnouin, T., Brin, A., Brustel, H., Dufrêne, M., Gosselin, F., … Bouget, C. (2012). A test for assessment of saproxylic beetle biodiversity using subsets of “monitoring species.” *Ecological Indicators*, *20*, 304–315. https://doi.org/10.1016/j.ecolind.2012.02.033

Staudinger, M. D., Carter, S. L., Cross, M. S., Dubois, N. S., Duffy, J. E., Enquist, C., … Turner, W. (2013). Biodiversity in a changing climate : A synthesis of current and projected trends in the US in a nutshell. *Frontiers in Ecology and the Environment*, *11*(9), 465–473. https://doi.org/10.1890/120272

Stork, N. E. (2010). Re-assessing current extinction rates. *Biodiversity and Conservation*, *19*(2), 357–371. https://doi.org/10.1007/s10531-009-9761-9

Stubler, A. D., Duckworth, A. R., & Peterson, B. J. (2015). The effects of coastal development on sponge abundance, diversity, and community composition on Jamaican coral reefs. *Marine Pollution Bulletin*, *96*, 261–270. https://doi.org/10.1016/j.marpolbul.2015.05.014

Svensson, J. R., Lindegarth, M., Siccha, M., Lenz, M., Molis, M., Wahl, M., & Pavia, H. (2007). Maximum species richness at intermediate frequencies of disturbance: Consistency among levels of productivity. *Ecology*, *88*(4), 830–838. https://doi.org/10.1890/06-0976

Van Der Heijden, M. G. A., Klironomos, J. N., Ursic, M., Moutoglis, P., Streitwolf-Engel, R., Boller, T., … Sanders, I. R. (1998). Mycorrhizal fungal diversity determines plant biodiversity, ecosystem variability and productivity. *Nature*. https://doi.org/10.1038/23932

Ward, D., & Stanley, M. (2004). The value of RTUs and parataxonomy versus taxonomic species. *New Zealand Entomologist*, *27*, 3–9. Retrieved from http://www.tandfonline.com/doi/abs/10.1080/00779962.2004.9722118

Ward, T., Vanderklift, M., Nicholls, A., & Kenchington, R. (1999). Selecting marine reserves using habitats and species assemblages as surrogates for biological diversity. *Ecol Appl*, *9*.

Weil, E., & Knowlton, N. (1989). A multi-character analysis of the Caribbean coral Montastraea annularis and its two sibling species M. faveolata and M. franksi. *Smithsonian*, *55*(September).

Figure 1. A map of Guana Island, British Virgin Islands, showing the eight study sites: (1) Grand Ghut, (2) Pelican Ghut, (3) Bigelow Beach, (4) Monkey Point, (5) White Bay, (6) Iguana Head, (7) Crab Cove, and (8) Long Point.

