**Temporal effectiveness of biodiversity surrogates in**

**coral reefs in the British Virgin Islands**

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Biological and Environmental Sciences

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# Abstract

Biodiversity is declining around the world, necessitating rapid identification of species distribution contractions and population declines to identify conservation priorities. Surrogates are increasingly being used to meet this challenge. A good surrogate is expected to be easier to monitor than the target component of biodiversity and meets the assumption that the target-surrogate relationship is constant over space and time. Our objective was to evaluate the spatio-temporal stability of surrogates in coral reef systems around using data from an ongoing 26-year monitoring program in the British Virgin Islands that has quantified the abundance of fish, coral, and sponge species at 8 sites. Of these taxa, corals are the most widely monitored and measures of coral cover are often assumed to be good surrogates for diversity of reef-associated taxa. We thus hypothesize that coral cover and rugosity will be good surrogates for fish, coral, and sponge species richness. We also investigated how the inclusion of recognizable taxonomic units (RTU’s) compares to species-level studies. We sought correlated relationships between the proposed surrogates and fish, coral, and sponge species richness. 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 surrogates that maintain stable relationships with target components of biodiversity over time can inform decisions regarding existing data from monitoring studies and the allocation of limited resources for 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). There are many aspects of biodiversity: landscape, ecosystem, taxonomic, and genetic diversity (Duelli & Obrist, 2003; Noss, 1990). Taxonomic diversity, particularly the diversity of species, is fundamental to understanding evolutionary patterns of distributions, interspecific interactions, and environmental conditions.Species diversity is, consequently, the most commonly measured component of biodiversity in ecological and conservation-related field studies. Species diversity is usually estimated in the field using measures such as species richness where richness is the count of species in a given area (Hamilton, 2005).

Unfortunately, a complete inventory of species present is unattainable in many ecosystems and, for taxonomic groups that can be inventoried in principle, monitoring strategies that could detect all species in a given habitat are often prohibitively expensive and time-consuming (Kati et al., 2004). In practice, surrogates are often used as simple indicators that provide an estimate of a target component of biodiversity (Magierowski & Johnson, 2006). Surrogates may be functional (e.g., structural complexity), taxonomic (e.g., species diversity), or landscape features (e.g., percent canopy cover; Paillet et al., 2018; Wessels, Freitag, & Van Jaarsveld, 1999). An effective surrogate takes less time, money, and experience to measure than the target and maintains a consistently strong correlation with the target over time and space (Magierowski & Johnson, 2006).

Ecological dynamics commonly change across spatial scales (Wiens, 1989), thus it is not surprising that most surrogate studies to date have investigated the effectiveness of surrogates across spatial scales (Kati et al., 2004). The prevalence of studies considering surrogates across spatial scales may also be due to the widespread use of surrogates to identify priority conservation areas; this task requires an understanding of how the size and dispersion of the areas being conserved will affect the dynamics between the surrogate and target (Margules, Pressey, & Williams, 2002; Padoa-Schioppa, Baietto, Massa, & Bottoni, 2006; Sarkar & Margules, 2002; T. Ward, Vanderklift, Nicholls, & Kenchington, 1999). However, few studies have explicitly investigated surrogate effectiveness across temporal scales, and those that have are typically quite short (e.g., 13 months and 1 year; Magierowski & Johnson, 2006; Rubal, Veiga, Vieira, & Sousa-Pinto, 2011). An effective surrogate must maintain a stable relationship with the target over time because environmental conditions vary and a species’ ability to respond to this variation across time changes. One of the few longer studies of ten years, which also included different geographic regions, concluded that percent canopy cover was a poor surrogate for bird richness (Pierson, Mortelliti, Barton, Lane, & Lindenmayer, 2016). Another ten year study identified a group of 35 surrogates that successfully predicted changes in the target assemblage of 98 benthic macroinvertebrate species in a temperate brackish system (Bevilacqua et al., 2018). The need for more studies that investigate the effectiveness of surrogates over time is evident.

Another widespread feature of surrogates for species diversity is a reduction of taxonomic resolution (Fontaine, Devillers, Peres-Neto, & Johnson, 2015; Musco, Mikac, Tataranni, Giangrande, & Terlizzi, 2011; Olsgard & Somerfield, 2000). Monitoring at the species level to estimate biodiversity often requires substantial taxonomic expertise (Hirst, 2008; Sebek et al., 2012). Moreover, some species that can be recognized in the lab using morphological features or genetic markers cannot be distinguished in the field during the monitoring surveys often used by specialists to estimate species richness. When individuals are not phenotypically distinguishable between taxonomic groups, studies sometimes use recognizable taxonomic units (RTU’s) that are defined by readily identifiable characteristics in the field (Derraik et al., 2002).

Coral reefs are biodiversity hotspots that are globally threatened due to environmental and anthropogenic factors, including ocean acidification, persistent high temperatures, and overfishing (T P Hughes, 1994; Terry P. Hughes et al., 2017). As such, many reefs are monitored across both spatial and temporal scales. The most commonly measured features of coral reefs are hard coral cover, structural complexity (rugosity), and coral richness. In fact, the temporal decline of these reef features is widely documented because they are so frequently measured (Alvarez-Filip, Carricart-Ganivet, Horta-Puga, & Iglesias-Prieto, 2013; Habibi, Setiasih, & Sartin, 2007; Stokes, Leichter, & Genovese, 2010). These measures make for good surrogates, as they are simple and affordable to measure, yet how effective they are over long temporal scales remains unknown.

Here, I evaluate the effectiveness of hard coral cover, rugosity, and coral richness as surrogates for species diversity of major taxonomic groups on coral reefs over time and space. Specifically, I use percent hard coral cover and rugosity as landscape feature surrogates, and coral richness as a taxonomic surrogate to predict species richness of corals, fishes, and sponges using 27 years of monitoring data from eight sites around Guana Island in the British Virgin Islands (Forrester et al., 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). I hypothesize that coral cover will be an effective surrogate for coral species richness because declines in coral cover have been associated with declines in coral diversity (Walton, Hayes, & Gilliam, 2018). Similarly, I hypothesize that coral cover will be an effective surrogate for predicting sponge species richness because declines in coral cover have been associated with increases in sponge cover (Ruzicka et al., 2013); this inverse relationship is most likely due to competitive interactions between sponges and corals. Coral cover is often positively associated with fish species richness (Jones, McCormick, Srinivasan, & Eagle, 2004; Pratchett, Hoey, Wilson, Messmer, & Graham, 2011), most likely because corals provide fish with food (directly in the case of corallivores and indirectly by providing habitat for prey) and protection from predators. Therefore, I hypothesize that coral cover will be an effective surrogate for fish species richness. I hypothesize that rugosity will be an effective surrogate for fish species richness because greater rugosity should provide a wider variety of structures that may be utilized by a greater diversity of fish species (Darling et al., 2017; Graham et al., 2006; Gratwicke & Speight, 2005; Newman et al., 2015) and fish may respond primarily to the structure of the reef rather than its biological features (Wilson et al., 2009; Wilson, Graham, Pratchett, Jones, & Polunin, 2006). Finally, I hypothesize that rugosity will be an effective surrogate for coral species richness because a greater number of coral species should increase the number of coral morphological types and increase rugosity (Alvarez-Filip, Dulvy, Côteé, Watkinson, & Gill, 2011; Newman et al., 2015).

# 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 1992-2018, except that sponges were not counted in 1992, 1996-1999, 2004, 1993 at Crab Cove, 2014 at Pelican Ghut, or in 2017 at Bigelow Beach and 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. 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.

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.

Rugosity was measured using the height difference method, where a diver records the difference between the top of a meter stick and the top of the reef at multiple points along the transects and uses these points to calculate a rugosity index.

*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. There were a total of 119 species of fishes, 27 recognizable taxonomic units (RTU’s) of hard corals, and 58 RTU’s of sponges (Forrester et al., 2015). 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.

## Data Cleaning

## Because there were 3-12 in a given year at a given site, 3 transects were randomly selected and coral cover, rugosity, coral richness, sponge richness, fish richness, and combined richness were calculated by averaging these values across the 3 randomly selected transects using Program R. Combined richness was only calculated for sites and years for which richness of all three taxonomic groups was available.

## Modeling

I ran linear regression models to predict coral richness (target) with rugosity and percent coral cover (surrogates). I also included models with site and time as additive and interactive effects where site is a categorical predictor of the 8 locations around the island and time is a continuous predictor of the 27 years. I repeated these steps for fish richness, sponge richness, and combined richness (targets) as dependent variables with rugosity, percent coral cover, and coral richness (surrogates) as predictors.

Preliminary investigation of the data suggested a nonlinear relationship between several of the dependent variables and their predictors. As a result, we fit simple logarithmic and power functions to see if nonlinearity might explain some of the variability within the data.

The linear, logarithmic, and power function models were compared using AIC for each of the dependent variables. Models were then evaluated using AIC weights, r-squared values, and log-likelihoods.

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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.

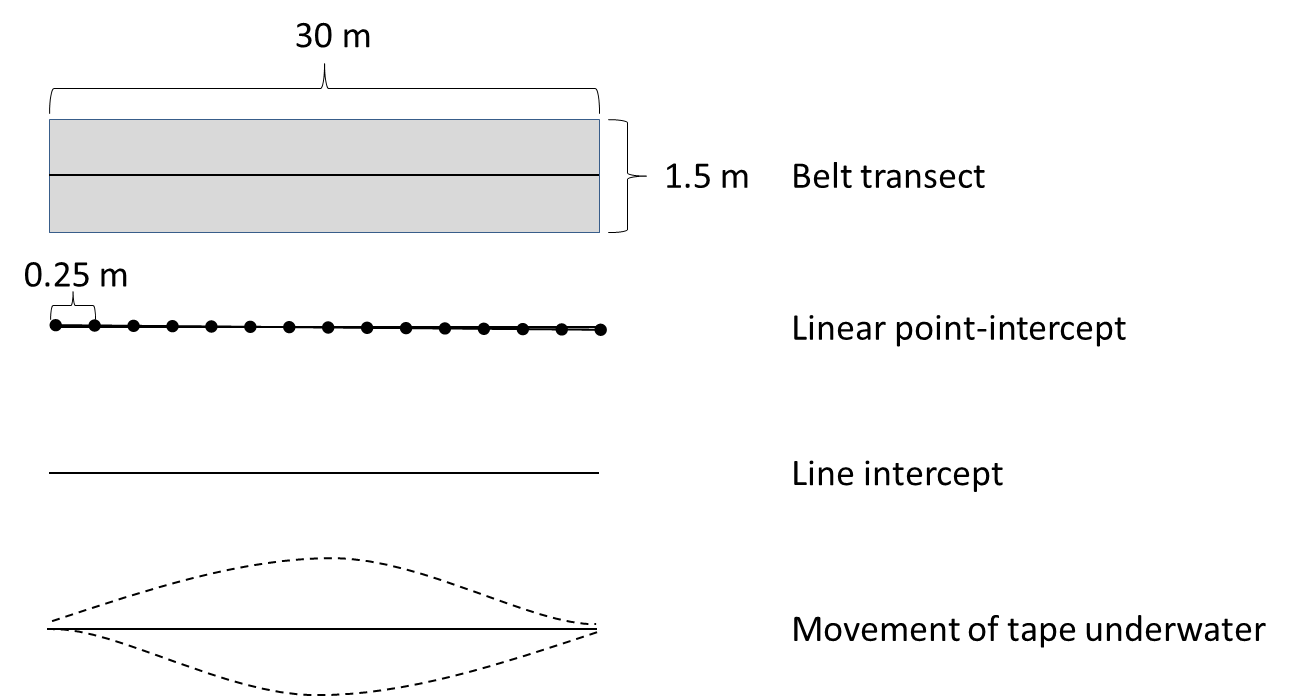


Figure 2. Preliminary depictions of the various transect methods used in this survey.