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

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**Statement of the Problem**

This study aims to investigate surrogate effectiveness of sponges, corals, and reef fishes in a tropical reef system over a span of 26 years. I will compare the effectiveness of RTU’s and functional groups. I will use data collected from reef monitoring in the British Virgin Islands.

**Justification for and Significance of the Study**

Biodiversity changes and declines associated with higher levels of anthropogenic stress disrupt community dynamics and are of great concern (Staudinger et al., 2013; Stork, 2010). However, because biodiversity 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 census-like proxies require large expenditures of time, effort, and taxonomic expertise, and are therefore often prohibitively expensive (Magierowski & Johnson, 2006).

Surrogates are more specific abiotic or biotic indicators/indicator groups that provide an estimate of a target aspect of biodiversity (Lambeck, 1997). A good surrogate meets the assumptions that the target-surrogate relationship remains constant over time and space. Many studies investigate the effectiveness of surrogates, some of which identify unsuccessful surrogates. For example, percent canopy cover was not a good surrogate to estimate bird richness in different geographic regions (Pierson, Mortelliti, Barton, Lane, & Lindenmayer, 2016). Other studies identify successful ones. For example, mollusk diversity serves as a good surrogate to estimate community diversity on the rocky shores of a marine park in Australia (Smith, 2005) and mycorrhizal fungal diversity acts as a good surrogate for plant diversity in lab and field experiments (Van Der Heijden et al., 1998).

Most surrogate studies to date investigate the spatial effectiveness of surrogates at different scales. This is likely because several authors suggest how surrogates may be used to identify priority conservation areas (Margules, Pressey, & Williams, 2002; Sarkar & Margules, 2002). A terrestrial study found hedgerow bird species abundances act as surrogates for landscape quality at a broad scale and for landscape structure at a local scale- guiding restoration efforts (Padoa-Schioppa, Baietto, Massa, & Bottoni, 2006). Spatial effectiveness has also been considered when comparing the use of surrogates such as habitat category, plants, fish, and invertebrates to identify marine reserves based on their ability to estimate all taxa (T. Ward, Vanderklift, Nicholls, & Kenchington, 1999).

Although the spatial assumption of surrogate effectiveness is frequently investigated, the lack of studies that investigate the temporal effectiveness of surrogates is often pointed out (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). Few studies are published on the topic and most of those take place in less than two years (Magierowski & Johnson, 2006; Rubal, Veiga, Vieira, & Sousa-Pinto, 2011). Percent canopy cover was found to be an inadequate surrogate when estimating bird population trends over a period of more than ten years (Pierson, Mortelliti, Barton, Lane, & Lindenmayer, 2016). Even more recently, a study identified surrogates over a five year period and examined the temporal effectiveness of the identified surrogates over the subsequent five years in a temperate brackish system (Bevilacqua et al., 2018).

In addition to spatial effectiveness, surrogate studies preferentially 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). Therefore, few studies investigate the use of groups such as recognizable taxonomic units (i.e. RTU’s[[1]](#footnote-1)) and functional groups when identifying potential surrogates. Effective surrogates identified using functional groups have been consistent with those using taxonomic designations (Rubal et al., 2011). However, functional and taxonomic diversity have been shown to provide different information when considering different scales (Törnroos, Nordström, & Bonsdorff, 2013).

Coral reefs are an example of an ecosystem altered by anthropogenic stressors. Due to persistent high temperatures in Australian waters, four times more reefs experienced extreme bleaching (>60% bleached) in 2016 than in 2002 (Hughes et al., 2017). Overharvesting in Indonesian reefs has led to declines in populations of economically and ecologically important fishes and invertebrates, including giant clams, as well as a 10% decrease in live cover from 1997-2006 (Habibi, Setiasih, & Sartin, 2007). Increased sedimentation rates in Jamaica have led to a decrease in the volume of sponges in coral reefs (Stubler, Duckworth, & Peterson, 2015).

Reef fishes, corals, and sponges are spatially dominant organisms that establish and maintain biodiversity by filling multiple functional roles in coral reefs (Angelini, Altieri, Silliman, & Bertness, 2018). Reef fish richness has been found to be a better surrogate than coral richness 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 the others, would provide valuable information to managers with limited monitoring resources.

This study will use data collected from the British Virgin Islands (Forrester et al., 2015) to investigate surrogate effectiveness of reef fishes, sponges, and corals over time. It will be the first study of surrogate effectiveness to consider 1) tropical coral reefs for a time span greater than 2 years, 2) the use of RTU's, and 3) a time span greater than 10 years. 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**

*Data Collection*

At 8 different locations around Guana Head Island in the British Virgin Islands, divers recorded percent cover for 27 recognizable taxonomic units (RTU’s) of corals (Forrester et al., 2015). They used the linear point-intercept method and recorded the substrate or coral group every 0.25 m for 3 30-m transects at each site between June and August from 1992-2016. 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. They used the line intercept method for 3 30-m transects between June and August from 1993-1995, 2000-2003, and 2005-2016 (Forrester et al., 2015). For both datasets, transects were set up at a depth of approximately 10-m. In 2011, transects for the sponge counts were conducted by two observers (E. MacLean and L. Jarecki), but in all other years, there was one observer (L. Jarecki). A percent cover of zero implies there were no individuals of that coral group represented on that transect. The 8 primary sites are White Bay, Muskmelon, Pelican Ghut, Crab Cove, Grand Ghut, Bigelow, Guana Head, and Monkey Point. In years when the primary site was unavailable for logistical reasons, an alternate site was surveyed. These sites are White Bay-alt, Monkey Pt area, White Bay E, and Bigelow-south. The sites were all fringing reefs and the only apparent difference among sites was the increased exposure to waves at the two sites to the north of the island. There are no negative values and data regarding sponges was not collected in 1993 at Crab Cove or 2014 at Pelican Ghut.

All sponges and corals were identified to species in the field where possible. However, for the following reasons several of the RTU’s include more than one species: (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 after the study began, and (3) field identification is impossible for some species because several species share morphological characteristics that make them indistinguishable from each other. For the former two cases, the lowest resolution RTU was used. For example, the coral *Montastraea annularis* was recognized to be three separate species (*M. annularis*, *M. faveolata*, and *M. franksi*) in 1994 (Weil & Knowlton, 1989), but because the study began in 1992, the aggregate group was used.

*Data Analysis*

For the analysis, I will include data from the 8 primary sites and reconcile data inconsistencies. All corals included in the analysis will be live hard corals in the order Scleractinia. All database management and analysis will be conducted in R (R Core Team, 2017).

Because corals and sponges are both taxonomically diverse groups and within these groups there are species with morphological similarities, some of the groups recorded may be considered recognizable taxonomic units (D. Ward & Stanley, 2004). In this study, the potential for observer error was minimized because the number of observers was small (3?? For corals and two for sponges), and because new observers’ counts were calibrated during a training period of at least 15 dives before their data were incorporated into the study. Because this study has one year for which two observers recorded sponge counts after calibration, there is a potential to investigate the reproducibility of the groups. For the analysis, I assume that within group variation is smaller than the spatial and temporal variation.

Because the diversity of functional groups has been shown to increase reef resilience (Nyström, 2006), species may be considered based on their functional role within the ecosystem. For sponges, the major functional roles consist of erosion, stabilization (accretion), bentho-pelagic coupling, and associations with other organisms (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 are typically determined by colony shape and structure and consist of massive, plate-like, bushy, columnar, unattached, encrusting, staghorn, and bottlebrush (Bellwood, Hughes, Folke, & Nyström, 2004). However, corals with different life history strategies have been shown to differ in their dominance on the reef (Bak & Engel, 1979).

For both RTU’s and functional groups, Bray-Curtis dissimilarity and nMDS will be applied.

**Resources Required**

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

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1. RTU’s are taxonomic units defined by readily identifiable characteristics in the field. For some groups, including corals and sponges, not all individuals can be recognized to species in the field, so this study uses RTU’s identified by SCUBA survey. [↑](#footnote-ref-1)