**Temporal effectiveness of biodiversity surrogates in**

**coral reefs in the British Virgin Islands**

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Biological and Environmental Sciences

MS Thesis Proposal

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**Abstract**

Biodiversity is declining around the world. There is an increasing need for rapid identification of species and biodiversity and conservation hotspots. One method of doing this is to employ surrogates, which are used to estimate characteristics, such as diversity, of a target species or group of species. A good surrogate is expected to be easier to monitor than the target species and meets the assumption that the target-surrogate relationship is constant over space and time. Our objective was to evaluate the temporal stability of surrogates in coral reef systems around Guana Island in the British Virgin Islands using data from an ongoing monitoring effort of fishes, corals, and sponges. Of these, corals are the simplest to monitor and we hypothesize coral richness will be a good surrogate for fish richness and sponge richness. We quantified sponges, corals, and fishes using transect methods for eight different sites around the island. We also investigated how the inclusion of recognizable taxonomic units (RTU’s) compares to species-level studies. We sought correlated relationships to identify taxonomic groups that act as surrogates for target groups. Our results provide insight on the use of surrogates in a coral reef ecosystem and on the inclusion of RTU’s in biodiversity studies. The identification of effective surrogates can inform the allocation of limited resources for conservation planning including decisions regarding existing data from monitoring studies and collection of future data.

**Introduction**

Biodiversity changes and declines associated with increasing levels of anthropogenic stress disrupt community dynamics and are of great concern because biodiversity contributes to ecosystem function (Emmett Duffy, 2009; Staudinger et al., 2013; Stork, 2010). However, true biodiversity often cannot be known in natural environments because detection is imperfect (Kéry et al., 2009). As a result, proxies such as species richness, evenness, and diversity indices have been developed that provide ways to estimate different aspects of biodiversity in the field. Richness is the count of species in a given area. Evenness is a measure of whether or not the number of individuals of each species is equivalent and may provide information about dominance-related patterns. There is contention over evenness in that it may be affected by trophic level, required resources/space, and reproductive frequency (i.e. just because there are more individuals than others doesn’t mean that the system is imbalanced)Diversity indices express both richness and evenness at the same time.

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).

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Surrogates are specific abiotic or biotic indicators/indicator groups that provide an estimate of a component of biodiversity (Lambeck, 1997). An effective surrogate takes less time, money, and experience to measure than the target. In addition, 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, a study greater than 10 years found percent canopy cover was a poor surrogate for bird population trends (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. 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 reef fish richness will be a good surrogate for the richness of corals and sponges over time. Reef fish richness has been shown to be a more effective surrogate for the diversity of corals and fishes than coral richness (Beger et al., 2003). I hypothesize that total coral cover will be an effective surrogate for diversity of fishes, corals, and sponges because coral-dominated reefs have the potential for a greater diversity in structural and resource-based niches (David R. Bellwood & Hughes, 2001). More specifically I hypothesize that total coral cover will be inversely related to sponge diversity due to competitive interactions between sponges and corals (Powell et al., 2014) and that total coral cover will be an effective surrogate for fish diversity because larger reefs provide more habitat for fishes (Darling et al., 2017; Pratchett, Hoey, Wilson, Messmer, & Graham, 2011). Similarly, I will test the hypothesis that the diversity of coral morphological groups will be a good surrogate for fish diversity because of the greater variety in size and shape of available refugia (Darling et al., 2017). I hypothesize that the richness of trophic groups of reef fish will be an effective surrogate for fish richness (Halpern & Floeter, 2008). I hypothesize that *Acropora* coral cover will be a good surrogate for the estimated abundance of large reef fishes (≥20 cm total length) over time (Kerry & Bellwood, 2012, 2015). Because of their relationships with maintaining reef structure, I hypothesize that cover of *Acropora* and *Montastrea* (now *Orbicella*) will be effective surrogates for the diversity of coral morphological groups and that cover of *Pocillopora* and *Porites* will be inversely related to the diversity of coral morphological groups (Alvarez-Filip, Carricart-Ganivet, Horta-Puga, & Iglesias-Prieto, 2013; Perry et al., 2015). Finally, I hypothesize that surrogate performance will be consistent over successive years of monitoring.

**Methods**

*Field study design*

There were eight study 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 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 1992-2016, except that sponges were not counted in 1992, 1996-1999, 2004, 1993 at Crab Cove, or in 2014 at Pelican Ghut. All surveys were conducted between June and August. Each year, fish densities, coral cover, and sponge abundances were measured using 3-12 transects (mean = 4.3) per site. Transects were 20-30.4 m long (mean = 29.95), and placed at selected locations within each site using a haphazard sampling approach.

*Survey methods*

For each transect at each site, corals, sponges, and fishes were sampled once per year using well-established methods (Fig. 2). 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 as the diver swam along the transect line. 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. Because fish were the only mobile organisms of the three taxonomic groups included in this study, the fish survey was conducted first for each transect in order to reduce the bias caused by “spooking” the fish (Emslie, Cheal, MacNeil, Miller, & Sweatman, 2018). The diver responsible for identifying corals used the linear point-intercept method and recorded the substrate or coral group every 0.25 m along the 30-m transect. There were 27 recognizable taxonomic units (RTU’s) of hard corals (Forrester et al., 2015). These point observations were later converted to surface area estimates of percent cover (Ohlhorst, Liddell, Taylor, & Taylor, 1988). Sponges were surveyed using the line intercept method in which any sponge that intercepted the transect was recorded. There were 58 RTU’s of sponges (Forrester et al., 2015).

Differences between observers can influence coral reef survey data, but are unlikely to influence the outcome of this study. Fish and sponge data were collected by a single respective 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.

*Different survey techniques*

If we had a line and sampled enough points along the line (i.e. as the number of points approaches infinity) then the linear point-intercept and the line intercept methods would yield the same results for abundance if we knew which points corresponded to which individuals. As of right now, the line intercept method yields counts of individuals (closer to the truth for abundance) and the linear point intercept method yields points covered by a given species which may be converted to percent cover using methods developed for vascular plants (closer to the truth for cover).

*Measures of diversity*

Richness, diversity indices, and evenness (rank-abundance) will all be used as measures of biodiversity. Species richness is the number of species in a given area. Evenness can give us information about dominance-related patterns. Diversity indices are one way to express dominance/evenness and richness at the same time.

*Sampling effort*

Species accumulation curves have demonstrated that richness may increase as sampling effort increases until an asymptote is reached that represents true site richness (i.e. the more you look, the more you’ll find). Our study aims to investigate how sampling effort may affect estimates of richness (RTU richness).

Because the data is recorded for each transect for each site in a given year, it must be consolidated to have one record per site per year (sampling unit). The minimum sampling effort is three transects. In cases when there are data for more than three transects at a site in a given year, a species accumulation curve will be simulated and if the difference in number of transects makes a difference in species accumulation, 3 transects will be selected at random (may go back and randomly select 3 different transects to see if results would change).

Two methods were used to consolidate counts within a given sampling unit: (1) sum counts over all transects and divide by the number of transects to get the count per transect and (2) sum counts within each transect and divide each sum by the length of the transect to get count per meter, then sum the counts per meter for all transects and divide by the number of transects to get the count per meter per transect (Equations 1 and 2)

1) 2)

Dividing by the number of transects assumes a linear relationship between effort and richness. Equation 2 accounts for differences in transect length.

*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.

Identification concerns are:

1)\*\*\*Changing taxonomy (e.g. lumping, splitting) Our study aims to investigate how changes in taxonomic groupings may affect estimates of richness

2) Coarseness of taxonomic level (e.g. species, family), which has often been the focus of taxonomic sufficiency studies

3) \*\*\*Ability to distinguish morphologically identical species that may not be taxonomically related (i.e. recognizable taxonomic units/RTUs) Our study aims to investigate how RTUs may affect estimates of richness

It is especially important to consider these potential issues for long-term studies because methods to distinguish species in the field may improve as our taxonomic understanding changes

\*\*\*look up similarities and differences between taxonomic levels studies comparing biological traist and how if you’re related, you’re probably more similar; aka phylogenetic constraints; how do they control for phylogeny?

We also have uneven taxonomic resolution. We have observed richness which, for corals and fishes, might be able to be reconciled to taxonomic richness, which could, in turn, be used to determine species richness. This observed richness 🡪taxonomic richness and taxonomic richness 🡪 species richness jump assumptions can be tested. For example, if site 1 has 10 RTUs and site 2 has 100 RTUs, does site 2 have 100 times more species than site 1? Or, can we collapse the taxonomic resolution to lowest common taxonomic id and retain the same richness relationships (is this even conceptually useful?)? This would require a simulation/extrapolation of RTUs to species. I could simulate this with fish, but I would need to mimic the pattern of taxonomic relatedness seen in sponges.

-Relationship between species richness and RTU richness: If a site has 10 RTUs, it could have 10 species OR, if 5 of these 10 have 2 species, then 10 RTUs might also be 15 species…basically we can say in this example that 10 RTUs can have 10-15 species

-simulate different outcomes with “observed” vs “truth” by lumping into RTUs OR we know RTUs and potential “truth” so first assume min and then max species richness to create bounds; want to determine if RTU richness is at least proportional to species richness BUT can we provide more? Maybe robust distributions (randomly change data uniformly between 2 and 10 to get estimates; r^2 from 0.5-1 good, but may range from 0-1); what might lead you to predict that this relationship is distorted between groups? Different may be due to close relationships, mimicry, or convergent evolution (see notes in red book)

*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 (D. R. 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).

*Modeling*

The relationships between fishes, sponges, and corals were modeled in a linear regression framework. A model with an interactive effect of sponges and corals was included in case fish do not have a preference for using the structural complexity of corals over sponges for shelter.

We will assume the species within an RTU have the same detectability, abundance, and distribution. An RTU may include one species that may be more common than the other included in the RTU, but we assume not.

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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.

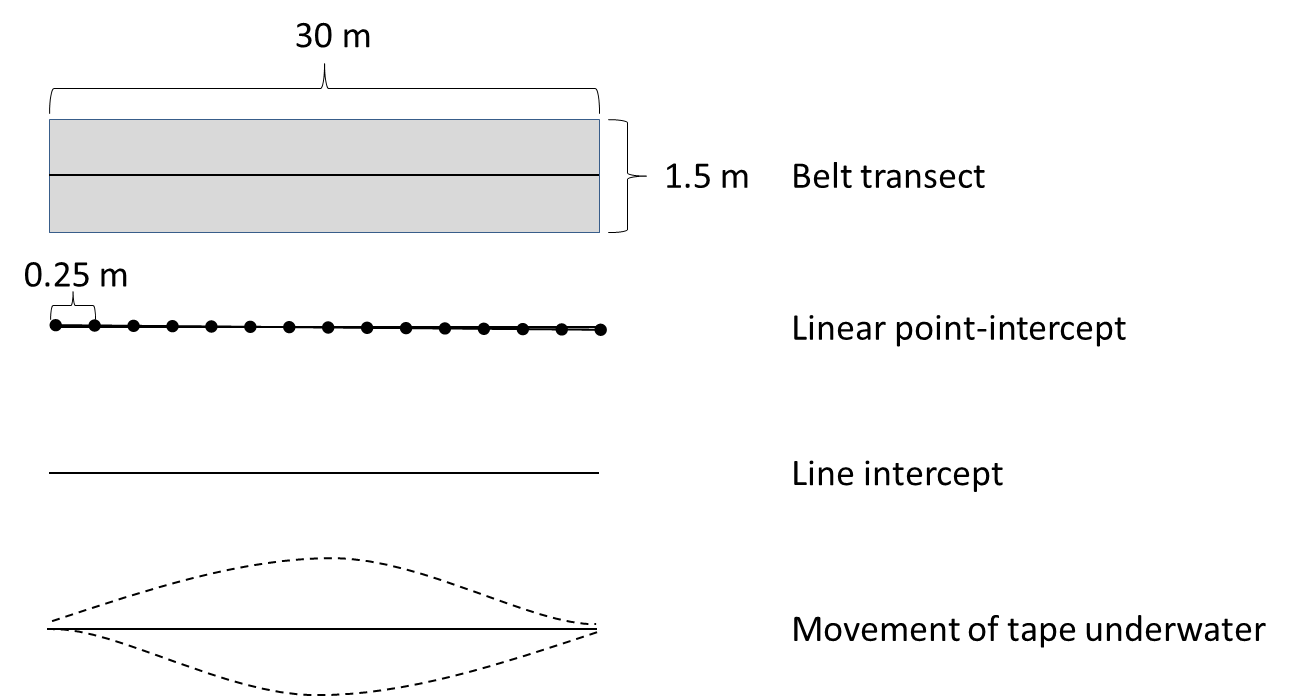


Figure 2. Preliminary depiction of the various transect methods used in this survey.