

## Research Article

# Long-Lasting Impacts of Beach Renourishment on nearshore Urban Coral Reefs: a Glimpse of Future Impacts of Shoreline Erosion, Climate Change and Sea Level Rise

Edwin A. Hernández-Delgado<sup>1-3\*</sup> and Bernard J. Rosado-Matías<sup>3</sup>

<sup>1</sup>Center for Applied Tropical Ecology and Conservation, University of Puerto Rico, Puerto Rico

<sup>2</sup>Department of Environmental Sciences, University of Puerto Rico, Puerto Rico

<sup>3</sup>Sociedad Ambiente Marino, Puerto Rico

**\*Corresponding author**

Edwin A. Hernández-Delgado, Center for Applied Tropical Ecology and Conservation, University of Puerto Rico, San Juan, 00931-3360, Puerto Rico, Tel: 1-787-764-0000 x188090#; Fax: 1-787-764-2610; Email: edwin.hernandezdelgado@gmail.com

**Submitted:** 01 January 2017

**Accepted:** 19 May 2017

**Published:** 26 May 2017

**Copyright**

© 2017 Hernández-Delgado et al.

**OPEN ACCESS****Keywords**

- Beach renourishment
- Climate change
- Community trajectory
- Sediment burial
- Urban coral reef
- Water quality decline

**Abstract**

Urban shoreline erosion mitigation through beach renourishment has often been dismissed as environmentally insignificant. Given predicted impacts of sea level rise (SLR) and increased shoreline erosion, such activities might become a common practice in the future. But its long-term impacts on adjacent coral reefs have remained poorly documented. Benthic community trajectories were addressed during a period of twelve years across a spatial gradient of sediment burial impacts by beach renourishment on a high-energy urban coral reef at La Marginal Beach, Arecibo, Puerto Rico. Impacts associated to beach renourishment, followed by long-term, slowly-evolving impacts associated to sediment bedload, increased turbidity, increased Arecibo River streamflow, urban polluted runoff discharges, high particulate organic carbon (POC) concentration, and coral mortality following massive coral bleaching in 2005 were addressed through long-term monitoring. There was an initial catastrophic loss in coral species richness, diversity index and percent living coral cover, and a rapid regime shift favoring dominance by macroalgae and other non-reef building taxa. Long-term chronic impacts arrested high impact sites to an early successional stage, and drove moderate and low impact sites to a similar stage of very low species diversity, colony abundance and reef growth. Such chronic changes in community trajectories represent a glimpse into potential future impacts of shoreline erosion, sediment bedload, increasing turbidity and coastal water quality decline associated to SLR. The combination of chronic coral reef decline resulting from beach renourishment, coastal pollution, turbidity, and sediment bedload may have critical long-term ecological implications for urban coral reef resilience, functions and benefits.

**ABBREVIATIONS**

CCA: Crustose Coralline Algae; H': Species Diversity Index; Ha: Hectares; HI: High Impact Site; J': Evenness; LI: Low Impact Site; MI: Moderate Impact Site; RDC: Recently Dead Coral; S: Species Richness; SLR: Sea Level Rise; SPR: Sand + Pavement + Rubble Benthic Category

**INTRODUCTION**

Shoreline erosion is a natural phenomenon magnified by the combined effects of habitat destruction, the construction of artificial structures, and sea level rise (SLR), which threatens a growing number of properties built on coastal shores. SLR is a global socio-economic and environmental concern, particularly for urban coastal societies and small island nations, with significant long-term physical and socio-economic impacts resulting from environmental degradation, shoreline erosion, and coastal flooding [1-7]. Modeling suggests that, regardless

of different potential climate change scenarios, there will be a significant coral reef decline and habitat loss across regional to global scales [8-12]. Coral reefs may often constitute the first line of defense against wave action, SLR, and coastal erosion. Losing coastal reef ecosystems may result in increased risks of coastal erosion. Current and forecasted global warming trends [3,13-16] could also magnify the risks of SLR in the near future, compromising the ecological benefits of coastal ecosystems [17]. SLR may cause multiple impacts to shallow coastal coral reefs. This could result from altered water quality and habitat decline, losing its multiple benefits, including buffering wave action. This could impact infrastructure adjacent to the shoreline and may affect coastal community livelihoods, risking climate change adaptation and mitigation capacity [7].

Coastal erosion threatens an increasing number of sandy beaches and the ecological, economic, social and cultural amenities they provide [18]. This may result in increased coastal vulnerability with variable impacts depending on local social

and economic conditions, built-environment, and physical characteristics [19]. Even sheltered coasts are being threatened by shoreline erosion [20]. Most mitigation measures to buffer impacts of coastal erosion have been overwhelmingly focused on engineering interventions [21], but the widespread construction of structures that alter sea surface circulation (e.g., breakwaters, spits, groins) [22], or fix the position of the shoreline undergoing retreat (e.g., seawalls, revetments) [22,23], has often resulted in increased sediment deposition in some areas and shoreline erosion rates in others. Frequent solutions for shoreline erosion mitigation have often involved sand replacement or renourishment [24,25]. However, these solutions are temporary and usually cost-prohibitive [18]. Annual expenditures and sand volumes showed an increasing trend in the U.S. even since 1990s, particularly on eastern sand bar islands [21,25,26]. Most of these activities can have long-term adverse ecological consequences, which have often been dismissed as environmentally insignificant, therefore, poorly documented in the scientific literature.

Enhanced recreational opportunities and socio-economic benefits have been commonly claimed as key justifications for beach renourishment, but rarely supported by increased numbers of visitors or by an enhanced local economy and livelihoods [27]. In contrast, beach renourishment can deteriorate coastal water quality by reducing water transparency and may not even prevent shoreline erosion [28]. Beach renourishment has also resulted in the physical disruption of macro-invertebrate and fish assemblages across shallow-water hard grounds, causing variable lethal and sub-lethal effects, and permanent benthic community regime shifts as a result of sediment bedload or burial, mechanical impacts, increased turbidity or a combination of these [29-32]. In addition, it has affected shorebird habitats on sandy beaches [33]. However, the long-term consequences of such impacts have still remained poorly studied.

Projected long-term trends of SLR suggest that coastal flooding in the near future could be more significant than previously modeled [34-36], and would require complex and expensive structural solutions to protect coastal infrastructure from erosion [37]. Under such conditions, mitigation of impacts might have magnified adverse ecological [38], environmental [39], and socio-economic consequences for societies [40]. This is particularly concerning for small tropical island nations, where socio-economic and trained human resources are often largely limited [7]. Solutions to buffer shoreline erosion may often require implementing very expensive dredge-and-fill programs to protect coastal development from shoreline erosion at the expense of adverse ecological impacts to adjacent sandy beaches [41,42], and to shallow coastal benthic ecosystems [30]. Nonetheless, mitigation has not always been successful and adverse impacts across adjacent benthic habitats are a concern which has remained poorly documented, particularly across nearshore high-energy habitats.

A regime shift can often be explained as a nonlinear system crossing a critical threshold [43], and could result from a combination of acute, chronic, press or pulse impacts. Such impacts could become more pronounced in shallow urban coral reefs under declining environmental conditions associated to periodic dredging and fill activities [44], due to long-term alterations of

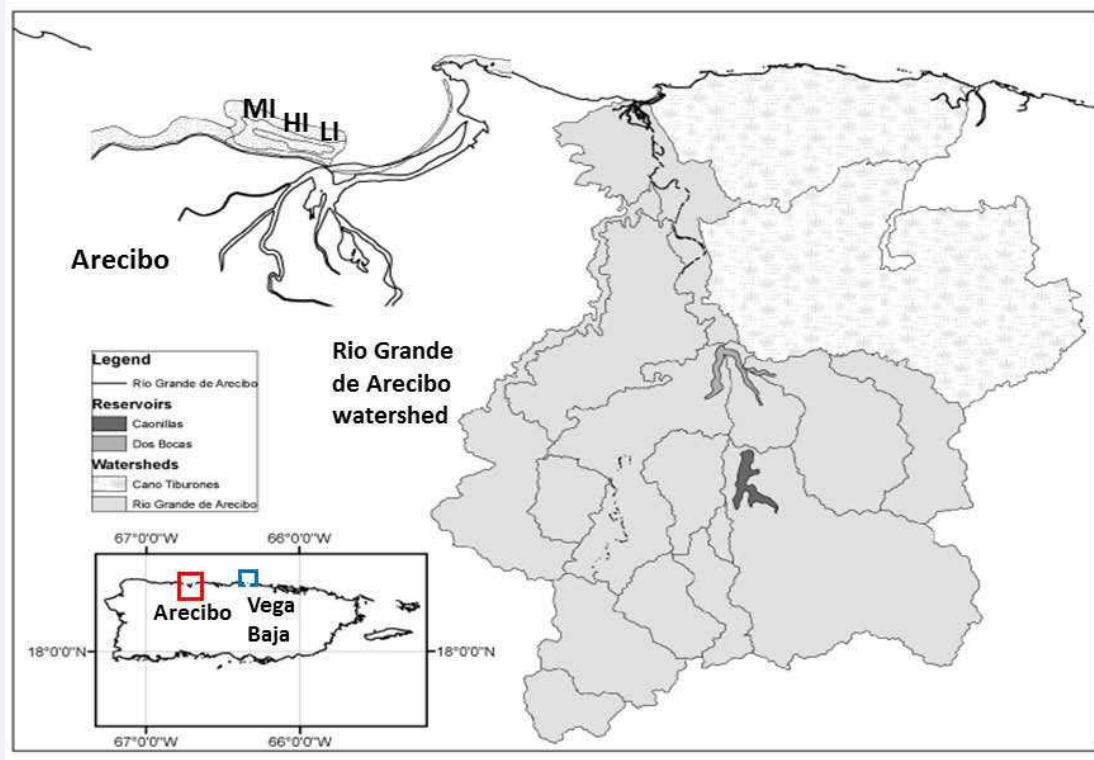
adjacent land use patterns [45-48], and due to long-term impacts of climate change and SLR. Dredging and sedimentation impacts can also be catastrophic for coral reef communities [49-53]. It can be argued that long-term impacts associated to beach renourishment on adjacent urban coral reefs may resemble the potential long-term impacts of shoreline erosion and sediment bedload associated to SLR and climate change. SLR impacts may include major shoreline erosion and a significant increase in chronic sediment spillover and bedload effects across shallow coral reefs adjacent to shore [54]. In this context, addressing the long-term trajectories of adjacent benthic communities following beach renourishment impacts may provide a glimpse of potential future impacts across adjacent urban coral reefs resulting from shoreline erosion, sediment bedload, and declining coastal water quality associated to SLR.

Monitoring of beach renourishment long-term ecological impacts has been frequently flawed by having poor sampling designs, failing to address natural spatial and temporal variation of ecological change, lacking proper controls, and by failing to support conclusions by statistical analyses [32]. Further, there is a major lack of available long-term assessments of ecological impacts across impacted benthic communities to determine effects on benthic community trajectories and address their natural recovery ability. This study highlights the long-term (12 years) benthic community trajectory of a high-energy urban coral reef impacted by a catastrophic beach renourishment activity at Arecibo Harbor, Puerto Rico, which strongly resembles the potential long-term predicted impacts from SLR across coastal urban coral reef ecosystems. This study investigated the immediate effects of reef burial by dredge spoil on a high-energy urban reef habitat, and addressed spatio-temporal variation in benthic community trajectory through a period of 12 years following sediment burial. Incidentally, this study also resulted in a long-term "natural" experiment to assess the impacts on coral reefs of heavy sediment bedload and water quality decline similar to what could be expected as a result of SLR and its concomitant increase in shoreline erosion and urban coastal flooding. The Arecibo Harbor case study constitutes a model to address the long-lasting impacts of beach renourishment and sediment bedload on near shore urban coral reefs. At the same time, it may provide a glimpse of potential long-term impacts of SLR and climate change in a coastal urban scenario.

## MATERIALS AND METHODS

### Study sites

This study was carried out at La Marginal reef, Arecibo, in the high-energy northern coast of Puerto Rico (18°28'32.29"N, 66°43'23.62"W), with a control site at Vega Baja (18°29'19.60"N, 66°25'19.70"W), 33 km east of the impact site (Figure 1). The eastern side of Arecibo Harbor has been protected from prevailing surge and northerly swells by a 366 m-long artificial breakwater built in 1944. The resulting altered local surface current patterns produce substantial accumulation of sediments within the harbor entrance that require periodic maintenance dredging. During June 2000, a total of 153,000 m<sup>3</sup> of shoal material were dredged from the harbor and used by the U.S. Army Corps of Engineers for a beach renourishment project at the nearby La Marginal



**Figure 1** Study site at La Marginal reef, Arecibo, Puerto Rico (red polygon): HI= high impact site, MI= moderate impact site, LI= low impact site. Control site (CT) is located at Vega Baja (blue polygon).

Beach. Dredged sediments were hydraulically pumped out onto a colonized high-energy emergent eolianite reef platform with the objective of constructing an artificial sandy beach to increase local recreational opportunities and socio-economic benefits. But spoil material shifted away with the strong littoral drift to the sublittoral zone and completely buried or significantly smothered out 5 ha of nearshore reef habitats, causing a widespread mortality of benthic and demersal fauna. Most of the spoil material (emergent and submerged) was washed out by the strong surge and littoral drift within two months, clearing most of the reef substrate, but creating a massive sandblasting effect across approximately 25 ha downstream.

La Marginal reef is characterized by a shallow and continuous high-energy emergent eolianite reef platform ranging from 0.2 to about 6 m in depth. However, there are some areas that have developed a system of colonized pavement and channels that extend down to about 5 to 8 m. These zones supported more coral growth and higher habitat heterogeneity than shallower low-relief zones. Data were collected at the 1.7 to 2.5 m depth zone, which was the most directly impacted by the initial burial and spillover of dredged spoil material. Data from the control site were obtained from a similar habitat and depth zone that has never been impacted by beach renourishment activities.

### Environmental variables

Environmental variables included the Río Grande de Arecibo mean monthly streamflow across two gauge stations at the Arecibo Observatory and at the Dos Bocas Reservoir. Data were

obtained for the period of 2000 to 2012 from US Geological Survey Caribbean Field Office. Water transparency was measured in triplicates using a 51 cm-diameter Secchi disk during each visit to study sites. Remotely-sensed data of chlorophyll-*a* concentration, particulate organic carbon (POC) and particulate inorganic carbon (PIC) were obtained from satellite data for the period of 2000 to 2012, from a polygon located between 18°30.0'N, 66°44.0'W, 18°30.0'N, 66°42.0'W, 18°28.5'N, 66°44.0'W, and 18°28.5'N, 66°42.0'W (=5.14 km<sup>2</sup>), and retrieved from NOAA's ERDDAP data set web link <http://coastwatch.pfeg.noaa.gov/erddap/index.html>.

### Benthic community sampling design

Two main types of variables were tested in this study: temporal and spatial effects. Data were collected in 2000 (two months after sediment burial), 2003, 2005, 2007, 2009, and 2012 across areas under three different levels of sediment burial impacts. These included: high impact (HI) - sediment disposal area, moderate impact (MI) - 200 m downstream of the sediment disposal western boundary, and low impact (LI) - 200 m upstream of the sediment disposal northeastern boundary. Data were also collected from a control site (CT) at Vega Baja, 33 km east of the impacted area. No sediments were directly deposited within the LI or MI, but there was high turbidity across all impacted sites. Sediment "sandblasting" effects due to strong currents, wave action and bedload eventually impacted the MI and LI sites during spillover effects.

Benthic community data were collected using 10 m-long

point-count line transects at 0.5 m point intervals (n=20 points per transect). Strong currents and prevailing 1.5 to 2.5 m-high surge prevented the use of longer transects. A total of 6 to 8 replicate transects parallel to the shoreline were haphazardly sampled across geo-referenced reef segments per site during each time interval. Community data included coral species richness, colony abundance, species diversity index (H'*c*) [55], species evenness (J'*c*) [56], percent living coral cover, and percent cover of macroalgae, algal turf, crustose coralline algae (CCA), golden algae (*Chrysozystis* spp.), zoanthids, sponges, acronym (RDC), cyanobacteria, and sand, pavement and rubble (SPR). The percent cover or bare substrate and of bleached and size-reduced colonies were also documented. All data were collected between 0600 and 0800 hr, when wind was low and swells were still marginally safe for field work.

### Statistical analyses

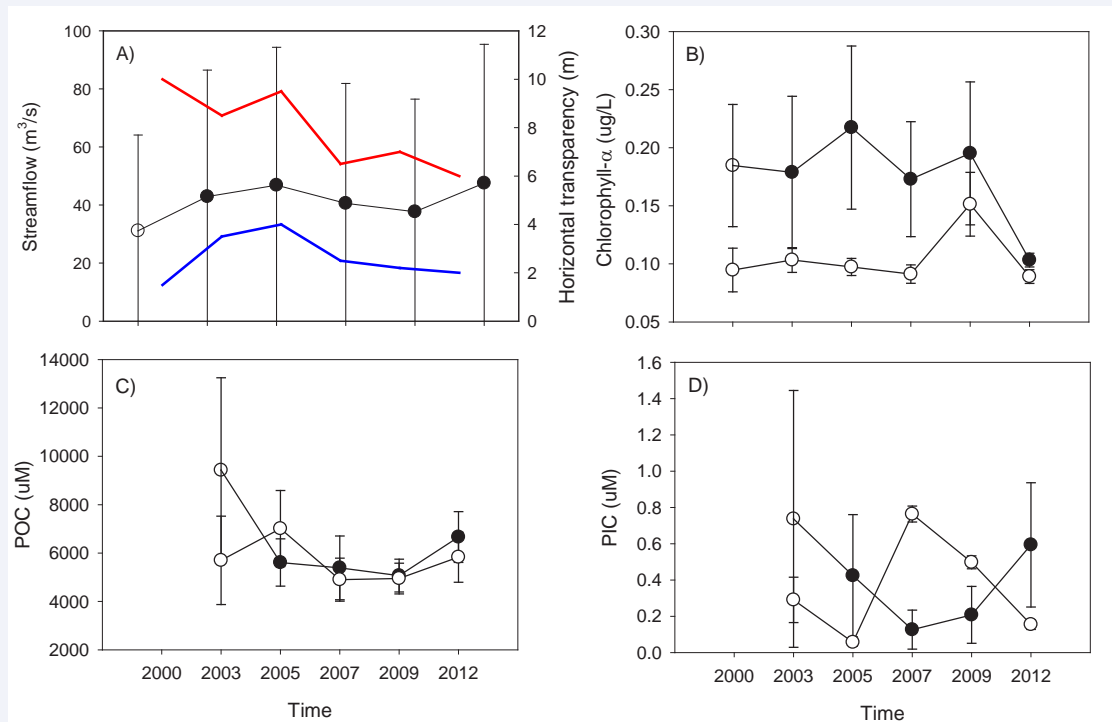
The null hypothesis of no significant difference in spatio-temporal patterns of coral reef benthic community structure among surveyed sites was tested using PRIMER v6.1.16 + PERMANOVA v1.0.6 statistical package (Plymouth Marine Lab, UK) [57], following a two-way permutational analysis of variance (PERMANOVA) [58]. Impact (HI, MI, LI, CT) and time (2000, 2003, 2005, 2007, 2009, 2012) were used as main variables. Also, impact x time interaction was tested. A similar approach was used to test for spatio-temporal patterns in species richness (S), H'*c*, J'*c*, and percent cover of selected benthic community components. Principal coordinate Ordination (PCO) was used to identify spatio-temporal patterns of coral reef benthic community

structure clustering, temporal trajectories, and indicator taxa of such clustering patterns for each impacted site. A total of 10,000 permutations were used for multivariate tests. A combination of linear and non-linear regression models was also fit using Sigma Plot v11.0 (Systat Software, Inc.) to test the relationship of selected coral reef parameters and Rio Grande de Arecibo streamflow. Regression analyses were also performed between selected coral reef benthic parameters and mean horizontal water transparency across study sites. Multivariate routine BEST was used to test for the best environmental variable combination to explain observed temporal variability in benthic community structure through the BVSTEP method [57].

## RESULTS AND DISCUSSION

### Environmental variables

Mean horizontal water transparency across the impacted site during 2000 was the lowest (1.5 m), with the highest value achieved during 2005 (4.0 m) (Figure 2a). Transparency declined during subsequent surveys to 2.0 m during 2012. Mean horizontal water transparency across the control site at Vega Baja during 2000 was 10 m, with a value during 2005 of 9.5 m. Transparency declined during subsequent surveys to 6.0 m during 2012. Such declines corresponded to increasing river local and urban discharge influences associated to increasing rainfall pulse events as reflected by streamflow. Mean annual streamflow at Río Grande de Arecibo ranged from 31.12 m<sup>3</sup>/s in 2000 to 47.56 m<sup>3</sup>/s in 2012 (Figure 2a). Other high mean annual peaks were documented in 2003 (42.91 m<sup>3</sup>/s), and in 2005 (46.83



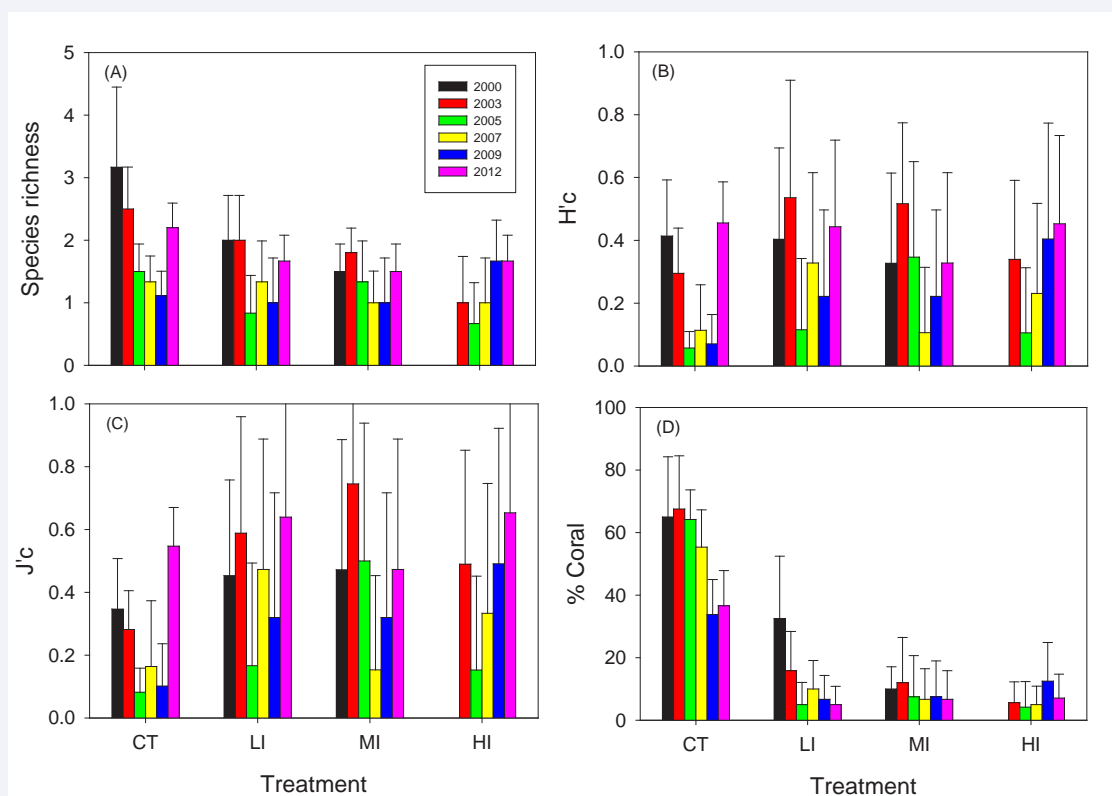
**Figure 2** Environmental parameters across the study sites. (A) Río Grande de Arecibo streamflow (m<sup>3</sup>/s) and Secchi disk horizontal water transparency (m): La Marginal – blue line, Vega Baja – red line; (B) Chlorophyll-*a* concentration (µg/L); (C) Particulate organic carbon (POC) concentration (µM); and (D) Particulate inorganic carbon (PIC) concentration (µM). Mean ± 95% confidence intervals. La Marginal – black dots, Vega Baja – white dots. No remotely sensed data existed for these parameters during year 2000.

m<sup>3</sup>/s). Mean annual chlorophyll-*a* concentration at the impact site ranged from 0.1033µg/L in 2012 to 0.2174 µg/L in 2005 (Figure 2b). Peak chlorophyll-*a* concentration corresponded to the second highest streamflow peak. Mean annual chlorophyll-*a* concentration at the control site ranged from 0.0891µg/L in 2012 to 0.1513µg/L in 2009 (Figure 2b). However, such chlorophyll-*a* concentrations were lower than those documented in previous studies in Puerto Rico [59], and were within the recommended maximum concentration of 0.3µg/L for coral reefs [60].

Mean annual particulate organic carbon (POC) concentration at the impact site ranged from 5,068.2µM in 2009 to 9,425.6µM in 2003 (Figure 2c). Highest POC concentration at the impact site corresponded to a year that showed two extreme rainfall events. Mean annual particulate organic carbon (POC) concentration at the control site ranged from 4,900.9µM in 2005 to 7,012.3 µM in 2007 (Figure 2c). Documented POC concentrations at the control site were lower than at the impact site. Nonetheless, POC concentrations at both sites were significantly higher in comparison to previous reports in the literature [61-63], and can be considered detrimental for coral reefs. Mean annual particulate inorganic carbon (PIC) concentration at the impact site ranged from 1.26 x 10<sup>-7</sup>µM in 2007 to 7.37 x 10<sup>-7</sup>µM in 2012 (Figure 2d). Highest PIC concentration at the impact site coincided with increasing river streamflow. Mean annual particulate inorganic carbon (PIC) concentration at the control site ranged from 6.42 x 10<sup>-7</sup>µM in 2007 to 2.26 x 10<sup>-6</sup> µM in 2003. PIC concentrations in coastal waters can often remain low [64].

### Benthic community variables

Coral species richness showed a significant decline ( $p < 0.0242$ ) through time and among impact levels (Table 1). Post-hoc analyses are all summarized in Table S1 in the Supplementary Material section. HI sites had no living corals in 2000 and showed only 1.67 species per transect in 2012 (Figure 3a). MI sites had 1.5 species in 2000, increased to 1.8 in 2003, but then remained at 1.5 species in 2012. LI sites had a mean of 2.0 species per transect in 2000, declining to 1.67 in 2012. CT sites supported a mean of 3.2 species in 2000, declining to 2.2 in 2012. H'*c* showed significant variation through time ( $p = 0.0032$ ), with two general patterns (Figure 3b). First, there was a significant increase in H'*c* across impact sites due to partial slow recovery of impacted coral colonies, followed by a net decline after the 2005 massive coral bleaching event and chronic water quality degradation. Secondly, H'*c* returned to original values by 2012 after some partial recovery of a few ephemeral coral species. No coral growth was observed at HI in 2000. Mean H'*c* in 2012 was 0.4527. Mean H'*c* at MI was 0.3269 in 2000 and 0.3277 in 2012. Mean H'*c* at LI was 0.4033 in 2000 and 0.4432 in 2012. Mean H'*c* at CT was 0.4134 in 2000 and 0.4554 in 2012. A nearly similar temporal pattern was documented in J'*c* ( $p = 0.0042$ ), but showing a significant increase through time (Figure 3c). No coral growth was observed at HI in 2000. Mean J'*c* in 2012 was 0.6530. Mean J'*c* at MI was 0.4716 in 2000 and 0.4728 in 2012. Mean J'*c* at LI was 0.4529 in 2000 and 0.6394 in 2012. Mean J'*c* at CT was 0.3464 in 2000 and 0.5468 in 2012.



**Figure 3** Benthic community parameters: (A) Coral species richness, (B) Coral species diversity index (H'*n*), (C) Coral species evenness (J'*c*), and (D) Percent live coral cover. Mean ± 95% confidence intervals. For site acronyms refer to Figure 1.

Percent living coral cover showed no temporal difference, but a highly significant difference ( $p < 0.0001$ ) between CT and impacted sites (Table 1 and Figure 3d). No corals were present at HI sites in 2000. Coral cover increased to 5.6% by 2003, and 12.5% by 2009, but declined to 7.1% in 2012, or a 43% loss in just three years. Coral cover at MI was 10% in 2000, and 12% in 2003, but declined to 7.5% in 2005 following the massive bleaching event and further to 6.7% in 2012, representing a 33% loss in 12 years. Coral cover at LI was 32.5% in 2000, dropping down to 15.8% in 2003 and to 5% in 2005, remaining unchanged by 2012. Living coral cover declined at LI by a factor of 51% by 2003, and by 85% after the 2005 massive bleaching event. Mean coral cover at CT was 65% in 2000, increasing to 68% in 2003. However, it showed a consistent decline to 55% in 2007, to 34% in 2009, and remaining at 37% in 2012, which represent a 44% decline within 12 years.

Spatio-temporal changes were documented in the four most important reef-building species across study sites. Elkhorn coral (*Acropora palmata*) was completely absent from impact sites, but highly abundant at CT sites (Figure 4a), with a percent relative cover of 58% during 2000, reaching up to 63% in 2005, and then declining to 31% during 2012, which represent a 51% loss. Brain coral (*Pseudodiploria strigosa*) showed a highly significant spatio-temporal fluctuation ( $p < 0.0001$ ) (Table 1). This species was completely absent from HI sites until 2012 (0.4%) (Figure 4b). MI sites had a mean 7.5% cover in 2000, but dropped to 1.3% in 2012, an 83% decline. LI sites had a mean 28% cover that dropped to 11% in 2003, and to 1.7% in 2005 after the massive bleaching and mass coral mortality event. It further declined to 0.8% in 2012. By 2003, *P. strigosa* loss reached 61%, 94% by 2005, and 97% by 2012. Mustard hill coral (*Porites astreoides*) showed a highly significant difference among sites ( $p = 0.0193$ ), but no significant temporal fluctuation (Figure 3c). HI sites had no colonies during 2000, but percent relative cover reached 6.7% in 2009, and then declined to 3.3% in 2012, a factor of 50%. MI showed a 0.8% cover in 2000, increasing to 3.3% in 2009, a 3-fold increase, but declined in 2012 to 2.5%. LI showed a 2.5% cover in

2000, increasing to 7.5% in 2007, but declined in 2012 to 2.1%, a 72% loss within only 5 years. This species remained virtually unchanged between 2000 and 2012 at CT with 2% relative cover. Fire coral (*Millepora complanata*) showed a non-significant spatio-temporal increase in percent relative cover (Figure 4d).

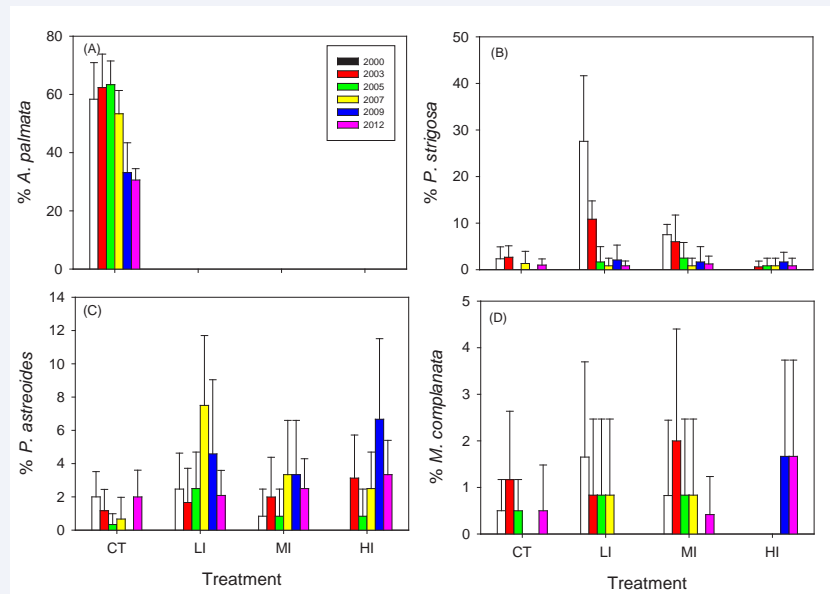
Macroalgal cover showed a significant increase ( $p < 0.0001$ ) through time, particularly across impacted sites (Table 1), shifting from 3% in 2000 to 74% in 2012 at HI, from 18 to 68% at MI, and from 10 to 71% at LI (Figure 5a). Macroalgal cover increased from 10 to only 14% across CT sites. Algal turf showed a significant decline through time across impacted sites ( $p < 0.0001$ ). Dynamics were also significantly different among sites ( $p = 0.0014$ ). Algal turf cover declined from 32% in 2000 to 10% in 2012 at HI, from 42 to 15% at MI, and from 32 to 13% at LI (Figure 5b). Algal turf just shifted from 15.7 to 16.3% across CT sites. CCA were significantly higher ( $p < 0.0001$ ) across CT sites than across impact sites, but showed no significant temporal trends (Figure 5c). CCA were almost absent across HI sites through 2009, and had 0.8% cover in 2012. No CCA were present at MI until 2009 (1.7%). CCA averaged 0.3% in 2000 at LI and 0.8% in 2012, while CT sites showed 2.3% in 2000 and 4.1% in 2012. Observed changes in algal communities suggest a significant regime shift through time following the sediment burial disturbance.

Golden noodle algae, *Chrysoecystis* spp. showed a significant increase in percent cover through time ( $p < 0.0001$ ), and significantly higher cover ( $p = 0.0017$ ) at CT sites (Table 1). *Chrysoecystis* spp. was absent at HI until 2012 (2.5%) (Figure 5d). It reached 3.3% at MI, 4.2% in LI, and 17% at CT during 2012. Cyanobacteria showed a non-significant spatio-temporal variation in percent relative cover, with mean values never exceeding 4.7% (Figure 5e). Percent cover of recently dead corals (RDC) showed a highly significant ( $p < 0.0001$ ) temporal variation, but no spatial variation. RDC reached 18% during 2000 at HI. There was also 1.7% of RDC during 2005 after the massive bleaching event. No other RDC were documented at HI. RDC

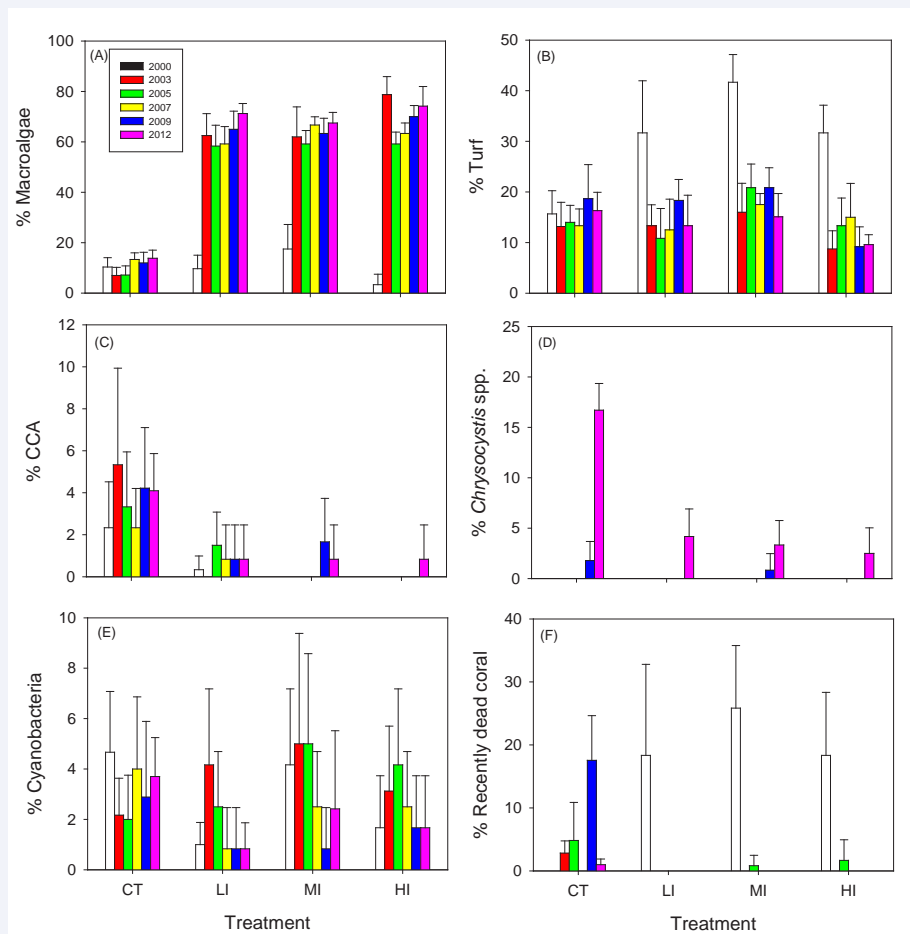
**Table 1:** Summary of the two-way PERMANOVA of selected benthic community parameters.

Variable	Time d.f.= 5,146*		Impact level d.f.= 3,148		Time x Impact d.f.= 23,128	
	Pseudo-F	p	Pseudo-F	p	Pseudo-F	p
Species richness (S)	2.62	0.0242	7.79	0.0242	4.17	<0.0001
Species diversity (H'c)	3.72	0.0032	0.59	**	1.69	0.0348
Evenness (J'c)	3.67	0.0042	0.52	**	1.52	**
% Coral	0.99	**	24.77	<0.0001	7.31	<0.0001
% <i>A. palmata</i>	0.05	**	796.4	<0.0001	164.1	<0.0001
% <i>P. strigosa</i>	7.67	<0.0001	7.87	<0.0001	7.22	<0.0001
% <i>P. astreoides</i>	1.60	**	3.36	0.0193	1.88	0.0136
% <i>M. complanata</i>	0.50	**	0.24	**	0.97	**
% Sponges	1.78	**	6.95	<0.0001	1.62	**
% Zoanthids	1.38	**	7.19	<0.0001	1.96	0.0119
% Macroalgae	11.58	<0.0001	38.20	<0.0001	50.34	<0.0001
% Algal turf	10.55	<0.0001	5.75	0.0014	5.47	<0.0001
% CCA	1.30	**	27.86	<0.0001	4.01	<0.0001
% <i>Chrysoecystis</i> spp.	43.61	<0.0001	5.46	0.0017	22.87	<0.0001
% Cyanobacteria	1.37	**	2.10	**	1.05	**
% RDC	14.75	<0.0001	2.14	**	18.90	<0.0001
% SPR	3.58	0.0035	11.78	<0.0001	8.88	<0.0001

\*d.f. : degrees of freedom; \*\*= not significant ( $p > 0.05$ )



**Figure 4** Percent relative cover of coral species: (A) Elkhorn coral, *Acropora palmata*, (B) Brain coral, *Pseudodiploria strigosa*, (C) Mustard hill coral, *Porites astreoides*, and (D) The word fire coral, *Millepora complanata*. Mean  $\pm$  95% confidence intervals. For site acronyms refer to Figure 1.



**Figure 5** Benthic community parameters: (A) Percent macroalgal cover, (B) Percent filamentous algal turf cover, (C) Percent crustose coralline algal cover (CCA), (D) Percent Golden noodle algae, *Chrysocestis* spp., cover, (E) Percent cyanobacterial cover, and (F) Recently dead coral cover (RDC). Mean  $\pm$  95% confidence intervals. For site acronyms refer to Figure 1.

reached 26% during 2000 at MI. There was also 0.8% RDC during 2005, and no other RDC after that. RDC reached 18% during 2000 at LI. No other RDC were documented at LI. Sporadic pulses of RDC were documented across *Acropora palmata*-dominated bottoms at CT sites, with 3% in 2003, 5% in 2005, 18% in 2009, and 1% in 2012. The 2009 incident across CT sites was associated to the illegal spilling of raw sewage along Vega Baja beach [65].

Percent zoanthid (Figure 6a) and sponge cover (Figure 6b) showed a significantly higher cover ( $p < 0.0001$ ) across impacted sites, but no significant temporal trends (Table 1). Percent cover of SPR showed significant decline through time ( $p = 0.0035$ ), as well as higher percent cover across impact sites (Figure 6c).

### Temporal trajectories of benthic community structure

There was a highly significant temporal variation in benthic community structure ( $p < 0.0001$ ), and across all combinations of pairs of years (Table 2). There was also a highly significant variation among impact levels ( $p < 0.0001$ ), including all combinations of impacts, and between impact sites and CT. Time x impact level interaction was also highly significant ( $p < 0.0001$ ). PCO analysis showed that benthic community trajectory at HI was initially dominated in 2000 by RDC, SPR and algal turf, right after reef bottom became exposed by strong surge and spoil material drifted away down current (Figure 7). Temporal changes in benthic community structure were largely explained in 2003 by the recovery of ephemeral *P. astreoides* and by a rapid increase in macroalgal growth on open reef substrates. During the sea surface warming episode in 2005 and 2007 benthic community within HI was largely explained by increasing ephemeral coral *Siderastrea radians*, zoanthids and cyanobacteria. By 2009, *P. astreoides* largely explained temporal variation, followed in 2012 by *M. complanata*, and in a lesser extent by the Encrusting gorgonian, *Erythropodium caribbaeorum* (vector not shown), a species that tend to dominate under significant disturbance. This PCO model explained 95.7% of the observed temporal trajectory.

Benthic community trajectory at MI showed that algal turf,

recently dead corals (RDC) and in a lesser degree *M. complanata* explained temporal variation in 2000 (Figure 8). Macroalgae, sand, pavement and rubble (SPR), and *Siderastrea radians* explained most of the variation observed in 2003. *Erythropodium caribbaeorum* explained most of the variation following mass bleaching and partial coral colony mortality in 2005, followed by macroalgae in 2007, then again in 2009 and 2012. This PCO model explained 91.9% of the observed temporal trajectory. Variation in benthic community trajectory at LI was mostly explained by RDC, *P. strigosa*, and *Agaricia agaricites* in 2000, followed by macroalgae in 2003 after benthic community degradation by sediment spillover and bedload effects (Figure 9). Sponges and SPR explained most of the observed variation in 2005 and 2007, followed again by macroalgae in 2012. This PCO model explained 83.9% of the observed temporal trajectory.

Benthic community trajectory at CT was initially explained in 2000 by *P. strigosa* and *P. astreoides*, followed in 2003, 2005, and 2007 by the decline observed in *A. palmata*, in 2009 by RDC, and in 2012 by increasing zoanthids and Golden noodle algae, *Chrysochysis* spp. (Figure 10). This PCO model explained 84.6% of the observed temporal trajectory.

### Impacts of environmental variables on benthic community trajectories

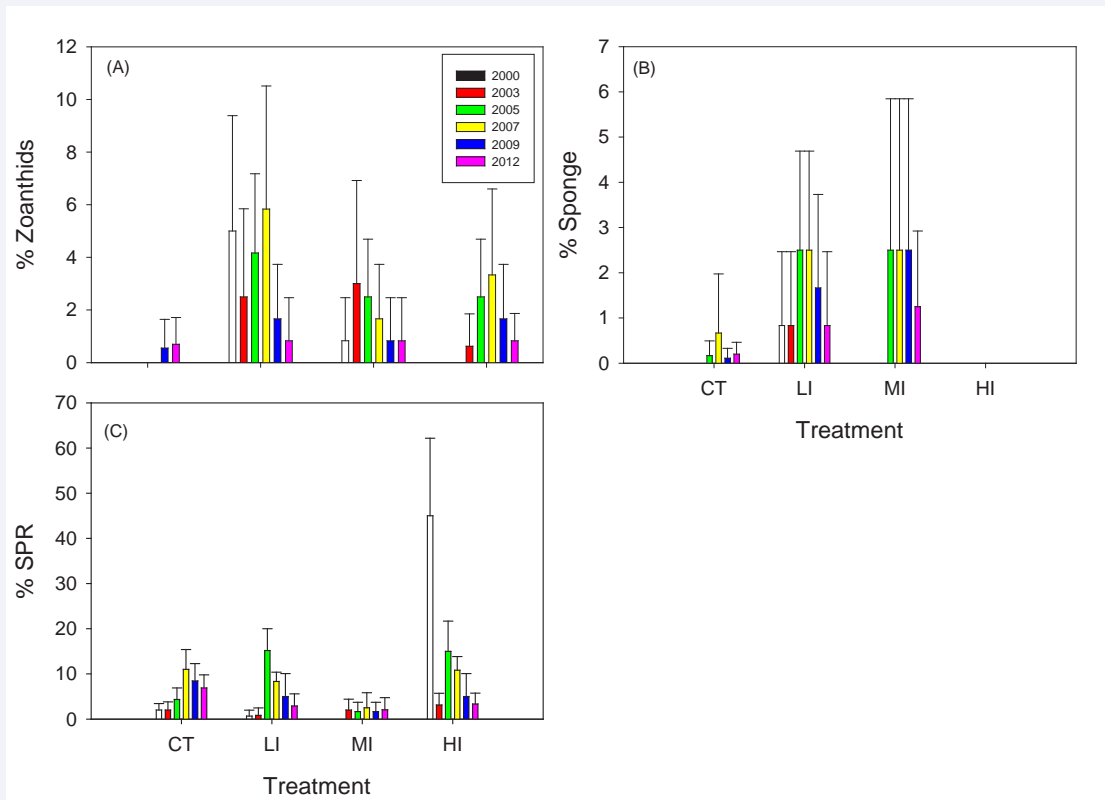
Cumulative rainfall across the Río Grande de Arecibo watershed resulted in a temporal increase in streamflow at the Arecibo Observatory sampling station from 2000 to 2012 through this study. However, annual variability was also becoming higher as a result from recurrent extreme rainfall events. In year 2000, maximum streamflow across the Río Grande de Arecibo watershed was  $231 \text{ m}^3 \text{ s}^{-1}$  in 2000, and  $338 \text{ m}^3 \text{ s}^{-1}$  in 2012. A similar trend was observed at the Dos Bocas Reservoir sampling station, with  $225 \text{ m}^3 \text{ s}^{-1}$  during 2000 and  $291 \text{ m}^3/\text{s}$  during 2012. Increasing mean and peak rainfall coincided with increasing sediment-laden runoff impacts to coastal waters across the impact site, although a correlation between cumulative streamflow and coastal water turbidity was short of significance ( $r^2 = 0.5938$ ,  $p = 0.0729$ ), probably as a result from high temporal variability in streamflow. However, increasing annual cumulative streamflow showed a significant negative regression with declining percent coral cover ( $r^2 = 0.7255$ ,  $p = 0.0313$ ), and declining percent relative cover of Brain coral, *Pseudodiploria strigosa* ( $r^2 = 0.7018$ ,  $p = 0.0374$ ), across LI sites, but showed no impacts across MI and HI sites. Declining water transparency resulted in a significant decline in percent *P. strigosa* cover ( $r^2 = 0.7808$ ,  $p = 0.0178$ ) across LI sites, but had no effect on other parameters across any of the impact sites. High variability in streamflow only reflects the increasing recurrence of extreme rainfall events. Such pulse events can be significant drivers of impacts on adjacent reefs. BVSTEP analysis showed that the combination of river streamflow, horizontal water transparency, POC, and PIC had the strongest correlation with temporal variation in benthic community structure at the impact site ( $\text{Rho} = 0.743$ ,  $p = 0.0110$ ). Horizontal water transparency, POC, and PIC had also the strongest correlation with temporal variation in benthic community structure at the control site ( $\text{Rho} = 0.587$ ,  $p = 0.0442$ ). This suggests that long-term degradation in coastal water quality at both, impact and control sites became a significant factor influencing benthic coral reef community trajectory. However, long-term river influences at the

**Table 2:** Summary of the two-way PERMANOVA of benthic community structure.

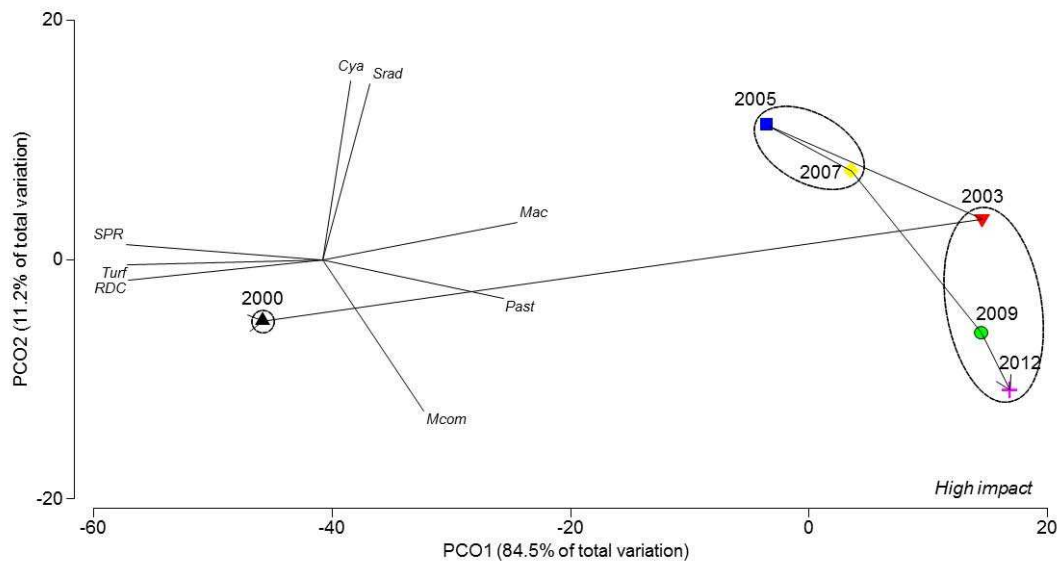
Variable (d.f.)*	Pseudo-F	P
Time (5,146)	6.44	<0.0001
Pairwise	<i>t</i>	<i>p</i>
2000 – 2003	3.39	<0.0001
2000 – 2005	3.12	<0.0001
2000 – 2007	3.91	<0.0001
2000 – 2009	2.95	<0.0001
2000 – 2012	4.23	<0.0001
Impact level (3,148)	33.43	<0.0001
Pairwise	<i>t</i>	<i>p</i>
CT – LI	8.05	<0.0001
CT – MI	8.42	<0.0001
CT – HI	7.97	<0.0001
LI – MI	1.57	0.0460
LI – HI	2.11	0.0031
MI – HI	2.55	0.0002
Time x Impact (23,128)	14.16	<0.0001

\*d.f. degrees of freedom

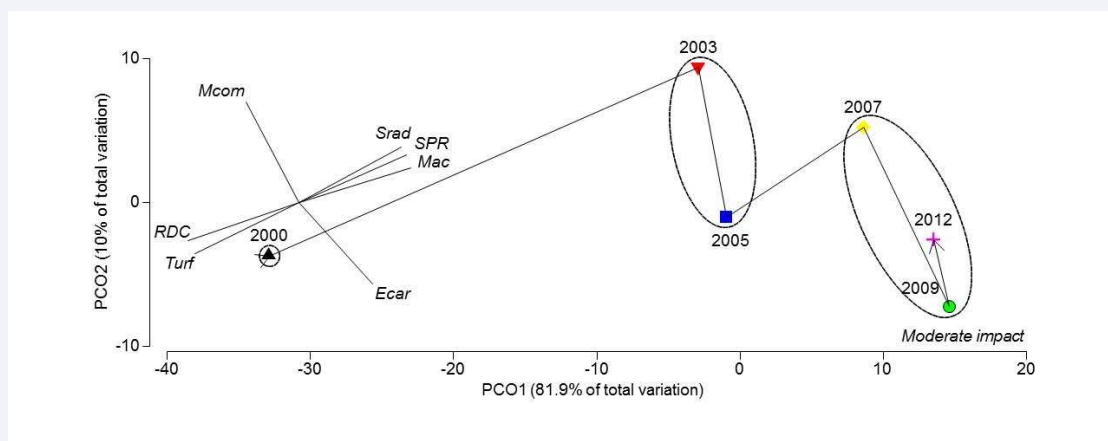




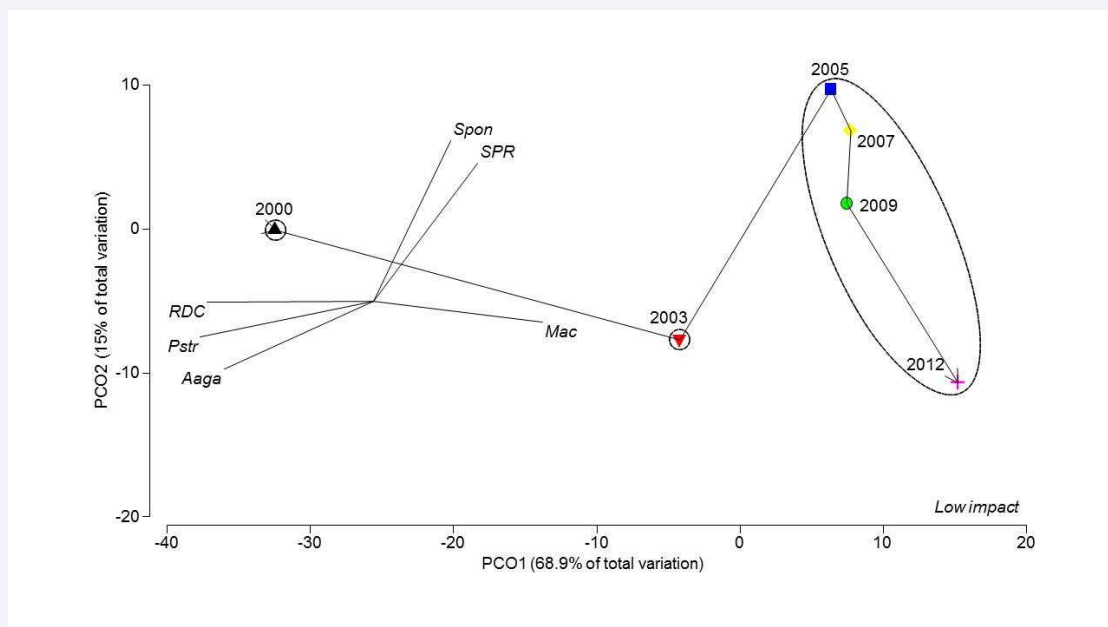
**Figure 6** Benthic community parameters: (A) Percentzoanthid cover, (B) Percentsponge cover, and (C) Percentsand, pavement and rubble cover (SPR). Mean±95% confidence intervals. For site acronyms refer to Figure 1.



**Figure 7** Principal Coordinate Ordination plot of HI benthic community trajectory. Dashed lines represent the 80% similarity cut-off level. Vectors based on a minimum correlation factor of 0.90. Model explains 95.7% of the observed variation in temporal patterns. Past= *Porites astreoides*, Srad= *Siderastrea radians*, Mcom= *Millepora complanata*, Cya= Cyanobacteria, Mac= Macroalgae, Turf= Filamentous algal turf, RDC= Recently dead coral, SPR= Sand, pavement, rubble.



**Figure 8** Principal Coordinate Ordination plot of MI benthic community trajectory. Dashed lines represent the 80% similarity cut-off level. Vectors based on a minimum correlation factor of 0.90. Model explains 91.9% of the observed variation in temporal patterns. Ecar= *Erythropodium caribbaeorum*. For other acronyms refer to Figure 7.



**Figure 9** Principal Coordinate Ordination plot of LI benthic community trajectory. Dashed lines represent the 80% similarity cut-off level. Vectors based on a minimum correlation factor of 0.90. Model explains 83.9% of the observed variation in temporal patterns. Aaga= *Agaricia agaricites*, Pstr= *Pseudodiploria strigosa*, Spon= Sponges. For other acronyms refer to Figure 7.

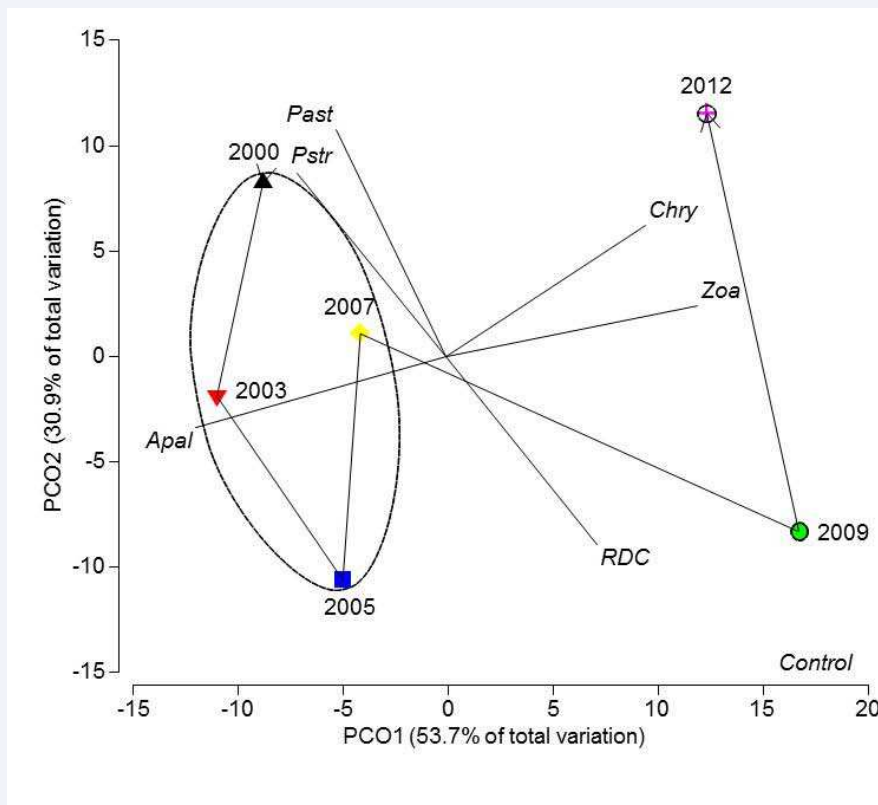
impact site had a stronger impact.

### Implications of beach renourishment on benthic community trajectories

Horizontal sediment transport in coral reefs can significantly alter benthic community composition [66] and affect long-term coral reef development [67-69]. Sediment transport modeling showed that under increasing SLR larger waves on shallow reefs would generate higher shear stress on benthic communities [70]. This would result in increased size and quantity of sediment being eroded and horizontally transported, in increased chronic turbidity, and in reduced light availability for photosynthesis, adversely impacting coral communities due to increased

sediment-induced stress. The observed benthic community degradation persisted over a time span of at least twelve years. PCO analyses showed that although minor recovery occurred within HI sites between 2000 and 2012, major degradation across MI and LI sites caused reef community trajectories to shift towards a nearly similar state as poorly recovered HI habitats, with dominance by non-reef building ephemeral coral species and macroalgae. Such altered trajectories in benthic community assemblages favoring non-reef building taxa were consistent with previous studies following impacts in coral reefs by water quality decline, fishing, declining herbivory, or coral bleaching [71-74].

Persistence of regime shifts often depend on the reef benthic



**Figure 10** Principal Coordinate Ordination plot of CT benthic community trajectory. Dashed lines represent the 80% similarity cut-off level. Vectors based on a minimum correlation factor of 0.90. Model explains 84.6% of the observed variation in temporal patterns. Chryso= *Chryso* sp. For other acronyms refer to Figure 7.

species composition, on coral assemblage functional redundancy, on individual species susceptibility to disturbance, on individual life history strategies of different coral species, and on the biogeographic location [75,76]. Coral reef communities recovered across a decadal scale or less following massive bleaching at the Arabian Gulf [77,78], Ryukyu Islands, Northwest Pacific [79], Lakshapweed Islands [76] and Chagos Archipelago [80], Indian Ocean, Palau, Micronesia [81], and Moorea, French Polynesia [82]. However, the wider Caribbean has mostly been characterized by a general lack of recovery following disturbance [72,73,83,84]. The Western Atlantic is characterized by low biodiversity and functional redundancy [85], a condition which can significantly impair rapid recovery of ecosystem resilience. Coral reef regime shifts favoring macroalgal dominance were suggested to be less abundant than previously considered [86]. However, this condition has been much more common across Caribbean reefs. Recovery may occur following an isolated stochastic disturbance such as a massive bleaching event or a hurricane, in the absence of other chronic disturbances. But the combined impact of chronic human-driven disturbances with acute massive bleaching events can produce cumulative or synergistic impacts on coral reef benthic community trajectories and ecosystem resilience which still remain poorly understood, and may result in slowly-evolving regime shifts which could be difficult to identify and reverse [87].

### Local and regional scale impacts on regime shifts

A regime shift may not always involve a large acute change

in ecosystem condition, but rather a long-term persistent shift in the trajectory and dynamics of the system, which often involves a change in its internal feedback mechanisms [88]. A regime shift favoring dominance by macroalgae and other non-reef building taxa was documented in this study across urban high-energy reefs. Initially, it involved a reef-scale, stochastic disturbance associated to sediment burial that extirpated a significant segment of benthic assemblages. It was followed by a long-term, slowly-evolving shift in the trajectory of extensive reef segments. This resulted in a permanent decline in percent coral cover, favoring dominance by macroalgae and ephemeral, disturbance-adapted, small-sized coral species. It is suggested that this change was the result of an additive combination or synergy of persistent local-scale and recurrent regional scale phenomena. Local-scale factors included the continuous sediment bedload impact that followed the original localized sediment burial incident and a persistent decline in coastal water transparency from a net temporal increase in rainfall and in sediment-laden runoff across the Río Grande de Arecibo watershed. In addition, chronic gray water and raw sewage pollution from multiple urban storm water outfalls, increasing POC and PIC, and the natural strong surge and hydrodynamic disturbance regime of local reef systems contributed to reef decline. Declining water quality associated to recurrent sewage and turbidity pulses has been shown to affect coral reefs across large spatial scales [59,89,90]. Elevated POC levels were experimentally shown to induce coral mortality and to accelerate the growth rate of microbes living in the corals'

surface mucopolysaccharide layer by an order of magnitude [61]. This suggests that coral mortality may occur due to a disruption of the balance between the coral and its associated microbiota. Elevated POC levels may cause coral reefs to shift further from coral to macroalgal dominance, therefore, POC concentrations should be closely monitored as part of standard long-term coral reef monitoring protocols.

Regional-scale phenomena also impacted the study sites and included persistent sea surface warming, the 2005 massive coral bleaching event and its subsequent mass coral mortality event across the northeastern Caribbean region [91-94]. It can be argued that the additive and synergistic impacts of these factors contributed to arrest natural recovery ability of HI sites to an early successional stage characterized by non-reef building taxa and macroalgae. In the other hand, the same conditions resulted in major decline of adjacent MI and LI sites, driving benthic communities back to a similar early successional stage as in HI sites. Further, this long-term degradation affected habitat quality and availability for fish assemblages resulting from the permanent filling of multiple small channels and reef crevices by coarse sand, cobbles and small boulders, which might have further pushed observed regime shifts through a combination of indirect feedback mechanisms.

The effect of chronic and acute stressors on coral reef state can form a strong synergy that can adversely influence resilience by intensifying a regime shift [95]. Pollution and climate change may alter the magnitude, frequency and duration of regime shifts, resulting in additive and often synergistic effects that can make coral reefs more vulnerable to change that previously could be absorbed, and could foster a shift towards a less desired state in their natural capacity to recover from disturbance and generate ecosystem services [96]. Coral reefs impacted by a combination of recurrent pulse and chronic factors have undergone permanent regime shifts favoring declining percent coral cover and increasing macroalgal dominance across the Southern Pacific [97], Indian [98,99], and Western Atlantic oceans [72,83,84,100,101], with limited natural recovery. Chronic stressors can foster increased macroalgal dominance, whereas acute coral bleaching events further push the system closer to such an alternate state. This occurs because resilience can change faster and much earlier than a change in reef state [95]. In this context, loss of resilience can go unnoticed for many years, resulting in slowly-evolving regime shifts which then become disproportionately more difficult to reverse [87], and result in the emergence of a novel ecosystem.

Novel Caribbean coral reef ecosystems have been largely characterized by declining reef-building coral assemblages [91,93,94,100,101], declining coral recruitment rates [102], and recent dominance by ephemeral coral species recruits [103]. This suggests a significant alteration of benthic community trajectories, particularly compromising net reef growth in the context of projected climate change, ocean acidification and SLR. This is consistent with observations of increasing significance of coral species life histories in determining dominance by stress-tolerant coral traits [104,105]. Indirect effects of degraded benthic spatial heterogeneity on herbivore fish assemblages can also reduce herbivory and alter fish community structure and functions across affected reefs [106], further pushing benthic

community regime shifts through a combination of indirect feedback mechanisms.

Disturbances that result in a state of low biotic spatial heterogeneity often precede a regime shift [107]. Chronic sediment bedload impacts can have a long-term sandblasting effect on coral assemblages which may significantly compromise net reef accretion in the future and exposing reefs to net bioerosion. Losing reef architecture may have profound long-term impacts on ecosystem processes such as nutrient recycling, fish productivity, sustaining biodiversity, and in providing multiple services [108-111], leading to net long-term declines in fish assemblages [112]. Following such disturbances, the system will shift into a state of macroalgal dominance and lower productivity, particularly under strong top-down (e.g., declining herbivory) or bottom-up (e.g., eutrophication) cascading effects. Strong persistent hydrodynamic disturbances may also affect coral species assemblages and community response to press and pulse disturbances [113], and in combination with bioerosion, may also affect reef structural complexity [110], further affecting natural recovery ability and future community trajectories. Similar combined mechanisms might have influenced benthic community trajectories after major beach renourishment stochastic impacts in this study, arresting reef natural recovery ability, resulting in a long-term slowly-evolving regime shift dominated by non-reef building taxa.

#### **A glimpse of future impact by climate change and SLR?**

Sea level is projected to significantly increase over the 21<sup>st</sup> century resulting in major coastal erosion [2,114], declining coral reefs, saltwater intrusion of groundwater [115], and displacement of human settlements from low-lying coastal areas [116]. Non-sustainable coastal development has already resulted in extensive urban and tourism construction across the coastal zone, largely affecting coastal reef systems and increasing increasing vulnerability to SLR [117]. Declining urban coral reefs are characterized by low species diversity and low percent living coral cover. High living coral cover [118], and benthic spatial heterogeneity [119], can reduce wave action by even an order of magnitude, creating calmer conditions in the backreef and fostering increased fish biodiversity. But losing percent living coral cover and spatial habitat heterogeneity, in combination with SLR, may result in long-term increased wave action. Waves are a significant destructive factor both, on exposed coral reefs and on exposed human communities [120,121]. Waves also influence sediment dynamics across coral reefs [122], and adjacent shorelines. A declining reef loses physical structure, and eventually its wave/swell buffering role. Waves, but in Puerto Rico particularly winter swells, can significantly impact coastal infrastructure, natural and cultural resources, and coastal-related economic activities. Projected SLR was modeled to generate stronger waves, increased surface currents, sediment resuspension and turbidity across shallow reefs [70]. Declining biodiversity can also increase the vulnerability of multiple coral reef functional groups and services [123], which can in turn increase the vulnerability of coastal human communities.

Under predicted SLR conditions beach renourishment could become a common shoreline stabilization management activity in the near future across many urban coastal areas, potentially

increasing impacts to adjacent shallow coral reefs. But modeling efforts to forecast potential large-scale and long-term impacts of such mitigation activities are still lacking. Spatial and temporal variation of cascade ecological effects of sediment bedload, habitat burial and turbidity associated to beach renourishment have remained largely obscure in the literature. Threshold and cross-scale effects (e.g., from physiological to ecosystem level impacts), as well as long-term coral reef benthic community trajectories following such impacts, have also remained poorly documented. This lack of information can make it difficult to predict potential long-term future impacts associated to sediment and turbidity dynamics across adjacent urban coastal reefs impacted by beach renourishment or by future shoreline erosion, increased sediment bedload, and turbidity associated to SLR.

Flooding of low-lying coastal urban and industrial areas will result in chronic unprecedented levels of pollution and eutrophication of coastal waters [124], potentially resulting in reduced water quality, hypoxia and habitat loss [7]. It can be argued that chronic shoreline erosion associated to climate change and SLR, in combination with impacts of eroded soils from flooded low-lying areas, will likely result in increased sediment spillover impacts across adjacent coastal coral reefs. If combined with increasing turbidity and urban pollution from coastal flooding and from land use change, this may produce chronic environmental impacts similar to those observed after this beach renourishment incident, potentially compromising benthic community trajectories in a nearly similar fashion.

There is evidence that some nearshore coral reefs can thrive under nearly continuous turbid conditions [125-128]. Some coral species can persist and produce an acclimatization response to high turbidity and sediment-induced stress, including the rapid replenishment of energy reserves during periods between sublethal turbidity events, shifting between phototrophy and heterotrophy, and by having rapid rates of photo-acclimation [129,130]. Nonetheless, rapid degradation of reef benthic habitat heterogeneity and the concomitant net elimination of microhabitats suitable for coral larval settlement due to sediment deposition, bedload or major algal overgrowth can be deleterious for community recovery [131,132]. Chronic impacts of horizontal sediment transport, sediment deposition, and eutrophication can be significant on scales of weeks to months [30], decadal [133], and even on century time scales [134-136]. Eutrophication has also been shown to trigger bleaching in corals [137,138], and in combination with other local stressors, can synergistically affect coral reef condition [139], and reduce its resilience to bleaching [140]. As a result, ecosystem engineer species and essential fish habitat functions for many economically valuable species can be severely eroded away or lost through slowly-evolving regime shifts. Such changes could be similar in nature to expected impacts on coastal reefs by climate change and SLR across urban scenarios, implying that slowly-evolving regime shifts could end up in a permanent alternate state which may compromise net reef accretion, ecological functions, ecosystem resilience, benefits and services.

## CONCLUSION

Burial of urban nearshore reefs by beach renourishment

had a variety of negative long-lasting impacts that resulted in a permanent regime shift favoring non reef-building taxa, and ephemeral, disturbance-adapted, small-sized corals. Impacts were initially characterized by a stochastic alteration in benthic community structure, followed by a slowly-evolving regime shift impacting adjacent benthic communities as a result from sediment bedload, arresting them on an early successional stage characterized by poor or none reef building activity. Beach renourishment in areas adjacent to coral reefs should not be dismissed as insignificant as many contracted environmental assessments often presume. Carry over and cross-scale effects of beach renourishment need to be addressed in the future during environmental impact evaluation prior to permitting procedures to help prevent ecological catastrophes.

This study also showed that other factors such as chronic coastal water quality decline associated to turbid river influences, polluted urban runoff, high POC, high SST, massive coral bleaching, post-bleaching coral mortality, and prevailing strong wave action could have also resulted in a combination of cumulative or synergistic effects that further contributed to the observed slowly-evolving regime shift. It arrested HI benthic community recovery trajectory to an early successional stage, but also drove MI and LI reef segments back to a nearly similar early successional stage within a period of twelve years following beach renourishment disturbance. Assessing benthic community trajectory is a critical tool to address the long-term recovery dynamics of coral reef benthic communities, can provide paramount information to managers to project and guide future reef rehabilitation and mitigation through ecosystem-based management, and should be incorporated as part of legal requirements during permitting procedures of similar projects in the future. Addressing such slowly-evolving changes in benthic community trajectories may also prove critical to address natural variability resulting from climate change-related phenomena.

Lessons learned from this glimpse of potential future climate change and SLR scenarios are also concerning as a slowly-evolving regime shift resulted in a significant change in benthic urban coral reef assemblages within only a decadal scale. Impacts on urban coral reefs can be severe and long-lasting, resulting in a benthic community trajectory leading to an arrested early successional stage, overall reef flattening, and in a permanent alteration to reef's biodiversity, ecosystem resilience, ecological functions, productivity, benefits and services. The fact that long-term persistent sediment bedload caused such broad impacts across reef segments adjacent to the sediment fill site reflects that even highly-localized beach renourishment activities can affect coral reefs across very large spatial scales, reflecting also potential future impacts by shoreline erosion, sediment bedload and chronic pollution associated to climate change and SLR. This case study represents a long-term natural experiment simulating what would be the impact of chronic sediment bedload and declining water quality on adjacent shallow urban coral reefs as a result of shoreline erosion and SLR.

Under such conditions, if impacts are similar to what was documented at La Marginal reef, net coral reef accretion would be severely compromised across very large spatial scales, losing reef's net wave action buffering function, and exposing in the

long run the shoreline and coastal infrastructure to further erosion, increasing the vulnerability of human settlements to SLR. The conservation and rehabilitation of shallow urban coral reefs have important implications for adapting and mitigating impacts by climate change and SLR on coastal human settlements. These potential effects should be taken into consideration when assessing the potential impacts and feasibility of beach renourishment activities to prevent irreversible ecological catastrophes in the future that could have unknown socio-economic and environmental consequences under projected climate change and SLR trends.

## ACKNOWLEDGMENTS

This publication was possible thanks to the support of the National Science Foundation (HRD #0734826) to the Center for Applied Tropical Ecology and Conservation, as well as by the support provided by the University of Puerto Rico Central Administration, both to E.A. Hernández-Delgado. We acknowledge the support provided by volunteer stakeholders from Surf and Environment Conservation Coalition and M. Sarriera, to A. Rodríguez and M. Sarriera during the early years of the study.

## CONFLICT OF INTEREST

All persons who meet authorship criteria are listed as authors, and both authors certify that they have participated sufficiently in the work to take public responsibility for the content, including participation in the concept, design, data collection, analysis, writing, or revision of the manuscript. Each author certifies that this material or similar material has not been and will not be submitted to or published in any other publication. The authors certify that they have no affiliations with or involvement in any organization or entity with any financial interest (such as honoraria, grants, membership, employment, consultancies, stock ownership, or other equity interest, or patent-licensing arrangements), or non-financial interest (such as personal or professional relationships, affiliations, knowledge or beliefs) in the subject matter or materials discussed in this manuscript.

## REFERENCES

- Nicholls RJ, Hoozemans FM, Marchand M. Increasing flood risk and wetland losses due to global sea-level rise: regional and global analyses. *Global Environmental Change*. 1999; 31: S69-87.
- Wong PP. Where have all the beaches gone? Coastal erosion in the tropics. *Singapore J Trop Geogr*. 2003; 24:111-132.
- Church JA, White NJ. A 20th century acceleration in global sea-level rise. *Geophys Res Lett*. 2006; L01602: 1-4.
- Dickson ME, Walkden MJ, Hall JW. Systemic impacts of climate change on an eroding coastal region over the twenty-first century. *Climatic Change*. 2007; 84: 141-166.
- Tibaldi C, Strauss BH, Zervas CE. Modelling sea level rise impacts on storm surges along US coasts. *Environ Res Lett*. 2012; 014032: 1-11.
- Bellard C, Leclerc C, Courchamp F. Potential impact of sea level rise on French islands worldwide. *Nature Conserv*. 2013; 5: 75-86.
- Hernández-Delgado EA. The emerging threats of climate change on tropical coastal ecosystem services, public health, local economies and livelihood sustainability of small islands: Cumulative impacts and synergies. *Mar Pollut Bull*. 2015; 101: 5-28.
- Buddemeier RW, Jokiel PL, Zimmerman KM, Lane DR, Carey JM, Bohling GC, et al. A modeling tool to evaluate regional coral reef responses to changes in climate and ocean chemistry. *Limnol Oceanogr Methods*. 2008; 6: 395-411.
- Buddemeier RW, Lane DR, Martinich JA. Modeling regional coral reef responses to global warming and changes in ocean chemistry: Caribbean case study. *Climatic change*. 2011; 109: 375-397.
- Hoeke RK, Jokiel PL, Buddemeier RW, Brainard RE. Projected changes to growth and mortality of Hawaiian corals over the next 100 years. *PloS One*. 2011; 6: e18038.
- Freeman LA, Kleypas JA, Miller AJ. Coral reef habitat response to climate change scenarios. *PloS One*. 2013; 8: e82404.
- Lane DR, Ready RC, Buddemeier RW, Martinich JA, Shouse KC, Wobus CW. Quantifying and valuing potential climate change impacts on coral reefs in the United States: Comparison of two scenarios. *PloS One*. 2013; 8: e82579.
- Arendt AA, Echelmeyer KA, Harrison WD, Lingle CS, Valentine, VB. Rapid wastage of Alaska glaciers and their contribution to rising sea level. *Science*. 2002; 297: 382-386.
- Meehl GA, Washington WM, Collins WD, Arblaster JM, Hu A, Buja LE, et al. How much more global warming and sea level rise? *Science*. 2005; 307: 1769-1772.
- Rahmstorf S. A semi-empirical approach to projecting future sea-level rise. *Science*. 2007; 315: 368-370.
- Nicholls RJ, Cazenave A. Sea-level rise and its impact on coastal zones. *Science*. 2010; 328: 1517-1520.
- Schlacher TA, Dugan J, Schoeman DS, Lastra M, Jones A, Scapini F, et al. Sandy beaches at the brink. *Div Distrib*. 2007; 13: 556-560.
- Shivlani MP, Letson D, Theis M. Visitor preferences for public beach amenities and beach restoration in South Florida. *Coastal Mgmt*. 2003; 31: 367-385.
- Boruff BJ, Emrich C, Cutter SL. Erosion hazard vulnerability of US coastal counties. *J Coast Res*. 2005; 21: 932-942.
- National Research Council, Ocean Studies Board. *Mitigating Shore Erosion Along Sheltered Coasts*. National Academies Press. 2007; 1-174.
- Valverde HR, Trembanis AC, Pilkey OH. Summary of beach nourishment episodes on the US east coast barrier islands. *J Coast Res*. 1999; 15:1100-1118.
- Fletcher CH, Mullane, RA, Richmond BM. Beach loss along armored shorelines on Oahu, Hawaiian Islands. *J Coast Res*. 1997; 13: 209-215.
- Kraus NC. The effects of seawalls on the beach: an extended literature review. *J Coast Res*. 1988; 4: 1-28.
- Davison AT, Nicholls RJ, Leatherman SP. Beach nourishment as a coastal management tool: an annotated bibliography on developments associated with the artificial nourishment of beaches. *J Coast Res*. 1992; 8: 984-1022.
- Haddad TC, Pilkey OH. Summary of the New England beach nourishment experience (1935-1996). *J Coast Res*. 1998; 14: 1395-1404.
- Trembanis AC, Pilkey OH, Valverde HR. Comparison of beach nourishment along the US Atlantic, Great Lakes, Gulf of Mexico, and New England shorelines. *Coastal Mgmt*. 1999; 27: 329-340.
- Silberman J, Klock M. The recreation benefits of beach renourishment. *Ocean Shoreline Mgmt*. 1988; 11: 73-90.
- García C, Servera J. Impacts of tourism development on water demand

- and beach degradation on the island of Mallorca (Spain). *Geograf Annaler: Ser A, Phys Geog.* 2003; 85: 287-300.
29. Lindeman KC. Comparative management of beach systems of Florida and the Antilles: applications using ecological assessment and decision support procedures. In Chambers G (ed.), *Managing Beach Resources in the Smaller Caribbean Islands.* 134-164. UNESCO Coastal Region and Small Island Paper. 1997 1: 1-269.
30. Lindeman KC, Snyder DB. Nearshore hardbottom fishes of southeast Florida and effects of habitat burial caused by dredging. *Fish Bull.* 1999; 97: 508-525.
31. Peterson CH, Hickerson DHM, Grissom-Johnson G. Short-term consequences of nourishment and bulldozing on the dominant large invertebrates of a sandy beach. *J Coast Res.* 2000; 16:368-378.
32. Peterson CH, Bishop MJ. Assessing the environmental impacts of beach nourishment. *Bioscience.* 2005; 55: 887-896.
33. Peterson CH, Bishop MJ, Johnson GA, D'Anna LM, Manning LM. Exploiting beach filling as an unaffordable experiment: benthic intertidal impacts propagating upwards to shorebirds. *J Exp Mar Biol Ecol.* 2006; 338: 205-221.
34. Overpeck JT, Otto-Bliesner BL, Miller GH, Muhs DR, Alley, RB, Kiehl JT. Paleoclimatic evidence for future ice-sheet instability and rapid sea-level rise. *Science* 2006; 311: 1747-1750.
35. Pfeffer, WT, Harper, JT, O'Neel S. Kinematic constraints on glacier contributions to 21st-century sea-level rise. *Science.* 2008; 321: 1340-1343.
36. Rahmstorf S. Sea-level rise: towards understanding local vulnerability. *Environ Res Lett.* 2012; 021001: 1-3.
37. Klein RJ, Nicholls RJ. Assessment of coastal vulnerability to climate change. *Ambio.* 1999; 28: 182-187.
38. Galbraith H, Jones R, Park R, Clough J, Herrod-Julius S, Harrington B, et al. Global climate change and sea level rise: potential losses of intertidal habitat for shorebirds. *Waterbirds.* 2002; 25:173-183.
39. Greene KE. Beach nourishment: a review of the biological and physical impacts (Vol. 7). *Atl States Mar Fish Comm.* 2002; 1-174.
40. Bosello F, Roson R, Tol RS. Economy-wide estimates of the implications of climate change: Human health. *Ecol Econ.* 2006; 58: 579-591.
41. Nelson WG. Beach nourishment and hard bottom habitats: The case for caution. *Proc 1989 Natl Conf Beach Preserv Technol Fl Beach Shore Preserv Assoc, Tallahassee, Florida.* 1990; 109-116.
42. Airolidi L, Abbiati M, Beck MW, Hawkins SJ, Jonsson PR, Martin D, et al. An ecological perspective on the deployment and design of low-crested and other hard coastal defence structures. *Coast Eng.* 2005; 52: 1073-1087.
43. Scheffer M, Carpenter SR. Catastrophic regime shifts in ecosystems: linking theory to observation. *TREE.* 2003; 18: 648-656.
44. Dodge RE, Rimas-Vaisnys J. Coral populations and growth patterns: responses to sedimentation and turbidity associated with dredging. *J Mar Res.* 1977; 35:715-730.
45. Nowlis JS, Roberts CM, Smith AH, Siirila E. Human-enhanced impacts of a tropical storm on nearshore coral reefs. *Ambio.* 1997; 26: 515-521.
46. Nemeth RS, Nowlis JS. Monitoring the effects of land development on the near-shore reef environment of St. Thomas, USVI. *Bull Mar Sci.* 2001; 69: 759-775.
47. Ramos-Scharrón CE, Amador JM, Hernández-Delgado EA. An interdisciplinary erosion mitigation approach for coral reef protection – A case study from the eastern Caribbean. 127-160. In, A. Cruzado (Ed.), *Marine Ecosystems*, InTech Publications. 2012.
48. Ramos-Scharrón C, Torres-Pulliza D, Hernández-Delgado EA. Watershed- and island-scale land cover changes in Puerto Rico (1930s-2004) and their potential effects on coral reef ecosystems. *Sci Total Environ.* 2015; 506-507.
49. Bak RPM. Lethal and sublethal effects of dredging on reef corals. *Mar Poll Bull.* 1978; 9: 14-16.
50. Brown BE, Howard LS. Assessing the effects of "stress" on reef corals. *Adv Mar Biol.* 1985; 22: 1-63.
51. Pastorok RA, Bilyard GR. Effects of sewage pollution on coral reef communities. *Mar Ecol Progr Ser.* 1985; 21:175-189.
52. Rogers CS. Responses of coral reefs and reef organisms to sedimentation. *Mar Ecol Prog Ser.* 1990; 62: 185-202.
53. Brown BE. Disturbances to reefs in recent times. 354-379. In, Birkeland C (ed.), *Life and Death of Coral Reefs.* Kluwer Academic Publishers, Boston, MA. 1997; 1-536.
54. Buddemeier RW, Smith SV. Coral reef growth in an era of rapidly rising sea level: predictions and suggestions for long-term research. *Coral Reefs.* 1988; 7: 51-56.
55. Shannon CE, Weaver W. *The Mathematical Theory of Communication.* University of Illinois Press, Urbana, IL. 1948; 1-117.
56. Pielou EC. Species diversity and pattern diversity in the study of ecological succession. *J Theor Biol.* 1966; 13: 370-383.
57. Clarke KR, Warwick RM. *Change in Marine Communities: An Approach to Statistical Analysis and Interpretation.* 2nd Ed. PRIMER-E, Ltd., Plymouth Marine Laboratory, UK. 2001.
58. Anderson MJ, Gorley RN, Clarke KR. *PERMANOVA + for PRIMER: Guide to Software and Statistical Methods.* PRIMER-E: Plymouth, U.K. 2008.
59. Díaz-Ortega G, Hernández-Delgado EA. Unsustainable land-based source pollution in a climate of change: a roadblock to the conservation and recovery of Elkhorn Coral *Acropora palmata* (Lamarck 1816). *Natural Resources.* 2014; 5: 561-581.
60. Lapointe BE, Clark MW. Nutrient inputs from the watershed and coastal eutrophication in the FloridaKeys. *Estuaries.* 1992; 15: 465-476.
61. Means JC, Sigleo AC. Contribution of coral reef mucus to the colloidal organic pool in the vicinity of Discovery Bay, Jamaica, WI. *Bull Mar Sci.* 1986; 39: 110-118.
62. Boyer JN, Fourqurean JW, Jones RD. Spatial characterization of water quality in Florida Bay and Whitewater Bay by multivariate analyses: zones of similar influence. *Estuaries.* 1997; 20: 743-758.
63. Kline DI, Kuntz NM, Breitbart M, Knowlton N, Rohwer F. Role of elevated organic carbon levels and microbial activity in coral mortality. *Mar Ecol Progr Ser.* 2006; 314:119-125.
64. Mook W, Koene BK. Chemistry of dissolved inorganic carbon in estuarine and coastal brackish waters. *Est Coastal Mar Sci.* 1975; 3: 325-336.
65. Hernández-Delgado EA, Hutchinson-Delgado YM, Laureano R, Hernández-Pacheco R, Ruiz-Maldonado TM, et al. Sediment stress, water turbidity and sewage impacts on threatened Elkhorn coral (*Acropora palmata*) stands at Vega Baja, Puerto Rico. *Proc Gulf Caribbean Fish Inst.* 2011; 63: 83-92.
66. Yoshioka PM, Yoshioka BB. Effects of wave energy, topographic relief and sediment transport on the distribution of shallow-water gorgonians of Puerto Rico. *Coral Reefs.* 1989; 8: 148-152.
67. Roy KJ, Smith SV. Sedimentation and coral reef development in turbid

- water: Fanning Lagoon. *Pac Sci* 1971; 25: 234-248.
68. Hubbard DK. Sedimentation as a control of reef development. *Coral Reefs*. 1986; 5: 117-125.
69. Abelson A, Shteinman B, Fine M, Kaganovsky S. Mass transport from pollution sources to remote coral reefs in Eilat (Gulf of Aqaba, Red Sea). *Mar Poll Bull*. 1999; 38: 25-29.
70. Storlazzi CD, Elias E, Field ME, Presto MK. Numerical modeling of the impact of sea-level rise on fringing coral reef hydrodynamics and sediment transport. *Coral Reefs*. 2011; 30: 83-96.
71. Hallock P, Schlager W. Nutrient excess and the demise of coral reefs and carbonate platforms. *Palaios*. 1986; 1: 389-398.
72. Hughe TP. Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. *Science*. 1994; 265: 1547-1551.
73. McClanahan TR, Muthiga, NA. An ecological shift in a remote coral atoll of Belize over 25 years. *Environ Conserv*. 1998; 25: 122-130.
74. Somerfield PJ, Jaap WC, Clarke KR, Callahan M, Hackett K, Porter J, et al. Changes in coral reef communities among the Florida Keys, 1996–2003. *Coral Reefs*. 2008; 27: 951-965.
75. Connell JH. Disturbance and recovery of coral assemblages. *Coral Reefs*. 1997; 16: S101-S113.
76. Arthur R, Done TJ, Marsh H, Harriott V. Local processes strongly influence post-bleaching benthic recovery in the Lakshadweep Islands. *Coral Reefs*. 2006; 25: 427-440.
77. Sheppard C, Loughland R. Coral mortality and recovery in response to increasing temperature in the southern Arabian Gulf. *Aquat Ecosyst Health Mgmt*. 2002; 5: 395-402.
78. Burt J, Bartholomew A, Usseglio P. Recovery of corals a decade after a bleaching event in Dubai, United Arab Emirates. *Mar Biol*. 2008; 154: 27-36.
79. Kayanne H, Harii S, Ide Y., Akimoto F. Recovery of coral populations after the 1998 bleaching on Shiraho Reef, in the southern Ryukyus, NW Pacific. *Mar Ecol Progr Ser*. 2002; 239: 93-103.
80. Sheppard CRC, Harris A, Sheppard ALS. Archipelago-wide coral recovery patterns since 1998 in the Chagos Archipelago, central Indian Ocean. *Mar Ecol Progr Ser*. 2008; 362: 109-117.
81. Golbuu, Y, Victor S, Penland L, Idip D Jr, Emaurois C, Okaji K, et al. Palau's coral reefs show differential habitat recovery following the 1998-bleaching event. *Coral Reefs*. 2007; 26: 319-332.
82. Adjeroud M, Michonneau F, Edmunds PJ, Chancerelle Y, Lison De Loma T, Penin L, et al. Recurrent disturbances, recovery trajectories, and resilience of coral assemblages on a South Central Pacific reef. *Coral Reefs*. 2009; 28: 775-780.
83. Gardner TA, Côté IM, Gill JA, Grant A, Watkinson AR. Long-term region-wide declines in Caribbean corals. *Science*. 2003; 301: 958-960.
84. Gardner TA, Côté IM, Gill JA, Grant A, Watkinson AR. Hurricanes and Caribbean coral reefs: impacts, recovery patterns, and role in long-term decline. *Ecology*. 2005; 86: 174-184.
85. Bellwood DR, Hughes TP, Folke C, Nyström M. Confronting the coral reef crisis. *Nature*. 2004; 429: 827-833.
86. Bruno JF, Sweatman H, Precht WF, Selig ER, Schutte VG. Assessing evidence of phase shifts from coral to macroalgal dominance on coral reefs. *Ecology*. 2009; 90: 1478-1484.
87. Hughes TP, Linares C, Dakos V, van de Leemput IA, van Nes EH. Living dangerously on borrowed time during slow, unrecognized regime shifts. *TREE*. 2013; 28: 149-155.
88. Möllmann C, Folke C, Edwards M, Conversi A. Marine regime shifts around the globe: theory, drivers and impacts. *Phil Trans Royal Soc B: Biol Sci*. 2015; 370: 1-7.
89. Bonkosky M, Hernández-Delgado EA, Sandoz B, Robledo IE, Norat-Ramírez J, Mattei H. Detection of spatial fluctuations of non-point source fecal pollution in coral reef surrounding waters in southwestern Puerto Rico using PCR-based assays. *Mar Poll Bull*. 2009; 58: 45-54.
90. Hernández-Delgado EA, Sandoz B, Bonkosky M, Mattei H, Norat J. Impacts of non-point source sewage pollution in Elkhorn coral, *Acropora palmata* (Lamarck), assemblages of the southwestern Puerto Rico shelf. *Proc 11th Int Coral Reefs Symp*. 2010; 747-751.
91. Miller J, Muller E, Rogers C, Waara R, Atkinson A, Whelan KRT, et al. Coral disease following massive bleaching in 2005 causes 60% decline in coral cover on reefs in the US Virgin Islands. *Coral Reefs*. 2009; 28: 925-937.
92. Eakin CM, Morgan JA, Smith TB, Liu G, Alvarez-Filip L, Baca B, et al. Caribbean corals in crisis: Record thermal stress, bleaching and mortality in 2005. *Plos One*. 2010; 5: e13969.
93. Hernández-Pacheco R, Hernández-Delgado EA, Sabat AM. Demographics of bleaching in the Caribbean reef-building coral *Montastraea annularis*. *Ecosphere*. 2011; 9: 1-13.
94. Edmunds PJ. Decadal-scale changes in the community structure of coral reefs of St. John, US Virgin Islands. *Mar Ecol Progr Ser*. 2013; 489: 107-123.
95. Bozec YM, Mumby PJ. Synergistic impacts of global warming on the resilience of coral reefs. *Phil Trans Royal Soc London B Biol Sci*. 2015; 370: 20130267.
96. Folke C, Carpenter S, Walker B, Scheffer M, Elmqvist T, Gunderson L, Holling CS. Regime shifts, resilience, and biodiversity in ecosystem management. *Ann Rev Ecol Evol Syst*. 2004; 35: 557-581.
97. De'ath G, Fabricius KE, Sweatman H, Puotinen M. The 27-year declines of coral cover on the Great Barrier Reef and its causes. *Proc Natl Acad Sci*. 2012; 109: 17995-17999.
98. Bruno JF, Selig ER. Regional Decline of Coral Cover in the Indo-Pacific: Timing, Extent, and Subregional Comparisons. *PLoS One*. 2007; 2: e711.
99. Ledlie MH, Graham NAJ, Bythell JC, Wilson SK, Jennings S, Polunin NVC, et al. Phase shifts and the role of herbivory in the resilience of coral reefs. *Coral Reefs*. 2007; 26: 641-653.
100. Rogers CS, Miller J. Permanent 'phase shifts' or reversible declines in coral cover? Lack of recovery of two coral reefs in St. John, US Virgin Islands. *Mar Ecol Progr Ser*. 2006; 306: 103-114.
101. Maliao RJ, Turingan, RG, Lin J. Phase-shift in coral reef communities in the Florida Keys National Marine Sanctuary (FKNMS), USA. *Mar Biol*. 2008; 154: 841-853.
102. Edmunds PJ, Elahi R. The demographics of a 15-year decline in cover of the Caribbean reef coral *Montastraea annularis*. *Ecological Monographs*. 2007; 77: 3-18.
103. Hernández-Delgado EA, González-Ramos CN, Alejandro-Camis PJ. Large-scale coral recruitment patterns in Mona Island, Puerto Rico: Evidence of shifting coral community trajectory after massive bleaching and mortality. *Rev Biol Trop*. 2014; 62: 49-64.
104. Darling ES, Alvarez-Filip L, Oliver TA, McClanahan TR, Côté IM. Evaluating life-history strategies of reef corals from species traits. *Ecol Lett*. 2012; 15: 1378-1386.
105. Darling ES, McClanahan TR, Côté IM. Life histories predict coral community disassembly under multiple stressors. *Glob Chang Biol*. 2013; 19: 1930-1940.



106. Cruz IC, Loiola M, Albuquerque T, Reis R, de Anchieta CC, Nunes J, et al. Effect of phase shift from corals to zoantharia on reef fish assemblages. *PLoS One*. 2015; 10: e0116944.
107. McManus JW, Polsenberg JF. Coral-algal phase shifts on coral reefs: ecological and environmental aspects. *Progr Oceanogr*. 2004; 60: 263-279.
108. Alvarez-Filip L, Dulvy NK, Gill JA, Côté IM, Watkinson AR. Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. *Proc Biol Sci*. 2009; 276: 3019-3025.
109. Alvarez-Filip L, Cote IM, Gill, JA, Watkinson AR, Dulvy NK. Region-wide temporal and spatial variation in Caribbean reef architecture: is coral cover the whole story? *Global Change Biol*. 2011a; 17: 2470-2477.
110. Alvarez-Filip L, Gill JA, Dulvy NK, Perry AL, Watkinson AR, Côté IM. Drivers of region-wide declines in architectural complexity on Caribbean reefs. *Coral Reefs*. 2011b; 30: 1051-1060.
111. Graham NAJ, Nash KL. The importance of structural complexity in coral reef ecosystems. *Coral Reefs*. 2013; 32: 315-326.
112. Alvarez-Filip L, Paddack MJ, Collen B, Robertson DR, Côté IM. Simplification of Caribbean reef-fish assemblages over decades of coral reef degradation. *PLoS One*. 2015; 10: e0126004.
113. Madin JS, Connolly SR. Ecological consequences of major hydrodynamic disturbances on coral reefs. *Nature*. 2006; 444: 477-480.
114. Gornitz V. Sea-level rise: A review of recent past and near-future trends. *Earth Surf Proc Landforms* 1995; 20: 7-20.
115. Ranjan P, Kazama S, Sawamoto M. Effects of climate change on coastal fresh groundwater resources. *Global Environ Change*. 2006; 16: 388-399.
116. Jallow BP, Toure S, Barrow MM, Mathieu AA. Coastal zone of The Gambia and the Abidjan region in Côte d'Ivoire: sea level rise vulnerability, response strategies, and adaptation options. *Clim Res*. 1999; 12: 129-136.
117. Hernández-Delgado EA, Ramos-Scharrón CE, Guerrero C, Lucking MA, Laureano R, Méndez-Lázaro PA, et al. Long-term impacts of tourism and urban development in tropical coastal habitats in a changing climate. In M. Kasimoglu (ed.), *Visions from Global Tourism Industry-Creating and Sustaining Competitive Strategies*. Intech Publications. Puerto Rico. 2012; 357-398.
118. Hearn C, Atkinson M, Falter J. A physical derivation of nutrient-uptake rates in coral reefs: effects of roughness and waves. *Coral Reefs*. 2001; 20: 347-356.
119. Stoddart JB, Rippe JP, Reidenbach MA. Structure and dynamics of turbulent boundary layer flow over healthy and algae-covered corals. *Coral Reefs*. 2016; 35: 1047-1059.
120. Stoddart D. Effects of Hurricane Hattie on the British Honduras reefs and cays, October 30-31, 1961. *Atoll Res Bull*. 1963; 95: 1-142.
121. Hernández-Avila M, Roberts H, Rouse L. Hurricane generated waves and coastal boulder rampart formation. *Proc 3rd Int Coral Reef Symp*. 1977; 2: 71-78.
122. Larcombe P, Ridd PV, Prytz A, Wilson B. Factors controlling suspended sediment on inner-shelf coral reefs, Townsville, Australia. *Coral Reefs*. 1995; 14: 163-171.
123. Micheli F, Mumby PJ, Brumbaugh DR, Broad K, Dahlgren CP, Harborne AR, et al. High vulnerability of ecosystem function and services to diversity loss in Caribbean coral reefs. *Biol Conserv*. 2014; 171: 186-194.
124. Rabalais NN, Turner RE, Díaz RJ, Justić D. Global change and eutrophication of coastal waters. *ICES J Mar Sci*. 2009; 66: 1528-1537.
125. Larcombe P, Ridd PV, Prytz A, Wilson B. Factors controlling suspended sediment on inner-shelf coral reefs, Townsville, Australia. *Coral Reefs* 1995; 14: 163-171.
126. Larcombe P, Woolfe KJ. Increased sediment supply to the Great Barrier Reef will not increase sediment accumulation at most coral reefs. *Coral Reefs*. 1999; 18: 163-169.
127. Guzmán HM, Guevara CA. Arrecifes coralinos de Bocas del Toro, Panamá. *Rev Biol Trop*. 1998; 46: 601-622.
128. Bak RPM, Meesters EH. Acclimatization/adaptation of coral reefs in a marginal environment. *Proc 9th Int Coral Reef Symp*. 2002; 1: 265-271.
129. Anthony KRN, Fabricius KE. Shifting roles of heterotrophy and autotrophy in coral energetics under varying turbidity. *J Exp Mar Biol Ecol*. 2000; 252: 221-253.
130. Anthony KRN, Larcombe P. Coral reefs in turbid waters: sediment-induced stresses in corals and likely mechanisms of adaptation. *Proc 9th Int Coral Reef Symp*. 2002; 1: 239-244.
131. Babcock R, Davies P. Effects of sedimentation on settlement of *Acropora millepora*. *Coral Reefs*. 1991; 9: 205-208.
132. Hunte W, Wittenberg M. Effects of eutrophication and sedimentation on juvenile corals. II. Settlement. *Mar Biol*. 1992; 114: 625-631.
133. Cortés J. A reef under siltation stress: a decade of degradation. A8-A14. In, Ginsburg RN (ed.), *Global Aspects of Coral Reefs: Health, Hazards and History*, Miami, FL. 1993.
134. McCulloch M, Fallon S, Wyndham T, Hendy E, Lough J, Barnes D. Coral record of increased sediment flux to the inner Great Barrier Reef since European settlement. *Nature*. 2003; 421: 727-730.
135. Perry CT, Smithers SG, Johnson KG. Long-term coral community records from Luggar Shoal on the terrigenous inner-shelf of the central Great Barrier Reef, Australia. *Coral Reefs*. 2009; 28: 941-948.
136. Carilli JE, Prouty NG, Huguen KA, Norris RD. Century-scale records of land-based activities recorded in Mesoamerican coral cores. *Mar Poll Bull*. 2009; 58: 1835-1842.
137. Vega-Thurber RL, Burkepile DE, Fuchs C, Shantz AA, McMinds R, Zaneveld JR. Chronic nutrient enrichment increases prevalence and severity of coral disease and bleaching. *Global Change Biol*. 2014; 20: 544-554.
138. Ennis RS, Brandt ME, Grimes KR, Smith TB. Coral reef health response to chronic and acute changes in water quality in St. Thomas, United States Virgin Islands. *Mar Poll Bull*. 2016; 111: 418-427.
139. Meesters EH, Bak RPM, Westmacott S, Ridgley M, Dollar S. A fuzzy logic model to predict coral reef development under nutrient and sediment stress. *Conserv Biol*. 1998; 12: 957-965.
140. Carilli JE, Norris RD, Black BA, Walsh SM, McField M. Local stressors reduce coral resilience to bleaching. *Plos One*. 2009; 4: e6324.
141. Carilli JE, Norris RD, Black BA, Walsh SM, McField M. effects of seawalls on the beach: an extended literature review. *J Coast Res*. 1988; 4: 1-28.

#### Cite this article

Hernández-Delgado EA, Rosado-Matías BJ (2017) Long-Lasting Impacts of Beach Renourishment on nearshore Urban Coral Reefs: a Glimpse of Future Impacts of Shoreline Erosion, Climate Change and Sea Level Rise. *Ann Mar Biol Res* 4(1): 1021.