RESEARCH ARTICLE

Coral Growth Assessment on an Established Artificial Reef in Antigua

Katherine Cummings, Anna Zuke, Bart De Stasio and Jason Krumholz

ABSTRACT

Anthropogenic pressure on coral reef ecosystems has increased the need for effective restoration and rehabilitation as a management tool. However, quantifying the success of restoration projects can be difficult, and adequate monitoring data are scarce. This study compared growth rates over a six-year period of three Caribbean coral species, staghorn coral (*Acropora cervicornis*), elkhorn coral (*Acropora palmata*), and thick finger coral (*Porites porites*), transplanted on an artificial reef off Maiden Island, Antigua, to literature values for the same species growing on naturally formed reefs in the Caribbean region. The average growth rate of staghorn coral was considerably lower than growth rates reported in the literature, while elkhorn and finger corals showed growth rates similar to literature values. The observed inter- and intraspecific differences may be caused by species-specific growth requirements and/or restoration site conditions, factors that should be taken into account when planning future projects involving coral transplant or rescue. This study also determined the analytical precision of a 'low tech' monitoring method using a basic underwater digital camera and the software program ImageJ to measure growth rates of corals. Measurement error between volunteer analysts receiving only minimal training was shown to be very small, ranging from 0.37–1.40% depending on the coral species. This confirms the validity of this basic technique, particularly in cases where data are sparse and resources for monitoring are extremely limited.

Keywords: artificial reef, Caribbean, coral transplant, growth rate, restoration

Coral reefs are experiencing precipitous declines worldwide, largely due to human impacts (Bellwood et al. 2004). Species more susceptible to environmental changes, such as staghorn coral (*Acropora cervicornis)* and elkhorn coral (*Acropora palmata)*, have experienced up to 98% reductions in local population size in the last 30 years (Aronson et al. 2008a,b). Overall, the Caribbean has experienced an 80% loss of coral cover since 1975 (Wilkinson 2004). In the face of such degradation, actively restoring coral reefs is frequently considered, but may not always be the best option (Moberg and Rönnbäck 2003, Krumholz et al. 2010, Forrester et al. 2011). Transplanting corals (especially imperiled corals which would otherwise die) onto restoration sites may preserve genetic variability and/or decrease the time for a restoration site to become viable habitat for many reef dwelling species (Lindahl 1998, Bowden-Kerby 2003) but long-term benefits are not well documented, and this process is time- and labor-intensive. Furthermore, when transplanting corals, it is important to consider environmental conditions and potential stressors at donor and restoration sites in order to maximize likelihood of success (Edwards and Clark 1999, Bowden-Kerby 2003, Krumholz et al. 2010).

This study aims to address how transplanted corals on artificial reefs compare to naturally recruited corals on naturally formed reefs in terms of growth. We provide a comparison of transplanted coral growth rates on artificial reefs to literature growth rates reported for corals recruited on naturally formed reefs. We determined growth rates of elkhorn, staghorn, and thick finger (*Porites porites*) coral colonies originally transplanted during 2004 on the Maiden Island reef restoration in Antigua and subsequently measured in 2010.

A common problem in assessing the success of this type of restoration is the lack of resources available to collect monitoring data. Most projects of this type are conducted using fixed pools of grant or mitigation funding, which often do not allow for longterm monitoring (e.g., Edwards and Gomez 2007, Sheppard et al. 2009). For this reason, while conducting this analysis, we also tested the viability of a basic low cost growth rate monitoring technique using a simple underwater digital camera and the free software program ImageJ to calculate linear extension. We test the reproducibility and inter-analyst comparability of photo surveys conducted by volunteers using this procedure to

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determine the precision of the technique and its use as an alternative to potentially more rigorous, but more time consuming methods.

Methods

Monitoring was conducted on Maiden Island, Antigua, at approximately 17.14° N, 61.76° W. This site was the subject of a large shoreline stabilization project in 2004, wherein 30 different coral species from imperiled colonies were transplanted onto approximately 3,500 concrete Reef Ball™ modules (J.C. Walch, Reef Ball Foundation [RBF], pers. comm.) in depths of 1–4 meters. The restoration utilized imperiled colonies and loose fragments of several species, which were collected from nearby reefs and fragmented into 1–3 cm nubbins. The nubbins were affixed in small hydrostatic cement 'plugs', and after a brief observation period in a nursery, were affixed to the Reef Balls™ using a two-part underwater epoxy to secure the plugs into receptors built into the Reef Balls™. Further details on the transplant method can be found in Barber et al. (2007).

In December 2010, a team of three divers photographed and documented colonies of the target species (elkhorn, staghorn, and thick finger coral) on Reef Balls™ off the southeast coast of Maiden Island. Elkhorn and staghorn were chosen because they are very important habitat-building corals experiencing severe declines in the Caribbean (Aronson and Precht 2001, Forester et al. 2011, Vardi et al. 2012), are expected to decline further under climate change (Albright et al. 2010), and are listed as endangered coral species according to the International Union for Conservation of Nature (IUCN) (Aronson 2008a,b). Finger coral was chosen because of its abundance in the surveyed area. All coral colonies were photographed with a 12MP DC 1200 Sea Life camera with Sea Life underwater housing. A numbered card was placed on the same plane as the coral colony to establish scale and reference frame for the image analysis software and later identification of individual colonies (Figure 1). Camera distance from the colonies was kept at approximately one meter perpendicular to the colony surface. Images that did not include the entire coral colony and of completely dead corals were omitted from the analysis. Of the remaining 61 colonies, 28 were staghorn, two were elkhorn, and 31 were finger coral. Images were analyzed using the ImageJ software package (ImageJ 1.43u, National Institutes of Health, USA).

Linear extension was calculated by measuring each colony from base to longest branch tip along the branch's centerline. Each of the two analysts measured multiple images (typically 2–3) of each colony. The average measurement of each analyst for each colony was determined, and the mean of the average measurement for both analysts was used for growth rate determination. To determine growth rates, we assumed an initial colony length of approximately 1.9 cm for all colonies because individual colonies were not identified during initial transplantation activities. This assumed initial length was based on the average size dictated for restoration activities by RBF (3/4 inch converted to the metric system; Barber et al. 2007). The growth rate was calculated as the difference in linear extension over the six years since planting. In order to compare these data to growth rates from natural reefs in the same region, data were compiled from a variety of primary literature sources (Table 1).

Figure 1. Example monitoring photo showing diver positioning numbered ID card with known width (as a frame of reference for imageJ) parallel to the primary plane of growth of the transplanted coral, in this case, elkhorn coral (*Acropora palmata*).

To determine the precision of using ImageJ software as a coral growth measurement tool, we estimated the measurement error between analysts. We determined measurement error as the difference between average measurements made by each analyst on a given colony, expressed as a percentage of the mean of the average measurements. The percent difference values were not normally distributed, so they were log-transformed prior to statistical analysis. Percent differences of the measurements of one user versus the measurements of the other were compared for staghorn and finger coral using two-tailed t-tests.

The field data were determined to have asymmetrical distributions; therefore, the data are displayed in box plots created using PASW (PASW v2010, IBM, Armonk, New York, USA). Statistical analyses were performed with StatPlus (StatPlus v2009, AnalystSoft, Alexandria, VA, USA) and SPSS (SPSS Ver. 19, IBM, Armonk, New York, USA).

Results

On Maiden Island, staghorn coral averaged 4.9 ± 1.8 cm yr^{-1} of growth (mean ± SD) with individual colony rates ranging widely from 1.67–7.93 cm yr^{-1} (Figure 2), considerably lower than observed literature values (mean: 10.8 ± 6.17 cm yr^{-1}; range: 2.52–26.4 cm yr^{-1}). The observed average was lower than 77% of the compiled Caribbean

Table 1. Summary of Caribbean literature growth rates (cm yr^{-1}) of staghorn coral (*Acropora cervicornis*), elkhorn coral (*Acropora palmata*), and thick finger coral (*Porites porites*). Note: All corals from Bowden-Kerby 2001 were transplanted fragments. The authors used two morphotypes of staghorn coral, one from a back reef environment (BRM) and one from a front reef environment (FRM).

Growing Environment	Growth Rate (cm yr^{-1})	Location	Reference
Staghorn coral (*Acropora cervicornis*)			
Turbid lagoonal reef	2.51 ± 2.2	Jamaica	Crabbe and Carlin 2007
	3.0–4.0	Bahamas	Becker and Mueller 2001
Shallow water	3.4–4.6	Key Largo, FL	Shinn 1966
	4.0	Dry Tortugas	Vaughan 1915
Turbid channel	4.3–4.9	Key Largo, FL	Shinn 1966
	5.1	Florida	Vaughan 1915
Fore reef slope	7.1 ± .65	St. Croix, USVI	Gladfelter et al. 1978
	4.0–11.5	Dry Tortugas/Bahamas	Vaughan 1915
Fragments in sand; BRM	7.1–11.3	Puerto Rico	Bowden-Kerby 2001
	10.0	Key Largo, FL	Shinn 1976
Back reef/lagoon	10.9–11.0	Key Largo, FL	Shinn 1966
Fringing reef, 5–8.5 m	10.95 ± 2.72	Jamaica	Crabbe and Carlin 2007
Fringing reef, 5–8.5 m	4.2–18.6	Jamaica	Crabbe 2009
Fragments in sand; FRM	10.1–12.2	Puerto Rico	Bowden-Kerby 2001
	11.5	Eastern Sambo, FL	Jaap 1974
Fragments above sand; BRM	9.0–15.9	Puerto Rico	Bowden-Kerby 2001
	14.5	Barbados	Bowden-Kerby 2001
	14.4	Barbados	Lewis et al. 1968
Back reef	16.0	Puerto Rico	Bowden-Kerby 2001
Fragments above sand; FRM	14.6–23.2	Puerto Rico	Bowden-Kerby 2001
Reef front	21.8	Puerto Rico	Bowden-Kerby 2001
	26.4	Jamaica	Lewis et al. 1968
Elkhorn coral (*Acropora palmata*)			
	5.2	Colombia	Garcia et al. 1996
Back reef/lagoon	5.5–5.8	St. Croix, USVI	Gladfelter et al. 1978
	2.0–11.0	Bahamas	Becker and Mueller 2001
	6.9	Florida	Lirman 2000
Fringing reef, 5–8.5 m	1.0–13.0	Jamaica	Crabbe 2009
	4.0–10.5	Dry Tortugas/Bahamas	Vaughan 1915
	4.7–10.2	St. Croix, USVI	Gladfelter et al. 1977
	7.6–8.8	Curaçao	Bak 1976
Shallow windward fore reef	4.7–8.7	St. Croix, USVI	Gladfelter et al. 1978
Fore reef slope	6.5–9.9	St. Croix, USVI	Gladfelter et al. 1978
	7.4–9.0	Curaçao	Bak et al. 2009
	8.8	Panama	Guzman et al. 1991
	8.8	Yucatan, Mexico	Padilla and Lara 1996
	8.3–9.7	Curaçao	Bak et al. 2009
	10.0	Florida Keys	Jaap 1974
Thick finger coral (*Porites porites*)			
	0.3	Eastern Caribbean	Vermeij 2006
	0.3–0.4	Eastern Caribbean	Johnson and Perez 2006
	0.3–0.6	Eastern Caribbean	Johnson and Perez 2006
	0.47–0.81		Johnson and Perez 2006
	1.2	Jamaica	Vaughan 1919
	0.6–2.1		Crossland 1981
	1.47	Dry Tortugas/Bahamas	Lozano Cortés (pers. comm.)
	1.7–2.9		Van Moorsel 1988
	3.7		Edmunds 2007

data (Table 1). Finger coral growth rates ranged from 0.21–2.21 cm yr^{-1} with a mean of 0.96 ± 0.6 cm yr^{-1} (Figure 2), very similar to a literature mean of 1.31 ± 1.11 cm yr^{-1}. The observed mean is also very close to the median literature value, exceeding four of nine literature values and falling within the range of a fifth value (Table 1). We were only able to estimate growth rate for two elkhorn colonies during the study, preventing a valid quantitative comparison; qualitatively, this coral was the fastest growing of the measured species. The two colonies were growing at very similar rates (9.6 ± 0.15 cm yr^{-1}; Figure 2) at the upper range of literature values (mean: 7.59 cm yr^{-1}; range: 5.2–10 cm yr^{-1}).

With regard to our test of the variability inherent in this monitoring technique, we found ImageJ easy to learn and that it facilitated rapid and precise measurements. Once volunteers were trained to consistently interpret the agreed upon definition of linear extension, our results showed very little variability in precision. After transformation back to original units, mean measurement errors between users were determined to be 1.24% for staghorn (95% confidence interval: 0.79%–1.92%; n = 23), 0.37% for elkhorn (0.01%–12.13%; n = 2) and 1.40% for finger coral (0.94%–2.10%; n = 27). There was no significant difference between measurement errors obtained for the different coral species by the two users (based on a t-test of measurement errors for staghorn and finger coral; t = 0.633, df = 43, p = 0.53). In terms of raw length, these measurement errors equate to 0.39 cm for staghorn, 0.22 cm for elkhorn, and 0.11 cm for finger coral. In all cases, this is far less than the variability in growth rates exhibited within and between species, and likely less than error introduced by other sources.

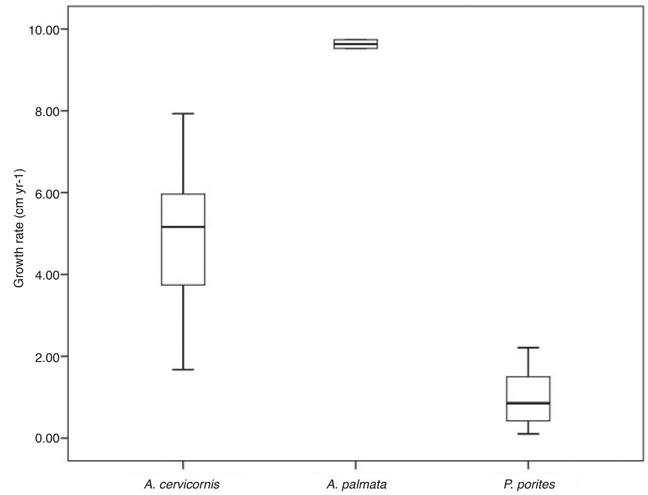

Figure 2. Growth rates (cm yr^{-1}) of species observed in this study. Staghorn coral (*Acropora cervicornis*), n = 23; elkhorn coral (*Acropora palmata*), n = 2; thick finger coral (*Porites porites*), n = 27.

Discussion

Growth Rates

In this study, the growth rates of elkhorn and thick finger coral were equivalent to literature values, while staghorn exhibited considerably lower growth rates than those reported in other studies. Although transplantation stress may be a factor causing lower than normal growth rates, RBF adheres to strict protocols and ethical guidelines when handling corals to reduce stress (Barber et al. 2007), and Forrester et al. (2011) show no adverse impact on growth rate from handling or transplanting stress on similar species in a similar system using similar techniques. These findings lead us to believe that the restoration environment likely played a relatively large role in the observed growth rate variation, though we acknowledge the possibility that other factors, such as temporal variability in growth rate (Bak et al. 2009) may also be of concern.

Environmentally sensitive species such as staghorn coral grow best in mid-depth waters (5–15 meters) on outer reef platforms (Shinn 1966, Acropora Biological Review Team 2005). Our observed growth rate is similar to reported rates for turbid lagoonal reefs or shallow water, less than ideal habitats for this species (e.g. Shinn 1966, Crabbe and Carlin 2007). However, elkhorn coral typically grows well in shallow, high-energy environments (Acropora Biological Review Team 2005) such as Maiden Island, so our result for this species is consistent with expectations given the limited sample size.

Thick finger coral grows well in numerous habitat types ranging from 0–35 meters in depth (Aronson et al. 2008c), which includes areas such as Maiden Island, so it is not surprising that our observed growth rates are very similar to literature values. However, coral colonies growing in shallow environments are more susceptible to certain stresses such as sedimentation and temperature change (Shinn 1966, Acropora Biological Review Team 2005).

Monitoring Techniques

In our study, the combination of low sample size, highly variable growth rates, and unknown size at transplant limit the scope of questions we can seek to answer. Many of the observed corals exhibited signs of stress, which included bleaching, broken fragments, competitive interactions, partially dead regions, or colony death, though the lack of detailed records makes it impossible to estimate transplant survival rate. Furthermore, a wide variety of external stressors including breakage due to boating, snorkeling, SCUBA diving, or fishing traffic, which have all been documented in the area and are well known to have negative impacts on reef health (Hawkins and Roberts 1993, Tratalos and Austin 2001, Lutz 2006), confound our ability to partition stressors on these corals into stress resulting from transplantation versus other uncontrolled stressors. In general, these types of problems are common to scientific attempts to assess restoration success, since funds for longterm monitoring are rarely included in restorations, and restorations are not often planned, implemented, and documented with scientific questions in mind (Edwards and Gomez 2007, Sheppard et al. 2009).

For this reason, we wanted to test the efficacy of the rapid inexpensive volunteer survey technique we chose to use. ImageJ has been used to determine coral growth in previous studies (e.g. Rasher et al. 2012), but we feel this type of analysis is still underutilized. We support the recent publications released by the Coral Reef Targeted Research and Capacity Building for Management Program, who published detailed guidelines on using image analysis software in various aspects of coral reef restorations (but not in linear extension determinations; van Woesik et al. 2009). Although we initially hypothesized that measurement error may be a concern for this type of study, mean colony measurement error between users was very small. As long as the reference metric, in this case the numbered card, is positioned perpendicular to the angle of the photograph and there is a clear understanding between analysts of what is being measured, this technique appears to have low sensitivity to sources of measurement error (even varying card angle 20° from perpendicular causes only about a 7% error). This precludes the need for establishing a correcting factor for this technique as is needed for roving fish counts between sites and analysts (e.g. Sale and Sharp 1983). This result is important because it validates the use of trained volunteers as a viable means of data collection, which could reduce cost and increase frequency of monitoring on this type of project, which is sorely needed.

Although we acknowledge that growth rate may not be the best indicator of reef health in areas with high sedimentation and eutrophication (Edinger et al. 2000), it is a relatively simple and easily collected parameter with abundant literature data with which to compare. Because Maiden Island is not near any sewage outfalls, and is no more or less susceptible to sedimentation than other similar shallow reef environments, it stands to reason that observed differences in growth rate between Maiden Island and naturally formed reefs in the region are most likely due to site-specific characteristics.

When quantifying the success of reef restorations or determining reef health, some authors employ coral growth and accretion (Guzman et al. 1994, Crabbe 2009), while others use diversity or a combination of other indicators (Edinger et al. 2000). Because the corals on Maiden Island were transplanted onto the site and thus do not reflect diversity of surrounding reefs, growth rate is the logical choice for this study. Linear extension was used over areal measurements of growth for several reasons. While areal growth is probably a "better" metric of the size increase and habitat provided by these transplanted corals, calculating areal growth introduces several additional sources for error, chiefly that volunteers conducting the monitoring must accurately and consistently identify and photograph the plane of maximum growth of each colony. In addition, individuals conducting the analysis must uniformly outline areas in the image processing software, which is much more time consuming and error prone than calculating linear extensions. Furthermore, it was much easier to find comparable literature rates for linear extension than areal growth. Thus, in attempting to maximize the efficiency of volunteer data collection, linear extension seems like the best choice.

Policy and Practice Implications

This study raises many questions about the efficacy of coral reef restorations that warrant further research. For threatened or endangered species like staghorn coral, is it more beneficial to transplant colonies only into ideal habitats, or should transplantation into marginal habitats be used, too? Although average growth rate is lower, these colonies may still contribute to natural recruitment through sexual reproduction. Furthermore, transplantation appears to have been beneficial in the specific case of the Maiden Island reef restoration. It is unlikely that natural recruits would have been able to outgrow macroalgae present at the site (Bowden-Kerby 2001, Bowden-Kerby 2003). Particularly in cases where donor colonies are imperiled (e.g. storm breakage or impending dredge or construction activity), this type of activity meets the principal precept to 'first, do no harm'. However, our research builds upon the body of literature which cautions that no two restorations are alike, and that proposed restorations should carefully consider whether water quality and environmental parameters are sufficient for coral health and survival before proceeding with this time consuming and potentially costly option.

In analyzing these data it becomes immediately clear that better data on size, quantity, and location of transplants at the time of transplantation would vastly improve our ability to assess restoration success. While collection of these data may seem tangential to the goals of a restoration project with limited resources (and as such are rarely collected), they are crucial to understanding how to improve efficiency of our efforts in future restorations. We also highlight the need for more effective longterm monitoring of restorations to critically assess the ability of these restorations and mitigation efforts to meet their longterm goals. Too often, monitoring is limited to short-term survival, and does not consider the longer term implications of a restoration. To that end, we also demonstrate the effectiveness of an inexpensive and easily learned monitoring technique, and show that no additional measurement error is imparted by using volunteers who have undergone only minimal training to collect and analyze data.

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Katherine Cummings (corresponding author), University of Florida, School of Natural Resources and the Environment, 103 Black Hall, Gainesville, FL 32611, katy.e.cummings@gmail.com.

Anna Zuke, Coral Cay Conservation, Montserrat Project, Taj Mahal, Old Towne, Montserrat.

Bart De Stasio, Lawrence University, Department of Biology, 711 E. Boldt Way, Appleton, WI 54911.

Jason Krumholz, NOAA, National Marine Fisheries Service, 212 Rodgers Ave, Milford, CT 06460.