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A Case Study in the Effectiveness of Marine Protected Areas (MPAs): the Islands of Bonaire and Curaçao, Dutch Caribbean

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A Dissertation

Presented to

The Faculty of the School of Marine Science

The College of William & Mary

In Partial Fulfillment

of the Requirements for the Degree of

Doctor of Philosophy

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by

Noelle J. Relles

2012

APPROVAL SHEET

This dissertation is submitted in partial fulfillment of

the requirements for the degree of

Doctor of Philosophy

Noelle J. Relles

Approved, by the Committee, May 2012

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DEDICATION

For the strongest and smartest woman I know, my mother, who always believed in me and encouraged me, even when I wanted to give up, she let me know that I could do anything I put my mind to. She taught me what it is to be a loving and supportive mother and I hope to be as great a mother as her to my daughter, Coral, who I also dedicate this dissertation to. I love you both and hope that I have and continue to make you proud.

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DISSERTATION ABSTRACT

The islands of Bonaire and Curaçao, Dutch Caribbean, were both mapped along their leeward coasts for dominant coral community and other benthic cover in the early 1980s. This mapping effort offers a unique baseline for comparing changes in the benthic community of the two islands since that time, particularly given the marked differences between the two islands. Bonaire is well-protected and completely surrounded by a marine protected area (MPA), which includes two no-diving marine reserves; additionally, Bonaire's population is only around 15,000. In contrast, the island of Curaçao is home to 140,000 inhabitants and marine protection is limited, with a reef area of 600 ha established as a "paper" park (i.e., little enforcement).

Video transects collected by SCUBA over the reefs were collected on Bonaire in January of 2008; when compared to data from 1985, coral cover had declined in the shallowest portion of the reef (< 5 m) and was mostly the result of declines in *Acropora* spp., whereas head corals increased. Transects closest to the no-diving marine reserves showed higher coral cover and diversity than transects located farther from the reserves.

Satellite remote sensing techniques were used to create landscape-scale reef maps along the leeward coasts of both islands, which could differentiate areas of high hard coral cover (> 20%), predominantly sand (> 50%) and areas where hard coral and sand were mixed with soft corals, sea whips and marine plants. These modern maps (2007-09) were groundtruthed using the video data collected on Bonaire for accuracy and then compared to the early 1980s maps of the reefs on both islands.

Bonaire experienced declines in coral cover overall and the remaining coral was increasingly patchy; however, changes in patch characteristics were not significant over the time period, but status as a marine reserve and the sheltering of the shoreline did appear to buffer against coral loss. Surprisingly, the island of Curaçao did not experience a decline in total coral cover, but did become increasingly patchy, significantly more so than Bonaire. The Curaçao Underwater Park afforded no additional protection against coral loss or fragmentation than an adjacent unprotected area of reef.

The difference between the two islands in coral loss versus fragmentation has the potential for a unique natural experiment to study the effects of habitat fragmentation in the absence of overall habitat loss at the landscape scale. The Bonaire National Marine Park could benefit by restricting visitors to its most frequented dive sites by increasing the cost of entry into a tiered pay system, thus generating more income for education and management of the park, as well as deterring some divers from these overused sites. Satellite remote sensing-derived maps are useful for rapid reef mapping and can be utilized for comparison to ancillary maps created by more traditional methods. Satellite-derived maps can only distinguish benthic habitats coarsely (3-4 habitat classes) and are

only as reliable as their source data, they benefit greatly from fieldwork to determine depth, geographic location, and benthic habitat cover in real time.

A Case Study in the Effectiveness of Marine Protected Areas (MPAs): the Islands of Bonaire and Curaçao, Dutch Caribbean

DISSERTATION INTRODUCTION

Coral Reef Ecosystems

Coral reefs serve as critical habitats for fishes and benthic invertebrates by providing concentrated food sources and protection from predation. Reefs also offer a number of services that are economically valuable to humans. They provide protected nurseries for fishes to grow to market-size. Reefs offer shoreline protection from hurricanes and have even been utilized for drug discovery, while offering a substantial draw for tourists, including, boaters, sport fishermen, snorkelers, and scuba divers.

Although corals start life as individual polyps, they form colonies as they grow. These colonies form massive reef structures that can be seen from outer space. Reef-building corals share a symbiotic relationship with dinoflagellate algae in the genus *Symbiodinium* called zooxanthellae. The zooxanthellae are photosynthetic, providing fixed carbon to the coral in exchange for protection from predation; zooxanthellae are also responsible for giving the corals their vibrant colors. For this reason, reefs need to be within the photic zone, able to receive enough light to facilitate photosynthesis, as well

C annual mean temperature isotherm north and south of the equator.

Recently, coral reefs have experienced dramatic phase shifts in dominant species due to intensified human disturbance beginning centuries ago (Jackson et al., 2001). Wilkinson (1998) concluded that 10% of the world's reefs had already been irretrievably destroyed, 30% more were critically endangered within the next two to ten years and another 30% were similarly threatened with destruction on a ten- to thirty-year time scale.

Increasing concentrations of atmospheric carbon dioxide, mostly the result of fossil fuel burning since the Industrial Revolution, have led to an increase in sea surface temperature by approximately 0.74°C over the 20th century and a sea level rise of 17 cm (IPCC Climate Change, 2001). Unfortunately, most corals live at their upper temperature maximum and that of their symbionts (Hoegh-Guldberg et al., 1999). *Symbiodinium* convert solar energy and nutrients into fixed carbon, which can provide more than 95% of the metabolic requirements of the host for some species of coral (Hoegh-Guldberg et al., 2007), making them essential to the coral's health. Stressed, overheated corals become pale or white when they expel their pigmented symbionts, a phenomenon known as coral bleaching. If thermal stress is severe and prolonged, most of the corals on a reef may bleach and many may die (Hughes et al., 2003). Some corals may survive and recover their dinoflagellate symbionts after mild thermal stress, but typically show reduced growth, calcification, and fecundity (Hoegh-Guldberg et al., 1999) and may experience greater incidences of coral disease (Harvell et al., 2002, Bruno et al., 2007).

Corals are also at increased risk for disease as a result of global climate change due to increased storm events and runoff. Rain and runoff are important vectors for disease in the marine environment and have been responsible for transferring terrestrial pathogens into the ocean (Kennedy et al., 2000). Reports on disease in the ocean are increasing in frequency and are thought to be tied to an increase in the frequency of El Nino Southern Oscillation (ENSO) events, coupled with trends in global warming (Harvell et al., 1999). Outbreaks of Caribbean yellow-band disease, once confined to summer, are now "permanent" and assaults have been quicker and deadlier than in the late 1990s, when the disease first appeared. The increased ferocity seems to be correlated with a rise in average minimum water temperatures (Jones, 2007).

Coral reefs are highly diverse marine communities constructed and dominated by sessile organisms that disperse via a planktonic larval stage in their early life history (Harrington et al., 2004). Post-larval settlement is an important part of the life history that can be limited by the availability of space on the reef. Crustose coralline algae are calcifying, encrusting organisms (Bak, 1976) that offer an optimal substrate for coral growth by excluding other space competitors (Harrington et al., 2004). Decreases in the rates of recruitment of coral throughout the Caribbean have been documented and are likely due to a combination of declining abundance of coralline algae and overall coral mortality, reducing the number of spawning individuals and therefore the number of larvae released, as well as the number that successfully recruit, per spawning season (Pennisi, 2007).

Grazing is an important aspect of reef life because corals compete for space with macroalgae. Algae are faster-growing than coral so it's important for grazers to keep the algal population in check until coral larvae can recruit, grow and occupy the substrate. *Diadema antillarum,* the black long-spined sea urchin, was an important herbivore on reefs throughout the Caribbean until its die-off from disease in the early 1980s, decreasing its population by two orders of magnitude (Lessios et al. 1984). In most places throughout the Caribbean, the die-off of *Diadema* was followed by a subsequent decline in coral cover, resulting in an observed transition from coral to algal dominance beginning in the 1980s (Jackson et al., 2001). On these reefs *Diadema* had become a keystone species because it was the sole herbivore as a result of overfishing of herbivorous fishes (Hughes 1994).

In the absence of severe human impacts, reefs readily reassemble after routine disturbances such as tropical hurricanes, but do not recover from chronic disturbances (Connell et al., 1997). The frequency and intensity of hurricanes may also increase in some regions, leading to a shorter time for recovery between occurrences (IPCC, 2001). Mumby (2006) found that reefs can withstand intense disturbances, such as severe hurricanes, when grazing is undertaken by both parrotfish and the urchin *Diadema.* When herbivores are lost from the system, disturbances from which corals are usually able to recover become more problematic, because faster-growing algae are able to exclude them (Pandolfi et al., 2003). Overharvesting of herbivorous fishes can impair the resilience of coral reefs. Resilience is defined as an organism or system's ability to recover following a disturbance; an impairment of the system's resilience can inhibit its recovery from bleaching and other disturbances, leading to a phase shift to algal-dominated reefs (Hughes et al., 2003). Human impacts and the increased fragmentation of coral reef habitat have preconditioned reefs for susceptibility, undermining reef resilience and making them much more susceptible to future climate change (Hughes et al., 2003).

The Island of Bonaire

The island of Bonaire is part of a chain of oceanic islands off the coast of South America including nearby Aruba and Curaçao. It lies in the southern Caribbean outside of the hurricane belt, 80 km north of Venezuela and 48 km east of Curaçao. Bonaire is only 290 km^2 in area with a population of about 13,000 (STINAPA, 2007) and an economy based primarily on diving tourism, which leads to a community invested in protecting the pristine nature of its reefs. This investment took the form of the establishment of the Bonaire National Marine Park in 1979, which surrounds the island down to a depth of 60 m. The regulations of the marine park include no fishing in its waters, a maintenance fee for underwater visitors, and a checkout dive required of scuba divers. These rules are enforced by Stichting Nationale Parken Bonaire, STINAPA, a non-governmental, not for profit foundation commissioned by the island government to manage the park.

A comprehensive atlas by Dr. Fleur van Duyl was completed in 1985 on the living reefs of Bonaire and Curaçao; the reef was mapped to a depth of 20 m, documenting the dominant community type and percent cover (van Duyl, 1985). More recently, Steneck et al., (2007) completed *A Report on the Status of the Coral Reefs of Bonaire in 2007 with Results from Monitoring 2003-2007*, which examined coral cover, macroalgae, herbivory, and coral recruitment at six sites along the leeward coast of Bonaire. A comparison of these six sites showed trends toward an increase in macroalgae, declining herbivory from parrotfish and an increase in damselfish populations over the four-year monitoring period (Steneck et al., 2007). Coral cover remains relatively high, with an average cover of nearly 50%, considerably higher than the Caribbean-wide average of nearly 10% (Gardner et al., 2003).

The island of Bonaire was affected by the *Diadema* die-off from disease in the 1980s, but populations are beginning to recover, increasing from zero to 0.3

individuals/(200 m^2) from 1999 to 2007 (Steneck et al. 2007). It is hypothesized that after the die-off of *Diadema*, healthy herbivorous fish populations kept the spatial coverage of macroalgae to a minimum. Steneck (2007) found a decrease in coralline abundance and an increase in macroalgal abundance over an eight-year survey, which included data for 1999 from the Atlantic and Gulf Rapid Reef Assessment (AGRRA). Steneck attributes these eight-year trends to the decline in parrotfish abundance. Mumby (2006) found that parrotfish can fulfill the ecological niche of *Diadema* as important grazers on the reef if their population size is large enough. For this reason, the importance of Bonaire's marine park as a no-fishing area sustains the populations of herbivorous fish while *Diadema* populations recover.

Conservation and Management of Coral Reefs

Responding to the global coral reef crisis requires active management of human activities that modify essential ecological processes (Bellwood et al., 2004). Reef managers and coastal resource policies must reduce the influence of local stressors such as declining water quality, coastal pollution, and overexploitation of key functional groups such as herbivores (Hughes et al., 2003). *Status of the Coral Reefs of the World: 2002* reported successes demonstrating our ability to reverse coral reef decline through the use of marine reserves, marine protected areas, and other initiatives that address multiple stresses, including fishing (Wilkinson, 2002). Marine Protected Area (MPA) is an umbrella term that can encompass various levels of marine protection. The Bonaire National Marine Park is an example of the strictest form of an MPA, a complete No-Take

Area (NTA). NTAs are a form of MPA which provide the most effective protection for extractive activities such as fishing, affording a spatial refuge for a portion of the stock from which larvae and adults can disperse to adjoining exploited areas (Lubchenco et al. 2003; Palumbi 2003). The rate of establishment and size of NTAs, as a tool for resilience management, needs to be greatly increased because even the largest NTAs in the world are not self-sustaining, because they are too small relative to the scale of natural and human disturbances, and to the dispersal distances of many larvae and migrating adults (Bellwood et al., 2004).

Long-term monitoring and comparisons of change over time in coral reef ecosystems can be an invaluable tool for reef management. An established NTA, like the Bonaire National Marine Park, can be used by decision-makers to test the long-term effectiveness of NTAs and evaluate the necessary spatial scale in order to maintain the system's resilience.

The Island of Curaçao

The island of Curaçao is located 48 km west of Bonaire in the Caribbean. It is slightly larger than Bonaire, 171 km^2, with nearly ten times the population, more than 130,000 inhabitants. Curaçao also offers a sharp contrast to Bonaire in its level of marine protection, with little legislation and virtually no enforcement.

The Curaçao Underwater Park was established in 1983. It encompasses 20 km of reef off of the southern coast of the island, from Oostpunt to Willemstad, but has no legislative support and therefore no legal protection. Park management is performed by the Caribbean Research and Management of Biodiversity (Carmabi, 2007) Foundation, a local NGO. Coral collection and spearfishing are banned, but enforcement is insufficient due to a lack of adequate financial support and staff.

The comprehensive maps completed in the 1980s by van Duyl for both islands offer a unique opportunity for comparison of change in benthic habitat type over time in a protected versus an unprotected reef. Results from this comparison could have important implications for the design and implementation of marine policy around the world based on its evaluation of the effectiveness of MPAs, specifically NTAs, of this size. The size and number of MPAs, as well as their level of restriction and regulation, are all important factors in the establishment of MPAs and enforcement of marine policy. Success in Bonaire relative to Curaçao has implications for these factors when considering establishment of MPAs elsewhere.

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CHAPTER 1

Coral cover and diversity on the leeward shore of Bonaire, Dutch Caribbean: changes since the early 1980s and current trends

Manuscript citation: Relles, NJ and MR Patterson. Coral cover and diversity on the leeward shore of Bonaire, Dutch Caribbean: changes since the early 1980s and current trends. Intended to be submitted to *Journal of Experimental Marine Biology and Ecology*.

Abstract

Bonaire, Dutch Caribbean, has a long-established Marine Protected Area (MPA) that has a high percentage of live coral cover, particularly compared to elsewhere in the Caribbean. The reefs along the leeward coast were extensively mapped in the early 1980s, shortly after the Bonaire Marine Park was established in 1979. Since this mapping, two no-diving marine reserves were established (1991) and management and maintenance of the park has increased. Underwater video data were collected in January 2008 at 10 sites at a resolution that made species identification possible. Percent coral cover has declined since the early 1980s, particularly in the assemblages dominated previously by *Acropora cervicornis*. These data showed a negative relationship between coral cover and diversity when compared to increasing distance from the no-diving marine reserves, but frequency of dive site use by sport divers showed no significant effect on coral cover or diversity.

Introduction

The island of Bonaire, Dutch Caribbean, is located in the southern Caribbean Sea, approximately 80 km north of Venezuela and is part of a seamount system that includes Aruba, Curaçao, and Los Roques. It represents a long-established Marine Protected Area (MPA) since its designation as such in 1979. The subtidal zone of the entire island, from the high water mark out to the 60 m depth contour, an area of 27 km^2, constitutes the Bonaire National Marine Park (BNMP), where fishing is limited to traditional methods including handline, hook and line, fish traps (limited in number), and nets (with size restrictions). Spear fishing is prohibited, and conch fishing requires a government permit that is only available to local residents. In 2008, a three km section of the coastline was designated a no-fishing zone. Underwater visitors are charged a nature fee that contributes to park infrastructure, including a series of moorings along the leeward coast so that boats do not need to anchor, as this is also prohibited in the park. Heavy fines and confiscation of equipment are associated with violations of the park rules and regulations, which are enforced by Stichting Nationale Parken Bonaire (STINAPA), a non-governmental, not-for-profit foundation commissioned by the island government to manage the park (STINAPA 2007). The BNMP also includes two marine reserves, established in 1991 along the northwestern coast, where anchor dropping, scuba diving, and snorkeling are prohibited, and fishing is limited to the traditional methods listed above, with the exception of fish traps.

A comprehensive atlas of the reefs of Bonaire and Curaçao was completed in 1985 by Dr. Fleur van Duyl, with data collection in the early 1980s; the reef on Bonaire was mapped in detail to a depth of 10 m, documenting the dominant community type and percent cover (van Duyl 1985). van Duyl used a categorical system of percent cover, which represented series of ranges (< 10%, 10-20%, 20-40%, and > 40%) for each of five major coral groups: *Acropora palmata* group, *A. cervicornis* group, foliate/finger coral group, head coral group, and head/foliate coral group. The foliate/finger coral group consisted primarily of the foliate corals *Agaricia* spp. and the hydrocoral *Millepora* spp., while the finger corals were represented predominantly by *Madracis mirabilis* and *Porites porites*. The head coral group was characterized by a dominance of head corals combined with an absence of foliate corals; the most common head coral species were *Montastraea annularis, M. cavernosa, Diploria strigosa, Siderastrea siderea, S. radians, Dichocoenia stokesii, Colpophyllia natans, P. astreoides, Meandrina meandrites,* and *Stephanocoenia intersepta*. The head/foliate coral group was characterized by head corals accompanied by the foliate *Agaricia* spp., with the dominant head coral *M. annularis.* Finger corals were also often present in this group (van Duyl 1985). More recently, Steneck et al. (2011) examined coral cover, macroalgae, herbivory, and coral recruitment at six sites along the leeward coast of Bonaire. Over a four-year period, the study showed trends of increased biomass of macroalgae, declining herbivory from parrotfish, and an increase in damselfish populations, which are known to be detrimental to the reef (Steneck et al. 2011). Steneck et al. (2007) concluded that coral cover remains relatively high, with an average cover of nearly 50% at 10 m deep; Bak et al. (2005) found lower coral cover, around 25% in permanent quadrats at 10-20 m depth along the leeward shore of Bonaire, but both of these estimates are considerably higher than the Caribbean-wide average of about 10% (Gardner et al. 2003).

As with most Caribbean islands, Bonaire is a popular tourist destination, and scuba diving tourism is the largest industry on the island. Nearly 60% of Bonaire's more than 70,000 tourists in 2008 were scuba divers (Bonaire Tourism 2008). Scuba diving is generally considered a non-destructive use of a reef; however, recent research has shown the effects of recreational scuba divers to be detrimental to the coral reef structure (Zakai and Chadwick-Furman 2002; Davis and Tisdell 1995; Hawkins et al. 1999). Dixon et al. (1993) posited that there is a critical threshold level when reef quality begins to suffer between 4000 and 6000 dives annually per site. Assuming dive tourists are evenly maximizing use of Bonaire's 86 dive sites, this works out to only between eight and twelve dives per tourist diver, based on the more than 42,000 divers that visited Bonaire in 2008, before reef degradation would be evident. However, the majority of divers on Bonaire make between 11 and 15 dives during their trip, and there is a large disparity in dive site usage, with the top 10 sites receiving approximately 35% of the total number of dives and the top 20 sites receiving more than 55% (Thur 2003).

Bonaire's approximately 15,000 residents reside in two main towns, Rincón and the capital of Kralendijk. Rincón is located inland, but Kralendijk is located along the coast in the center of the leeward shore and is the area of the majority of development (Figure 1). The number of residents on Bonaire has increased since the time of van Duyl's mapping from 9,000 inhabitants (Central Bureau of Statistics 1981); development and urbanization are known to threaten reefs through increased sedimentation and nutrient runoff as a result of human activities such as pollution and dredging (Hughes 1994; Hughes et. al. 2003). Despite many empirical studies on reserves, only a few have included data collected before and after reserve establishment (Willis et al. 2003), which are the most appropriate for investigating the impact of reserve establishment (Claudet et al. 2006). Although the reserves on Bonaire are considered marine reserves by STINAPA Bonaire, the only difference between the reserves and the rest of the park is a lack of underwater visitors and fishing traps; traditional fishing is allowed in the reserves, as outlined above.

In this study coral cover and diversity were used as positive metrics of reef health. While cover is generally agreed upon as a good measure of a healthy reef, the use of diversity remains debatable (Ives and Carpenter 2007). However, we chose to use diversity as a positive sign of reef health because diversity increases the resiliency of an ecosystem due to functional redundancy (McCann 2000). This study compared coral cover in 10 transects from 2008 to cover in the early 1980s calculated from van Duyl's categorical coral cover system. This study also examined coral cover and diversity at the same 10 sites in 2008 as a function of distance from the nearest marine reserve, distance from the capital city, and frequency of use by divers.

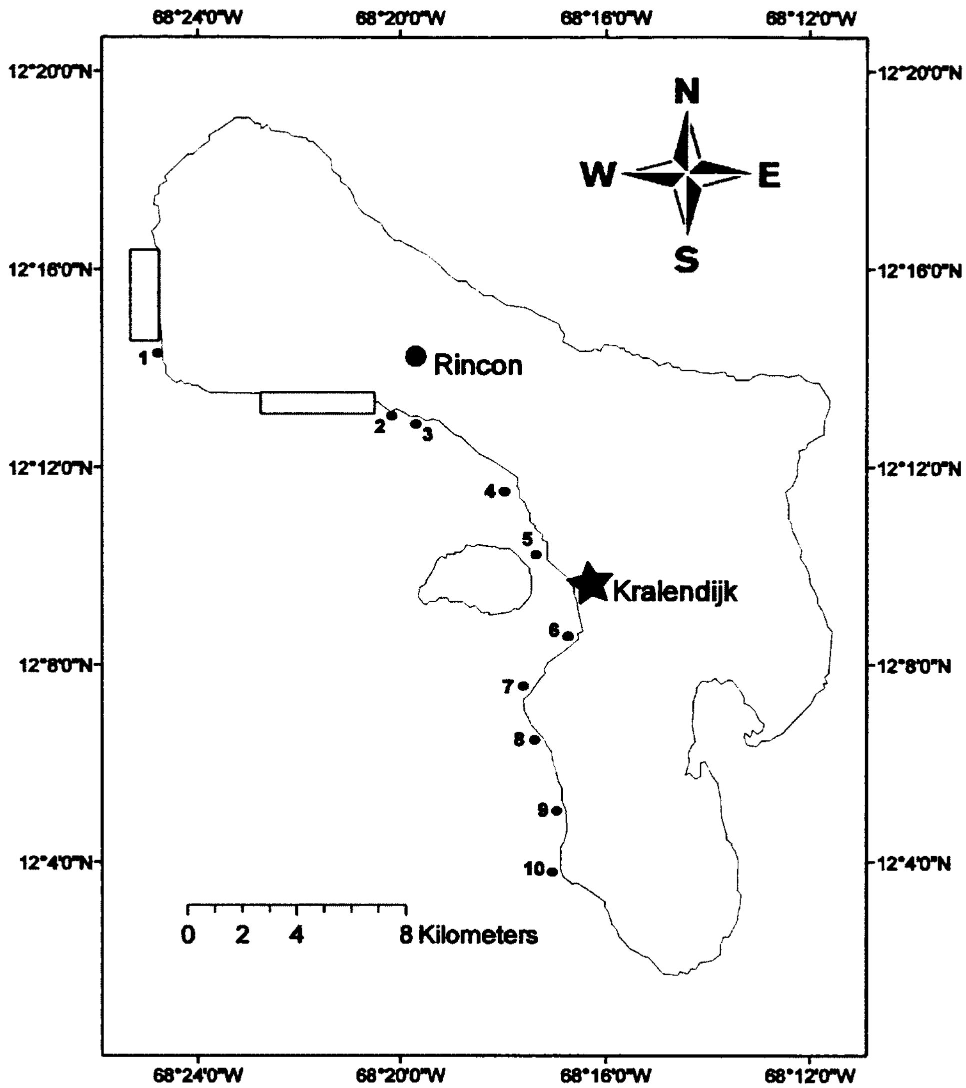

Figure 1. The island of Bonaire, Dutch Caribbean, is located in the southern Caribbean Sea approximately 80 km north of Venezuela. The Bonaire National Marine Park includes two no-diving marine reserves on the northwestern coast shown as grey rectangles. The inland city of Rincon, black circle, and the capital city of Kralendijk, black star, are depicted. Black circles along the western coast show the location of the 10 sites analyzed for coral cover and diversity.

Materials and Methods

In January 2008, underwater video of the reef was collected using scuba along shore-perpendicular transects beginning at the shoreline and extending to a depth of 20 m at 10 sites distributed on the leeward coast of the island. Divers used a Canon PowerShot SD950 IS digital camera, on video setting, in an underwater housing held perpendicular to the substrate while swimming approximately 2 m from the bottom. To record geographic location, divers towed a surface-floating Garmin GPS, (WAAS-enabled) using a caving reel. The GPS was synched in time with the video camera, as were the divers' Oceanic® dive computers, which recorded depth. Ten transects, one per dive site, which represent coverage along a north-south gradient, were analyzed for coral cover and diversity (Figure 1; Table 1).

Table 1. Names and characteristics of sampling sites shown in Figure 1. At sites marked with an asterisk, only every fourth image in the transect was analyzed, and in those without an asterisk, every image was analyzed, as discussed in the text.

Site	Transect Name	Latitude (°N)	Longitude (°W)	Coastline distance from nearest MR (km)	Coastline distance from Capital (km)	No. of Images Shoreward	No. of images Seaward
1	Nukove	12.241528	68.413056	0.49	20.49	51	18
2	Ol' Blue*	12.214811	68.337239	1.86	10.56	11	4
3	1000 Steps	12.210329	68.323082	3.68	8.64	107	28
4	Andrea II*	12.189278	68.297417	7.36	5.03	14	9
5	Buddy's Reef	12.170639	68.288576	9.64	2.76	40	26
6	18 Palms	12.138077	68.276903	13.55	1.23	30	23
7	Bachelor's Beach*	12.125706	68.287779	15.55	3.19	18	8
8	Hilma Hooker	12.104282	68.288754	18.14	5.9	103	32
9	Jeannie's Glory*	12.086361	68.283111	20.38	8.05	24	12
10	Pink Beach	12.064490	68.282003	22.87	10.54	103	57

Videos were separated into still-frame digital photographs that had a resolution of 1024 x 768 pixels and covered an area approximately 1.0 m^2. The photographs were separated within the video transect such that the upper boundary of one image represented the bottom boundary of the next image acquired. The number of images per transect varied greatly, ranging from 63 to 158, as a result of variable distances from the coastline at which a depth of 20 m was reached. For six of the 10 transects, indicated by an asterisk in Table 1, every image from a video was analyzed between 0 and 20 m depth. For the remaining four transects, only every fourth image was analyzed after determining there was no significant difference (Mann-Whitney Rank Sum Test, $U_{161,161} = 12245.5$, $p = 0.39$) between analyzing every fourth image (Mean = 17.13) and analyzing every image for coral cover (Mean = 15.68). The images were analyzed for coral cover and diversity using Coral Point Count with Excel® Extensions (CPCe; Kohler and Gill 2006). CPCe is a standalone Visual Basic program that overlays a matrix of randomly distributed points on an image. The substrate-type lying beneath each point was visually identified by the user and recorded by the program with a resolution high enough to identify coral to the species level, namely 200 points per image. CPCe automatically generates analysis spreadsheets in Microsoft Excel® based upon the supplied species and substratum codes in a given image or series of images, and computes percent coral cover and diversity, using the Shannon diversity index, in a given image or an entire transect (Kohler and Gill 2006).

Transects were divided into two sections: a portion from the shore to the reef-front drop-off (shoreward; < 5 m deep), and the remainder seaward of the reef-front (seaward; 5-20 m deep). The distinction between the two parts of each transect was based on the morphology of the reef surrounding the island; the reef-front drop-off consistently occurs around the 5 m depth contour and a visible change in cover occurs at this point. This morphology was previously described as "a submarine terrace stretching across a distance of 50-100 m from the coast to where there is an 8-12 m drop-off" (Bak 1976). Coastline distances were measured in Google Earth 5.0, using the path tool, with a caliper size (distance between path points) that varied from 20 to 200 m, depending on the curvature of the coastline.

Although this study used 10 transects representing over 700 images, these images are not considered to be completely independent samples because the presence of coral in one image is inherently related to the presence of coral in surrounding images. Although the photos within a given transect are technically spatially correlated subsamples, because the transects are shore-perpendicular, each frame represents a location further from the influence of the shore, at a deeper location, etc. Pseudoreplication is a common and nearly unavoidable problem in studies exploring geospatial trends and we felt that this problem should not preclude an analysis that used the entirety of the "signal" at each location. While the 10 transects analyzed here represent a north-south gradient along the leeward coast of the island, the farthest north site does not extend beyond the farthest northwest marine reserve due to difficulty accessing the five dive sites located along the northwest shore, within the Washington Slagbaai National Park. As with previous researchers, including van Duyl (1985), the reefs on the windward coast were not surveyed because strong wave energy prevents the development of well-structured reefs and it is very dangerous and difficult to dive there under most weather conditions. All sites represented here lie along the coast of the main island of Bonaire, whereas dive sites on the small uninhabited island of Klein Bonaire, which lies less than 1 km offshore of Kralendijk, are not considered. Although several of these represent popular dive sites, sites on Klein can only be reached by boat and each individual site receives less than 2.04% of the total dives on Bonaire (15th most popular), whereas six of the sites analyzed receive between 2.31 and 5.20% of the dives (1st-11th most popular; Thur 2003).

The high-resolution 2008 data set was analyzed using the GENMOD procedure in SAS 9.2 (SAS Institute, Inc.), which utilizes a generalized linear model (GLIM; Nelder and Wedderburn 1972) to determine the effect of reef position (shoreward *vs.* seaward), coastline distance from the marine reserves, and coastline distance from development (the capital city, Kralendijk) on total coral cover and coral species diversity. The data were not normally distributed, so a Poisson distribution was found to best fit the data using Akaike information criterion (AIC). Distance to a marine reserve was measured from the sampling location to the edge of the nearest of the two reserves. Averages and standard errors were calculated based on the percent cover and diversity of each image in a given transect on the shoreward or seaward reef.

A general linear model (GLM) in Minitab15 was used for a comparison to historical data, which was completed to determine changes on the reef over time, using the van Duyl (1985) atlas. The 2008 coral cover data were converted to van Duyl's categorical system of percent cover (henceforth referred to as "lower resolution" data), which represented series of ranges (categories: 1 = < 10%, 2 = 10-20%, 3 = 20-40% and 4 = > 40%) for each of five major coral groups: *Acropora palmata* group, *A. cervicornis* group, foliate/finger coral group, head coral group, and head/foliate coral group. After assigning the numbers 1 to 4 to each group for both the van Duyl and the data collected for this study, the categories were totaled to calculate a measure of total coral cover in the early 1980s and 2008 for each of the 10 transects analyzed. A GLM with year as the predictor variable and total coral cover and coral cover for each group as the responses was analyzed on this lower-resolution data set to test for significant differences over time at the 10 sites.

Frequency of use (%) at dive sites was taken from Thur (2003), who presented dive frequency as the percent of dives that occurred at that site, based on a survey of divers in 2001. Frequency of use was regressed against coral cover and species diversity using a linear regression in Minitab 15 to determine the relationship between dive site use and these dependent variables. Use was also compared to distance from the no-diving reserve and the capital city using a linear regression.

Results

The interaction between coastline distances from the marine reserve and development was only significant when the effect of development was taken into consideration before the interaction term (GLIM, $X^2 = 97.98$, $p < 0.0001$), but not when distance from the marine reserve was considered first (GLIM, $X^2 = 0.49$, $p = 0.48$), indicating that the effect of the marine reserve was greater than the effect of development. Live coral cover (Figure 2A.; GLIM, $X^2 = 174.55$, $p < 0.0001$) and coral diversity (Figure 2B.; GLIM, $X^2 = 55.41$, $p < 0.0001$) significantly decreased in dive sites farther removed from the marine reserve in the high resolution data set from 2008. Coastline distance from the capital city, a surrogate for development, also significantly affected coral cover (Figure 3A.GLIM, $X^2 = 4.26$, $p = 0.04$), but only in the shoreward reef (GLIM, $X^2 = 6.8$, $p = 0.01$), such that increasing distance from development was correlated with increasing coral cover. Distance from development did not significantly impact coral diversity (Figure 3B. GLIM, $X^2 = 1.13$, $p = 0.29$).

When considering the lower-resolution data set from 2008, and comparing coral cover over time since the early 1980s, position on the reef (shoreward vs. seaward) was only a significant factor when considering total coral cover in the 2008 data set (Figure 4; GLM, $F_{(16,19)} = 17.84$, $p = 0.001$), and not in the early 1980s data. Year impacted total coral cover (Figure 4; GLM, $F_{(36,39)} = 7.8$, $p = 0.008$), and specifically cover of the *A. cervicornis* group (Figure 5; GLM, $F_{(36,39)} = 63.44$, $p < 0.0005$), with significantly less cover in 2008 compared to the period of van Duyl's survey (early 1980s). The *A. palmata* group was significantly impacted by development (GLM, $F_{(36,39)} = 4.42$, $p = 0.043$), but this trend was only apparent in the early 1980s data in the shoreward reef (GLM, $F_{(16,19)} = 5.3$, $p = 0.035$) with significantly higher coral cover farthest from the capital city.

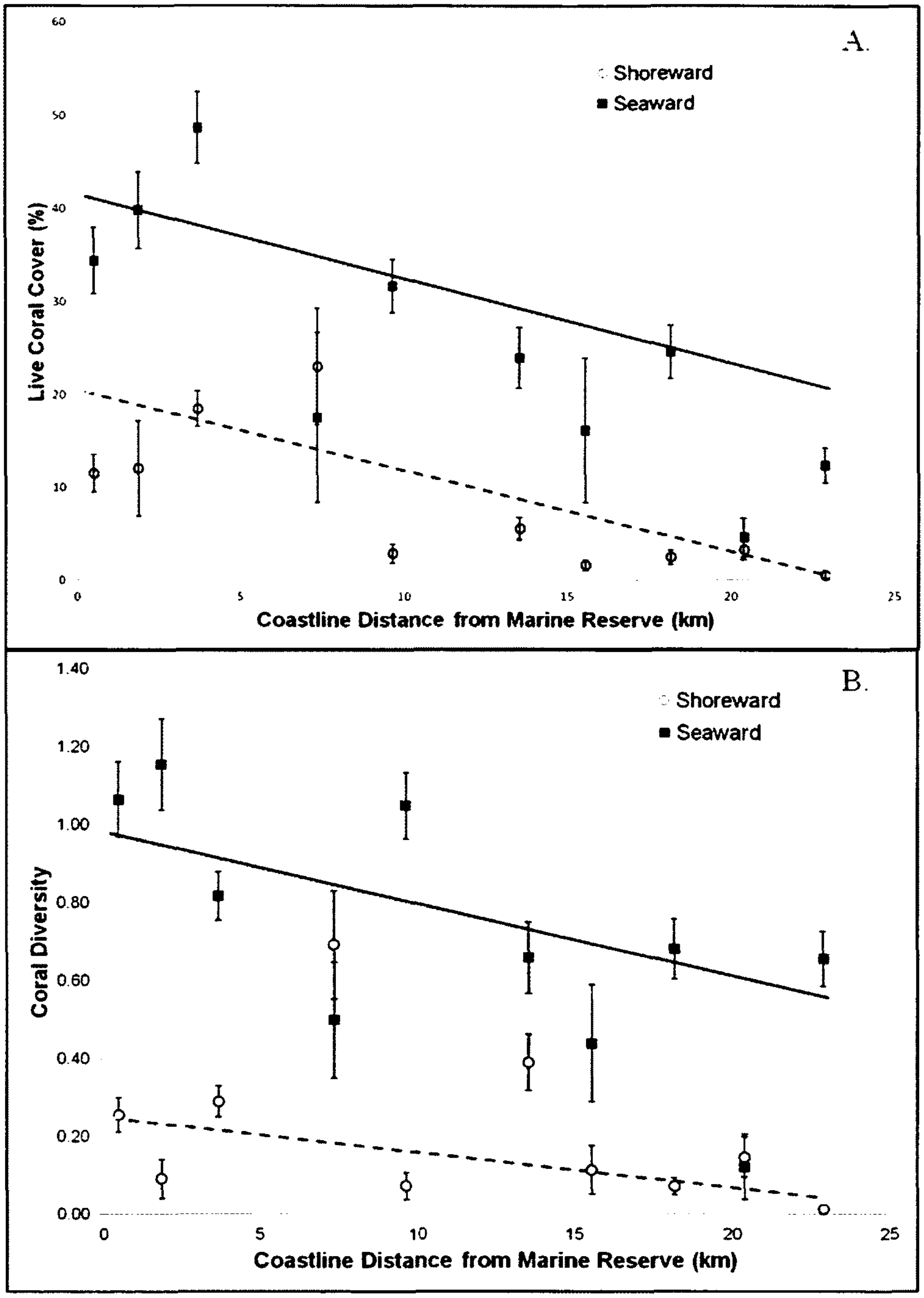

Figure 2. A. Linear regressions between percent live coral cover (y) and distance from the marine reserves (x) in the zones shoreward (< 5 m deep) and seaward (5-20 m deep) of the reef drop-off, from data collected in 2008. Points represent the geometric mean of percent cover at a given site and error bars represent the 95% confidence interval in either the shoreward or seaward reef. Both regressions show a negative relationship in the

shoreward ($\log(y) = -0.118x - 1.55$, GLIM, $X^2 = 104.74$, $p < 0.0001$) and seaward ($\log(y) = -0.054x - 0.747$, $X^2 = 71.4$, $p < 0.0001$) reef. B. Linear regressions between coral diversity, measured using the Shannon index (y), and distance (x) from the marine reserve in the zones shoreward (< 5 m deep) and seaward (5-20 m deep) of the reef drop-off, from data collected in 2008. Points represent the geometric mean of diversity at a given site and error bars represent the 95% confidence interval in either the shoreward or seaward reef. Both regressions show a negative relationship in the shoreward ($\log(y) = -0.075x - 1.024$, GLIM, $X^2 = 46.53$, $p < 0.001$) and seaward ($\log(y) = -0.025x + 0.005$, $X^2 = 18.83$, $p < 0.0001$) reef.

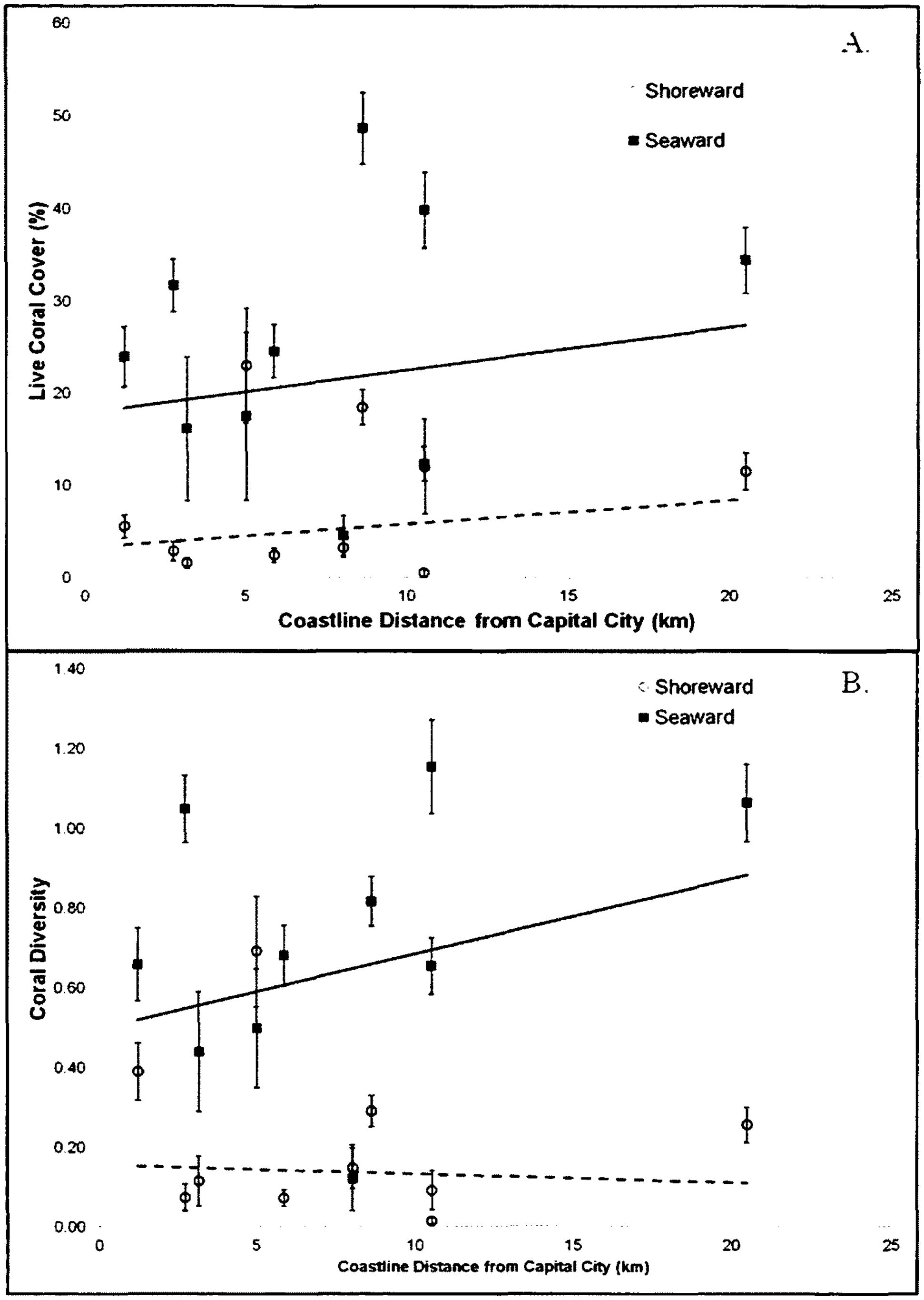

Figure 3. A. Linear regressions between percent live coral cover (y) and distance from the capital city (x) in the zones shoreward (< 5 m deep) and seaward (5-20 m deep) of the reef drop-off, from data collected in 2008. Points represent the geometric mean of percent cover at a given site and error bars represent the 95% confidence interval in either the

shoreward or seaward reef. Both regressions show a **negative** relationship in the shoreward ($\log(y) = 0.0195x + 0.532$, **GLIM, $X^2 = 104.74$, $p < 0.0001$**) and seaward ($\log(y) = 0.00847x + 1.265$, **$X^2 = 71.4$, $p < 0.0001$**) reef. B. Linear regressions between coral diversity, measured using the Shannon index (y), and distance (x) from the capital city in the zones shoreward (< 5 m deep) and seaward (5-20 m deep) of the reef drop-off, from data collected in 2008. Points represent the geometric mean of diversity at a given site and error bars represent the 95% confidence interval in either the shoreward or seaward reef. Both regressions show a **negative** relationship in the shoreward ($\log(y) = -0.00732x - 0.804$, **GLIM, $X^2 = 46.53$, $p < 0.001$**) and seaward ($\log(y) = 0.012+ 0.300$, **$X^2 = 18.83$, $p < 0.0001$**) reef.

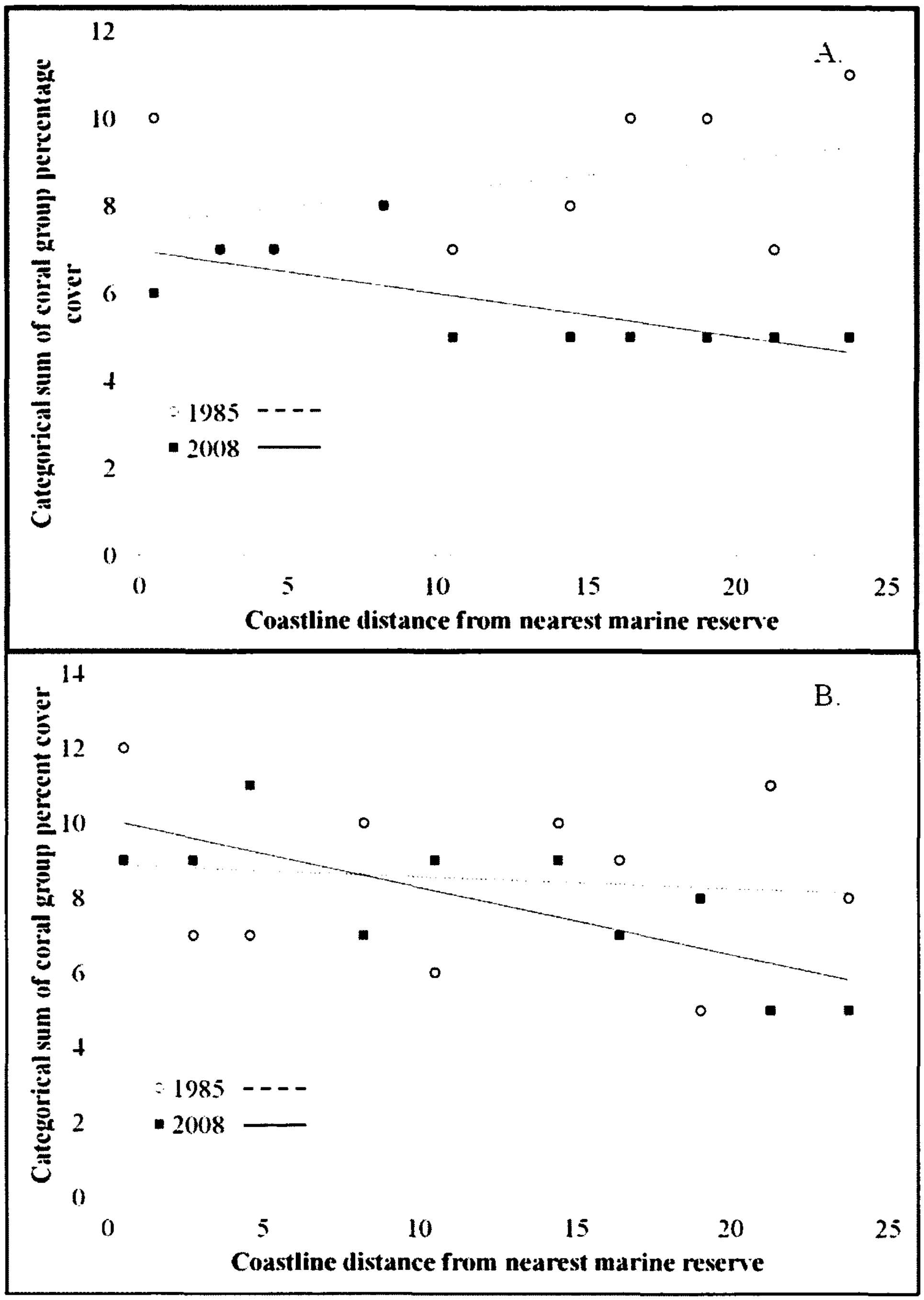

Figure 4. A. Linear regression between distance from the marine reserve (x) and coral cover (y) in the shoreward reef using van Duyl's categorical system where 1= < 10%, 2=10-20%, 3=20-40% and 4= > 40% for each of five major coral groups: *Acropora palmata* group, *A. cervicornis* group, foliate/finger coral group, head coral group, and

head/foliate coral group. The numbers 1-4 for each group were totaled to calculate an amount of coral cover for both data collected in the early 1980s ($y = 0.08x + 7.58$, Linear regression, $r^2 = 0.15$, $F_{1,8} = 1.32$, $p = 0.283$) and 2008 ($y = -0.10x + 6.98$, $r^2 = 0.48$, $F_{1,8} = 7.44$, $p = 0.026$). Year was a significant factor when considering live coral cover in the shoreward reef (GLM, $F_{(1,18)} = 19.24$, $p < 0.0005$). B. linear regressions in the seaward reef between distance from the marine reserve (x) and coral cover (y) in the seaward reef using van Duyl's categories. The categories 1-4 for each group were totaled to calculate regressions for both early 1980s ($y = -0.07x + 9.65$, Linear regression, $r^2 = 0.06$, $F_{1,8} = 0.10$, $p = 0.757$) and 2008 ($y = -0.18x + 10.21$, Linear regression, $r^2 = 0.54$, $F_{1,8} = 10.93$, $p = 0.011$). Year (early 1980s *vs.* 2008) was not a significant factor when considering live coral cover in the seaward reef (GLM, $F_{(1,18)} = 0.41$, $p = 0.531$).

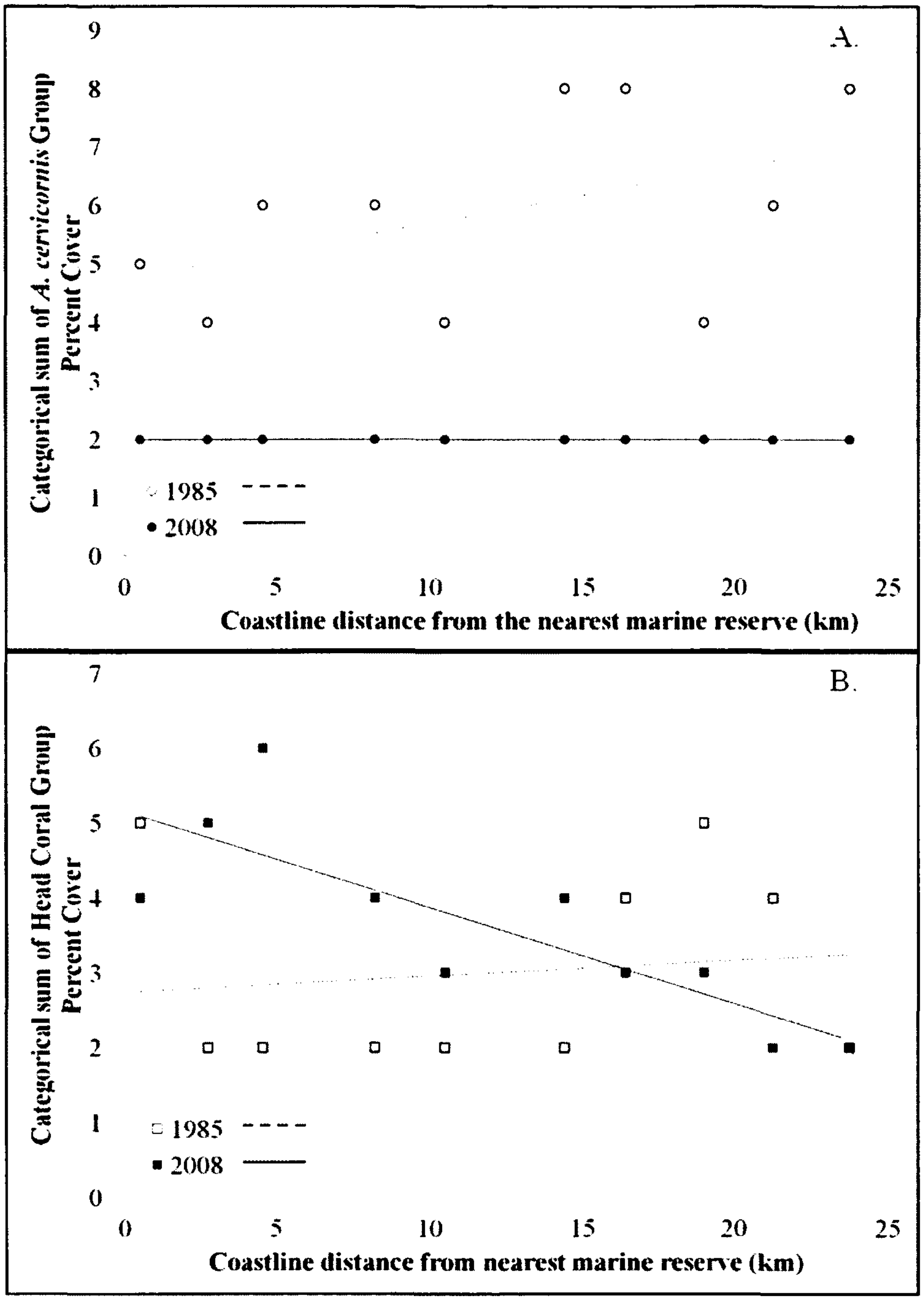

Figure 5. A. Linear regression between distance from the marine reserve (x) and *Acropora cervicornis* cover (y) in the early 1980s (dotted line; y = 0.10x + 4.74, Linear regression, $r^2 = 0.22$, $F_{1,8} = 2.21$, $P = 0.175$) and 2008 (solid line: y = 2.00). Year was a significant factor in this group (ANOVA, $F_{(16,19)} = 17.52$, $P = 0.001$). B. Linear

regression between distance from the marine reserve (x) and head coral group cover (y) in the early 1980s ($y = -0.02x + 2.74$, $r^2 = 0.02$, $F_{1,8} = 0.13$, $P = 0.723$) and 2008 ($y = -0.13x + 5.14$, $r^2 = 0.674$, $F_{1,8} = 16.54$, $P = 0.004$). Year was also significant in this group (Figure 4; ANOVA, $F_{(16,19)} = 7.1$, $P = 0.017$).

Using the lower-resolution 2008 data set to compare to van Duyl (1985), distance from the marine reserve was significantly related to total coral cover (Figure 4; GLM, $F_{(16,19)} = 19.56$, $p < 0.0005$), with increasing cover closer to the reserve, which was not the case in the early 1980s (Figure 4; GLM, $F_{(16,19)} = 1.23$, $p = 0.284$). The significance of position on the reef (shoreward vs. seaward), and distance from the reserve on coral cover in 2008 was significant only in the head coral group (Figure 5; GLM, $F_{(16,19)} = 16.11$, $p = 0.0001$; $F_{(16,19)} = 16.35$, $p = 0.0001$, respectively).

In the shoreward reef (< 5 m), year significantly impacted total coral cover, with an overall decrease in total coral cover between the early 1980s and 2008 (Figure 4A; GLM, $F_{(16,19)} = 18.36$, $p = 0.001$). Most of the decline seen over time in the shoreward reef was in the cover of the *Acropora cervicornis* group (Figure 5A.; GLM, $F_{(16,19)} = 76.63$, $p < 0.0005$). In the seaward reef (> 5 m), distance from the marine reserve had a significant effect on the head/foliate coral group (data not shown; GLM, $F_{(16,19)} = 7.09$, $p = 0.017$), with more live cover of this group closer to the reserve. Year, early 1980s *vs.* 2008, significantly impacted the seaward reef, showing declines in the *A. cervicornis* group (Figure 5A.; GLM, $F_{(16,19)} = 17.52$, $p = 0.001$) and increases in the head coral group (Figure 5B.; $F_{(16,19)} = 7.1$, $p = 0.017$).

Frequency of use by divers was not correlated with coral cover in the shoreward (regression, $F_{(8,9)} = 0.00$, $p = 0.0.978$) or seaward (regression, $F_{(8,9)} = 1.96$, $p = 0.199$) reef. Use also did not affect coral species diversity in the shoreward (regression, $F_{(8,9)} = 0.35$, $p = 0.568$) or seaward (regression, $F_{(8,9)} = 1.43$, $p = 0.266$) reef. Use was also not correlated with how far a site was from the marine reserve (regression, $F_{(8,9)} = 0.02$, p =

0.893) or the capital city (regression, $F_{(8,9)} = 1.87$, $p = 0.209$) in either the shoreward or seaward reef position.

Discussion

The current study represents a unique analysis of coral cover over time, before and after the establishment of marine reserves, which are closed to underwater visitors. In 2008, higher coral cover and diversity were both correlated with increasing proximity to these marine reserves. Coral cover at the same locations prior to reserve establishment did not historically show a similar trend, particularly in the shoreward reef (< 5 m depth; Figure 3). The most substantial decline in coral was in the *Acropora cervicornis* group, with nine out of 10 shoreward sites and seven out of 10 seaward sites experiencing decreases in *A. cervicornis* cover over time. Declines in *A. cervicornis* were accompanied by increases in the head coral group in three shoreward sites and six seaward sites and increases in the head/foliate coral group in four shoreward sites and five seaward sites. Hawkins et al. (1999) found lower coral cover, but higher diversity close to moorings used by divers, whereas this study found a positive correlation between cover and diversity. Changes in reef composition off the coast of Bonaire have shown a shift from dominance by *Acropora* spp. to dominance by larger head corals, particularly *Montastraea* spp.

Urbanization and development were negatively correlated with live coral cover in the shoreward reef (< 5 m), but did not seem to impact cover in the seaward reef (> 5 m) or coral diversity in either portions of the reef. This is likely due to a combination of factors, including proximity to runoff in the shoreward reef, which can lead to turbid water and sedimentation, inhibiting photosynthesis by zooxanthellae, and also brings in nutrients, which contribute to algal overgrowth. Hurricanes are also well-known to significantly impact shallow reefs (Michot et al. 2002); although Bonaire is considered to be well-south of the hurricane belt, hurricanes Ivan and Felix impacted the island of Bonaire in 2004 and 2007, respectively, as did Hurricane Lenny in 1999 (Bries et al. 2004). Evidence of storm damage to *Acropora cervicornis* is evident to snorkelers and divers as coral rubble of this species litters the sediment in the shallower reef. While *Acropora* spp. was dominant in the shallow reef of Bonaire, and throughout the Caribbean, when van Duyl (1985) collected data in the early 1980s, a subsequent dieoff of acroporids occurred after the outbreak of white band disease, also in the early 1980s (Rosenburg 2004).

Proximity to the no-diving marine reserves was most strongly correlated with the metrics of reef health, coral cover, and diversity. This implies that underwater visitors throughout the rest of the marine park might be negatively impacting the reef structure. However, data collection by van Duyl (1985) was conducted in the early 1980s, coincident with the die-off of the sea urchin *Diadema antillarum*, an important herbivore on reefs throughout the Caribbean. In the absence of large herbivorous fish, e.g., as a result of overfishing, *Diadema antillarum* had come to fill the ecological niche of dominant herbivore throughout most of the Caribbean. A disease outbreak in the 1980s killed off large populations of *Diadema antillarum*, the result of which was overgrowth of coral by macroalgae in the absence of herbivores (Bellwood et al., 2004). Bonaire was not immune to the urchin dieoff, but significant populations of herbivorous fish kept macroalgal overgrowth from negatively impacting its reefs (Steneck et al. 2007), as occurred elsewhere in the Caribbean (Sotka and Hay 2009). The grazing sea urchin *D. antillarum* remains relatively rare (0.03 urchins m^{-2}) and thus has little or no functional importance as an herbivore on Bonaire at this time (Steneck et al. 2005). While the present study found a low correlation between coral cover and diversity with proximity to the capital city, which was used to represent development, and therefore urbanization and potential input of nutrients, it is important to note that long-term monitoring of Bonaire's near-shore nutrients is ongoing and there is concern that levels are close to thresholds that may predict future reef degradation (Wieggers 2007).

Although the Bonaire National Marine Park represents one of the longest-established examples of marine protection, it highlights a recurring problem in the ambiguity of what defines a marine protected area (MPA) or a marine reserve (Agardy et al. 2003). Currently the most commonly used definition of an MPA is provided by the International Union for Conservation of Nature (IUCN, 1994): "any area of inter-tidal or sub-tidal terrain, together with its overlying water and associated flora, fauna, historical, or cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (Kelleher and Kenchinton 1992). The Scientific Consensus Statement on Marine Reserves and Marine Protected Areas (National Center for Ecological Analysis, 2001) distinguishes between marine reserves and marine protected areas, the former being exclusive of all fishing, disruptive or extractive use (with the exception of scientific research), and the latter referring to multiple-use areas with mixed harvest, restricted harvest, and/or complete prohibition areas (Agardy et al. 2003). Although the BNMP considers the two no-diving areas to be marine reserves, based on this definition and because fishing is only limited to traditional methods within the reserves, rather than banned completely, Bonaire's reserves do not represent true marine reserves.

While this research is far from conclusive, given the increases in diving tourism since Dixon et al. (1993) and its proposed critical dive threshold, it is likely that Bonaire is surpassing the suggested limit of 4,000 to 6,000 dives per site per year and will soon show signs of decreased reef quality as a result. This fact, combined with the correlation found in this study between coral cover and diversity and proximity to the no-diving reserves highlights the importance of management to all threats to the reef structure, including those that are generally considered non-destructive, such as regulated sport diving. Surprisingly, frequency of use by divers was not correlated with coral cover, coral species diversity, proximity to the no-diving reserve or proximity to the capital city. This suggests that some mechanism other than, or more likely in addition to, frequency of use by divers is negatively impacting coral cover and diversity, particularly along the southern leeward coast of the island. This finding also suggests that choice of dive site by divers is driven by some factor other than coral cover and diversity, perhaps by ease of entry, presence of large schools of reef fish, or some other unique feature of a site as in the Hilma Hooker shipwreck, which is the most popular dive site on the island of Bonaire (Thur 2003).

The data presented here represent a unique study of coral cover over a nearly quarter century time scale, during which marine protection was variable. While the results of this study are by no means conclusive, they can be used by managers of the Bonaire National Marine Park (BNMP) to focus on areas of concern and adjust management priorities accordingly, particularly toward collection of new data, which may complement the findings reported here. Although the BNMP's marine reserves differ from those defined by the National Center for Ecological Analysis (2001) described above, this study represents a unique analysis of a marine protected area area before and after its establishment. The discrepancy of definitions here highlights the importance of standards to define areas of varying level of marine protection.

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CHAPTER 2

Creating landscape-scale maps of coral reef cover for marine reserve management from high resolution multispectral remote sensing

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Abstract

New methods are needed for the study of coral reefs as they are changing rapidly. Satellite remote sensing has become a common method for benthic mapping with advances in satellites and sensors and as methods are devised to account for atmospheric and water column effects. Images from the QuickBird satellite have proven useful in reef mapping. Sand was distinguished from coral with an overall accuracy of 75% on the shallow reef off of Bonaire. Coral and sand had user accuracies of 50% and 90%, respectively. Increased samples of field-collected data would further increase the accuracies of such classifications.

Introduction

Coral reefs are critical ecosystems, home to the largest concentrations of biodiversity in the ocean (Roberts et al. 2002). However, coral organisms of the order Scleractinia, which create the underlying reef structure, are specialists, with a narrow range of temperature and depth tolerances, leaving them susceptible to threats associated with global climate change (Hughes et al. 2003). In addition, increases in destructive physical processes, such as ocean acidification, hurricanes and direct human activities, weaken the reef matrix. Weakening of the reef helps contribute to a competitive advantage to macroalgae, which can overgrow the slow-growing coral organism, leading to a phase shift to a less-favorable, algae-dominated, habitat structure (Hughes 1994). Destructive processes often lead to reef habitat loss, resulting in fragmentation of the remaining habitat (Jones and Syms 1998), which can presumably lead to reductions in dispersal of gametes and larvae between smaller, more isolated patches of reef.

Percent cover is frequently used as a metric of reef health with healthy reefs exhibiting greater than 50% live coral cover (Gardner et al. 2003). Quadrat counts of coral percent cover are often extrapolated to assess the health of large reef areas. To map larger reef areas, aerial photographs are often used, which requires mosaicing multiple photographs that can be quite expensive to acquire. Mumby et al. (2004) had some success calculating percent live cover of coral from airborne remote sensing, but Mishra et al. (2005; 2006a) concluded that further processing (i.e., water column correction) to remotely sensed images is necessary for live coral cover mapping. More recently, Autonomous Underwater Vehicles (AUVs) have been used in reef mapping, which can potentially be a more cost-effective method than either aerial photography or the quadrat method, which requires extensive diver surveys (Patterson and Relles 2008). New methods are being developed to increase the time- and cost-efficiency of acquiring reef habitat maps; this is especially important when the reef system is changing very rapidly as a result of multiple high-frequency stressors.

Satellite remote sensing has typically been used for mapping of surface ocean parameters such as sea surface temperature and chlorophyll-*a* (Lagerloef et al. 1995) and is now being applied to benthic habitat mapping (Mumby et al. 1997; Mishra et al. 2006b). Remote sensing techniques have been applied to coral reef management for nearly four decades (Smith et al. 1975) and remote sensing derived maps of coral reefs can reveal patterns in substratum type, but not generally in coral species (Green et al. 2000). Mumby et al. (1997) compared the effectiveness of mapping using images acquired from four different satellites and one hybrid satellite against aerial photography and compact airborne spectrographic imager (CASI) imagery at various levels of resolution and found that for coarse (4 habitat classes) and intermediate (6 classes) resolution, satellite sensors compared favorably to aerial photography. The most widely used satellite sensor, Landsat TM, was found to be more accurate than aerial photography. However, at fine-scale resolution (nine habitat classes), aerial photography was more accurate than all satellite sensors (Mumby et al. 1997).

High-resolution multispectral data from the QuickBird (QB) satellite is highly suited to creating maps of tropical benthic community type (seagreass, sand, algae, and coral) after appropriate atmospheric and water column corrections (Mishra et al. 2006b). The QB sensor system has three visible bands centered at 485 nm (blue), 560 nm (green), and 660 nm (red), which can be used for shallow marine applications, and a near infrared band (830 nm) (Mishra et al. 2006b). Classification algorithms, such as the Iterative Self Organizing Data algorithm (ISODATA; Jensen 2005), are able to distinguish between these different benthic habitat types based on corrected seabed reflectance in the four spectral bands (Mishra et al. 2006b). Despite the higher resolution of QB, this method has been unsuccessful at distinguishing between coral species. However, this method has numerous advantages in extent, speed, and cost of data acquisition over other methods (e.g. towed cameras and divers with underwater video). With suitable background data, large areas can be analyzed relatively quickly in remote locations without necessitating travel and requiring very little equipment.

Mishra et al. (2006b) assessed the utility of the higher spatial resolution of the QB data for mapping benthic communities and found improved classification accuracies over studies utilizing other commercial sensors. This study applied Mishra et al. (2006b) atmospheric and water column correction procedures used on Roatan Island Honduras, covering less than 10 km of coastline, to a unique site of larger geographic extent in order to produce an up-to-date map of coral cover along the leeward coast of the island of Bonaire, Dutch Caribbean. The resulting map will be useful for assessing the efficacy of the marine protected area (MPA) that surrounds the island of Bonaire.

Materials and Methods

Study Site

The island of Bonaire, Dutch Caribbean (Figure 1), is located approximately 80 km north of the coast of Venezuela and is a part of the ABC island chain, which includes the islands of Aruba and Curaçao. Bonaire is a relatively small island, with a land area of only 294 km^2, 32 km long and 11 km wide at its widest point. The population of Bonaire is around 15,000 (Bonaire government). It also includes the small, uninhabited island of Klein Bonaire to the west as can be seen in Figure 1. The reefs surrounding Klein Bonaire and off the leeward coast of the main island were extensively mapped in the early 1980s into dominant community-type and percent cover out to 10 m deep (van Duyl 1985). Bonaire represents a long-established MPA, with various levels of protection beginning in 1979 and subsequently increasing until the entire island was protected from the shoreline to the 60 m depth contour. The park is managed by the non-governmental, not for profit foundation Stichting Nationale Parken Bonaire (STINAPA Bonaire), which enforces the laws of the park, provides information, education and outreach, and also conducts monitoring and research in the park waters.

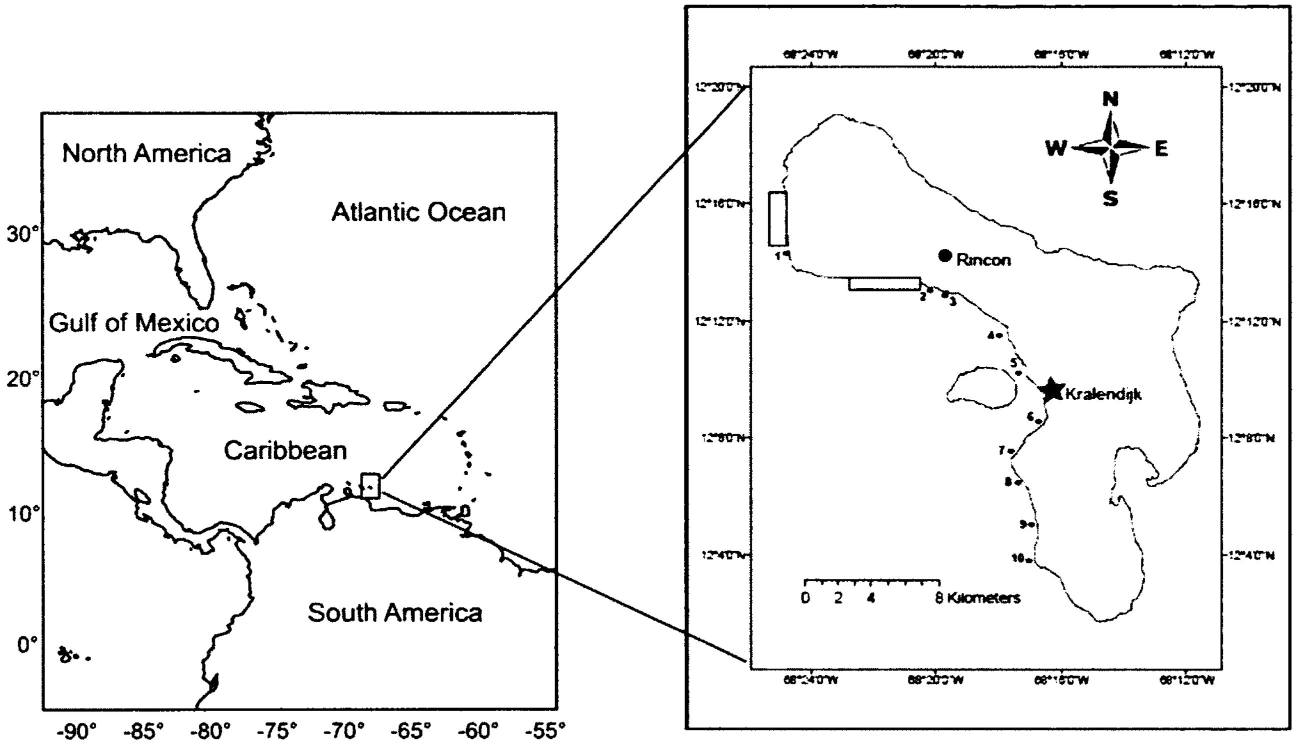

Figure 1. The island of Bonaire, Dutch Caribbean (left), is located in the southern Caribbean Sea (right). The capital city and main port, Kralendijk, is signified by the black star, while the only other major city, Rincon, is signified by the black circle. Numbered points along the coast indicate areas where video data was collected (Table 1). Gray rectangles indicate the location of no-entry marine reserves.

In Situ Data

In situ data was collected off the leeward coast of Bonaire. Underwater video was collected by SCUBA divers, swimming shore-perpendicular at approximately two meters above the bottom using a Canon PowerShot SD950 IS digital camera in an underwater housing on video setting. To record geographic location, divers used a surface-floating Garmin eTrex Navigator GPS on a reel, with WAAS enabled, synchronized in time with the video camera and the divers' Oceanic® Pro Plus 2 dive computers, which recorded depth to the nearest half meter. Videos were separated into still-frame digital photographs with a resolution of 1024 x 768 pixels, each measuring approximately one square meter. The photographs were separated within the video transect such that the upper boundary of one image represented the bottom boundary of the preceeding image. The resolution of the images was sufficient for species-level identification of corals and other macroinvertebrates. The images were analyzed for coral cover and diversity using Coral Point Count with Excel® Extensions (Figure 2; CPCe; Kohler and Gill 2006). CPCe is a standalone Visual Basic program that overlays a matrix of randomly distributed points onto the photograph. The substratum type or species lying beneath each point was then visually identified by the user and recorded by the program. CPCe automatically generates analysis spreadsheets in Microsoft Excel® based upon the supplied species and substratum codes in a given image or series of images, and computes percent coral cover (Kohler and Gill 2006; Table 1). Two hundred points per image were used in all analyses (Relles and Patterson Chapter 1). The resulting spreadsheet, which calculated percent coral cover, was fused with GPS and depth data to determine depth and geographic location for the analyzed images using Eonfusion (Myriax Pty. Ltd.).

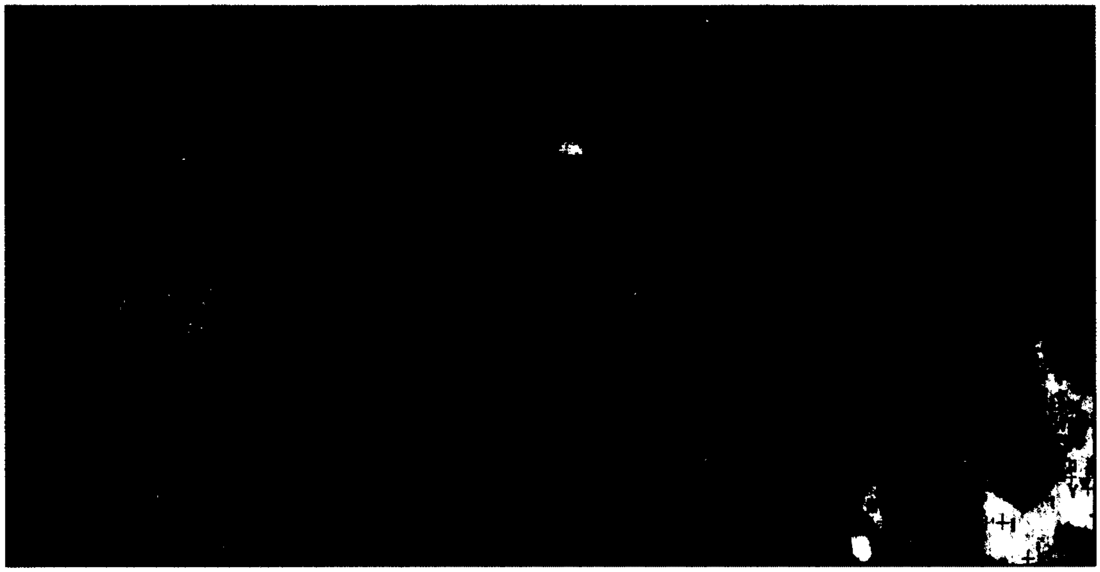

Figure 2. Still-frame digital photographs grabbed from video with a resolution of 1024 x 768 pixels, approximately one meter by one meter squared, overlain with 200 random points. The image on the left represents an area of 40.4% live coral cover and 58.6% sand, pavement or rubble. The image on the right, taken from a deeper depth, represents an area of 29.4% live coral, 48.9% dead coral with algae and 15.2% sand, pavement, or rubble.

Table 1. Site numbers from Figure 1 with site names, geographic location and total number of images analyzed per site. Average percent coral and sand cover over all the images from a site are given.

Site #	Site Name	Latitude (°N)	Longitude (°W)	Total number of images analyzed	Coral cover (%)	Sand cover (%)
1	Nukove	12.241528	68.413056	69	17.53	6.06
2	Ol' Blue	12.214811	68.337239	15	19.46	38.90
3	1000 Steps	12.210329	68.323082	135	24.58	35.56
4	Andrea II	12.189278	68.297417	23	20.87	23.22
5	Buddy's Reef	12.170639	68.288576	66	14.21	52.95
6	18 Palms	12.138077	68.276903	53	13.54	14.94
7	Bachelor's Beach	12.125706	68.287779	26	6.07	45.56
8	Hilma Hooker	12.104282	68.288754	135	7.71	56.89
9	Jeannie's Glory	12.086361	68.283111	36	3.69	53.87
10	Pink Beach	12.064490	68.282003	160	4.66	44.56

QuickBird Satellite Imagery

Three QuickBird (QB) multispectral images were acquired (DigitalGlobe Inc.) for the island of Bonaire. The images used represent the most recent QB satellite images available at the time the study took place; the northwest coast image was collected on 10^{th} December 2008, the central west coast, including the island of Klein Bonaire, was collected on 10^{th} January 2009 and the image of the southwest coast and southern tip was collected on 6^{th} September 2008. Little change is assumed to have occurred during the four-month period between acquisition times. Tidal stage at the time of acquisition was insignificant; time of day was taken into consideration as the mean sun elevation angle reported in the QB metadata (meanSunEl) and used to calculate the solar zenith angle (θ_0).

QB high-resolution satellite images are delivered as radiometrically corrected image pixels (eMap International). Corrected counts are specific to the QB instrument, so the imagery must be converted to spectral radiance before radiometric/spectral analysis (Krause 2003). Conversion required multiplying the radiometrically corrected image pixels by the absolute radiometric calibration factors (referred to in the QB metadata as absCalFactor), which vary by band and are delivered with every QB product as part of the image metadata files (extension .IMD). The absCalFactor for each band was multiplied by every pixel in that band of the image so that the images represented radiance values.

When extracting aquatic information, it is useful to eliminate all upland and terrestrial features, as well as boats, piers, and clouds (Jensen et al. 1991). Working in the NIR band (band 4), a binary mask was created and then applied to all four QB channels. The resulting "land-mask" restricts the spectral range of brightness values to the aquatic features of interest and allows for more detailed feature discrimination. While clouds were successfully masked out as a result of their brightness, cloud shadows on the water had radiance values similar to the surface of the water and were not possible to remove. However, most cloud shadows occurred in deeper water (> 30 m), whereas habitat mapping was restricted to areas less than 10 m deep.

First-order atmospheric correction

Once the QB images were converted to spectral radiance and all irrelevant features were removed using the binary mask, a first-order atmospheric correction based on Mishra et al. (2005) was performed to remove the contributions of Rayleigh and aerosol scattering to the total radiance. In a single scattering approach, the radiance received by the sensor at the top of the atmosphere (L_t) in a spectral band centered at a wavelength λ_i can be divided into the following components (Gordon et al. 1983):

$$L_t(\lambda_i) = L_r(\lambda_i) + L_a(\lambda_i) + T(\lambda_i)L_g(\lambda_i) + t(\lambda_i)L_w(\lambda_i) \qquad (1)$$

where,

$L_r(\lambda_i)$ and $L_a(\lambda_i)$ represent the radiances generated along the optical path in the atmosphere by Rayleigh and aerosol scattering, respectively;

$L_g(\lambda_i)$ is the sun-glint component;

$L_w(\lambda_i)$ is the desired water-leaving radiance;

T is direct atmospheric transmittance; and

t is diffuse atmospheric transmittance of the atmosphere.

According to Gordon and Voss (1999), for areas around the sun-glint pattern, $T(\lambda_i)L_g(\lambda_i)$ is so large that the imagery is virtually useless and must be discarded (Mishra et al. 2005). The QB images have negligible sun-glint effects, so $T(\lambda_i)L_g(\lambda_i)$ was removed from the equation and Rayleigh and aerosol scattering components were approximated for the first-order atmospheric correction (Mishra et al. 2006b). Consequently, Equation (1) can be written as:

$$L_t(\lambda_i) = L_r(\lambda_i) + L_a(\lambda_i) + t(\lambda_i)\, L_w(\lambda_i) \qquad (2)$$

Rayleigh scattering is the elastic scattering of light or other electromagnetic radiation by particles much smaller than the wavelength of the light, which may be individual atoms or molecules. It takes place in the atmosphere 2 to 8 km above the ground and is also referred to as molecular scattering (Jensen 2005). In calculations for Rayleigh scattering, $L_r(\lambda_i)$ is equal to the Rayleigh path radiance and $t(\lambda_i)$ is equal to the diffuse transmittance for all input channels (bands). The input channels, or bands, for the QB satellite are equal to 0.485, 0.560, 0.660 and 0.830 μm.

Rayleigh Path Radiance

Rayleigh path radiance can be computed using Gordon and Clark (1981):

$$L_r(\lambda_i) = (F_0'(\lambda_i)\ \omega_{0Y} \tau_Y P_Y) \ / \ 4\pi \cos\theta_v \qquad (3)$$

where,

$F_0'(\lambda)$ is the instantaneous extraterrestrial solar irradiance, $F_0(\lambda)$, which is reduced by two trips through the ozone layer and is computed by:

$$F_0'(\lambda) = F_0(\lambda) * e^\wedge(\tau_{Oz}(1/\cos\theta_v + 1/\cos\theta_0)) \qquad (4)$$

$F_0(\lambda)$ values are taken from Nickel and Labs (1984).

ω_0 is single scattering albedo (equal to 1);

τ_Y is Rayleigh optical thickness;

τ_{Oz} is Ozone optical thickness;

P_Y is Rayleigh scattering phase function;

θ_v is the satellite viewing zenith angle (OffNadirViewAngle in QB metadata); and

θ_0 is the solar zenith angle (equal to 90 degrees minus the meanSunEl in QB metadata).

The value of Rayleigh optical thickness (τ_Y) at any atmospheric pressure P is given by Hansen and Travis (1974):

$$\tau_Y = (P/P_0)\,[\,0.008569\lambda^{-4} * (1 + 0.0113\lambda^{-2} + 0.00013\lambda^{-4})\,] \qquad (5)$$

where,

λ = wavelength in µm; and

P_0 = standard atmospheric pressure of 1013.25 mb

P was local ground pressure on the dates that the satellite images were taken and was found online for Bonaire's Flamingo airport (El Tiempo 2011).

The Rayleigh scattering phase function accounts for the direct scattered light and the scattered light that is specularly reflected at the sea/air interface (Doerffer 1992). The Rayleigh scattering phase function is given by Doerffer (1992):

$$P_Y(\theta_\pm) = \tfrac{3}{4}\,(1 + \cos^2\theta_\pm) \qquad (6)$$

where,

$\theta_\pm$ is the forward/backward scattering angle and it is related to the sensor viewing and solar illumination directions through:

$$\cos\theta_\pm = \pm\cos\theta_0 \cos\theta_v - \sin\theta_0 \sin\theta_v \cos(\varphi_v - \varphi_0) \qquad (7)$$

where,

θ_0 is the solar zenith angle (equal to 90 degrees minus the meanSunEl in QB metadata);

φ_0 is the solar azimuth angle (meanSunAz in QB metadata);

θ_v is the satellite viewing zenith angle (OffNadirViewAngle in QB metadata); and

φ_v is the satellite viewing azimuth angle (meanSatAz in QB metadata).

A diagram of the relevant sun and satellite angles with respect to the pixel [image] of interest is shown in Figure 3.

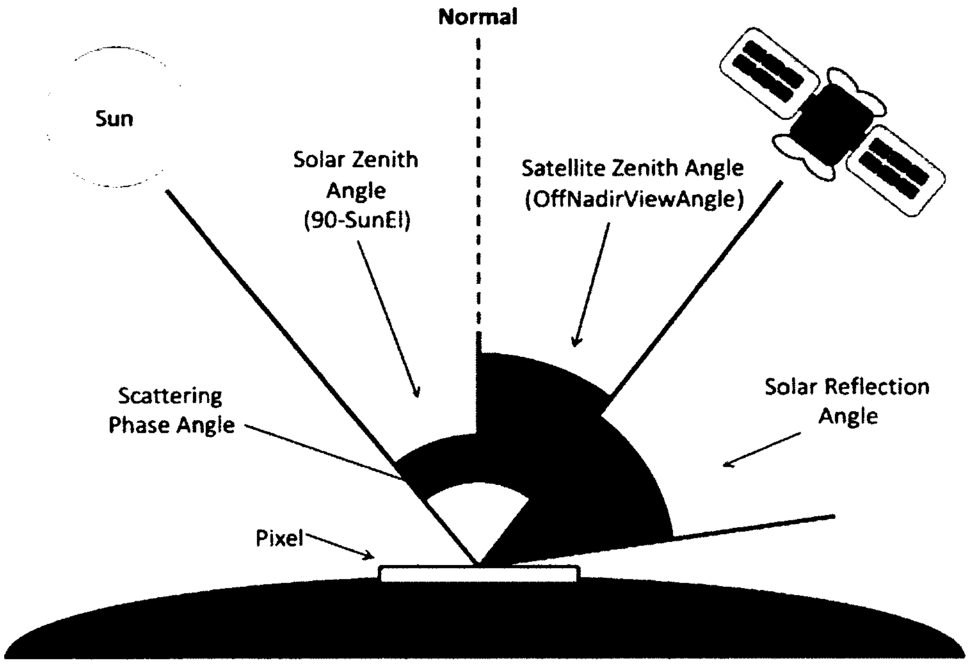

Figure 3. Relevant satellite and sun angles for calculating Rayleigh path radiance ($L_r(\lambda_i)$).

Aerosol Path Radiance

Aerosol scattering is the scattering of radiative energy by processes at the aerosol and molecular level, by particles larger than the wavelength of light, which takes place in the lower 4.5 km of the atmosphere and is also referred to as Mie or particle scattering (Jensen 2005); the variable L_a represents aerosol scattering. In the NIR band (band 4), optically deep, clear water pixels have no contribution from bottom reflectance or other reflectance, such as from suspended sediment or chlorophyll-*a* in the water column. The value of these pixels is close to zero, owing to absorption by water, and any difference from zero is a result of aerosol scattering. The contribution of aerosol scattering for these pixels was computed as:

$$L^0_a(\lambda_4) = L^0_t(\lambda_4) - L^0_r(\lambda_4) \qquad (8)$$

where,

$L^0_a(\lambda_4)$ is the contribution of aerosol scattering over the clear water pixels at band 4;

$L^0_t(\lambda_4)$ is the total radiance observed over the clear water pixel at band 4; and

$L^0_r(\lambda_4)$ is the contribution of Rayleigh scattering over the clear water pixels at band 4.

Aerosol path radiance at any wavelength, $L_a(\lambda_i)$, can be calculated from $L^0_a(\lambda_4)$ and is given as:

$$L_a(\lambda_i) = S(\lambda_i, \lambda_4)\, L^0{}_a(\lambda_4) \qquad (9)$$

where,

S is a ratio related to the optical properties of the aerosol through:

$$S(\lambda_i, \lambda_4) = \varepsilon(\lambda_i, \lambda_4) * (F_0'(\lambda_i) / F_0'(\lambda_4)) \qquad (10)$$

Averaging the value of the pixels over a significant area (81 pixels) gave an L_a value for the entire image. Although L_a is technically wavelength dependent, it was assumed that the aerosols were homogenously distributed over the entire area of interest as in Mishra et al. (2005), such that $L^0{}_a(\lambda_4)$, computed over the clear water is assumed to be constant over the entire scene. Hence, the aerosol path radiance for the three visible bands was computed using equation (9) as well.

Diffuse transmittance

Diffuse transmittance ($t(\lambda_i)$) is defined as the water leaving radiance in a particular viewing direction (θ_0, θ_v) transmitted to the top of the atmosphere and is given by:

$$t(\lambda i) = exp\,[\, - ((\tau_r(\lambda)/2) + (\tau_{O_3}(\lambda)) * ((1 / cos\theta_0) + (1 / cos\theta_v))\,]$$

(11)

where,

$\tau_r\,(\lambda)$ is Rayleigh optical thickness; and

$\tau_{O_3}\,(\lambda)$ is ozone optical depth, and ozone optical depth for a concentration of DU (Dobson units or milli-atmosphere centimeters) at the latitude and longitude of the image on the date the image was collected, which were determined using data from the Ozone Monitoring Instrument (OMI) on the Aura spacecraft (NASA 2009a) and is given by:

$$\tau_{O_3}\,(\lambda) = k_{O_3}\,(\lambda)\ (DU/1000) \qquad (12)$$

where,

$k_{O_3}(\lambda)$ is ozone absorption coefficient taken from Leckner (1978).

The desired water-leaving radiance ($L_w\,(\lambda_i)$) at a specific wavelength was computed by rewriting equation (2) as:

$$L_w\,(\lambda_i) = [\,L_t\,(\lambda_i) - L_r\,(\lambda_i) - L_a\,(\lambda_i)\,] \,/\, t\,(\lambda_i) \qquad (13)$$

Once the Rayleigh path radiance ($L_r(\lambda_i)$), aerosol scattering ($L_a(\lambda_i)$), and diffuse transmittance ($t\,(\lambda_i)$) were calculated for each of the four wavelengths of the QB image, all three were subtracted from each pixel of each band of the image. An example of an image before and after atmospheric correction can be seen in Figure 4.

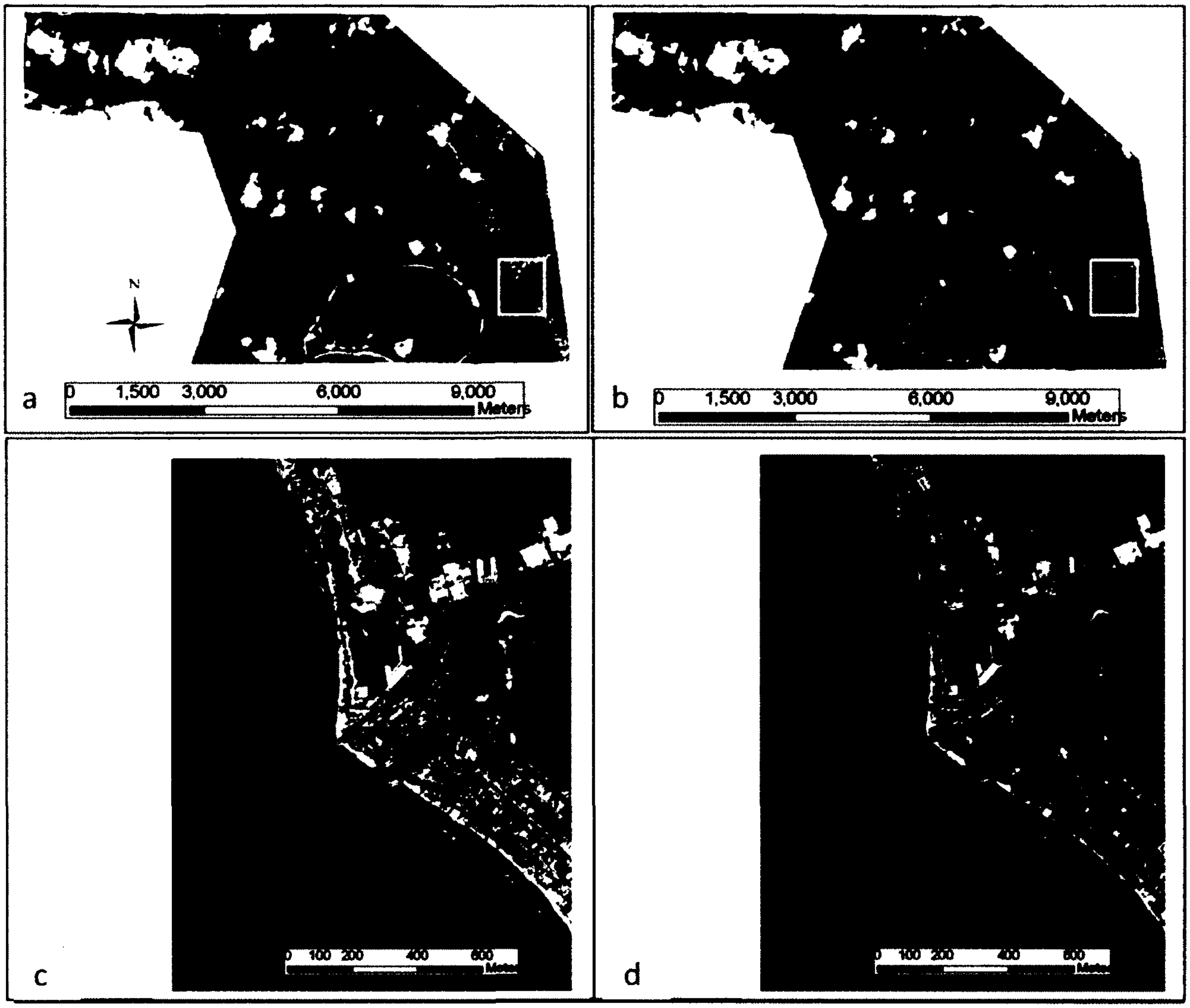

Figure 4. QuickBird image of central Bonaire before (a & c) and after (b & d) atmospheric correction. c & d are zoomed-in areas near the capital city and main port of Kralendijk.

Shallow Marine Bathymetry Estimation

One major difficulty in using satellite images to infer seabed habitat is the effect of variable water depths on bottom reflectance (Mumby et al. 1997). When attempting to discern information about the benthos, it is necessary to remove water column attenuation effects, just as it is necessary to remove effects from the atmosphere. The effects of variable depth were accounted for using the model derived by Lyzenga (1978; 1981; Mumby et al. 1997; Mishra et al. 2006b). In order to remove water column effects from each pixel, the depth at each pixel must be known (Lee et al. 1994), in the absence of reliable bathymetric data, this can be accomplished by creating a bathymetry raster at the same spatial extent as the study site using reflectance information from areas of known benthos with associated bathymetry measurements. With respect to bathymetric data, Bonaire has only nautical charts, which are extremely coarse resolution. The charts provide very little detail in the shallow reef system, where benthic mapping is most useful. While Mishra et al. (2006b) used seagrass benthos in their analyses, the Bonaire study site does not have large areas of seagrass, but is predominantly made up of coral and sand bottom-types. Because sand is an extremely bright bottom-type it is known to underestimate the water depth and therefore, not well suited to be used in this analysis. van Duyl (1985) used coral cover percentage groups of 10-20%, 20-40%, and greater than 40%; however, 20% represented a reasonable threshold above which the sensor could detect coral cover. Therefore, for this study, locations with greater than 20% coral cover were identified with their corresponding depth and geographic location (latitude and longitude) along the entire leeward coast. These points (N=103) were then used in a regression analysis with the water leaving radiance values at each of the pixels in the green band (560 nm). The green band absorbs more than the blue band with depth, but less than the red band, and was found to have the highest correlation with depth for the study site. When the radiance in the green band for areas with greater than 20% coral cover was plotted against depth, a logarithmic trend was found (Figure 5; $R^2 = 0.66$). This relationship was used to produce a depth raster by applying the equation to the atmospherically corrected green band (Figure 6a-b). In order to ensure the validity of the resulting depth raster, points of known depth were plotted against the depth estimates of the raster (Figure 7; $y = 1.0x + 5.95$, $R^2 = 0.58$, N=129) with estimated and actual depths of greater than 20 m removed based on the depth limitations of benthic habitat mapping (Mumby et al. 1998). Three sites with complex reef topography, where transects were not perpendicular to the shore, were excluded from analysis (Andrea II, Jeannie's Glory, and south of Oil Slick Leap).

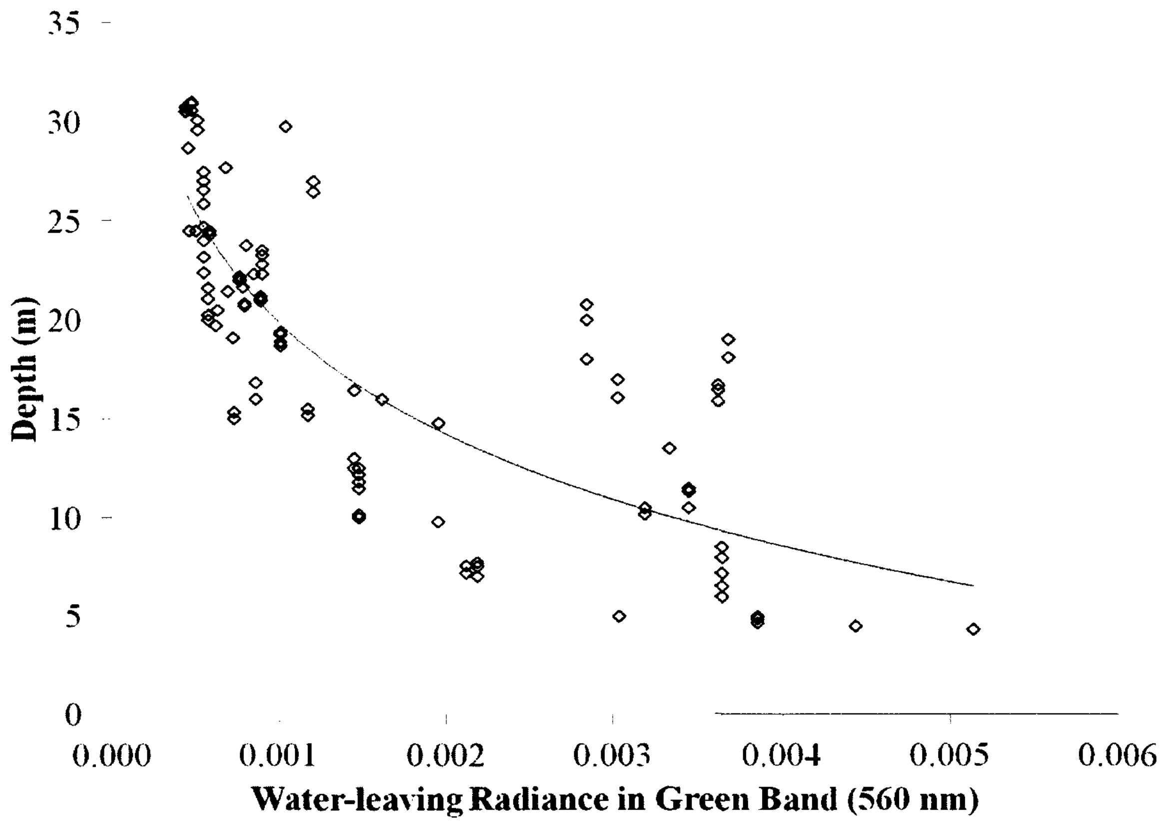

Figure 5. Model calibration dataset showing the regression between known depths and radiance in the green band (560 nm) over pixels of greater than 20% live coral cover ($y = -8.16\ln(x) - 36.49$, $R^2 = 0.66$).

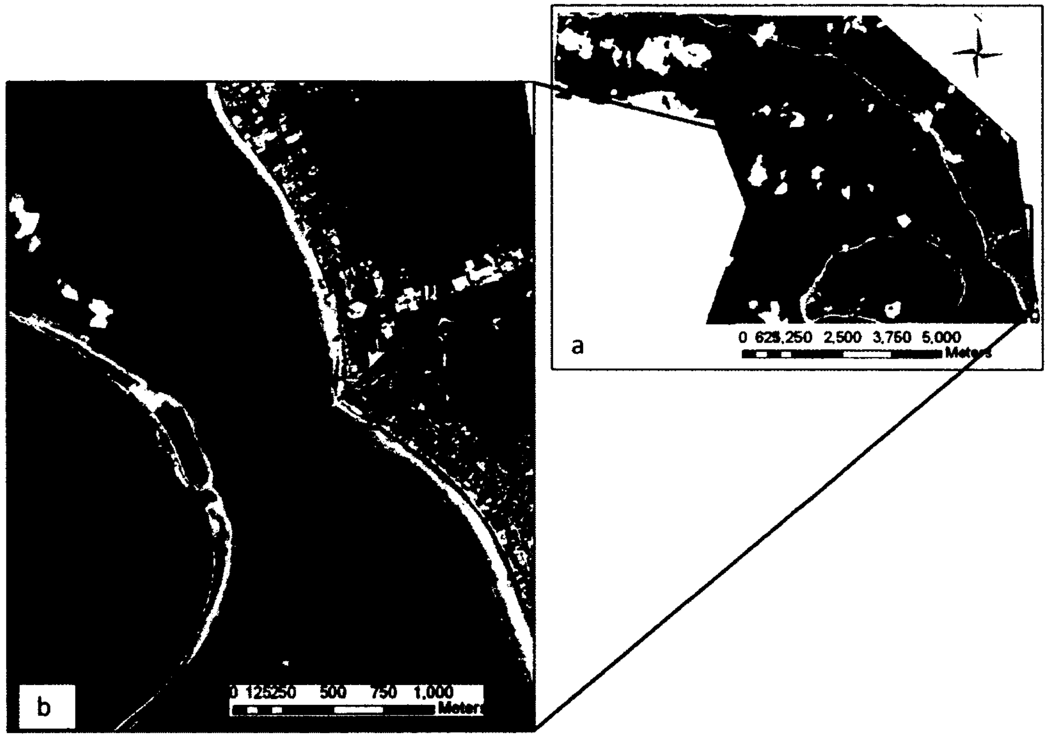

Figure 6. Applying the equation between depth and radiance in the green band (560 nm) to the green band of each image produces a depth raster of the entire coast (a) and a close-up of the coast around the capital and main port of Kralendijk and the small island of Klein Bonaire, masking out deeper depths (b).

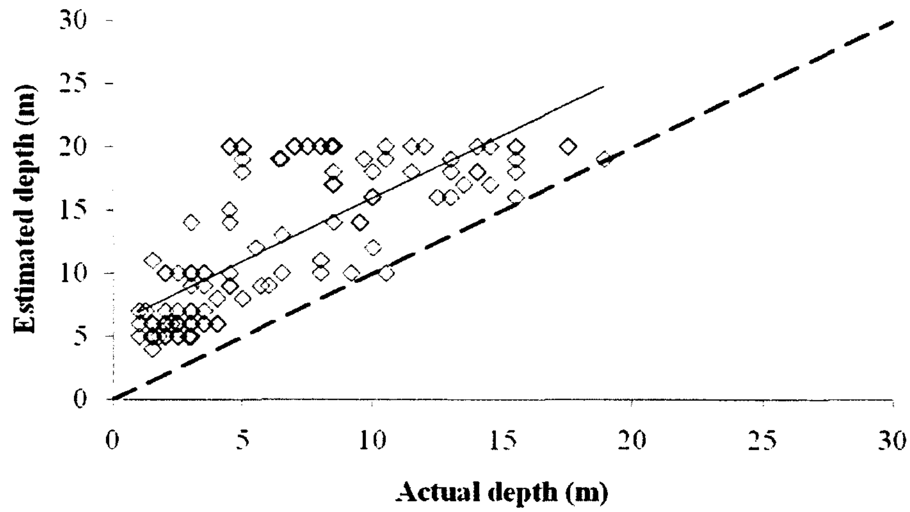

Figure 7. (a) Plot of actual versus estimated depths, dashed line represents a 1:1 relationship ($y = 1.0x + 5.95$, $R^2 = 0.58$).

Deriving Bottom Albedo

After the first-order atmospheric correction, the resulting images represent the remote sensing reflectance (R_{rs}), which is the ratio of water-leaving radiance (L_w) to downwelling radiance (E_d) just above the surface. R_{rs} is an apparent optical property controlled by the absorption and scattering properties of the constituents in the water and the bottom albedo:

$$R_{rs}(\lambda) = R^{w}{}_{rs}(\lambda) + R^{b}{}_{rs}(\lambda) \qquad (14)$$

where $R^{w}{}_{rs}(\lambda)$ is the remote sensing reflectance from the water column and $R^{b}{}_{rs}(\lambda)$ is the remote sensing reflectance from the bottom. $R^{b}{}_{rs}(\lambda)$ is the parameter of interest for benthic habitat mapping, but $R^{w}{}_{rs}(\lambda)$ can be readily calculated once water depth is known and can then be removed from the image (Lee et al. 1994):

$$R^{w}{}_{rs}(\lambda) \approx 0.05 * [\, b_b(\lambda) / (a(\lambda) + b_b(\lambda))] * [1 - e^{\{-3.2(a(\lambda) + b_b(\lambda))*H\}}] \qquad (15)$$

where $a(\lambda)$ is the total absorption coefficient at λ in m^{-1}, $b_b(\lambda)$ is the backscattering coefficient at λ in m^{-1}and H is the depth of the water in meters, values for which can be taken from the depth raster created in the previous section. Backscattering is a combination of the backscattering by particles (b_{bp}) and the backscattering by water molecules (b_{bw}; Morel 1974), such that:

$$b_b(\lambda) = b_{bp}(660) * (660/\lambda)^{\eta} + b_{bw}(\lambda) \qquad (16)$$

On Bonaire $\eta = 0.5$ was used, as the seas surrounding the island are Case-1 water, water in which the concentration of phytoplankton is high compared to nonbiogenic particles (Morel and Prieur 1977). $b_{bp}(660)$ can be estimated by assuming there is no contribution from the bottom to the upwelling signal in band 3 (660 nm) because absorption of the red band is large (> 0.4 m^{-1}) and dominated by water molecules (Mishra et al. 2006b). Total absorption can be calculated using the following equation, modified from Austin and Petzold (1986):

$$a(\lambda) = M(\lambda)[a(485) - a_w(485)] + a_w(\lambda) \qquad (17)$$

where M is a statistically-derived attenuation coefficient taken from Austin and Petzold (1986) and a_w is the pure-water absorption coefficient (Pope and Fry 1997), $a(485)$ can be calculated using the following equation:

$$a(485) = [(a(440) - a_w(440)) / M(440)] + a_w(485) \qquad (18)$$

and $a(440)$ can be calculated empirically (Lee et al. 1998):

$$a(440) = 10\text{^}[-0.619 - 1.969\{\log_{10}(R_{rs}(485) / R_{rs}(560)\} + 0.790*\{\log_{10}(R_{rs}(485) / R_{rs}(560)\}^2 \qquad (19)$$

For Case-1 waters values for total absorption ($a(\lambda)$) and total backscatter ($b_b(\lambda)$) for each QB wavelength are calculated (Table 2). Given the depth raster calculated by the shallow marine bathymetry estimation method, H is also known, so R_{rs}^{w} can now be calculated using equation 15.

Rasters of R_{rs}^{w} for each band (485, 560 and 660) of each satellite image are subtracted from the R_{rs} rasters (post first-order atmospheric correction images) for all three bands of each image to give R_{rs}^{b} values, remote sensing reflectance from the bottom in each band:

$$R^{b}{}_{rs}(\lambda) = R_{rs}(\lambda) - R^{w}{}_{rs}(\lambda) \qquad (20)$$

The three bands can then be recombined (bands 1, 2 and 3), creating a true color R_{rs}^{b} image. At this point benthic information could be extracted using standard image processing procedures.

Table 2. Absorption and backscattering coefficients from Mishra et al. (2006b; Table 2).

Wavelength (nm) (λ)	Total Absorption Coefficients (m^{-1}) ($a(\lambda)$)	Total Backscattering Coefficients x 10^{-3} (m^{-1}) ($b_b(\lambda)$)
485	0.0366	1.2540
560	0.0743	0.6740
660	0.4266	0.3310

Image Classification

Once the R_{rs}^{b} image for the entire island was created, the area along the coast that was mapped in the early 1980s (< 10 m; van Duyl 1985) was extracted for classification. Initially an unsupervised classification was performed on the image of the remote sensing reflectance from the bottom (R_{rs}^{b}) comprised of the three bands (red, blue and green), placing the pixels into 10 different classes based only on the optical properties of the pixel. After the classification, areas of known benthos were extracted and the classes those areas were place in by the unsupervised classification was determined. The classes of known benthos were either greater than 20% coral or greater than 50% sand. Areas of high coral cover fell into two classes, while sand fell into three classes, and four classes had areas of sand and coral and were therefore considered a mixed sand/coral class. The sand/coral mixture class contained some mixture of less than 20% hard coral and less than 50% sand, additional cover was attributed to the presence of octocorals, marine plants, or dead coral with algae (Relles and Patterson Chapter 1). The final remaining class included areas of deeper water (> 10 m). Classes were combined in the signature file, a set of statistics that is created when clustering, which defines the cluster, such that only four classes remained: deep water, coral, sand, and sand/coral (Figure 8). A supervised classification using a multivariate discriminant function (Mather 1987), was then performed on the image of the remote sensing reflectance from the bottom (R_{rs}^{b}) comprised of all three bands (red, blue and green) using this modified signature file. This function creates a supervised learning algorithm based on the signature file to produce an inferred function, called a classifier. In this case, a statistical algorithm used the range of radiance values derived from the sample data in the training classes (Mather 2004). Two classification schemes were used (i.e., two different signature files), with Classification #1 (C1), which was more likely to classify an image as either sand or coral and Classification #2 (C2), which was more likely to put a pixel into the sand/coral class. The resulting classifications were ground truthed using 364 points of known bottom-type of either coral (140) or sand (224).

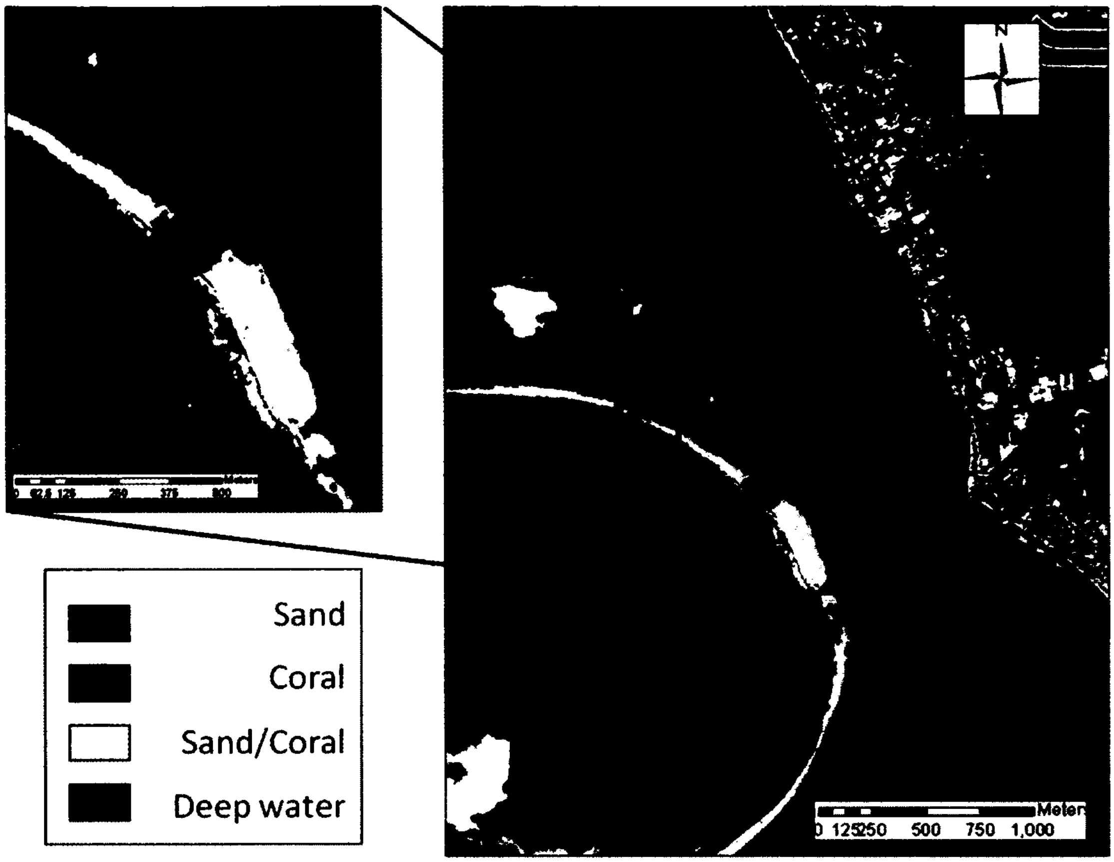

Figure 8. Color coded map showing dominant benthic habitat classes: sand, coral, and sand/coral mixture, as well as areas deeper than 20 m, which were unable to be classified (a) and a close-up of the coast around the capital and main port of Kralendijk and the small island of Klein Bonaire to the west (b).

Results

The satellite images from before and after the first-order atmospheric correction were visually distinct (Figure 4). A model validation compared the radiance values over sand and coral bottom-types in each of the four wavelengths, before and after atmospheric correction, and showed a greater than 99% reduction in radiance values (Figures 4a-d and 9a-d). Differences in radiance values were particularly pronounced in the blue (485 nm) and infrared bands (830 nm), depending on the depth and bottom-type.

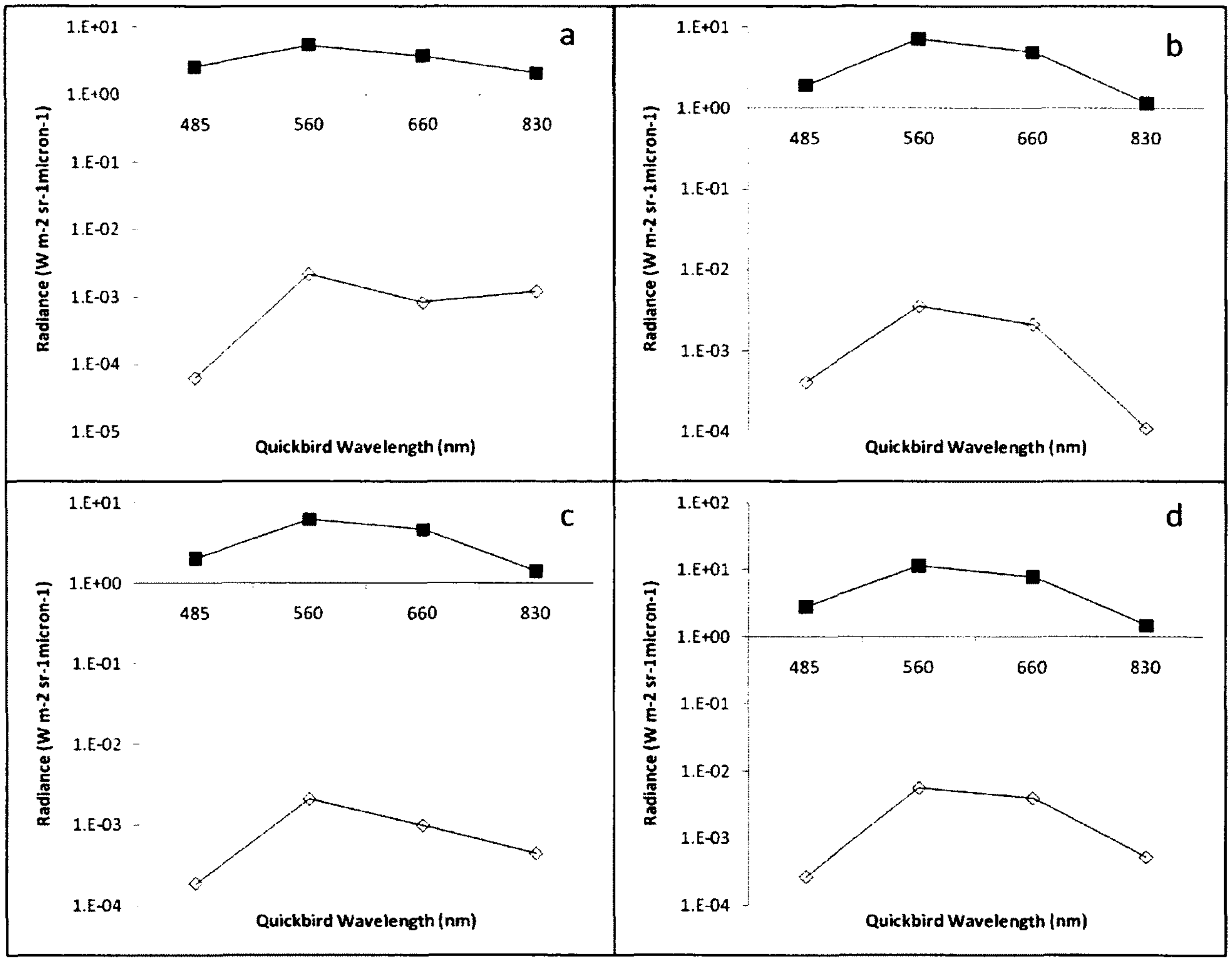

Figure 9. Radiance values before (solid black square) and after (open diamond) first-order atmospheric correction in all four QuickBird spectral bands: (a) on central Bonaire over a pixel of 67% coral in 7.5 m of water, (b) on central Bonaire over a pixel of 100% sand in 5.4 m of water, (c) on southern Bonaire over a pixel of 54% coral cover in 7.5 m of water and (d) on southern Bonaire over a pixel of 100% sand in 3 m of water.

Mishra et al. (2006b) found the highest correlation between depth and radiance in the blue band (485 nm) for their study site; in this study radiance in the green band (560 nm) had the highest correlative relationship with depth, showing a logarithmic relationship (Figure 5). As opposed to Mishra et al. (2006b), this study used areas of coral cover greater than 20% to calibrate the depth equation, due to a lack of high-density seagrass areas. Validation of the resulting depth raster showed a positive correlation ($R^2 = 0.58$) between estimated and actual depth (Figure 7; $y = 1.0x + 5.95$). Although Figure 7 shows an offset of 5.95 m from the 1:1 trend line, subtracting 5.95 from the depth raster resulted in overcorrection of R_{rs}^{w} into negative values.

Remote sensing reflectance from the water column (R_{rs}^{w}) varied by band, with comparable ranges in the blue (485 nm) and green band (560nm), and higher values in the red band (660 nm). Subtracting R_{rs}^{w} values from the satellite images post-atmospheric correction for each band (R_{rs} values; equation 14), gave remote sensing reflectance from the bottom values (R_{rs}^{b}), which was used for classification. R_{rs}^{b} values varied accordingly by band, with highest values in the green band (Figure 9a-d).

Several types of assessments can determine the accuracy of a classification. Overall accuracy is simply the sum of correctly labeled test sites divided by the total number of test sites, while user accuracy is the probability that a classified pixel actually represents that category on the ground (Mumby et al. 1997). The two classifications tested in this study had overall accuracies of 71% (C1) and 72% (C2). C1 correctly classified sand 97% of the time, but coral only 17%, while C2 correctly classified them 94% and 36%, respectively. As evidenced by its increased ability to classify coral, C2 was considered the better classification method. The main difference between the two classifications was that C2 was more likely to place a pixel in the sand/coral mixture class, as opposed to discriminating into either sand or coral. The low user accuracy in the coral group was likely due to variation at only one site, where all 38 points with coral greater than 20% were incorrectly categorized as sand. When this site was removed from the analysis, the user accuracy for the coral class rose to 50% using the C2 classification method; the exclusion of the site was considered valid in this case based on the disparate nature of the high coral cover sites and the fact that they were surrounded mostly by areas of high sand cover, with average live coral cover of less than 10% at depths less than 20 m (Relles and Patterson Chapter 1; Table 1). This study assumed that areas with coral cover greater than 20% at this site were surrounded by areas of high sand cover, obscuring the radiance values and subsequent classification. This assumption was based on the first author's experience when collecting *in situ* data on SCUBA and previously analyzed work (Relles and Patterson Chapter 1). After removal of the Pink Beach site, overall accuracy increased to 75% using C2 and user accuracy for coral pixels increased to 50%, while user accuracy for sand dropped only slightly to 91% (Table 3 for accuracies in both classifications before and after Pink Beach Site removal). The C2 classification system found coral cover to be approximately 28% along the entire leeward coast of the island.

Table 3. Overall accuracy and user accuracies for sand and coral classes using both classifications (C1 and C2), before and after removal of the Pink Beach site.

	Overall Accuracy	User Accuracy (Sand)	User Accuracy (Coral)
C1-before	71%	97%	17%
C2-before	72%	94%	36%
C1-after	74%	96%	41%
C2-after	75%	91%	50%

Discussion

First-order atmospheric correction and removal of water column contributions to satellite remotely sensed images provided visually distinct before and after images, which varied by band (wavelength) and bottom-type. Reductions in radiance due to removal of atmospheric effects in this study were comparable to previous work by Mishra et al. (2006b). However, where previous studies found the blue band (485 nm) to be most useful for shallow-marine bathymetry estimation, this study found water leaving radiance in the green band (560 nm) to have the highest correlation with depth, likely because first-order atmospheric correction resulted in the least reduction in radiance in this band, and the blue and red bands showed larger reductions. The resulting correlation between estimated and actual depth was also lower than in previous work using seagrass bottom, which is likely due to a combination of using coral bottom, which presumably has much larger variability in reflectance values due to the variety in coral pigmentation, and the greater variability of depth in general on the island of Bonaire when compared to the island of Roatán, Honduras (Mishra et al. 2006b). Subtracting the y-intercept of 5.95 m from the estimated versus actual depth resulted in over-correction of the remote sensing reflectance from the water column, likely due to wave action at shallow depths, which can obscure radiance values. The highly variable bottom-types used in this depth calibration may have contributed to the trend of overestimation in depth validation. However, this study showed that the method of correction and subsequent classification presented in Mishra et al. (2005; 2006b) can be adapted for other tropical marine environments.

Mapping at coarse resolution (i.e. four habitat classes; sand, coral, algae, seagrass), Mumby et al. (1997) found overall accuracies ranging from <60% to 73% using various satellite images, slightly lower than the 75% overall accuracy found in this study, which used three habitat classes (sand, coral, and sand/coral mixture). Mumby et al. (1997) commented on the difficulty in distinguishing between algae and seagrass. For the purposes of this study and owing to the low occurrence of both, these classes were included in the sand/coral mixture class when they occurred at all; dead coral and/or rubble overgrown with algae can also be confused with algae and seagrass. Fortunately, the island of Bonaire boasts a large population of herbivorous fish that keep algal overgrowth to a minimum (Steneck et al. 2007). This study applied methods using QB images to a different, larger geographic area than used in the original study (Mishra et al. 2006b), and also had higher overall accuracy than the Landsat TM or SPOT XS sensors evaluated by Mumby et al. (1997). Overall and user accuracy are ultimately determined by the number of samples and reliability of field-collected data, the overall and user accuracies could be further increased with more field samples of known depth and benthos to use to create the depth raster and signature file for classification.

The most significant advantage of satellite remote sensing to create reef maps is the amount of area that can be covered in a short amount of time. Earlier work on Bonaire has covered less than 100 one meter-squared quadrats and 10 m long transect lines (Steneck et al. 2007) or video transects (N=10) approximately one meter wide by no more than 300 meters long (Relles and Patterson Chapter 1). The current map covered approximately 695 ha of reef along the entire (50+ km) leeward coast. Relles and Patterson (Chapter 1; Table 1) found average coral cover to be much lower (~13%) than in the present study (28%), likely due to the locations of their transects, which excluded the small, uninhabited island of Klein Bonaire, where high levels of coral cover were found in this analysis and elsewhere (Steneck et al. 2007). Steneck et al. (2007) reports higher coral cover (close to 50%), with a study area restricted to depths less than 10 m.

Threats associated with climate change and other region-wide problems, such as overfishing and disease, are affecting reefs on a landscape-scale, so changes in reef cover and structure should be measured at the landscape-scale accordingly. Large-scale maps allow managers to visualize the spatial distribution of habitats, aiding the planning of marine protected areas and allowing the degree of habitat fragmentation and overall loss to be monitored (Mumby and Harborne 1999). The ability to make landscape-level maps of coral cover is important for conservation efforts and of particular interest to government officials and MPA managers. Coastal habitat maps are a fundamental requirement in establishing coastal management plans (Cendrero 1989). Landscape-level maps can be created from satellite images taken at various time periods and be used to compare changes in coral cover at different snapshots in time. Change detection techniques (Jensen 2005) can be employed to look at changes in coral cover, or other substrata of interest, on a pixel-by-pixel basis, while looking at total changes at the landscape level. Using these techniques, satellite remote sensed maps can be compared to one another from varying dates or to ancillary maps, like the van Duyl (1985) atlas, offering a more realistic baseline against which to compare. Identifying known areas of reef degradation is useful for identifying particular areas of management concern and directing conservation efforts where coral cover has suffered over time or as a result of a specific disturbance, such as a hurricane. Maps before and after disturbance are particularly useful for identifying declines in coral cover as a result of localized disturbance versus regional or global-scale threats, such as those associated with climate change.

The intent for the maps created here is for use by the STINAPA Bonaire to inform management decisions in the Bonaire National Marine Park. Park management actively maintains a series of moorings along the leeward coast, for use by boats as an alternative to anchor dropping, which is prohibited. Included in the park are two no-entry marine reserves, where snorkeling and scuba diving is also not allowed. It is the intention that the maps created here will be compared to earlier maps of the reefs in the park to identify areas of management concern and to direct conservation efforts. This will be the focus of a future manuscript.

Conclusion

Satellite Remote Sensing of the benthos should be restricted to water depths of less than 25 m (Mumby et al. 1997) and areas of known depth should be used to create reliable depth estimates in order to account for water column effects. Remote sensing techniques do not preclude the necessity of field-collected data; samples of reliable depth and bottom-type are ultimately used to create the classification system, so any increase in their number and accuracy will lead to an overall increase in the accuracy of the classification itself.

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CHAPTER 3

Change detection in a Marine Protected Area (MPA) using satellite remote sensing: coral loss and fragmentation on Bonaire

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Abstract

The island of Bonaire, Dutch Caribbean, is a long-established Marine Protected Area (MPA), the reefs of which were extensively mapped in the early 1980s to a depth of 10 m. Satellite remote sensing techniques were used to construct reef maps for 2008-09. Percent coral and sand cover were compared to the earlier maps. Coral habitat fragmentation metrics were also compared over this time interval. Overall, coral cover has declined during the past three decades, being replaced by sand, but the decline has not been as drastic as elsewhere in the Caribbean. Changes in coral fragmentation of the reef habitat were not associated with exposure along the coastline, however, coral cover was maintained in sheltered areas, whereas it declined along exposed shorelines. One of two no-diving marine reserves showed increases in coral cover accompanied by decreases in the number of patches of coral habitat over the time period, while the second reserve exhibited the opposite trend. Advances in satellite remote sensing techniques allow for a more rapid assessment of reef trends at the landscape level. Landscape-scale maps are useful for identifying areas of coral decline and focusing marine park management efforts.

Key Words Marine Protected Area (MPA), coral reef, landscape-scale, fragmentation

Introduction

The island of Bonaire, Dutch Caribbean, represents a unique, long-established Marine Protected Area (MPA). The Bonaire Marine Park (BMP) was established in 1979, after a series of other marine conservation measures, including increasing protection for turtles (1961), prohibitions on spear fishing (1971) and protection for corals, living or dead, from removal, damage by anchor dropping, and contact from divers (1975). Although the BMP was originally funded by the World Wildlife Fund Netherlands, the Dutch Government, the Government of the Netherlands Antilles, and the Island Government of Bonaire, the park ran into funding problems during the 1980s (STINAPA 2011). In 1991, the lack of funding was dealt with by requiring scuba divers to purchase a dive tag at a rate of 10 USD per calendar year to dive in park waters. Two marine reserves were simultaneously established that excluded underwater visitors, and the BMP was given full protection out to the 60 meter depth contour. The fee applied only to non-resident, tourist divers and in 2005 this "nature fee" was increased to 25 USD for divers and began including snorkelers at a rate of 10 USD. In 1999, the BMP gained national status as a park of the Netherlands Antilles and became the Bonaire National Marine Park (BNMP), which came with the advantage of increased opportunities for funding (STINAPA 2011). In addition to enforcing marine park rules and regulations, the BNMP's management body, STINAPA, is also involved in ongoing research, specifically monitoring stony coral and other invertebrates and fish, through the Atlantic and Gulf Rapid Reef Assessment (AGRRA) program and the Reef Environmental Education Foundation (REEF; STINAPA 2011).

In 1985, Dr. Fleur van Duyl published the *Atlas of the Living Reefs of Curaçao and Bonaire (Netherland Antilles)*, comprehensively mapping the coral reefs off the leeward coasts of Bonaire and Curaçao, an island to the west of Bonaire (Figure 1). The map classified the subtidal substratum into dominant benthic community types out to 10 m depth based on data collected by low-altitude aerial photography and ground truthed extensively by scuba diving in the early 1980s. van Duyl (1985) found coral to be the dominant bottom-type, making up around 62% of the benthos off the leeward side of Bonaire in shallow water (< 10 m). The total amount of coral cover was made up of nearly 40% *Acropora cervicornis* and 40% head coral, with the most common head corals being *Montastraea annularis, M. cavernosa, Diploria strigosa, Siderastrea siderea, S. radians, Dichocoenia stokesii, Colpophyllia natans, Porites astreoides, Meandrina meandrites,* and *Stephanocoenia intersepta.* Head corals in this group were sometimes accompanied by the foliate *Agaricia* spp. and finger corals were also often present in this group (van Duyl 1985). Since van Duyl's work was completed, cover of *A. cervicornis* has dropped to nearly zero, but increases in the head coral group have occurred at various locations along the coast (Relles and Patterson Chapter 1).

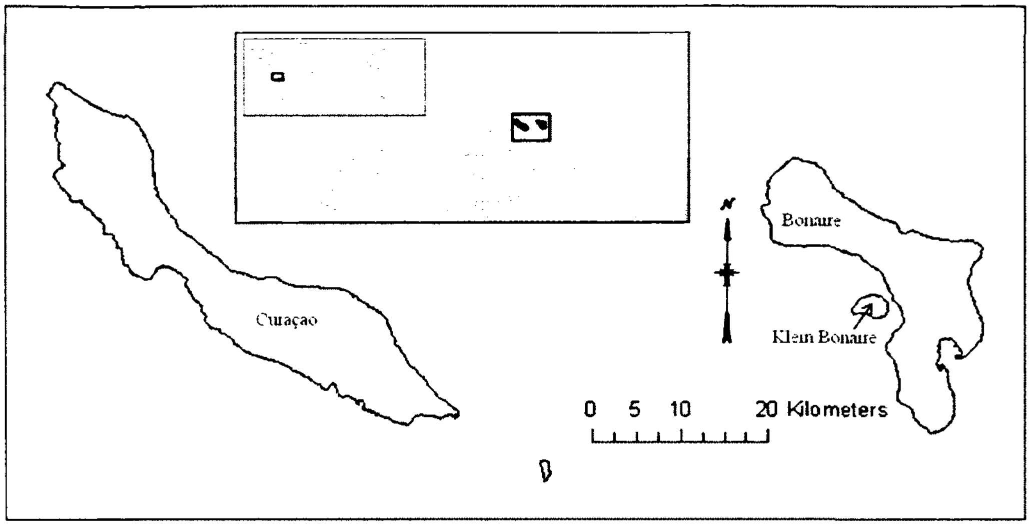

Figure 1. The islands of Bonaire and Curaçao are located in the Dutch Caribbean, about 80 km north of the coast of Venezuela.

Satellite remote sensing has proven to be an effective technique for creating benthic habitat maps, similar to van Duyl (1985), in coral ecosystems at coarse habitat resolution (3-4 bottom-type classes) and less than 20 m deep (Mishra et al. 2006; Mumby et al. 1997; Mumby et al. 1998; Relles et al. 2012, Chapter 2). Temporal change detection techniques (Jensen 2005) can then be employed to compare changes in coral cover, or other substrata of interest, on a pixel-by-pixel basis, while computing total changes at the landscape level. Habitat fragmentation can be analyzed by measuring changes in reef patch sizes, total area and perimeter-to-area ratios (PARA), and patch isolation, which is the distance between patches (McGarigal & Cushman 2002). Decreases in patch size and increases in the isolation of patches lead to reductions in population connectivity and are of particular concern for small reef-dwelling organisms with limited adult ranges and could potentially affect reproduction or dispersal (Schroeder 1987). Coral reefs, like most habitats, offer a number of advantages to their denizens, including protection from predation and a location to forage and find mates. The complex structure of coral reefs provides the physical habitats and shelter sites that accommodate many size classes of associated organisms. Small-sized invertebrates and many species of fishes are particularly characteristic of coral colonies, rubble drifts, and algal turfs (Choat & Bellwood 1991).

Habitat loss and habitat fragmentation are separate issues and must be dealt with accordingly (Fahrig 1997); fragmentation is more than just the loss of habitat, but loss such that small, isolated patches are created, changing the properties of the remaining habitat (van den Berg et al. 2001). In the case of coral reefs, we are concerned about habitat loss *and* fragmentation. The advantage of satellite remote sensing combined with change detection techniques is that total loss versus fragmentation can be rapidly quantified using Geographic Information Systems (GIS). Because fragmentation is a landscape-level process, fragmentation measurements are correctly made at the landscape scale (McGarigal et al. 2002; Fahrig 2003). The ability to make landscape-level maps of coral cover is important for conservation efforts and of particular interest to government officials and marine protected area (MPA) managers. Coastal habitat maps are thus a fundamental requirement in establishing coastal management plans for systems like coral reefs (Cendrero 1989; Relles et al. 2012, Chapter 2).

In this study, a recent (2008-09), satellite-derived, map of the reefs of Bonaire (Relles et al. 2012, Chapter 2) is compared to the habitat maps (van Duyl 1985) from data collected in the early 1980s to identify areas of coral habitat loss and fragmentation. Specifically, areas within the no-diving marine reserves and comparable sites that permit snorkelers and divers are compared to determine whether the lack of underwater visitors has had a significant positive impact on coral cover. The effect of coastline exposure on coral cover and fragmentation at exposed versus sheltered sites along the coast is also compared to evaluate the potential effects of storm damage on the reef habitat. This particular study is unique because it evaluates changes in the reef at the landscape-level whereas most previous research has been conducted at a finer-scale, from line transects and quadrats (Steneck et al. 2011; Bak et al. 2005).

Methods

Baseline Data

The island of Bonaire is located in the southern Caribbean Sea, approximately 80 km off the coast of Venezuela (12°10' N 68°17' W). This study focuses on the reefs of the leeward coast of Bonaire, including the accompanying uninhabited island to the west, Klein Bonaire (Figure 1). Maps of dominant coral community type and other bottom-types (e.g., sand, rubble, shore zone, and marine plants) were mapped in the early 1980s (van Duyl 1985) using aerial photographs and scuba diving to a depth of 10 m. The atlas was digitized into images (TIFF format) and subsequently georectified using ArcGIS 9.3 (ESRI). To align the maps with the coast on the satellite images, between 12 and 18 control points were identified using the georeferencing tool in ArcGIS, that allowed features identified by van Duyl (1985; Figure 2) to be aligned to the satellite images (e.g., distinct features of coastal morphology, piers and other permanent structures; Figure 3). Based on the control points, ArcGIS computed spatial residual error values, a measure of the fit between the true location on the image itself and the transformed locations of the output control points. Control points with the highest levels of error were then removed until the total root mean square (RMS) error, a statistical measure of the magnitude of variability between the shape of the original file and the shape of the georeferenced file, was less than 9 (usually between 6 and 9), without dropping the total number of control points below six. The resulting benthic habitat maps were saved as raster files. Polygon vector shapefiles were drawn manually around each of van Duyl's original 51 bottom-types using the editor function of ArcGIS 9.3 (ESRI). After creation of these polygons based on the van Duyl (1985) maps, bottom-types were reclassified into the coarser class distinctions of the satellite classification (coral, sand, and sand/coral) as outlined in Table

1 for comparison, and converted to raster files of comparable resolution. While van Duyl mapped areas of coral with percent cover ranging from 10-20, 20-40, and >40%, satellite data were only classified as coral if they contained greater than 20% coral cover. For this reason, areas considered to be 10-20% coral by van Duyl's classification were included in the sand/coral mixture class.

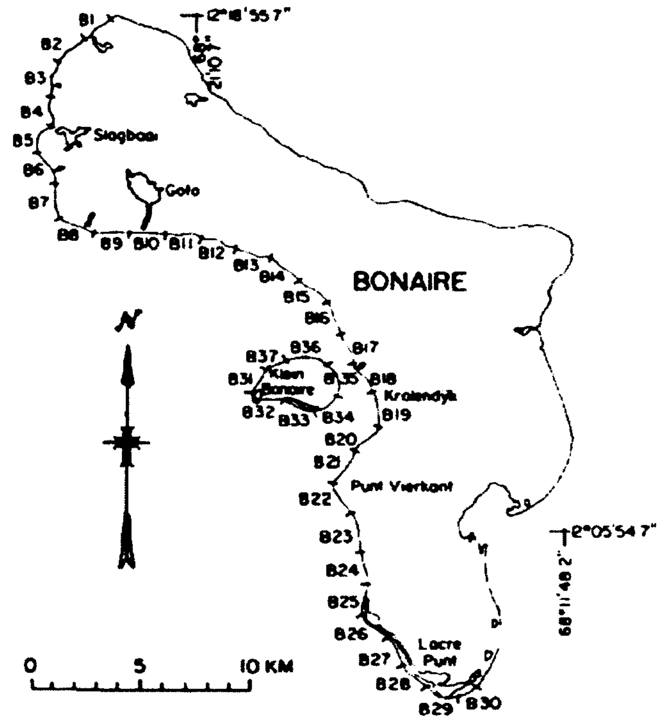

Figure 2. Locations of the 37 maps created by van Duyl (1985) along the coasts of Bonaire and Klein Bonaire.

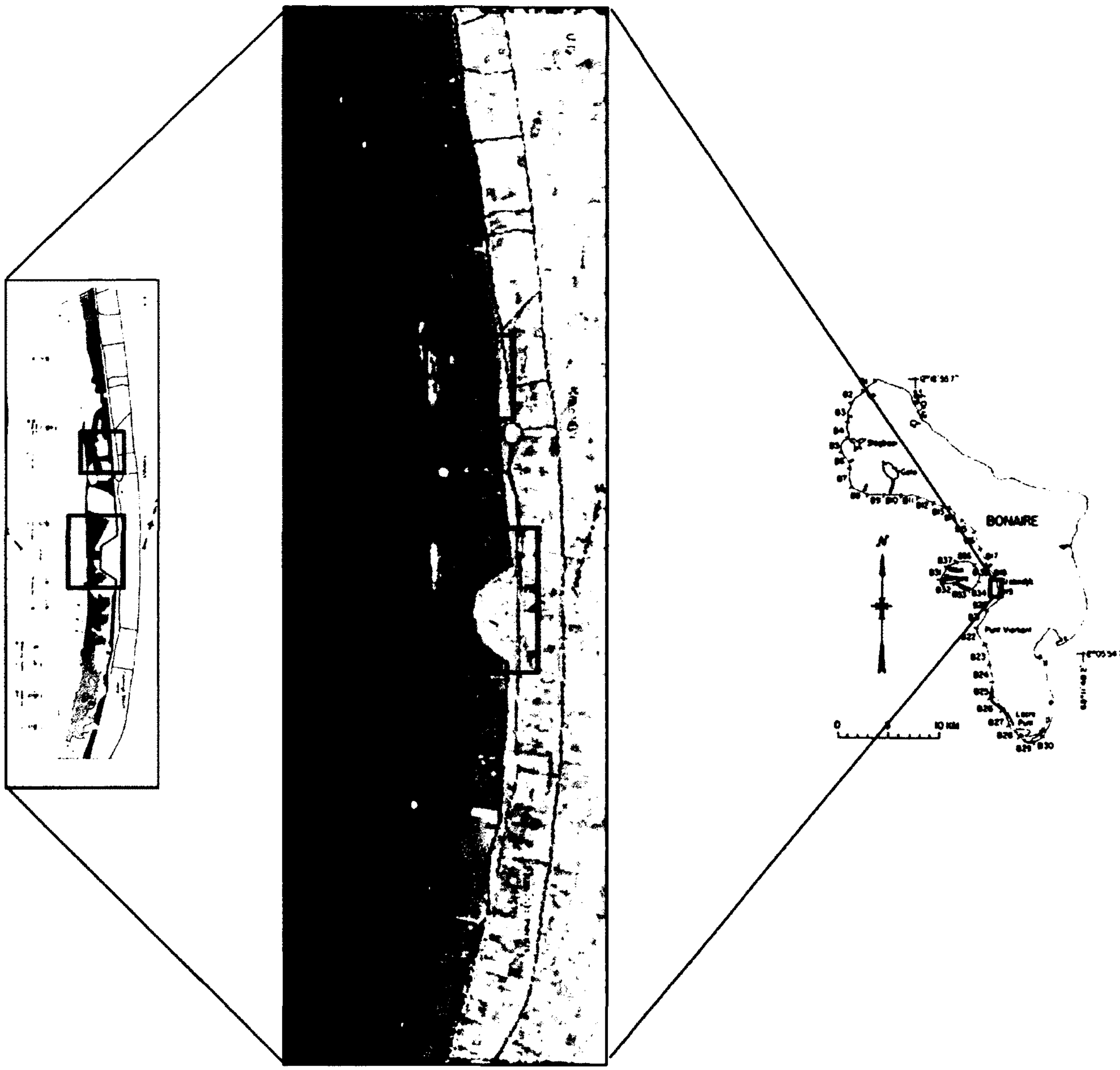

Figure 3. An example of one of van Duyl's (1985) maps (B19.; left rectangle) georeferenced with a modern satellite image (2009; right rectangle) in ArcGIS9.3 using the presence of piers (open black rectangles) along the coast. The map on the right indicates the relative location along Bonaire's shoreline.

Table 1. van Duyl's classifications on the left reclassified into the coarser classification system discernible by the satellite remote sensing method for the 2008-09 maps.

Van Duyl's class	**Coarse class system**
Sea Whip	Sand/Coral
Acropora cervicornis	Coral
Acropora palmata	Coral
Finger/Foliate coral group	Coral
Head coral group	Coral
Plant	Sand/Coral
Rubble	Sand
Sand	Sand
Shorezone	Sand

Satellite-derived data

Benthic habitat maps were created from satellite remote sensing data using a classification technique described below. Multi-spectral, high-resolution (2.4 m pixel) images from the QuickBird (QB) satellite from 2008 and 2009 were prepared and analyzed using methods outlined in Relles et al. (2012, Chapter 2). This required a first-order atmospheric correction, which removed the effects of particles smaller than the wavelength of light (Rayleigh scattering) and aerosols, and a water column correction, which removed the effects of particulates and chlorophyll in the water. After these corrections were performed, the benthos were classified into either sand, coral, or a sand/coral mixture class based on habitat types that were ground truthed using data collected in the field by SCUBA; QB imagery was proven useful for such coarse classifications (3-4 classes) in coral reef habitat (Mishra et al. 2006; Relles et al. 2012, Chapter 2). Details on the algorithms for atmospheric and water column collections, as well as the classification system, are described in detail in Relles et al. (2012, Chapter 2). The coral class included areas where coral cover was greater than 20%, while the sand class had greater than 50% sand cover. The sand/coral mixture class contained some mixture of less than 20% hard coral and less than 50% sand with the additional cover attributed to the presence of octocorals, marine plants, or dead coral with algae (Relles and Patterson Chapter 1). A total of 6.8 km^2 of reef along more than 50 km of leeward coastline was mapped out to a depth of approximately 10 m. For the purposes of classification and data manipulation, areas of coral were coded as 1, areas with the sand/coral mixture bottom-type were coded as 2, and areas of sand were coded as 3.

Harmonization of data

Prior to comparison of the two data sets, it was necessary to adjust the spatial resolution such that both data sets had the same resolution as the lowest resolution data set, in this case the van Duyl maps from 1985. The minimum mapping unit (MMU), which represents the minimum size of a polygon delineated by van Duyl (1985), and presumably the smallest habitat area discernible in the aerial photographs used to create the maps, was 9 m x 9 m (81 m^2). Both the early 1980s and the classified 2008-09 satellite images were then down-resolved from the original 2.4 m x 2.4 m pixels (5.8 m^2) by resampling the 2.4 m^2 pixels into 9.6 m^2 pixels using a majority rule.

Change Detection

The categories of coral, sand/coral, and sand were represented numerically as 1, 2, and 3, respectively, for the van Duyl data set, hereafter referred to as 1980s, and 10, 20 and 30, respectively, for the satellite data set, hereafter referred to as 2008-09. For classification and change detection purposes, areas of coral were thus given a value of 1 in 1980s and 10 in 2008-09, areas of sand/coral mixture were assigned a value of 2 in 1980s and 20 in 2008-09 and areas of sand a value of 3 in 1980s and 30 in 2008-09. This representation of the data allowed addition of the two data sets together on a [recalculated] pixel-by-pixel basis for the purpose of change detection. Because of this coding convention, progression from the ones column to the tens column of the resulting sum would represent the change in bottom-type from 1980s to 2008-09 (Table 2).

Changes were quantified as positive, negative, or neutral/no change. Change was considered positive when a pixel that was something other than coral changed to coral. It was also considered positive when an area previously dominated by sand became an area of sand/coral mixture. Negative changes occurred when coral changed to anything that was not coral, including when an area of sand/coral mixture changed to exclusively sand.

Table 2. Change values calculated in ArcGIS representing changes in bottom type between the early 1980s and 2008-09, distinguishing positive, negative and no change.

Value	Change (From-to)	Change (type)
11	Coral to Coral	None
12	Sand/coral to Coral	Positive
13	Sand to Coral	Positive
21	Coral to Sand/coral	Negative
22	Sand/coral to Sand/coral	None
23	Sand to Sand/coral	Positive
31	Coral to Sand	Negative
32	Coral/sand to Sand	Negative
33	Sand to Sand	None

Patch Dynamics

Raster data for both years were analyzed using FragStats 3.3 (McGarigal et al. 2002), which calculated patch, class, and landscape metrics. A patch is defined as an area of similarly-classified pixels, using an eight-cell rule that takes into consideration all eight adjacent cells, including the four orthogonal and four diagonal neighbors, to determine patch membership. The classes in this case were coral, sand, and sand/coral mixture, as described above. In addition to calculating the number and size of patches, including total patch area and perimeter-to-area ratios (PARA), a contiguity index (CONTIG) and Euclidean Nearest Neighbor (ENN) value were also calculated. CONTIG is quantified in FragStats by convolving a 3 x 3 pixel template with a binary digital imagine in which the pixels within the patch of interest are assigned a value of 1 and the background pixels (all other patch types) are given a value of zero. Template values of 2 and 1 are assigned such that orthogonally contiguous pixels are weighted more heavily than diagonally contiguous pixels; the contiguity value for a pixel is the sum of the products of each template value and the corresponding input image pixel value within the nine cell neighborhood. CONTIG values range between zero and one, with large contiguous patches resulting in larger values, as opposed to smaller, more disparate patches (McGarigal et al. 2002). The isolation of patches of coral was measured using the ENN approach, the shortest straight-line distance between the focal patch and its nearest neighbor of the same class (McGarigal et al. 2002). Patch, class, and landscape metrics for the two data sets, 1980s and 2008-09, were analyzed statistically using the Mann-Whitney Rank Sum Test (non-normal data).

Klein Bonaire Coastline Exposure

The small island of Klein Bonaire, located just west of the main island of Bonaire, is uninhabited. The western portion of the island is exposed to incoming waves and storm energy, while the eastern portion is sheltered by the main island. The island was divided as shown in Figure 4B and the two halves were statistically analyzed using a two-way ANOVA to compare the patch statistics described above for the exposed *vs.* sheltered halves between the 1980s and 2008-09.

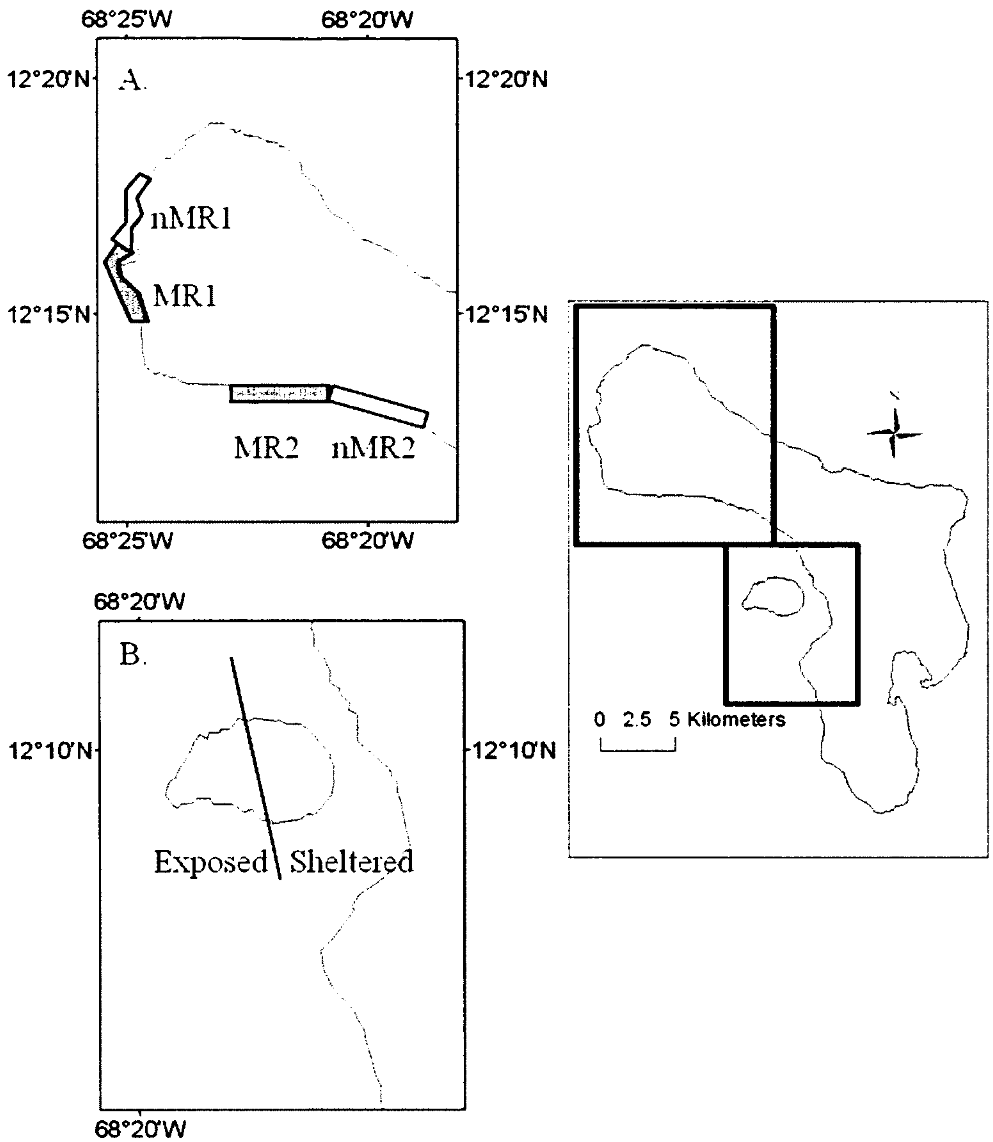

Figure 4. The locations of the exposed (MR1) and sheltered (MR2) marine reserves and adjacent exposed (nMR1) and sheltered (nMR2) non-reserve sites (A.). Klein Bonaire divided into the sheltered and exposed sides of the island (B.).

No-diving marine reserves

The farthest northwest marine reserve closed to divers was designated marine reserve number one (MR1) and was considered an exposed site because its position along the coastline left it potentially more exposed to storms. A comparable site of equal size and adjacent to MR1 was identified as nMR1 and considered to be an exposed site in a similar area along the coast that was not closed to divers and other underwater visitors. The second marine reserve is located farther south along the coast and is sheltered by the northwestern portion of the island and was designated MR2. A comparable site of equal size to the east of MR2 was designated as the non-reserve, sheltered site (nMR2; Figure 4A). These four sites, MR1, MR2, nMR1, and nMR2, were compared using a three-way ANOVA to look at the patch statistics described above and compare marine reserve to non-reserve, exposed versus sheltered sites, and the earlier, 1980s data to the 2008-09 satellite data. MR1, nMR1, MR2 and nMR2 are shown in Figure 4 and their bounding coordinates are listed in Appendix A.

Results

Baseline data

In the early 1980s, 707 hectares of reef offshore of the leeward coast of Bonaire was mapped (van Duyl 1985). Sixty-two percent of this area represented greater than 20% hard coral cover at the time (441 ha), while areas of high sand cover made up almost

32% (226 ha). The remaining 6% was composed of a sand/coral mixture (40 ha), which included soft corals, as well as dead coral covered with algae, and other marine plants.

Satellite-derived data

In 2008-09, 695 hectares of reef were mapped; the disparity in area mapped was due to cloud cover in the satellite images. Slightly greater than 30% of the 92.2 m^2 pixels represented areas of greater than 20% hard coral cover (210 ha). Sandy bottom (>50% cover) dominated over 50% of the reef, approximately 370 ha, while the remaining 17% of the reef (115 ha) was covered by a sand/coral mixture, often accompanied by octocorals (e.g., sea whips and gorgonians), dead coral covered with algae, and marine plants.

Several types of metrics can determine the accuracy of a classification. Overall accuracy is simply the sum of correctly labeled test sites divided by the total number of test sites, while user accuracy is the probability that a classified pixel actually represents that category on the ground (Mumby et al. 1997). The overall accuracy of the classification scheme was 75% (N = 364), with user accuracies of 91% and 50% for sand and coral classes, respectively (Relles et al. 2012, Chapter 2).

Change Detection

Areas of no change made up 45% of the total reef area and negative change occurred on 43% of the reef, while areas of positive change were only found on 12%.

For areas previously dominated by coral, 47% became sand, while 36% stayed coral. The largest percentage of sand pixels from the 1980s data set remained sand in 2008-09 (66%), while 18% became coral and 15% changed to a sand/coral mixture. The largest percentage of the sand/coral mixture pixels changed to sand in 2008-09 (43%), 24% remained sand/coral, while 34% changed to coral (Figure 5).

The northwest coast, most of which is uninhabited because it includes Washington Slagbaai National Park, experienced the highest area of negative change (51%), with 60% of coral pixels changing to the sand/coral mixture class or to just sand. While 82% of the area was represented by coral in the 1980s (133 ha), coral dropped to 38% of the area in 2008-09 (61 ha; Figure 5A.), and sand went from 16% of the area (26 ha) to 55% (89 ha; Figure 5C.). The sand/coral mixture class also experienced an increase from 2% (3 ha) to 7% (11 ha) from the early 1980s to 2008-09 (Figure 5B.).

The coast of the central part of the island, which includes one of the no-diving marine reserves (MR2) and the capital city (Kralendijk), extends 16 km along the coastline to the north of the city, experienced the highest level of positive change (33%) and the lowest level of negative change (26%) of the four areas. A larger portion of this coastal area was coral in the early 1980s (92 ha) than in 2008-09 (74 ha; Figure 5A.). The sand/coral mixture increased from 11 ha in the early 1980s to 41 ha in 2008-09 (Figure 5B.). Correspondingly, the amount of sand cover in the area declined from 40 ha to 28 ha (Figure 5C.).

The southwest coast is also sparsely inhabited and consists mostly of salt pans for the island's sea salt industry. In the early 1980s more than 50% of this portion of the coastline was covered in coral (158 ha), which dropped to 19% in 2008-09 (52 ha; Figure

5A.). Correspondingly, sand cover increased from 38% (109ha) to nearly 70% (190 ha: Figure 5C.).

The uninhabited island of Klein Bonaire, located approximately 1 km west of Bonaire, experienced declines in coral cover from 52% (59 ha) to 20% (23 ha) over the time period (Figure 5A.). Thirty-four percent of the coral pixels changed to sand, while 35% changed to the sand/coral mixture, resulting in an increase in the sand/coral mixture class from 3% (4 ha) to 24% (27 ha; Figure 5B.), and the sand class from 45% (51 ha) to 56% (63 ha; Figure 5C.).

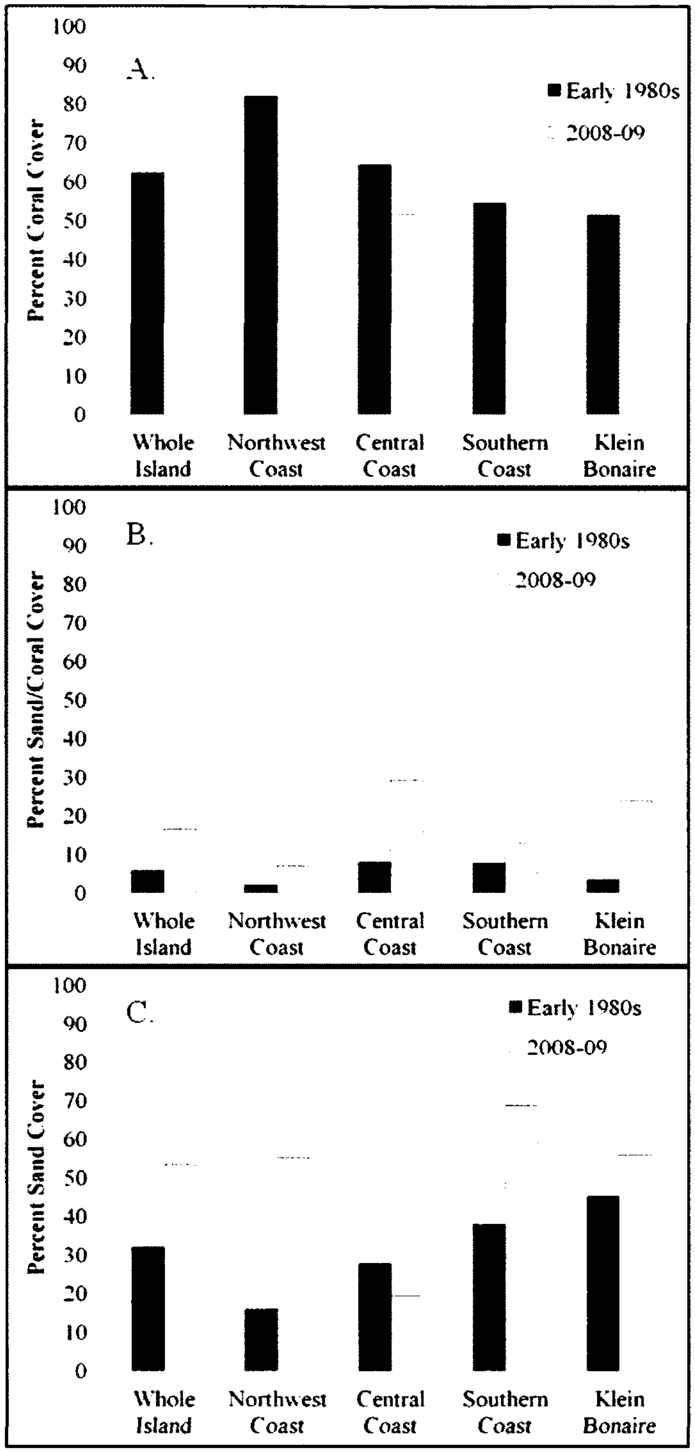
A.
Early 1980s
2008-09
Percent Coral Cover
100
90
80
70
60
50
40
30
20
10
0
Whole Island
Northwest Coast
Central Coast
Southern Coast
Klein Bonaire
B.
Early 1980s
2008-09
Percent Sand/Coral Cover
C.
Early 1980s
2008-09
Percent Sand Cover

Figure 5. Changes in percent coral cover (A.), sand/coral mixture (B.) and sand (C.) between the early 1980s and 2008-09 on the entire island of Bonaire, the northwest coast, the central coast, the south coast and the uninhabited island of Klein Bonaire.

Whole Island Patch Dynamics

While the percentage of area covered by coral declined from 62% in the 1980s to 30% in 2008-09 (Figure 5A.), the number of patches of coral increased from 72 to 221 (Figure 6). Patch area, perimeter-to-area ratio (PARA), contiguity index (CONTIG) and Euclidean Nearest Neighbor (ENN) distance were all found to fail the test of normality (Shapiro-Wilkes), so they were compared using the non-parametric Mann-Whitney Rank Sum Test. Mean patch size decreased from 6.12 ha to 0.95 ha ($U_{221,72} = 6035.00$, $p = 0.002$). The PARA increased from 2247.87 to 2827.34 ($U_{221,72} = 5838.50$, $p < 0.001$). The CONTIG decreased from 0.41 to 0.30 ($U_{221,72} = 5959.50$, $p = 0.001$). The Euclidean Nearest Neighbor (ENN; $U_{221,72} = 7826.50$, $p = 0.83$) values were not significantly different between years.

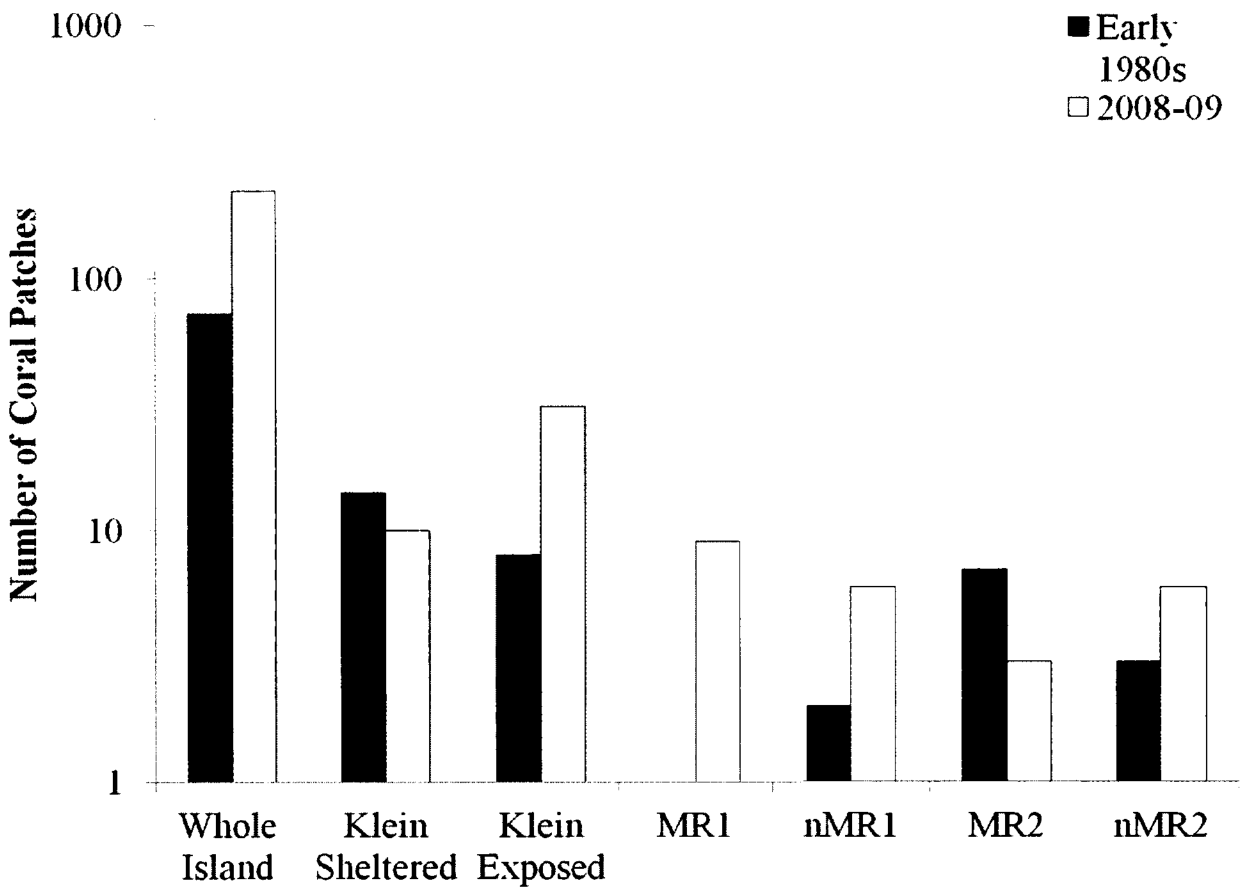

Figure 6. Changes in the number of coral patches between the early 1980s and 2008-09 on the main island, on the sheltered and exposed sides of Klein Bonaire, and in the marine reserves and adjacent non-reserve sites. Note: log scale and only one coral patch in the early 1980s in MR1.

Effect of exposure on Klein Bonaire

From the early 1980s to 2008-09, the sheltered, eastern portion of Klein Bonaire declined from 37% coral to 27% (Figures 7A and 9.). Between the two time periods the sand/coral mixture class increased from 6% of the total area to 19% (Figures 7B. and 9), while sand declined slightly from 57% to 54% (Figures 7C. and 9). The exposed, western side of Klein Bonaire initially had a higher percentage of coral cover than the eastern side (63%), which declined to less than 15% in 2008-09 (Figures 7A. and 10). This was accompanied by an increase in sand from 35% to 57% and an increase in the amount of area covered by a sand/coral mixture from 1.5% to 28% (Figures 7B., 7C., and 10). The exposed side of the island increased in the number of patches of coral from 8 to 31, whereas the sheltered side of Klein experienced a decline in the number of coral patches from 14 to 10 (Figure 6).

The patch metrics area, PARA, CONTIG and ENN were not normally distributed, so the Mann-Whitney Rank Sum Test and Kruskal-Wallis nonparametric tests were used. Between the two time periods, only ENN distance was significantly different on the exposed side of the island ($U_{31,8} = 50.5$, p = 0.01; Figure 8D.). There was no significant effect of year or exposure (H = 5.35, df = 3, p = 0.15, adjusted for ties) on patch area (Figure 8A.), PARA (H = 6.15, df = 3, p = 0.11, adjusted for ties; Figure 8B.), or CONTIG (H = 6.15, df = 3, p = 0.11, adjusted for ties; Figure 8C.). ENN was significantly different between the groups (H = 7.99, df = 3, p = 0.05, adjusted for ties) so the Mann-Whitney Rank Sum Test was used to determine which year and exposure combinations were significantly different from one another; because there were six pairwise comparisons of the four combinations (only one patch in MR1 in the early

1980s, therefore no ENN value) the alpha level of significance was adjusted accordingly by dividing it, 0.05, by 6, resulting in an α of 0.0083 (Bonferroni adjustment). A pairwise comparison found none of the combinations of year and exposure to be significantly different given this adjusted α.

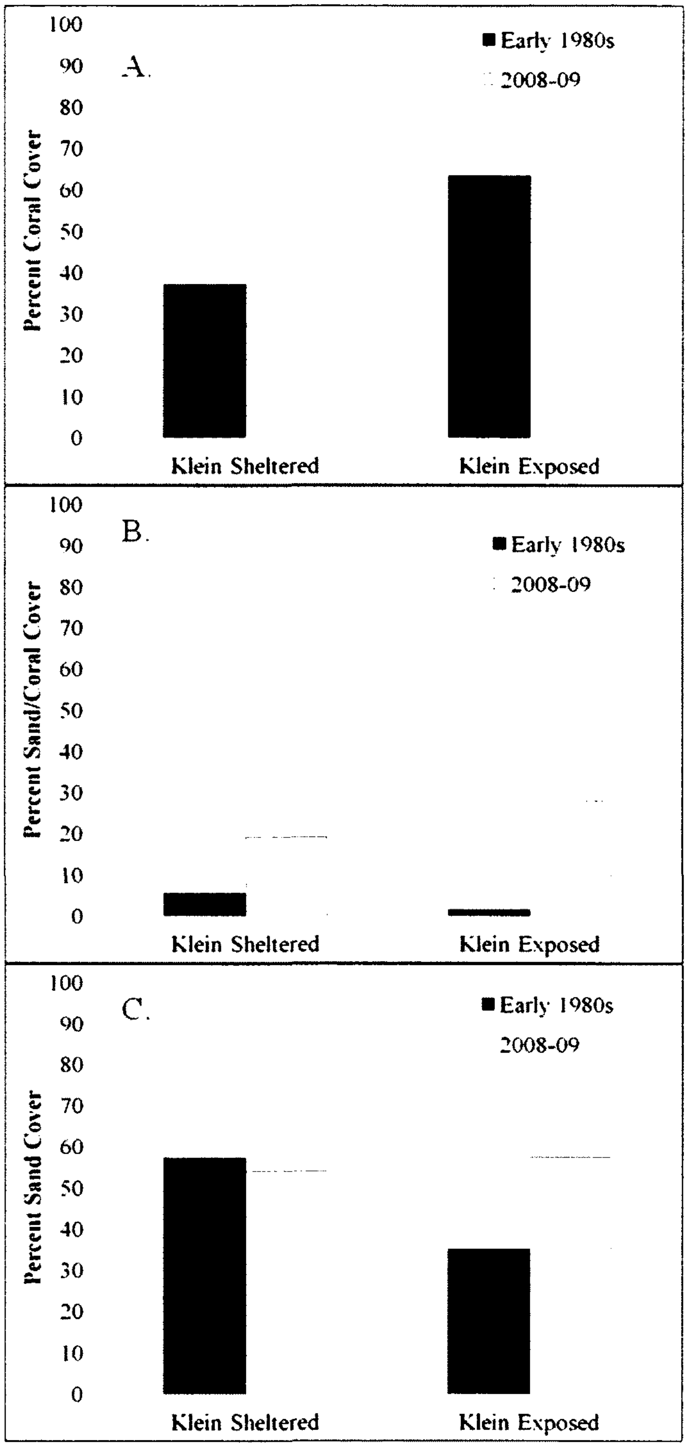

Figure 7. Changes in percent coral cover (A.), sand/coral mixture (B.) and sand (C.) between the early 1980s and 2008-09 on the sheltered and exposed portions of Klein Bonaire.

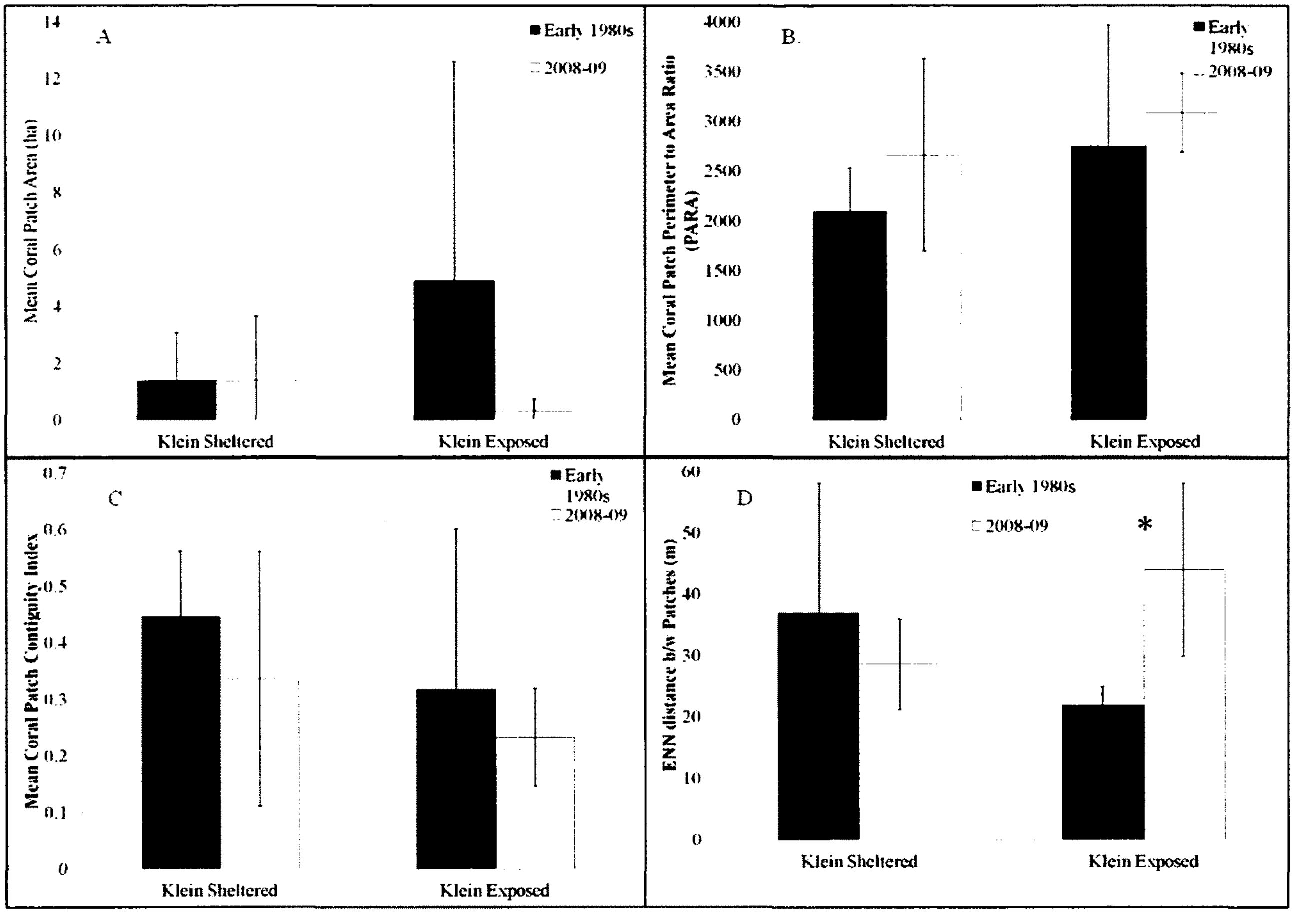

Figure 8. Changes in the patch parameters area (A.), PARA (B.), contiguity index (C.) and ENN distance (D.) between the early 1980s and 2008-09 on the exposed and sheltered sides of the island. Only the change in ENN distance over the time period on the exposed side of the island was significant ($U_{31,8} = 50.5$, p = 0.01). * indicates a significant difference between time periods. Error bars indicate +/- two standard errors.

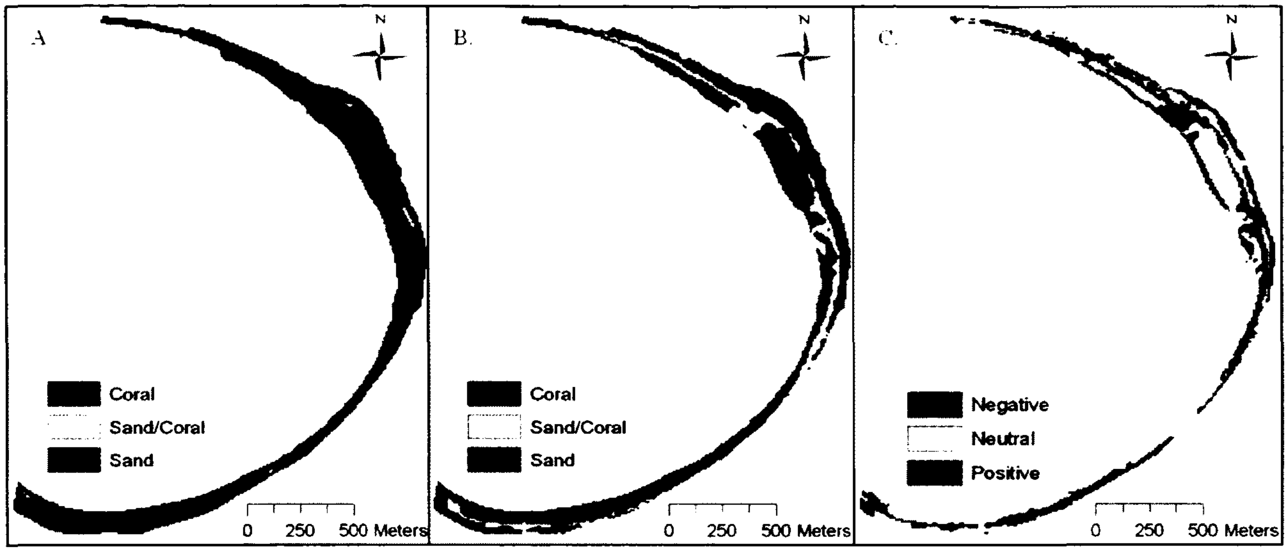

Figure 9. Coral, sand/coral mixture, and sand classes on the eastern, sheltered coast of Klein Bonaire in the early 1980s (A.) and 2008-09 (B.) C. The negative, neutral and positive change values over the time period along the exposed coast. (Location of sheltered coast of Klein Bonaire shown in Figure 4B.)

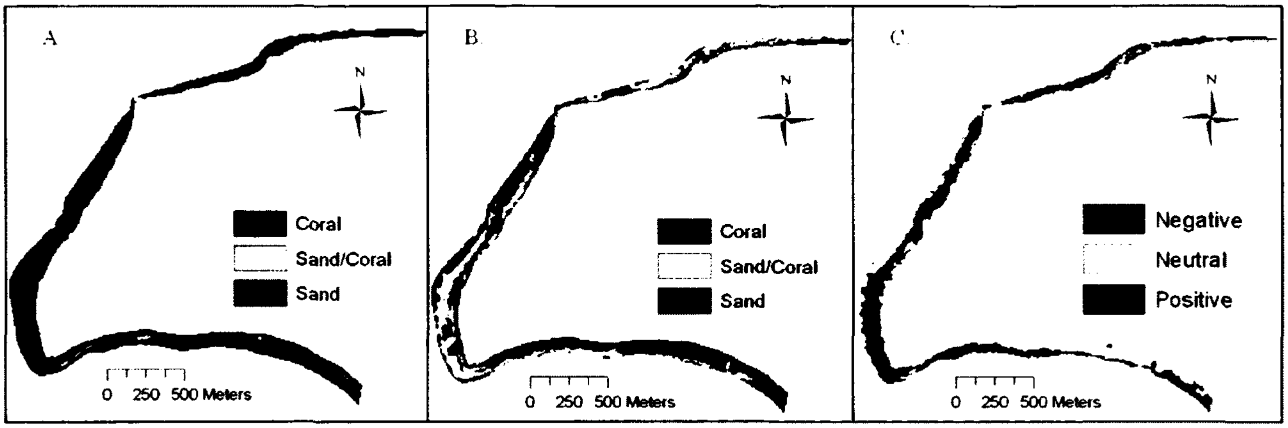

Figure 10. Coral, sand/coral mixture, and sand classes on the western, exposed coast of Klein Bonaire in the early 1980s (A.) and 2008-09 (B.) C. The negative, neutral and positive change values over the time period along the exposed coast. (Location of exposed coast of Klein Bonaire shown in Figure 4B.)

No-diving marine reserves

Out of the four areas, MR1, MR2, nMR1, and nMR2, only the sheltered, no-diving reserve site (MR2) experienced a positive increase in coral cover over the time period, going from 66% coral to greater than 83% (Figures 11A. and 13). This was accompanied by a decline in sand from 31% in the early 1980s to 6% in 2008-09 (Figures 11C. and 13). MR1, nMR1, and nMR2 all experienced declines in coral cover and increases in sand (Figures 11, 13, and 14). MR2 also experienced a decrease in the number of patches of coral within the reserve over time from 7 to 3 (Figure 6), which was accompanied by increases in the mean patch area and PARA (Figures 12A. and 12B.). MR1, nMR1, and nMR2 all experienced the opposite trends in number of patches and mean patch area, but mean PARA increased in MR1 in all three (Figures 6 and 12). All four areas experienced declines in CONTIG (Figure 9C.). Mean ENN values decreased in MR2 and nMR1, but increased in nMR2 (Figure 9D.). In the early 1980s there was only one large coral patch in MR1 so there is no ENN value. Patch PARAs and CONTIGs were not significantly different as a result of year, exposure, status as a marine reserve or any combination of the three (Table 3). Area and ENN values all failed the ANOVA assumption of normality, so the non-parametric Kruskal-Wallis Test was performed on the data. The mean patch area was not significantly impacted by year, exposure or status as a marine reserve ($H = 4.69$, $df = 7$, $p = 0.70$, adjusted for ties). ENN was significantly different between the groups ($H = 16.68$, $df = 6$, $p = 0.01$, adjusted for ties) so the Mann-Whitney Rank Sum Test was used to determine which year, exposure, and marine reserve status combinations were significantly different from one another; because there were 21 pairwise comparisons of the seven combinations (only one patch in MR1 in the early 1980s, therefore no ENN value) the alpha level of significance was adjusted accordingly by dividing it, 0.05, by 21, resulting in an α of 0.0024 (Bonferroni adjustment). In the early 1980s the ENN value on the non-reserve, exposed site was significantly higher than the sheltered reserve site in the early 1980s (t-test = -5.79, df = 7, $p < 0.001$) and in 2008-09 (t-test = -10.446, df = 3, $p = 0.002$; Table 4).

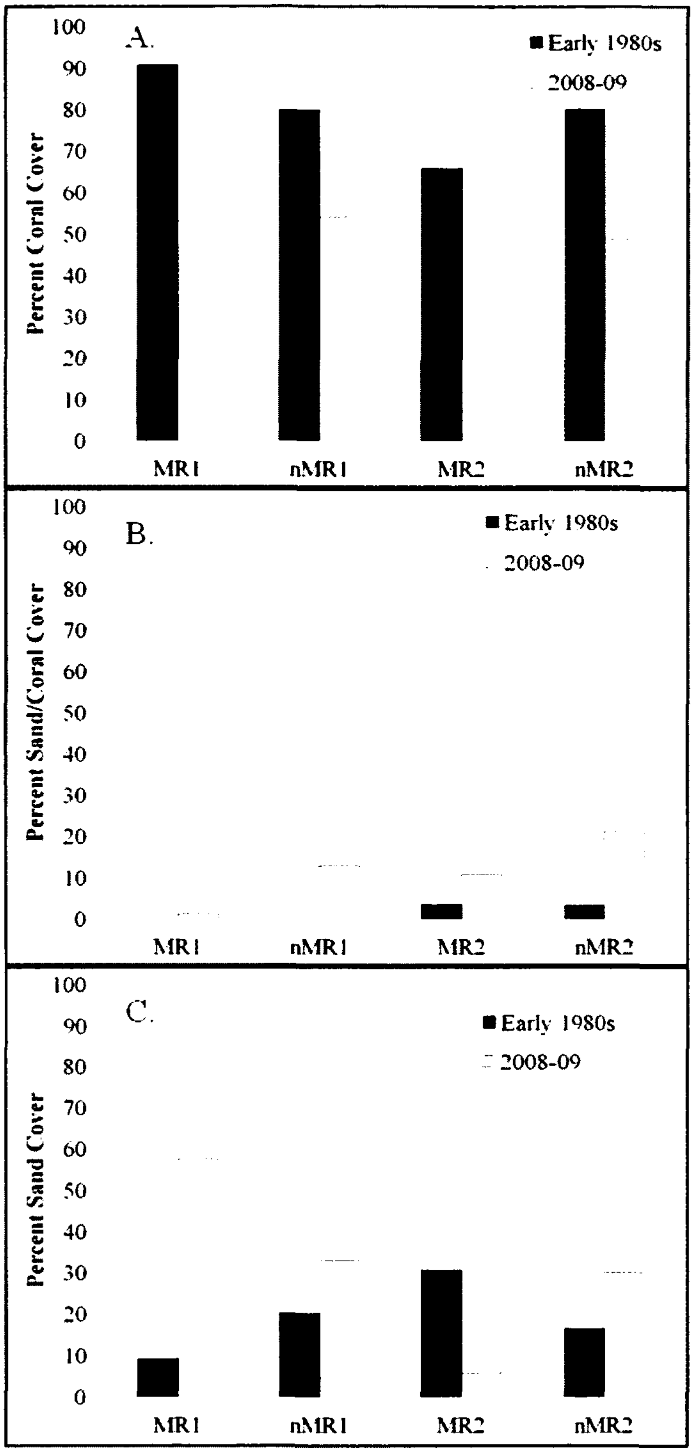

Figure 11. Changes in percent coral cover (A.), sand/coral mixture (B.) and sand (C.) between the early 1980s and 2008-09 in the two marine reserves (MR1 and MR2) and

unprotected adjacent areas (nMR1 and nMR2) on Bonaire. Note: there was no sand/coral mixture class in MR1 or nMR1 in the early 1980s.

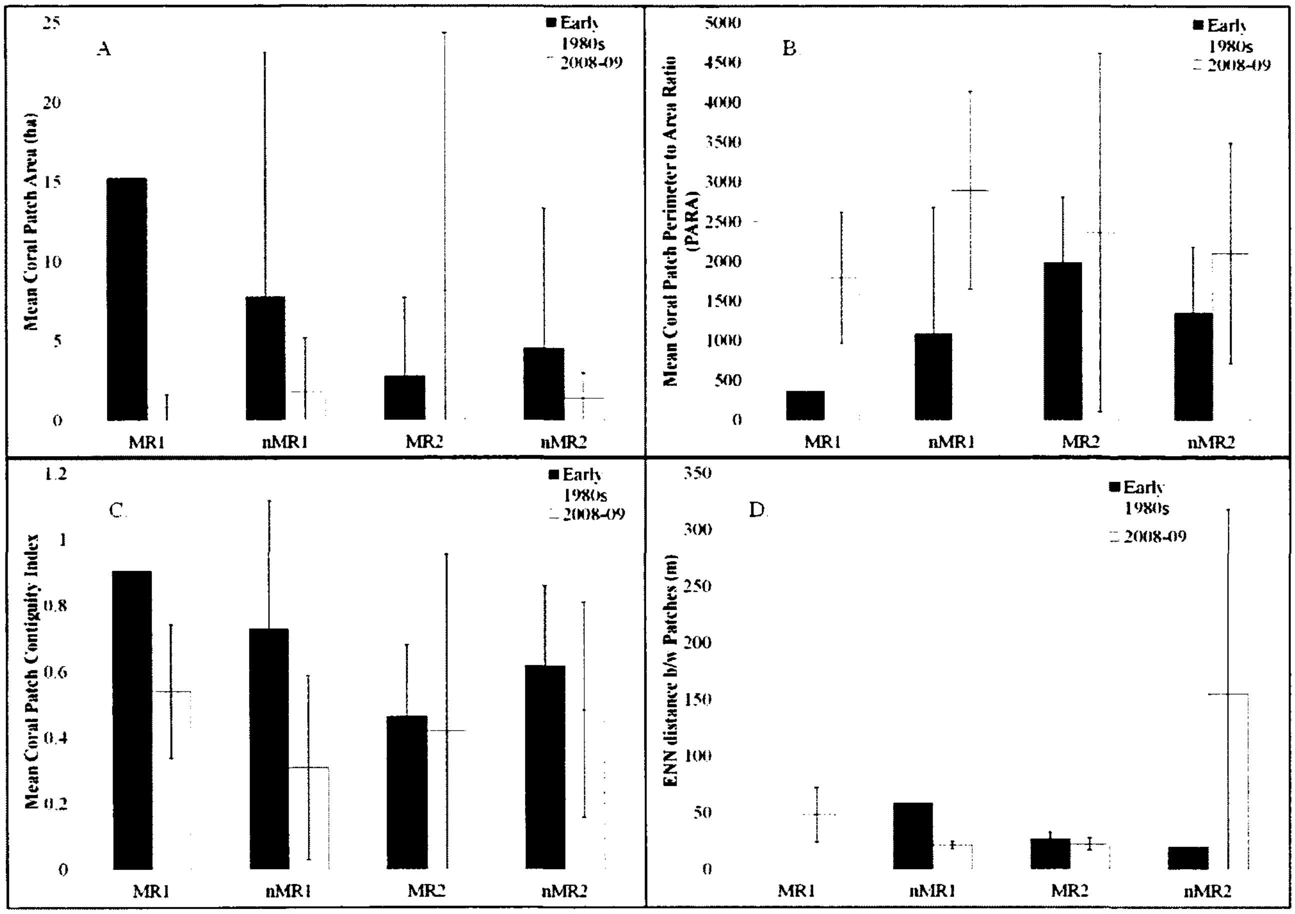

Figure 12. Changes in the patch parameters area (A.), PARA (B.), contiguity index (C.) and ENN distance (D.) between the early 1980s and 2008-09 on the exposed and sheltered sides of the island. Only the change in ENN distance over the time period on the exposed side of the island was significant ($U_{31,8} = 50.5$, $p = 0.01$). Error bars indicate +/- two standard errors.

Table 3. F-stats and p-values resulting from a three-way ANOVA with year, exposure and marine reserve status as predictors for the variables PARA and CONTIG (df = 29).

	PARA		CONTIG	
	F	P	F	P
Year	3.63	0.07	3.07	0.09
Exposure	0.53	0.47	0.81	0.38
Reserve	0.17	0.69	0.12	0.74
Year x Exposure	0.86	0.36	1.20	0.28
Year x Reserve	0.11	0.74	0.07	0.80
Exposure x Reserve	1.40	0.25	1.29	0.27
Year x Exposure x Reserve	0.00	1.0	0.00	0.95

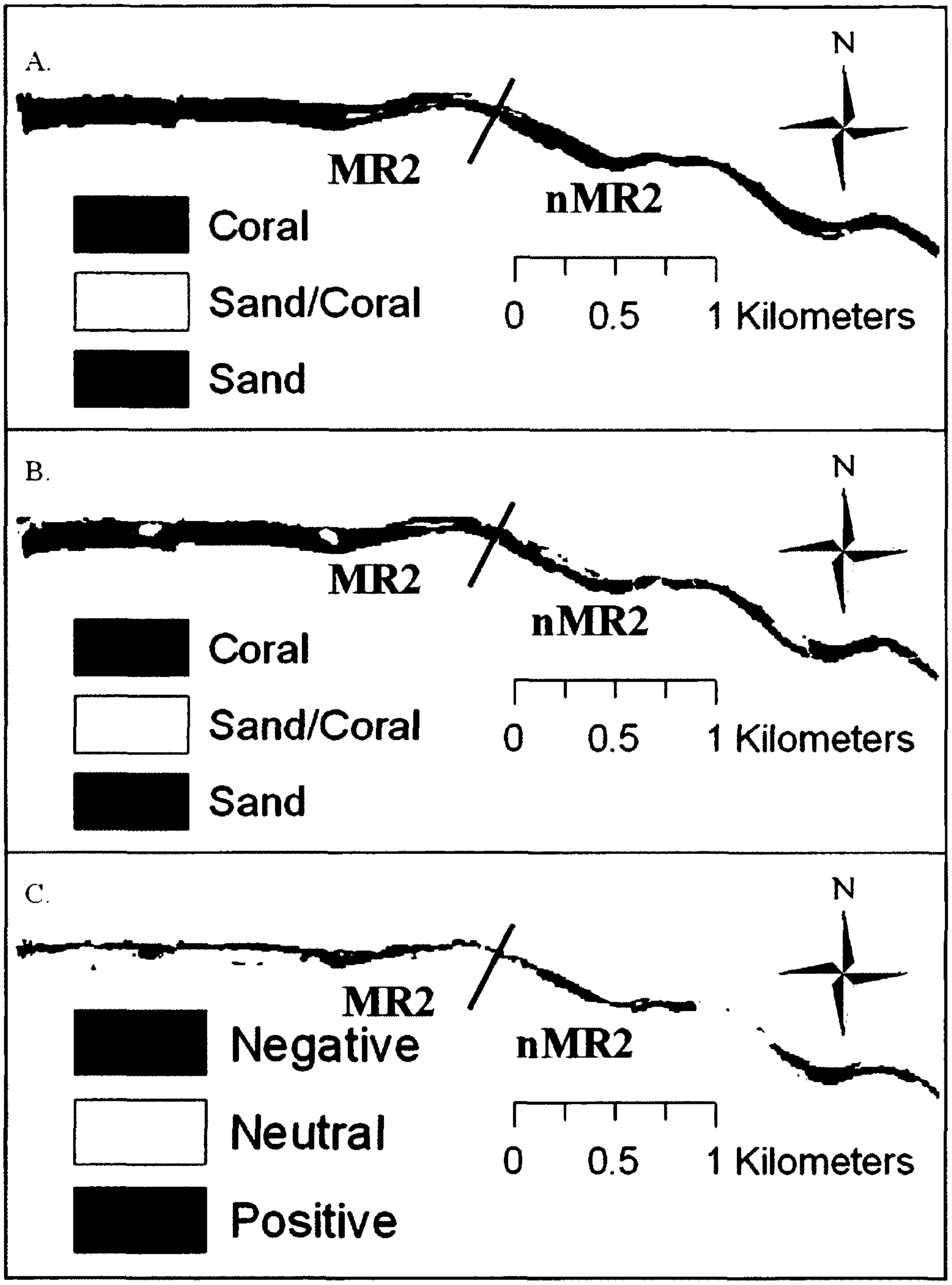

Figure 13. Coral, sand/coral mixture, and sand classes in the sheltered MR2 and nMR2 in the early 1980s (A.) and 2008-09 (B.) C. The negative, neutral and positive change values over the time period in MR2 and nMR2. (Locations of MR2 and nMR2 shown in Figure 4A.)

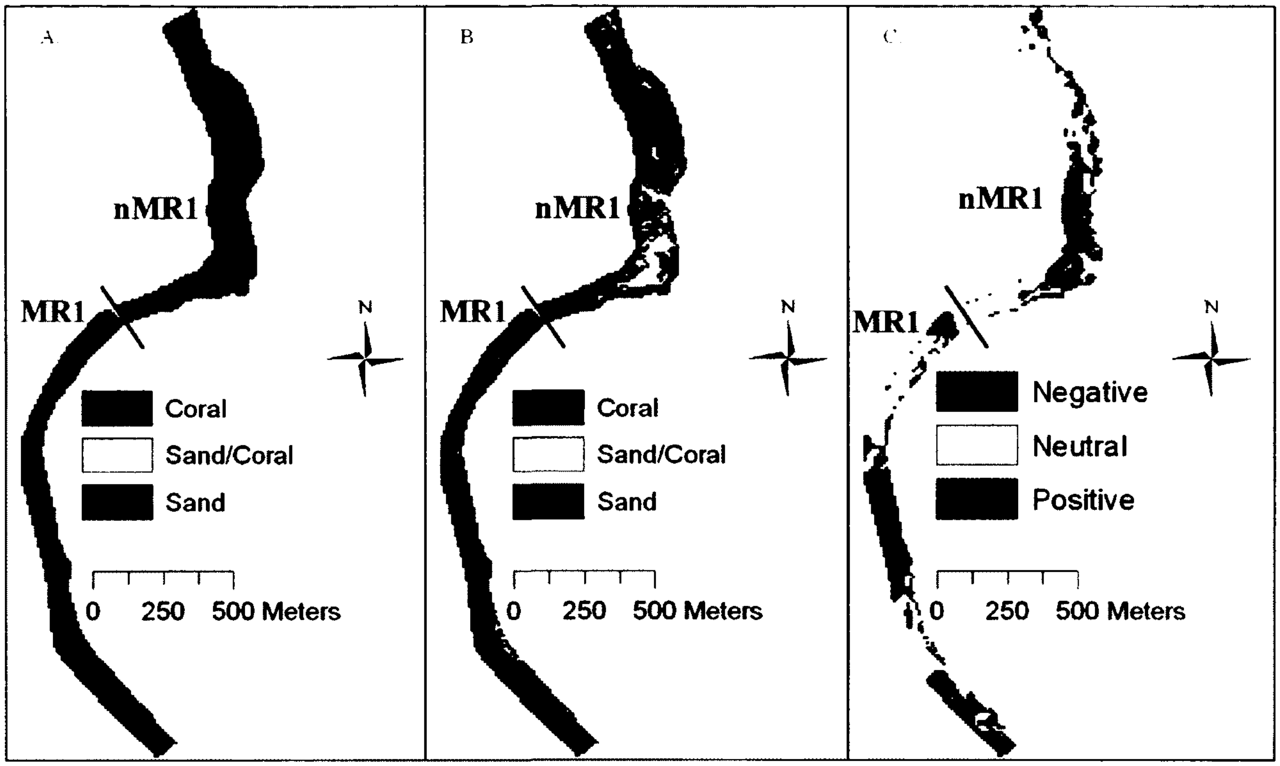

Figure 14. Coral, sand/coral mixture, and sand classes in the exposed MR1 and nMR1 in the early 1980s (A.) and 2008-09 (B.) C. The negative, neutral and positive change values over the time period in MR1 and nMR1. (Locations of MR1 and nMR1 shown in Figure 4A.)

Discussion

Although coral cover shallower than 10 m on Bonaire declined from 62% to only 30% between the early 1980s and 2008-09, Bonaire's reefs are experiencing less severe declines in coral cover than elsewhere in the Caribbean, which has seen declines of 80% in hard coral cover from about 50% to 10% in three decades (Gardner et al. 2003). This finding of current coral cover on Bonaire is similar to Steneck et al. (2011), who found 34-38% live cover at quadrats in 10 m of water. Deeper than 10 m, Bak et al. (2005) reported ~20% coral cover within permanent quadrats at 10-20 m depth. Areas of previously high coral cover examined in this study were replaced mostly by the sand class and the remaining coral has become increasingly patchy, with more small, less contiguous coral patches. We suggest this is largely the result of hurricanes that affected the island over the time period, which resulted in the breaking of large branching corals, particularly *Acropora palmata* and *A. cervicornis*, the rubble created by which can be seen by divers and snorkelers, particularly in the shore zone and shallow reef (< 5 m). Although Bonaire is technically outside of the hurricane belt, it has been affected by hurricanes that have passed north of the island, particularly Hurricane Lenny in 1999, which moved in a rare west-to-east direction across the Caribbean, which resulted in coral toppling, fragmentation, tissue damage, bleaching, and smothering along the coast of Bonaire (Bries et al. 2004). A subsequent study comparing QB satellite images from the years surrounding the hurricanes could potentially test this hypothesis. Fortunately, Bonaire has not seen an overgrowth of macroalgae in areas where coral has been lost and replaced by sand and rubble (Kramer 2003; pers. obs.), as has been documented elsewhere in the Caribbean (Bellwood et al. 2004; Bruno et al. 2009). It is important to note that this study concerns only the shallow reef, less than 10 m; while previous work using permanent quadrats has reported declining coral cover in the shallow reef (10-20 m), the deeper reef (30-40 m) has maintained the same level of coral cover over a 30 year time frame on Bonaire (Bak et al. 2005)

Although most of the reef experienced declines in coral cover, a non-negligible 16% of the reef did experience positive changes toward higher coral cover, and a large amount, 40%, remained unchanged between the early 1980s and 2008-09. It was initially surprising to us that the largest amount of positive change in coral cover was concentrated along the middle of the leeward coast, where the capital city of Kralendijk is located and most of the population resides. In contrast, the much less inhabited northern and southern leeward coasts experienced high levels of negative change. We expected to see more negative impacts concentrated around the population center due to nutrient inputs, sedimentation, and runoff as a result of development. A possible reason these negative impacts were not found where expected is that mapping by van Duyl (1985) may have occurred after damage had already taken place as a result of rapid building and development of the capital city of Kralendijk. Present development in the area is possibly more controlled and environmentally responsible. In addition, the area of the coastline sheltered by the neighboring island of Klein Bonaire and the sheltered shore of Klein were not found to have experienced as drastic a decrease in coral cover as the exposed side and they also became less patchy over the time period, with fewer and larger patches of coral. The negative changes elsewhere along the coast could be a result of local currents, which were not measured in the present study, or the exposure of the coastline resulting in increased vulnerability to storm damage. Given our findings, we suspect that coral cover may be more heavily impacted by storms, such as the previously mentioned hurricanes, than development on this small island of only around 15,000 inhabitants. The fact that the sheltered marine reserve and sheltered side of Klein Bonaire both experienced decreases in the number of coral patches and increases in patch area supports this hypothesis. Status as a marine reserve and sheltering from storm exposure may buffer against coral fragmentation. Work on cold and deep-water reefs, mainly *Lophelia pertusa*, has found that as a coral patch grows over time, water circulation can be cut off from the centre of the patch or colony, causing mortality and forming a ring-shaped colony known as a "Wilson-ring" (Wilson 1979; Rogers 2004; Wheeler et al. 2007). No such literature discusses the degradation of tropical coral patches over time. Surrounding Klein Bonaire the majority of coral loss is in the shallowest portions of the reef, along the shoreline (Figures 9 and 10). Elsewhere, long stretches of coral patches have been broken up along the coast over time, as in MR1, MR2 and nMR2 (Figures 13 and 14), and coral in the shallowest part of the reef, along the coast, has been lost.

It is not surprising that increases in the number of patches of coral were accompanied by overall declines in cover and decreases in the size of individual patches. Patches with small nearest neighbor distances are typically situated in landscapes containing more habitat than are patches with large nearest neighbor distances, so this measure of isolation is generally related to amount of total habitat in the landscape (Fahrig 2003), but is not always necessarily the case. ENN showed positive changes by declining over time in the sheltered marine reserve and in the exposed non-reserve site, but the sheltered non-reserve site experienced an increase in this value, with a larger number of smaller coral patches spaced farther apart from one another. Fragmentation *per se* implies a larger number of smaller patches; however, the fact that these changes, as well as the change in CONTIG and ENN values were not significant suggests that habitat fragmentation is less of an issue on Bonaire than habitat loss in general. Fahrig (2003) suggests that the term "fragmentation" be limited to the breaking apart of habitat, independent of habitat loss, this can happen on a reef when a large coral patch breaks apart at the center, but gains area along the outside edges, resulting in no net loss of total habitat; empirical evidence to date suggests that the loss of habitat has large negative effects on biodiversity. Recent studies have shown that a diversity of impacts can result from habitat fragmentation, it is unknown whether such impacts are the result of fragmentation itself, the total loss of habitat during fragmentation, degradation of the habitat after the fragments are isolated, or the effect of isolation itself (Caley et al. 2001). Most studies of habitat fragmentation in the marine environment have been in seagrass habitats (Eggleston et al. 1998, Hovel & Lipcius 2001, 2002). Shrimp are more abundant in small patches of seagrasses because a large PARA is important for feeding (Eggleston et al. 1998) and a greater number of invertebrate taxa occur in larger patches of seagrass habitat (Bowden et al. 2001). Other studies have reported reduced survival in fragmented habitats as a result of increased exposure to predators along the edges of habitat patches, i.e., a large PARA (Brittingham and Temple 1983, Andrén and Angelstam 1988). These effects of fragmentation likely vary greatly by species (Eggleston et al. 1998), particularly between invertebrates and fishes. Although loss of coral habitat on Bonaire is undoubtedly occurring, and the remaining available habitat is being broken into smaller patches, it is not possible to separate the effects of loss from fragmentation. Fahrig (2003) suggests that the effects of fragmentation *per se* may be greater in tropical systems than in temperature systems, but this prediction remains to be tested. Caley et al. (2001) represents an experimental study on a coral reef at a fine spatial scale and found habitat degradation to have a much greater detrimental impact than fragmentation, and the effects of fragmentation in the absence of loss and degradation to be either neutral or positive, and provides a useful and complementary approach to experiments at macro-landscape scales. Unfortunately, landscape-level analyses of coral cover are lacking and do not lend well to experimental manipulation, particularly given the current fragile state of coral reef ecosystems.

Landscape-level analyses such as this are useful for evaluating the success of marine policy and focusing future management decisions on areas of concern, as coral reef ecosystems continue to change faster than our current abilities to measure those changes. As a well-managed and long-established Marine Protected Area (MPA), the island of Bonaire seems to be doing better than elsewhere in the Caribbean. The work completed here can potentially be used to establish additional no-diving marine reserves by identifying areas that have maintained relatively high coral cover or have experienced increases in coral over the time period and also identify areas of concern that have not fared as well and may warrant future increased protection. Figure 9 shows an area on the east side of Klein Bonaire where some positive changes occurred and coral cover is relatively high, which may be an ideal place to establish an additional no-diving reserve to maintain this high level of coral. Given the increases in frequency and intensity of tropical storms and hurricanes in the Caribbean (Saunders and Lea 2008) this location would also be ideal as the island of Klein Bonaire shelters the shoreline. Threats to the coral reef habitat do not recognize the imaginary lines outlining an MPA; as threats are occurring at local, regional and global scales, they must be managed as such.

Conclusion

Although the island of Bonaire has suffered less drastic declines in coral cover than elsewhere in the Caribbean, habitat loss, independent of fragmentation, has been occurring since the early 1980s with coral habitat being replaced by sand. The remaining reef is becoming increasingly patchy, with large coral patches breaking apart into smaller, more disparate ones. Shoreline exposure and status as a no-diving marine reserve did not predict successful preservation of the reef habitat. This suggests that some other factor is driving declines in coral, perhaps at a larger scale.

It is unfortunate that more landscape-scale maps of coral reefs are not created, as they can be extremely useful for identifying areas of coral decline, as well as areas where coral cover has remained relatively high. Identifying areas of decline and resiliency are both useful for management, to concentrate efforts to protect threatened areas, as well as to conserve areas that are doing relatively well. The maps created here will be used by the management body of the Bonaire National Marine Park, STINAPA. Their monitoring efforts can focus on specific areas of concern as identified through the change detection techniques used herein.

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CHAPTER 4

Coral reef habitat fragmentation in protected vs. unprotected reef systems in the Dutch Caribbean

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Abstract

The islands of Bonaire and Curaçao in the Dutch Caribbean were both mapped along their leeward shore out to the 10 m depth contour in the early 1980s. Satellite techniques were used to create comparable maps for the 2007-09 time period and the maps were analyzed for changes in coral cover over the time period. The two islands were compared to one another for changes in coral cover including changes in patchiness of the reef habitat and their respective marine protected areas (MPAs), which vary in level of protection, were also compared. Although the island of Bonaire is more heavily protected than Curaçao, coral cover on Bonaire declined since the initial mapping while it remained the same on Curaçao. The coral habitat on both islands became increasingly patchy and fragmented, accompanied by an overall loss of habitat on Bonaire, but in the absence of overall loss on Curaçao. The Curaçao Underwater Park (CUP), which lacks as stringent enforcement as that on Bonaire, does not seem to be faring any better or worse than an adjacent unprotected reef with respect to overall coral cover. However, the CUP has shown a significant decrease in the area of individual patches and a related increase in patch perimeter-to-area ratios when compared to no-diving marine reserves on Bonaire. The difference between Bonaire and Curaçao could be due to the disparate effect of hurricanes on their respective coastline orientations. The results of this study have implications for future studies assessing the effects of fragmentation with and without overall loss of habitat.

Keywords marine reserve, Curaçao, Bonaire, habitat fragmentation, MPA

INTRODUCTION

The islands of Bonaire and Curaçao are part of the Dutch Caribbean and are located in the southern Caribbean Sea (Figure 1). In 1979 Bonaire established a Marine Protected Area (MPA), from the high-water mark out to the 60 m depth contour, making it one of the longest-established MPAs in the Caribbean. Bonaire's economy relies heavily on tourism and nearly 70% of tourists were SCUBA divers in 2008 (STINAPA Annual Statistics Report 2008). The Bonaire National Marine Park (BNMP) is completely self-sustained through fees paid by divers and non-divers to use the park and is managed by Stichting Nationale Parken Bonaire (STINAPA Bonaire). Within the BNMP it is prohibited to remove anything, alive or dead, with the exception of fish caught using traditional fishing practices. Spear fishing is also prohibited and good diving practices are required to maintain the integrity of the reef structure and its role as habitat. A series of permanent moorings are maintained by STINAPA for use by dive boats to avoid anchor-dropping, which is also prohibited. The BNMP also includes two no-entry marine reserves where underwater visitors are not permitted. Around the time the marine park was established Bonaire was home to a population of around 9,000 (1981 Census CBS-Netherlands Antilles), which grew to around 15,000 in 2008 (DEZA 2008).

In contrast to Bonaire, the island of Curaçao has a population of over 140,000 (Meteorological Service 2010) and has a more diverse economy, which does not rely entirely on tourism. The island is home to the Caribbean Research and Management of Biodiversity (CARMABI), a non-profit foundation established in 1955 as a marine research institute (CARMABI 2011), but statistics on the number of divers and dive site usage are not available. In 1983, the Curaçao Marine Park, also known as the Curaçao Underwater Park (CUP), was established from the high water mark out to the 60 m depth contour along 22 km of coastline. The protected area includes 600 ha of reef and 436 ha of inner bays (Figure 2). The park is currently a "paper" park, with no specific restrictions on activities in its waters and not officially stipulated as a marine park by law; it was a relatively unaffected area, valuable aesthetically and for its biological diversity (CARMABI 2011).

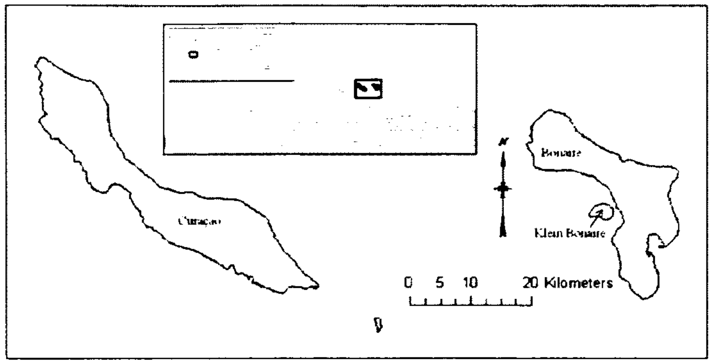


Figure 1. The islands of Bonaire and Curaçao are located in the Dutch Caribbean, about 80 km north of the coast of Venezuela.

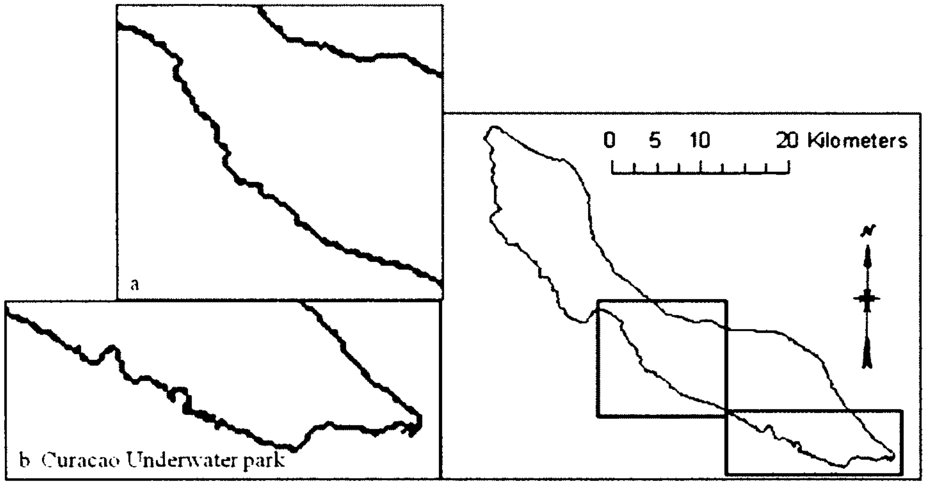


Figure 2. The locations of a. an area that is not a marine park, but is of comparable coastal size to b. the Curaçao Underwater Park on the island of Curaçao.

The reefs off the leeward coasts of both Bonaire and Curaçao were mapped in the early 1980s into dominant coral community type out to 10 m deep (van Duyl 1985). On Bonaire, van Duyl's mapping found coral to be the dominant bottom-type, making up around 62% of the benthos (Relles and Jones Chapter 3). The total amount of coral was made up of nearly 40% *Acropora cervicornis* and 40% head coral, with the most common head coral species being *Montastraea annularis, M. cavernosa, Diploria strigosa, Siderastrea siderea, S. radians, Dichocoenia stokesii, Colpophyllia natans, Porites astreoides, Meandrina meandrites* and *Stephanocoenia intersepta*. Head corals in this group were sometimes accompanied by the foliate *Agaricia* spp. and finger corals were also often present in this group (van Duyl 1985). Since van Duyl's mapping work was completed, *A. cervicornis* has dropped to nearly 0%, but significant increases in the head coral group have occurred at various locations along the coast (Relles and Patterson Chapter 1).

Recently, satellite remote sensing techniques have been used to evaluate coral cover at the landscape-scale in shallow reefs (Relles et al. 2012, Chapter 2; Mishra et al. 2005; Mishra et al. 2006a & b). These techniques can further be used to evaluate changes in coral cover over time in specific areas of interest. In general, areas closed to diving within the BNMP did not fare significantly better in terms of overall change in coral cover and coral fragmentation from the early 1980s until 2008-09 than did comparable areas where diving was permitted (Relles and Jones Chapter 3). However, the no-diving area that is located along a more sheltered coastline did experience an increase in coral cover over the time period, with fewer and larger patches of coral habitat. The sheltered shoreline of the uninhabited island of Klein Bonaire exhibited fewer and larger patches of coral as well (Relles and Jones Chapter 3). Scuba diving is generally considered a non-consumptive use of a reef because one diver's use of the resource does not prohibit use by future users, and it is therefore generally considered non-destructive, particularly when compared to other threats to the reef structure. However, recent research has shown the effects of recreational scuba divers to be detrimental to the coral reef structure (Zakai and Chadwick-Furman 2002; Davis and Tisdell 1995; Hawkins et al. 1999). This study looked at how coral cover has changed since van Duyl (1985) on the island of Curaçao using satellite remote sensing and geographic information system (GIS) techniques and specifically compared changes on Curaçao and in the Curaçao Underwater Park to the heavily-protected MPA that surrounds the island of Bonaire and its no-diving marine reserves.

MATERIALS AND METHODS

Study Sites

The island of Bonaire is located in the southern Caribbean Sea, approximately 80 km off the coast of Venezuela (12°10' N 68°17' W), while the island of Curaçao is located approximately 40 km west of Bonaire (12°7' N 68°56' W). This study focuses on the reefs off of the entire leeward coast of both Bonaire and Curaçao, including the small, uninhabited island of Klein Bonaire to the west of the main island of Bonaire (Figure 1).

Marine Protected Areas (MPAs)

The Curaçao Underwater Park (CUP) is located along the southern leeward coast of Curaçao along approximately 22 km of coastline out to the 60 m depth contour (Figure 2b), but for the purposes of this study was only mapped out to the 10 m depth contour. A comparable area of unprotected reef consisting of an adjacent 22 km of reef was used for comparison.

On Bonaire the furthest northwest marine reserve closed to divers was designated marine reserve number one (MR1) and the second marine reserve is located farther south along the coast and is sheltered by the northwestern portion of the island and was designated MR2 (Figure 3). The marine reserves on Bonaire were compared to the changes since van Duyl's mapping (1985) in the CUP.

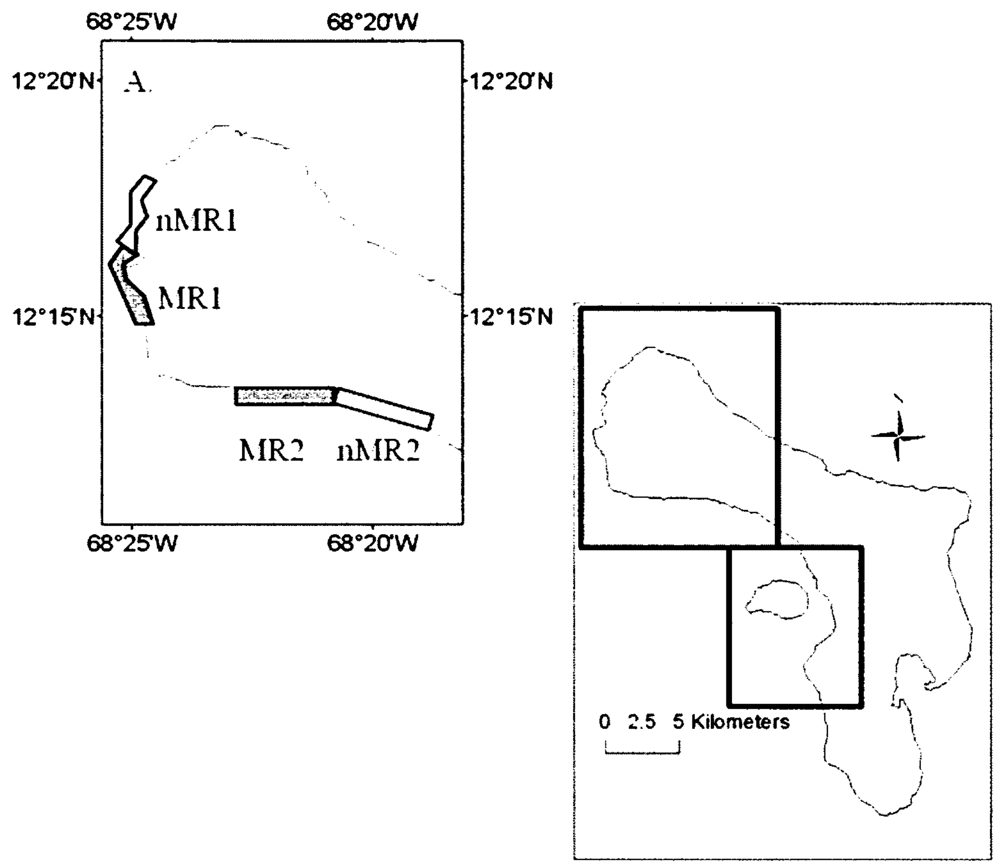


Figure 3. The locations of MR1 and MR2 marine reserves on Bonaire.

Baseline data

Maps of dominant coral community type and other bottom-types (e.g. sand, rubble, shore zone, and marine plants) on both islands were mapped in the early 1980s (van Duyl 1985) out to 10 m deep using aerial photographs and scuba diving. van Duyl (1985) described the shore zone map unit as being of heterogeneous composition, comprising composite patches with sand, coral debris, beach rock, and hard bottoms, often partly covered with encrusting organisms. The atlas was digitized into TIFF images and subsequently georectified using the geographic information system (GIS) software, ArcGIS 9.3 (ESRI). To align the maps with the coast on the satellite images, between 12 and 18 control points were identified using the georeferencing tool in ArcGIS, based on the relative location identified in van Duyl (1985; Figure 4) and shoreline features (e.g., coastal morphology, piers, and other permanent features; Figure 5). Based on the control points, ArcGIS computed residual error values, a measure of the fit between the true location on the image itself and the transformed locations of the output control points. Control points with the highest levels of error were then removed until the total root mean square (RMS) error, a statistical measure of the magnitude of variability between the shape of the original file and the shape of the georeferenced file, was less than 9 m (usually between 6 and 9 m), without dropping the total number of control points below six. The resulting benthic habitat maps were saved as raster files. Polygon vector shape files were drawn manually around each of van Duyl's bottom-types using the editor function of ArcGIS 9.3. After creation of these polygons based on the van Duyl (1985) maps, bottom-types were reclassified into the coarser class distinctions of the satellite classification (coral, sand, and a sand/coral mixture described below) as outlined in Table 1 for comparison, and converted to raster files of comparable resolution. While van Duyl mapped areas of the reef with coral bottom cover percentages ranging from 10-20, 20-40, and >40%, the satellite data set only classified pixels as coral if they contained greater than 20% coral. For this reason, areas considered to be 10-20% coral by van Duyl's classification were included in the sand/coral mixture class, which is a larger, more inclusive class that includes areas of soft coral cover, less than 20% hard coral, sea whips, and marine plants.

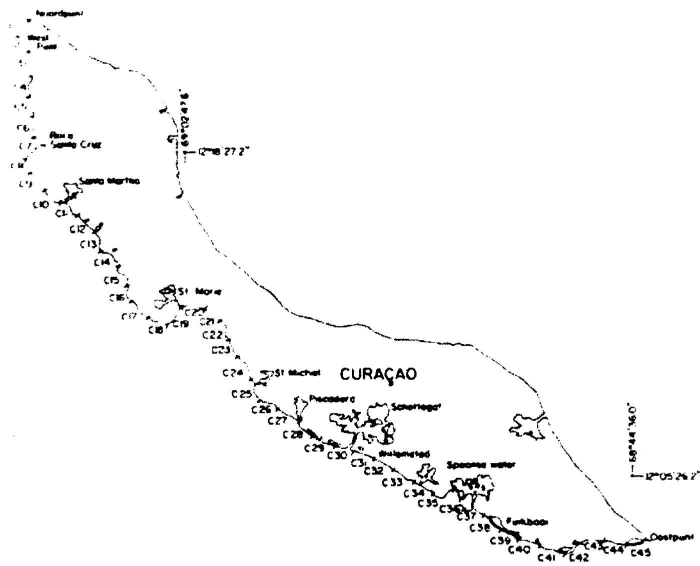


Figure 4. Relative location of the 45 maps created by van Duyl (1985) along the coast of Curaçao.

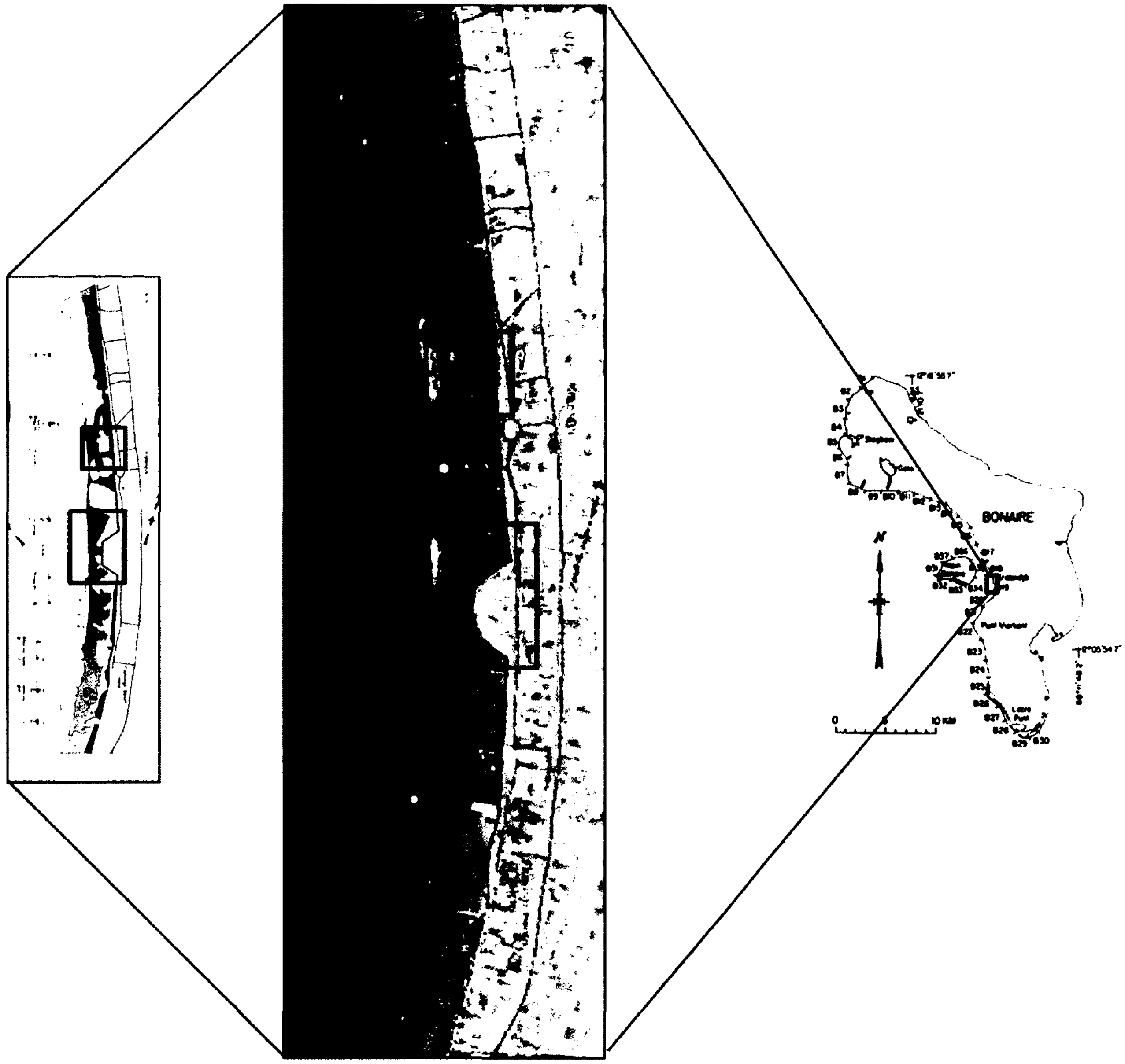


Figure 5. An example of one of van Duyl's (1985) maps (B19.; left rectangle) georeferenced with a modern satellite image (2009; right rectangle) in ArcGIS9.3 using the presence of piers (open black rectangles) along the coast. The map on the right indicates the relative location along Bonaire's shoreline.

Table 1. van Duyl's classifications on the left reclassified into the coarser classification system discernible by the satellite remote sensing method for the 2007-09 maps.

Van Duyl's class	**Coarse class system**
Sea whip	Sand/Coral
Acropora cervicornis	Coral
Acropora palmata	Coral
Finger/Foliate coral group	Coral
Head coral group	Coral
Plant	Sand/Coral
Rubble	Sand
Sand	Sand
Shorezone	Sand

Satellite-derived data

Benthic habitat maps were created from satellite remote sensing data using a classification technique described below. Multi-spectral, high resolution (2.4 m edge pixel) images from the QuickBird (QB) satellite from 2007, 2008, and 2009, hereafter referred to as 2007-09 data, were prepared and analyzed using methods outlined in Relles et al. (2012, Chapter 2), with modifications from P. Dash (Appendix B) used on the Curaçao data. This required a first-order atmospheric correction, which removed the effects of air molecules (Rayleigh scattering) and aerosols, and a water column correction, which removed the effects of particles in the water. It was necessary to develop a unique depth equation for Curaçao, using depth data from Sams et al. (2012) and radiance in the blue band (Figure 6). After these corrections were applied, the benthos were classified into either sand, coral, or a sand/coral mixture class based on habitat types that were ground truthed in the field by SCUBA using 200 random points overlaying 1 m x 1 m images of the bottom. Details on the algorithms for atmospheric and water column collections, as well as the classification system, are described in detail in Relles et al. (2012, Chapter 2). QB imagery has proven useful in coral reef habitat mapping at coarse habitat resolutions (3-4 classes; Mishra et al. 2006; Relles et al. 2012, Chapter 2) and was used to classify the 2007-09 benthos into coral, sand, and a sand/coral mixture class. The coral class included areas where coral cover was greater than 20%, while the sand class had greater than 50% sand cover. The sand/coral mixture class contained some mixture of less than 20% hard coral and less than 50% sand. Additional cover could be attributed to the presence of octocorals, marine plants, or dead coral with algae (Relles and Patterson Chapter 1). Several types of metrics can determine the accuracy of a classification. Overall accuracy is simply the sum of correctly labeled test sites divided by the total number of test sites, while user accuracy is the probability that a classified pixel actually represents that category on the ground (Mumby et al. 1997). The overall accuracy of the classification scheme was 75% (N = 364), with user accuracies of 91% and 50% for sand and coral classes, respectively (Relles et al. 2012, Chapter 2). For the purposes of classification and data manipulation, areas of coral were coded as 1, areas with the sand/coral mixture bottom-type were coded as 2, and areas of sand were coded as 3. Off the coast of Bonaire a total of 695 ha of reef along more than 50 km of leeward coastline and off of Curaçao 707 ha of reef along more than 60 km of leeward coast were mapped out to a depth of 10 m.

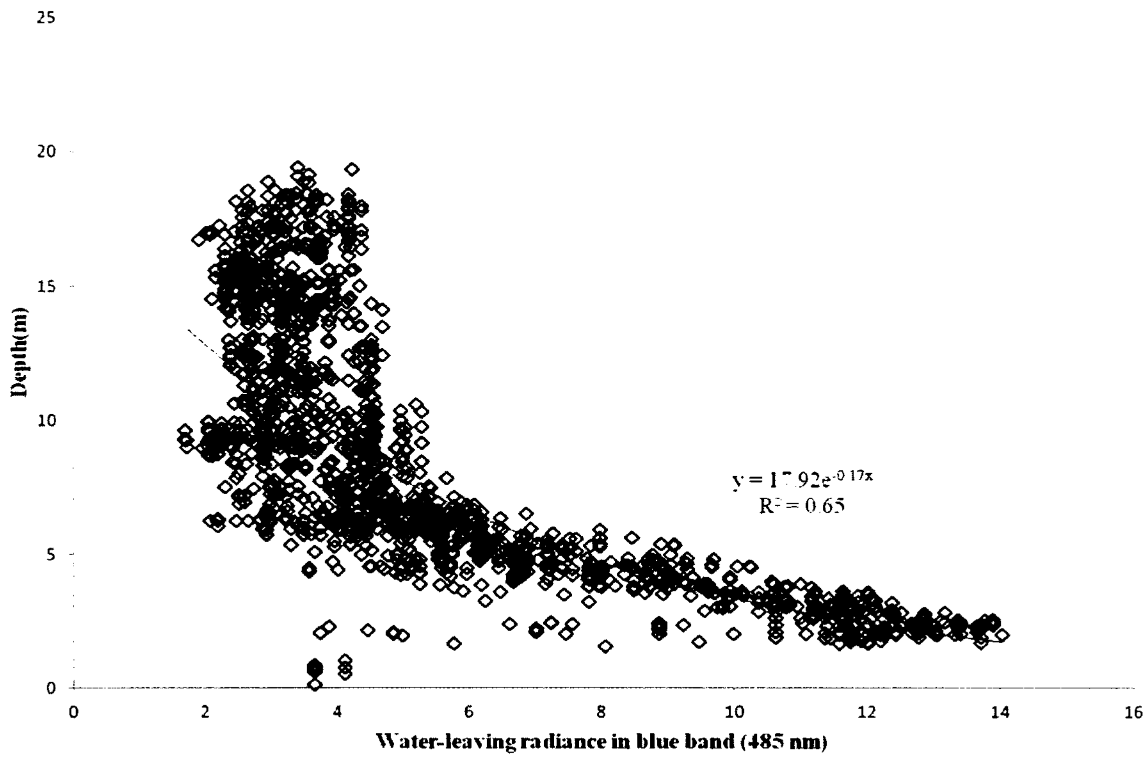


Figure 6. Model calibration dataset showing the regression between known depths and radiance in the blue band (485 nm) over pixels of greater than 20% live coral cover (y = 17.92e^(-0.17x), $R^2 = 0.65$).

Harmonization of data

Prior to comparison of the data sets from the early 1980s and 2007-09, it was necessary to adjust the resolution such that both data sets had the same resolution as the lowest resolution data set, in this case the van Duyl maps from 1985. The minimum mapping unit (MMU), which represents the minimum size of a polygon delineated by van Duyl (1985), and presumably the smallest habitat area discernible in the aerial photographs used to create the maps, was 9 m x 9 m (81 m^2). Both the early 1980s and the classified 2007-09 satellite images were then down-resolved from the original 2.4 m x 2.4 m pixels (5.8 m^2) by resampling the 2.4 m^2 pixels into 9.6 m^2 pixels using a majority rule.

Change Detection

The categories of coral, sand/coral, and sand were represented numerically as 1, 2, and 3, respectively, in the ones column for the satellite data set, hereafter referred to as 2007-09, and the tens column for the van Duyl data set, hereafter referred to as 1980s, with zero as a placeholder in the ones column. For classification and change detection purposes areas of coral were thus given a value of 1 in 2008 and 10 in 1985, areas of sand/coral mixture were assigned a value of 2 in 2008 and 20 in 1985 and areas of sand a value of 3 in 2008 and 30 in 1985. This representation of the data made it so that the two data sets could be added together and progression from the tens column to the ones column of the resulting sum would represent the change in bottom-type from 1985 to

2008 as can be seen in Table 2. Changes were quantified as positive, negative, or neutral. Change was considered positive when a pixel that was something other than coral changed to coral, it was also considered positive when an area previously dominated by sand became an area of sand/coral mixture. Negative changes were when coral changed to anything that was not coral, including when an area of sand/coral mixture changed to exclusively sand.

Table 2. Change values calculated in ArcGIS representing changes in bottom type between the early 1980s and 2008-09, distinguishing positive, negative and no change.

Value	Change (From-to)	Change (type)
11	Coral to Coral	None
12	Sand/coral to Coral	Positive
13	Sand to Coral	Positive
21	Coral to Sand/coral	Negative
22	Sand/coral to Sand/coral	None
23	Sand to Sand/coral	Positive
31	Coral to Sand	Negative
32	Coral/sand to Sand	Negative
33	Sand to Sand	None

Patch Dynamics

Raster data for both years and for both islands were analyzed using FragStats 3.3 (McGarigal et al. 2002b), which calculated patch, class and landscape metrics. A patch is defined as an area of similarly classified pixels, using an eight cell rule that takes into consideration all eight adjacent cells, including the four orthogonal and four diagonal neighbors, to determine patch membership. The classes in this case were coral, sand, and sand/coral mixture, as described above. In addition to calculating the number and size of patches, as well as relevant statistics, a contiguity index (CONTIG) and Euclidean Nearest Neighbor (ENN) value were also calculated. CONTIG is quantified in FragStats by convolving a 3x3 pixel template with a binary digital imagine in which the pixels within the patch of interest are assigned a value of 1 and the background pixels (all other patch types) are given a value of zero. Template values of 2 and 1 are assigned such that orthogonally contiguous pixels are weighted more heavily than diagonally contiguous pixels; the contiguity value for a pixel is the sum of the products of each template value and the corresponding input image pixel value within the nine cell neighborhood. CONTIG values range between zero and one, with large contiguous patches resulting in larger values, as opposed to smaller, more disparate patches (McGarigal et al. 2002). The isolation of patches of coral was measured using the ENN approach, the shortest straight-line distance between the focal patch and its nearest neighbor of the same class (McGarigal et al. 2002). Patch, class, and landscape metrics for the two data sets, 1980s and 2007-09, were analyzed statistically using the Mann-Whitney Rank Sum Test (non-normal data). Comparisons over time were made along the leeward coasts of both islands. A more detailed review of the changes observed only along Bonaire can be found in Relles & Jones (Chapter 3). The area that comprises the Curaçao Underwater Park was compared to an adjacent area along Curaçao's coast of equal size (Figure 2), that is not considered a marine park, as well as to Bonaire's two no-diving marine reserves.

RESULTS

Curaçao

In the early 1980s, 875 hectares of reef offshore of the leeward coast of Curaçao were mapped (van Duyl 1985). On Curaçao, nearly 34% of that area represented greater than 20% hard coral cover at the time (296 ha). Large areas of sand made up 39% (345 ha) and the remaining 27% was made up of a sand/coral mixture (234 ha), which included soft corals, as well as dead coral covered with algae, and other marine plants.

In 2007-09, 707 ha of reef were mapped on the leeward coast of Curaçao, the difference between the amount of area here and in the early 1980s can be attributed to cloud cover in the satellite data. Coral (>20% hard coral cover) covered 34% of the area (241 ha) on Curaçao, with sand (>50% cover) declining to only 20% of the area (141 ha). The largest percentage of the area, 46% on Curaçao was made of the sand/coral mixture class (325 ha), often accompanied by octocorals (e.g., sea whips and gorgonians), dead coral covered with algae, and marine plants.

Bonaire

In the early 1980s, 707 hectares of reef offshore of the leeward coast of Bonaire was mapped (van Duyl 1985). Sixty-two percent of this area represented greater than 20% hard coral cover at the time (441 ha), while areas of high sand cover made up nearly 32% (226 ha). The remaining 6% was composed of a sand/coral mixture (40 ha), which included soft corals, as well as dead coral covered with algae, and other marine plants.

In 2008-09, of 695 hectares of reef mapped, slightly greater than 30% of the 92.2 m^2 pixels represented areas of greater than 20% hard coral cover (210 ha). Sandy bottom (>50% cover) dominated over 50% of the reef, approximately 370 ha, while the remaining 17% of the reef (115 ha) was covered by a sand/coral mixture, often accompanied by octocorals (e.g. sea whips and gorgonians), dead coral covered with algae, and marine plants.

Change Detection

Curaçao

Areas that experienced no change over the time period made up the majority of the reef along Curaçao (47%; 343 ha), followed by areas of positive change on 34% of the reef (243 ha) and only 19% experiencing negative changes. Of areas previously dominated by hard coral, 58% remained coral, with 34% changing to the coral/sand mixture class and 7% changing to just sand. Areas previously dominated by a mixture of less than 20% hard coral and less than 50% sand mostly stayed a mixture (50%), 34% changed to more than 20% hard coral, and 16% changed to greater than 50% sand. Areas covered by more than 50% sand in the early 1980s changed mostly to a mixture of sand and coral (51%), 34% stayed sand, and 15% changed to more than 20% hard coral cover (Figure 7).

Looking specifically at the Curaçao Underwater Park (CUP), equal portions of the reef, 39%, underwent no change and positive change, while 22% experienced negative change. Of pixels previously dominated by coral, 47% stayed coral, while the other portion changed to the sand/coral mixture (45%) or to sand (8%). Most of what was the sand/coral mixture in the early 1980s remained as such in 2007-09 (57%), only 4% changed to sand, and 38% experienced a positive change to coral. The largest percentage of sand pixels in the CUP positively changed to either coral (19%) or the sand/coral mixture (61%), with only 20% remaining as sand (Figure 7).

The unprotected area adjacent to the CUP, and equal in length to it, also experienced equal parts neutral and positive change, with 45% of each and 10% negative change. Seventy-one percent of coral pixels remained coral, 25% changed to a mixture of sand and coral and 4% changed to sand. Of the sand/coral mixture pixels from the early 1980s, 39% changed to coral, 14% to sand and 47% remained a sand/coral mixture. Thirty-four percent of area previously dominated by sand remained sand, but the rest experienced positive change to the sand/coral mixture (52%) or to coral (14%; Figure 7).

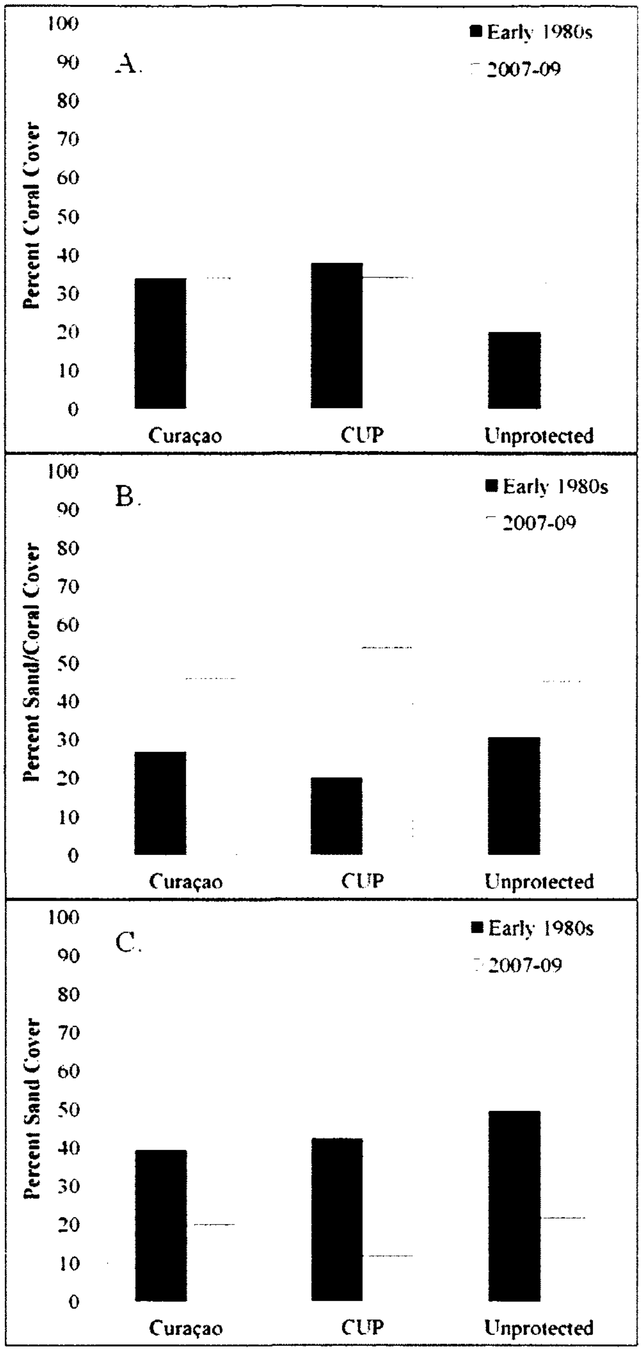


Figure 7. Changes in percent coral cover (A.), sand/coral mixture (B.) and sand (C.) between the early 1980s and 2008-09 on the island of Curaçao as a whole, in the Curaçao Underwater Park (CUP) and an adjacent area of unprotected reef equal in length along the coastline.

Bonaire

Areas of no change made up 45% of the total reef area and negative change occurred on 43% of the reef, while areas of positive change were only found on 12% of the reef. For areas previously dominated by coral, 47% became sand, while 36% stayed coral. The largest percentage of sand pixels from the 1980s data set remained sand in 2008-09 (67%), while 18% became coral and 15% became a sand/coral mixture. The largest percentage of the sand/coral mixture pixels changed to sand in 2008-09 (43%), 24% remained sand/coral and 34% became coral (Figure 8).

In the exposed marine reserve on the island of Bonaire (MR1) coral cover decreased over the time period from almost 91% in the early 1980s to less than 42% in 2007-09. Sand increased from only 9% to greater than 57%. The sand/coral mixture class was nonexistent in the early 1980s in MR1, but took up 1% of the area in 2007-09. In contrast to MR1, the sheltered marine reserve (MR2) increased in coral cover over the time period from 66% to greater than 83%. Sand decline from 31% to 6% and the sand/coral mixture class increased slightly from 4% to 11% (Figure 8).

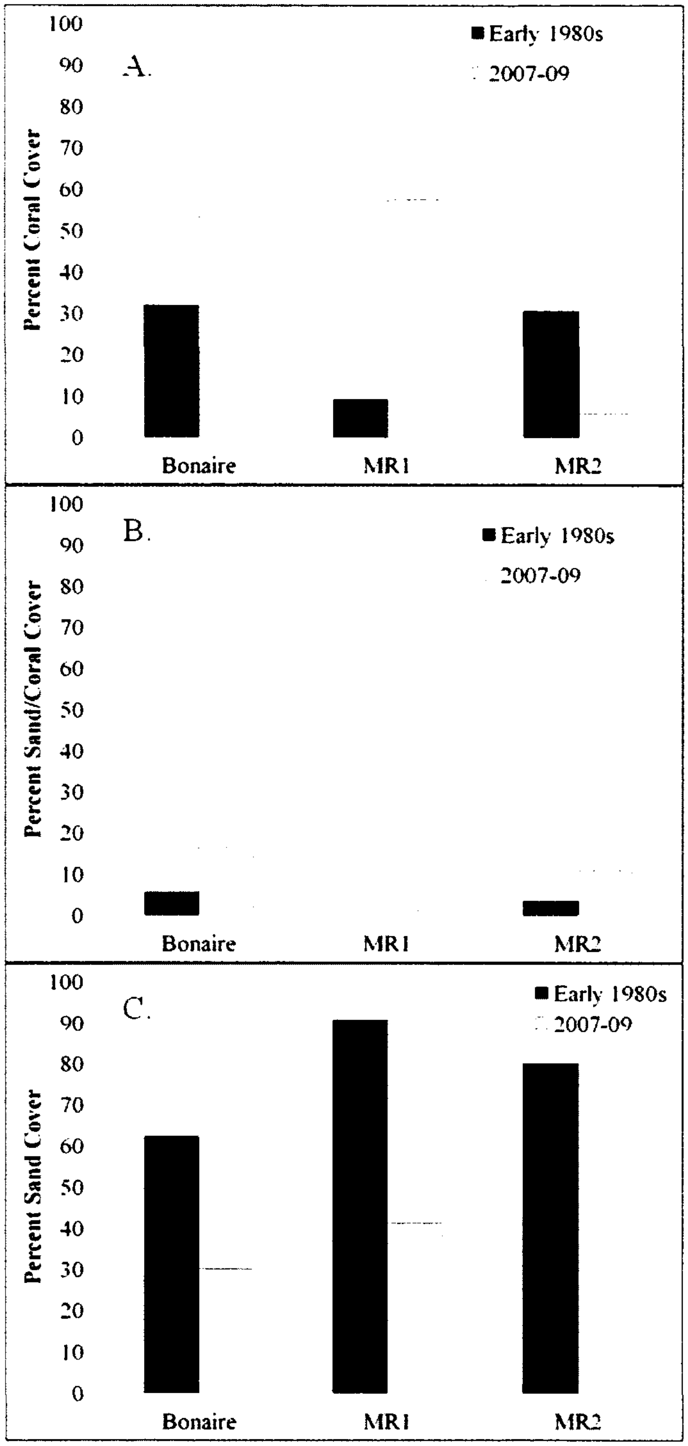


Figure 8. Changes in percent coral cover (A.), sand/coral mixture (B.) and sand (C.) between the early 1980s and 2008-09 on the island of Bonaire as a whole, in the exposed marine reserve site (MR1) and the sheltered marine reserve site (MR2).

Whole Island Patch Dynamics

Curaçao

The percentage of area covered by greater than 20% coral remained the same at 34% from the early 1980s to 2007-09, while the number of patches of coral increased from 259 to 495 (Figure 9.) Patch area, perimeter-to-area ratio (PARA), contiguity index (CONTIG) and Euclidean nearest neighbor (ENN) all failed the ANOVA assumptions of normality and equal variances. The Mann-Whitney Rank Sum Test found the decline in patch area from 1.14 ha to 0.52 ha to be significant ($U_{495,259} = 40933.5$, $p < 0.001$; Figure 10A.). The increase in PARA from a mean value of 2205.86 m ha^{-1} to 3052.17 m ha^{-1} was significant ($U_{495,259} = 37544.5$, $p < 0.001$; Figure 10B.), as was the decline in CONTIG from 0.44 to 0.24 ($U_{495,259} = 36805.5$, $p < 0.001$; Figure 10C.). The decrease in ENN from 35.42 m to 33.88 m was also significant ($U_{495,259} = 58182.5$, $p = 0.04$; Figure 10D.).

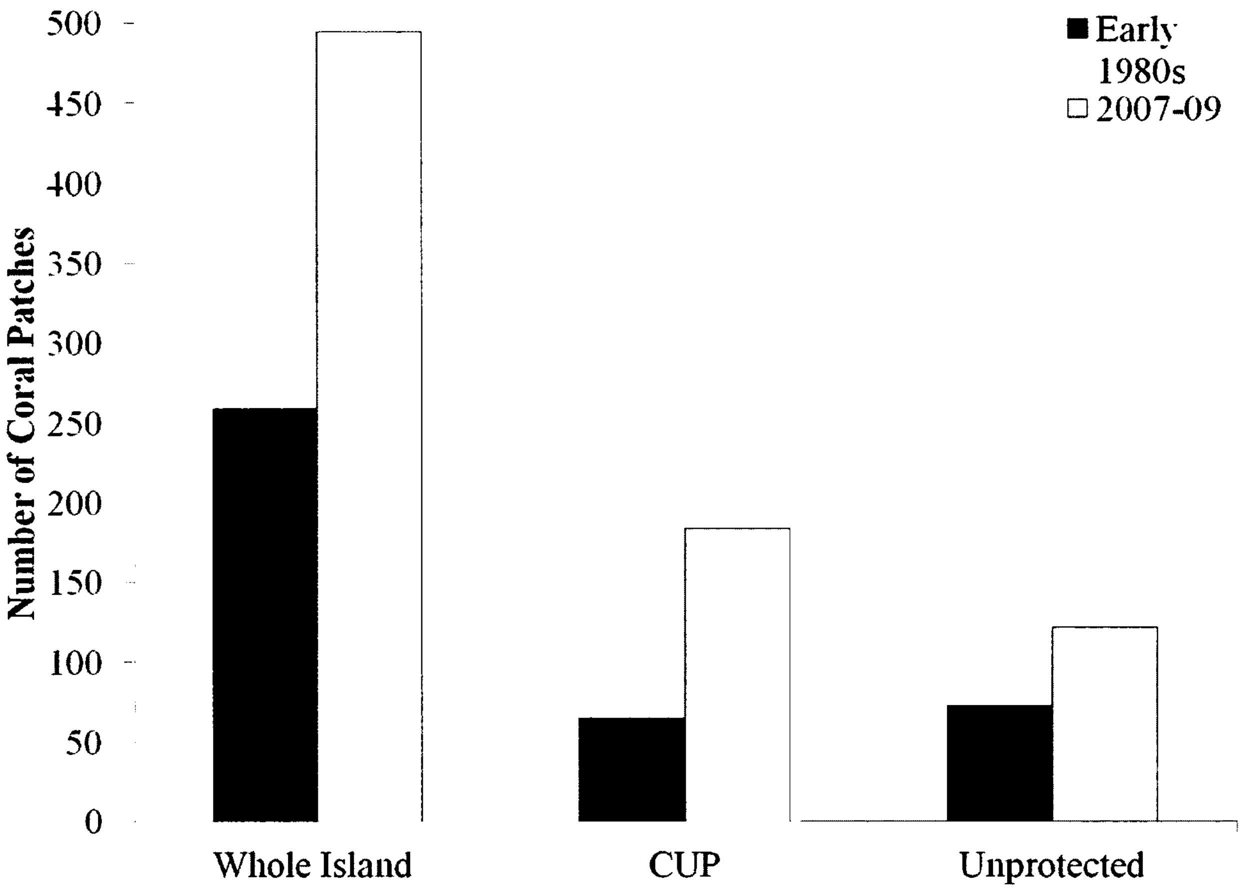


Figure 9. Changes in the number of coral patches between the early 1980s and 2007-09 on the whole island of Curaçao, in the Curaçao Underwater Park, and in the adjacent unprotected reef.

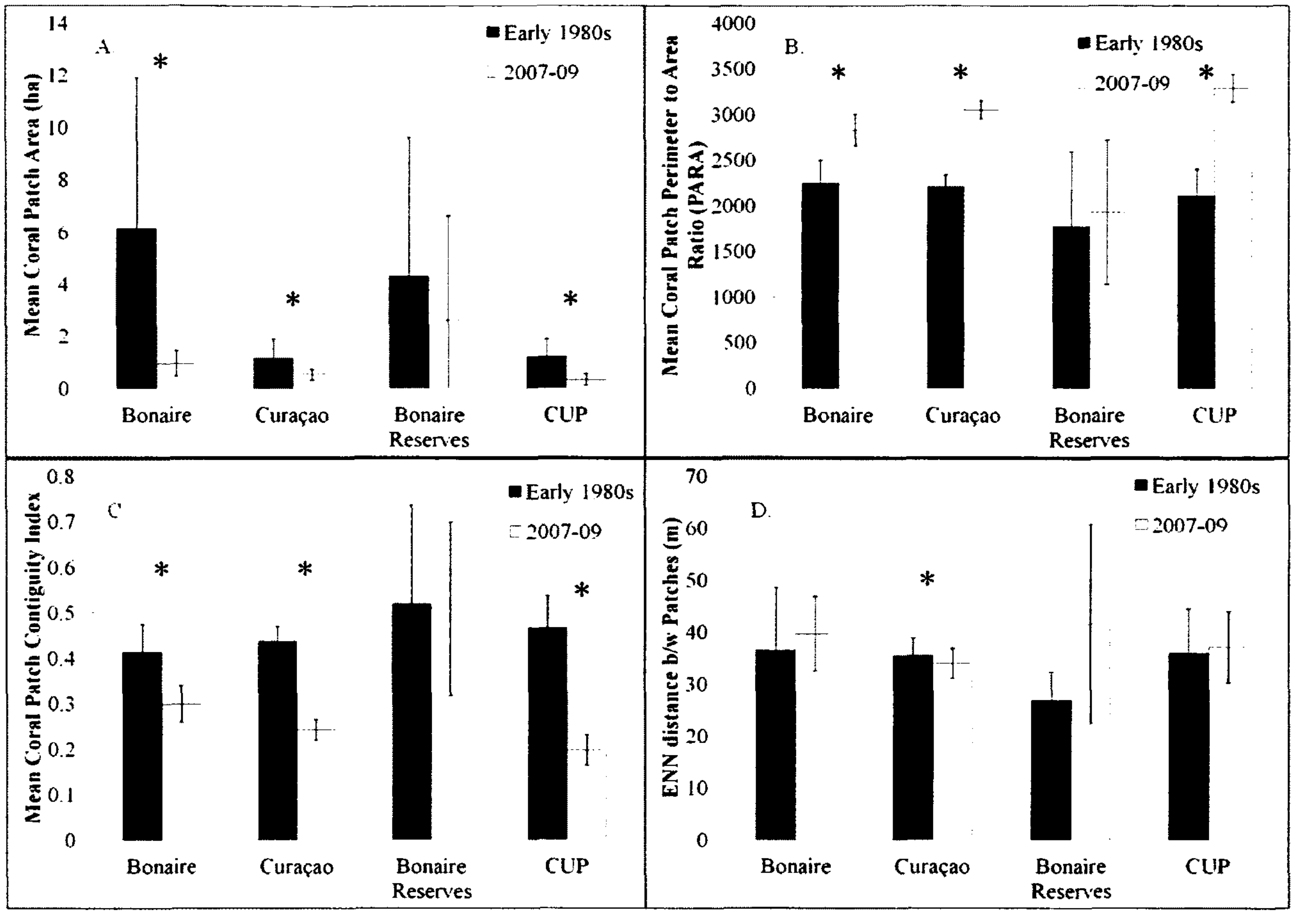


Figure 10. Changes in the patch parameters area (A.), PARA (B.), contiguity index (C.) and ENN distance (D.) between the early 1980s and 2008-09 on the islands of Bonaire and Curaçao, in the two no-diving marine reserves on Bonaire, and in the Curaçao Underwater Park. * indicates a significant difference between time periods. Error bars indicate +/- two standard errors.

Bonaire

While the percentage of area covered by coral declined from 62% in the 1980s to 30% in 2008-09, the number of patches of coral increased from 72 to 221 (Figure 11). Patch area, PARA, CONTIG and ENN were all found to fail the test of normality (Shapiro-Wilkes), so they were compared using the non-parametric Mann-Whitney Rank Sum Test. Mean patch size decreased from 6.12 ha to 0.95 ha ($U_{221,72} = 6035.00$, p = 0.002; Figure 10A.). The PARA increased from 2247.87 to 2827.34 ($U_{221,72} = 5838.50$, $p < 0.001$; Figure 10B.). The CONTIG decreased from 0.41 to 0.30 ($U_{221,72} = 5959.50$, p = 0.001; Figure 10C.). The Euclidean Nearest Neighbor (ENN; $U_{221,72} = 7826.50$, p =0.83; Figure 10D.) values were not significantly different between years.

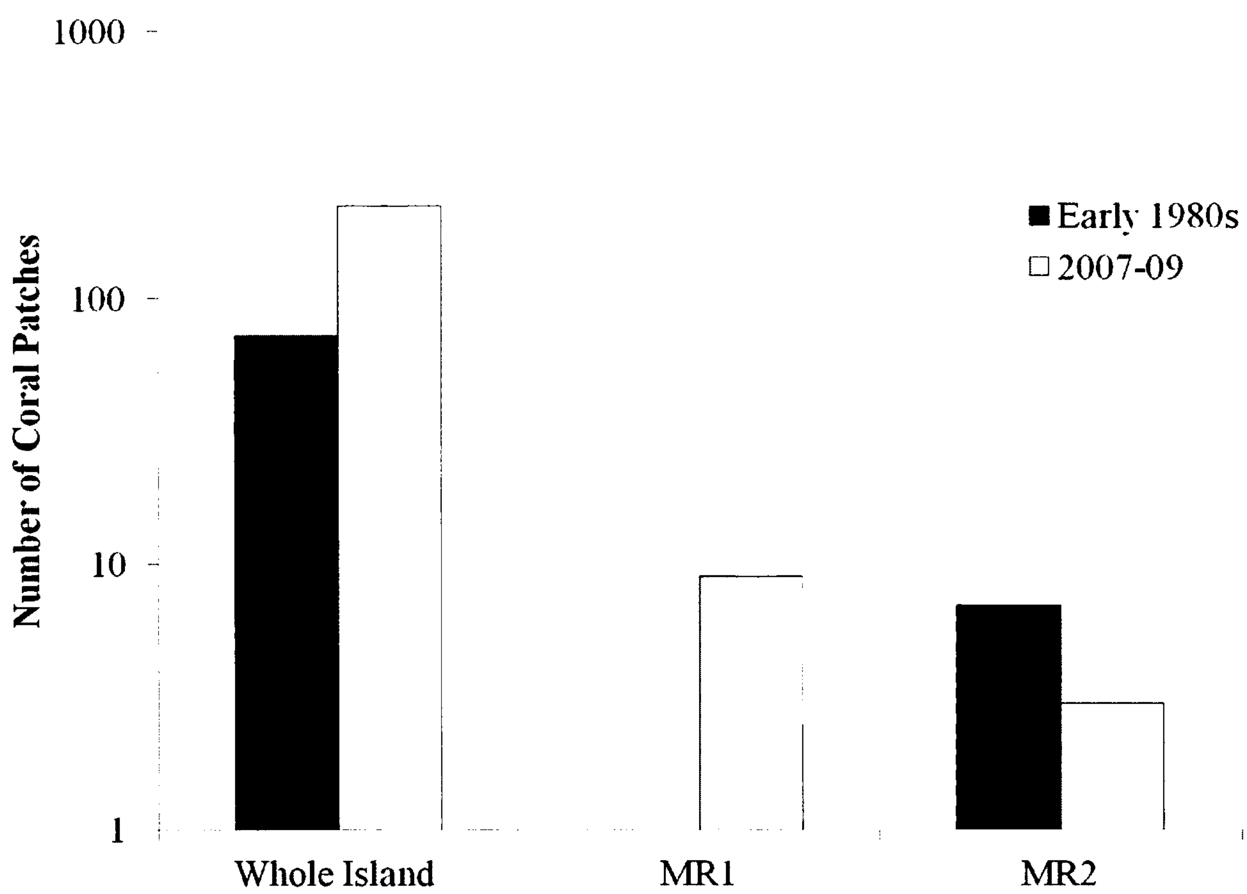


Figure 11. Changes in the number of coral patches between the early 1980s and 2008-09 on the main island, and in the marine reserves on Bonaire. Note: log scale and only one coral patch in the early 1980s in MR1.

Curaçao vs. Bonaire

In the early 1980s there were 72 coral patches along the leeward coast of Bonaire and surrounding Klein Bonaire, which increased to 221 total coral patches in 2008-09 (Figure 11). Separating the patch metrics of area, PARA, CONTIG and ENN into groups based on island and year gave the following four combinations: Curaçao 1980s ($N = 259$), Curaçao 07-09 ($N = 495$), Bonaire 1980s ($N=72$), and Bonaire 07-09 ($N = 221$). The data were not normally distributed so they were analyzed using the Kruskal-Wallis Test, which found that area, PARA, and CONTIG were significantly different among the four groups ($p < 0.00005$), but ENN was not ($H = 6.93$, $DF = 3$, $p = 0.07$; Figure 10; Table 3). The Mann-Whitney Rank Sum Test was used to determine which island and year combinations were significantly different from one another for each of the variables area, PARA and CONTIG; because there were six pairwise comparisons of the four combinations, the alpha level of significance was adjusted accordingly by dividing it, 0.05, by six, resulting in an α of 0.0083 (Bonferroni adjustment). For the three variables Area, PARA and CONTIG all comparisons were significant ($p < 0.0083$; Table 4) except for the pairwise comparison of Bonaire to Curaçao in the early 1980s and Bonaire to Curaçao in 2007-09.

Table 3. The results of the non-parametric Kruskal-Wallis test comparing the two different islands between years with H factors reported after adjusting for ties. Significant p-values are indicated with an *, df = 3.

	Area	PARA	CONTIG	ENN
H	75.62	100.81	102.40	6.93
P-value	<0.00005*	<0.00005*	<0.00005*	0.074

Table 4. The results of the Mann-Whitney Rank Sum Test pairwise comparisons.

Significant p-values at the $\alpha = 0.0083$ level are indicated with an *.

	Area		PARA		CONTIG	
	U-stat	p-value	U-stat	p-value	U-stat	p-value
Bon 80s vs. Bon 07-09$_{(72,221)}$	6035.0	0.002*	5838.5	<0.001*	5959.5	0.001*
Bon 80s vs. Cur 80s$_{(72,259)}$	9082.0	0.74	9039.5	0.69	8811.5	0.48
Bon 80s vs. Cur 07-09$_{(72,495)}$	11996.0	<0.001*	10700.0	<0.001*	10985.5	<0.001*
Bon 07-09 vs. Cur 80s$_{(221,259)}$	21077.5	<0.001*	20541.5	<0.001*	20289.0	<0.001*
Bon 07-09 vs. Cur 07-09$_{(221,495)}$	51386.5	0.19	50011.5	0.06	50252.5	0.08
Cur 80s vs. Cur 07-09$_{(259,495)}$	40933.5	<0.001*	37544.5	<0.001*	36805.5	<0.001*

Marine Protected Area (MPA) Patch Dynamics

Curaçao

In the early 1980s there were 96 coral patches in the Curaçao Underwater Park (CUP), which increased to 421 patches in 2007-09 (Figure 9). This was accompanied by no change in overall coral cover, which remained at 35% (Figures 7A. & 13). The unprotected area adjacent to and of comparable size to the CUP experienced an increase in coral cover from 19% to 34% (Figures 7A. & 14), which included an increase in the number of patches from 108 to 214 (Figure 9). Separating the patch metrics of area, PARA, CONTIG, and ENN into groups based on year and location gave the following four combinations: CUP 1980s ($N = 96$), CUP 07-09 ($N = 421$), unprotected 1980s ($N = 108$), and unprotected 07-09 ($N = 214$). The data were not normally distributed so it was analyzed using the Kruskal-Wallis Test, which found that area, PARA, and CONTIG were significantly different among the four groups ($p < 0.00005$), but ENN was not ($H = 3.53$, $DF = 3$, $p = 0.317$; Table 5; Figure 12). The Mann-Whitney Rank Sum Test was used to determine which location and year combinations were significantly different from one another for each of the variables area, PARA and CONTIG; because there were six pairwise comparisons of the four combinations the alpha level of significance was adjusted accordingly by dividing it, 0.05, by six, resulting in an α of 0.0083 (Bonferroni adjustment). For all three variables, all pairwise comparisons were significantly different from one another ($p < 0.00005$) except CUP 1980s was not significantly different from unprotected 1980s and CUP 2007-09 was not significantly different from unprotected 2007-09 ($p > 0.0083$; Figure 12; Table 6).

Table 5. The results of the non-parametric Kruskal-Wallis test comparing the CUP to the adjacent unprotected reef between years with H factors reported after adjusting for ties. Significant p-values are indicated with an *, df = 3.

	Area	PARA	CONTIG	ENN
H (adjusted for ties)	66.73	78.86	78.37	9.96
P-value	<0.00005*	<0.00005*	<0.00005*	0.02*

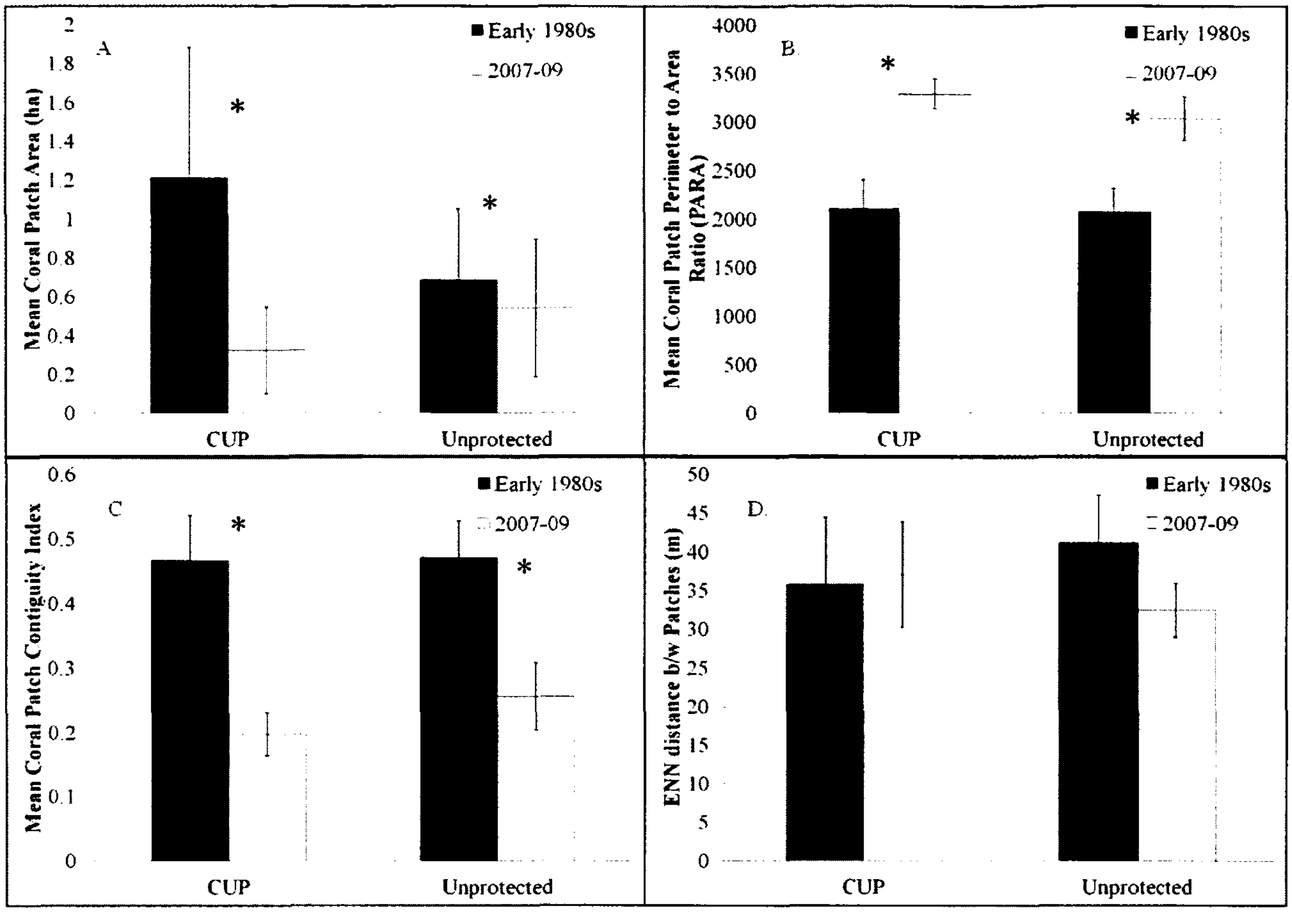
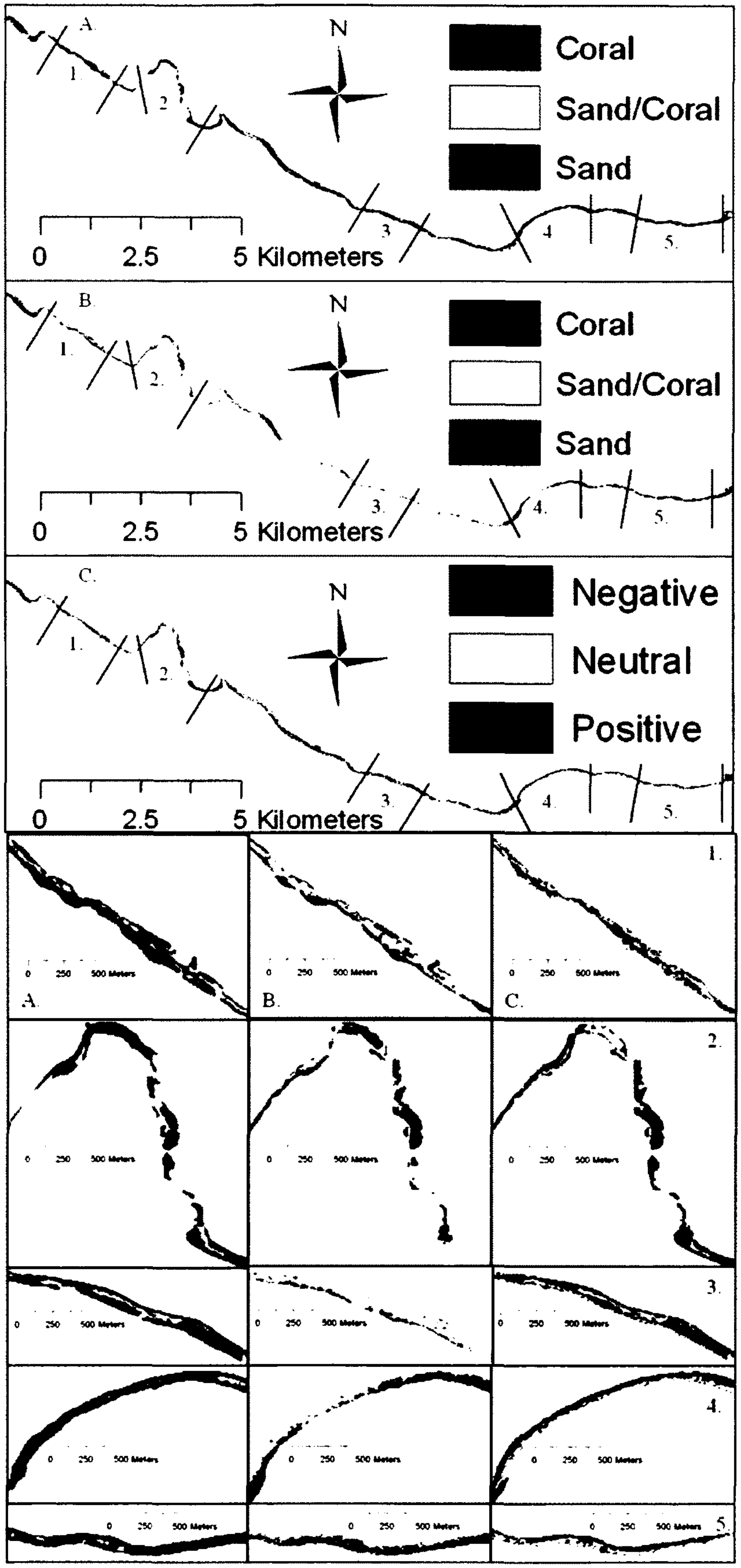


Figure 12. Changes in the patch parameters area (A.), PARA (B.), contiguity index (C.) and ENN distance (D.) between the early 1980s and 2008-09 in the Curaçao Underwater Park and an adjacent area of unprotected reef. * indicates a significant difference between time periods. Error bars indicate +/- two standard errors.

Table 6. The results of the Mann-Whitney Rank Sum Test pairwise comparisons.

Significant p-values at the $\alpha = 0.0083$ level are indicated with an *.

	Area		PARA		CONTIG	
	U-stat	p-value	U-stat	p-value	U-stat	p-value
CUP 80s vs. CUP 07-09$_{(96,421)}$	3082.5	<0.001*	2760.5	<0.001*	2732.0	<0.001*
CUP 80s vs. Unprotected 80s$_{(96,108)}$	2370.5	1.0	2312.5	0.80	2322.5	0.83
CUP 80s vs. Unprotected 07-09$_{(96,214)}$	2369.5	<0.001*	2343.5	<0.001*	2297.0	<0.001*
CUP 07-09 vs. Unprotected 80s$_{(421,108)}$	3137.0	<0.001*	2807.5	<0.001*	2815.5	<0.001*
CUP 07-09 vs. Unprotected 07-09$_{(421,214)}$	10569.5	0.37	10292.0	0.19	10392.0	0.26
Unprotected 80s vs. Unprotected 07-09$_{(108,214)}$	2545.0	<0.001*	2515.5	<0.001*	2483.5	<0.001*



A.
N
Coral
Sand/Coral
Sand
0 2.5 5 Kilometers
B.
C.
Negative
Neutral
Positive

Figure 13. A portion of the Curaçao Underwater Park in the early 1980s (A.) and 2007-09 (B.). The negative, neutral and positive change values over the time period (C.). The numbers 1-5 in the maps on the bottom indicate close-ups of areas as indicated on the larger map on top.

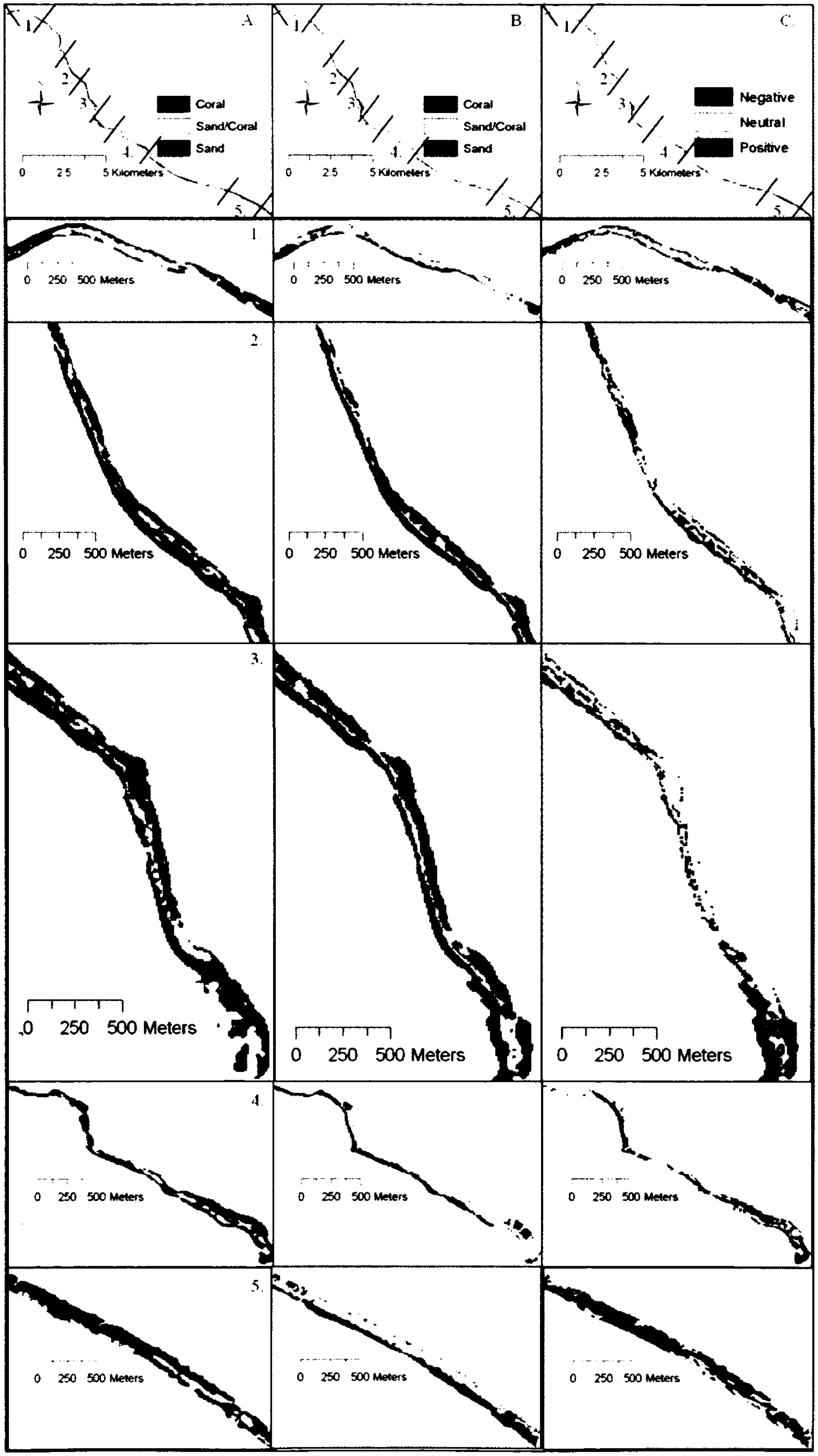

A.
B.
C.
Coral
Sand/Coral
Sand
Negative
Neutral
Positive
0 2.5 5 Kilometers
0 250 500 Meters
1.
2.
3.
4.
5.

Figure 14. A portion of the unprotected reef adjacent to the Curacao Underwater Park in the early 1980s (A.) and 2007-09 (B.). The negative, neutral and positive change values over the time period (C.). The numbers 1-5 in the maps on the top indicate close-ups of areas as indicated on the maps on the bottom.

Curaçao vs. Bonaire

In the early 1980s there were 6 coral patches within the two marine reserves on Bonaire, which increased to 21 total coral patches in 2007-09 (Figure 11). Separating the island and year combinations into groups gave the following four combinations: CUP 1980s ($N = 96$), CUP 07-09 ($N = 421$), Bonaire 1980s ($N=6$), and Bonaire 07-09 ($N = 21$) on which to compare the patch metrics area, PARA, CONTIG, and ENN. The data were not normally distributed so they were analyzed using the Kruskal-Wallis Test, which found that area, PARA, and CONTIG were significantly different between the four groups ($p < 0.00005$), but ENN was not ($H = 4.08$, $DF = 3$, $p = 0.253$; Table 7). The Mann-Whitney Rank Sum Test was used to determine which island and year combinations were significantly different from one another for each of the variables area, PARA and CONTIG; because there were six pairwise comparisons of the four combinations the alpha level of significance was adjusted accordingly by dividing it, 0.05, by six, resulting in an α of 0.0083 (Bonferroni adjustment). For the three variables Bonaire 1980s and CUP 1980s were significantly different from CUP 07-09, with area and CONTIG significantly less and PARA significantly greater. For the variable PARA Bonaire 07-09 was also significantly different from CUP 07-09, with PARA significantly greater in the CUP in 2007-09 than on Bonaire (Table 8; Figure 10).

Table 7. The results of the non-parametric Kruskal-Wallis test comparing the protected areas on two different islands between years with H factors reported after adjusting for ties. Significant p-values are indicated with an *, df = 3.

	Area	PARA	CONTIG	ENN
H	42.02	56.84	54.79	1.02
P-value	<0.00005*	<0.00005*	<0.00005*	0.80

Table 8. The results of the Mann-Whitney Rank Sum Test pairwise comparisons. Significant p-values at the $\alpha = 0.0083$ level are indicated with an *. Note: when comparing the marine reserves on Bonaire in the early 1980s to 2007-09 the data were normally distributed for the variables PARA and CONTIG so a simple t-test was used and is reported as opposed to the *U*-stat.

	Area		PARA		CONTIG	
	U-stat	p-value	U-stat	p-value	U-stat	p-value
CUP 80s vs. CUP 07-09$_{(96,421)}$	3082.5	<0.001*	2760.5	<0.001*	2732.0	<0.001*
CUP 80s vs. Bon Reserves 80s$_{(96,6)}$	253.5	0.92	220.5	0.49	229.0	0.59
CUP 80s vs. Bon Reserves 07-09$_{(96,21)}$	387.0	0.97	342.0	0.50	347.0	0.55
CUP 07-09 vs. Bon Reserves 80s$_{(421,6)}$	356.0	0.01	231.5	<0.001*	260.5	0.001*
CUP 2007-09 vs. Bon Reserves 07-09$_{(421,21)}$	0.0	<0.001*	490.0	<0.001*	514.0	0.001*
Bon Reserves 80s vs. Bon Reserves 07-09$_{(6,21)}$	48.0	0.97	t = 0.26	0.80	t = 0.07	0.95

DISCUSSION

Coral cover on Bonaire was initially higher than on Curaçao during the time of van Duyl's analysis (1985) and while it declined on Bonaire since then, it remained the same on Curaçao in the shallow (< 10 m) reef area mapped here. Coral cover on both islands became increasingly patchy since the initial mapping, with significantly larger, more contiguous patches in the early 1980s, compared to smaller patches, with greater perimeter-to-area ratios and lower contiguity indices in 2007-09.

On the island of Curaçao, the designation of the Curaçao Underwater Park as a "paper park" is apparent; although it experienced only a slight decline in coral cover, an adjacent area of reef that was not designated as a marine park experienced an increase in cover over the time period. The patchiness of the reef in the CUP was not significantly different from the unprotected area in the early 1980s or 2007-09, but both became increasingly patchy, with smaller, less-contiguous patches that had greater perimeter-to-area ratios. The same trends in patches were seen in Bonaire's no-diving marine reserves, but the changes were not significant (Relles and Jones Chapter 3). On Curaçao the fragmentation of the reef in these two areas appears to be occurring in the absence of overall coral loss, whereas on Bonaire habitat fragmentation is less of an issue than habitat loss in general (Relles and Jones Chapter 3). The results of fragmentation have been studied in various marine habitats; the loss of kelp habitat for reef fishes is more important than fragmentation (Deza and Anderson 2010). The literature is inconclusive on the effects of habitat fragmentation without loss, in most cases it is difficult to separate the two; Caley et al. (2001) suggests that rather than negative, habitat fragmentation in the absence of loss is usually either positive or neutral. Given the existing baseline data on Bonaire and Curaçao, and the fragmentation with and without loss on the islands, respectively, this might represent an ideal location to test the effect of fragmentation *per se* on the quality of coral habitat, which Fahrig (2003) suggests may be greater in tropical than in temperate systems.

Although coral habitats on Bonaire and Curaçao, both inside and outside of designated marine protected areas (MPAs), are becoming more fragmented, the remaining patches do not appear to be increasingly isolated as indicated by the lack of significance in changes in the Euclidean Nearest Neighbor (ENN) distance between patches. Most studies of habitat fragmentation in the marine environment have been in seagrass habitats (Eggleston et al. 1998, Hovel & Lipcius 2001, 2002). Shrimp are actually more abundant in small patches of seagrasses because a large PARA is important for feeding (Eggleston et al. 1998) and a greater number of invertebrate taxa occur in larger patches of seagrass habitat (Bowden et al. 2001). The PARA of the patches was much greater in the CUP in 2007-09 than previously in the CUP and on Bonaire at any point, which could be a positive change for some organisms, but the large PARA value in the CUP in 2007-09 was accompanied by extremely small patches of coral (mean = 0.32 ha). Other studies have reported reduced survival in fragmented habitats as a result of increased exposure to predators along the edges of habitat patches, i.e., a large PARA (Brittingham and Temple 1983, Andrén and Angelstam 1988). These effects of fragmentation likely vary greatly by species (Eggleston et al. 1998), particularly between invertebrates and fishes. On coral reef habitats specifically, Caley et al. (2001) reported the positive effects of fragmentation on a commensal crab species; by partitioning corals into a larger number of distinct patches, a greater number of territorial individuals were able to occupy the same amount of total habitat. More recently, Bonin et al. (2011) also concluded positive effects of coral reef fragmentation, particularly accompanied by total habitat loss, on the abundance and species richness of reef fishes. They hypothesize this was the result of decreased predation and competition for shelter, but caution that future studies are necessary and should consider that the relative magnitude of habitat loss and fragmentation effects may depend on the time elapsed since the disturbance that caused them because the effects of fragmentation are dependent on the interaction between the extent of fragmentation and the home range size of the species considered at a given point in its development (Bonin et al. 2011).

While it is surprising to the us that the highly protected area of Bonaire experienced a decline in coral cover and unprotected, more densely populated Curaçao did not, it is important to note that cover on Bonaire was higher at the time of the initial study by van Duyl (1985). This may be because development had already occurred on Curaçao by the time of van Duyl (1985), as indicated by relatively little increase in population, so the damage from those activities was already apparent on the reef, whereas Bonaire's population increased from 9,000 inhabitants to 15,000 over the time period (1981 Census CBS-Netherlands Antilles). Although dive statistics are not readily available on Curaçao, an organization of divers that identifies reef fish species has shown that recreational divers have logged 17 times more hours on Bonaire than on Curaçao (Reef Environmental Education Foundation 2012), suggesting a larger prevalence of diving and the potential for diver-associated damage to the reef on Bonaire.

Another potential reason for the decline in coral on Bonaire and not on Curaçao could be due to hurricanes that passed near both islands, which lie along the southern edge of the hurricane belt, between van Duyl's initial mapping in the early 1980s and the satellite images (2007-09). Most notably, Hurricane Lenny passed north of the islands on a west-to-east track in 1999. Damage on both islands from the heavy waves associated with the storm is documented by Bries et al. (2004), particularly in the shallow reef (<10 m), which is coincident with the area assessed in the present study. Bries et al. (2004) also concluded that sites with a N-S, NW-SE, and NNW-SSE coastal orientation were more heavily damaged than sites with an E-W to WNW-ESE orientation. Comparing coastlines, most of Bonaire's coast is oriented in the north-south (N-S) direction, whereas a part of Curaçao, specifically the southern half of the leeward coast, is oriented west-northwest to east-southeast (W-NW:E-SE); this area includes the CUP and the majority of the adjacent unprotected area analyzed here. Looking specifically at the portion of Bonaire's coast that is oriented east to west (E-W) and includes the second no-diving marine reserve (MR2), there is actually an increase in pixels representing coral from the early 1980s to 2007-09 (Figure 15A. and B.) and very little negative change in coral cover (Figure 11C.). Conversely, looking at the northern leeward coast of Curaçao, which is oriented in a north-south (N-S) direction, there is less coral in 2007-09 (Figure 16B.) when compared to the early 1980s (Figure 16A.) and more negative change overall (Figure 16C.) particularly when compared to the east-west (E-W) portion of Bonaire (Figure 15), the Curaçao Underwater Park (Figure 13) and the unprotected reef adjacent to the CUP (Figure 14). Additional hurricanes affected the islands in 2004 and 2007 and on Bonaire the rubble from *Acropora palmata* and *A. cervicornis* can be seen in the very shallow reef (<5 m) (Relles and Jones Chapter 3). It is worth noting that coral cover on both islands is much greater than elsewhere in the Caribbean where hard coral cover has declined from around 50% to 10% over the time period (Gardner et al. 2003).

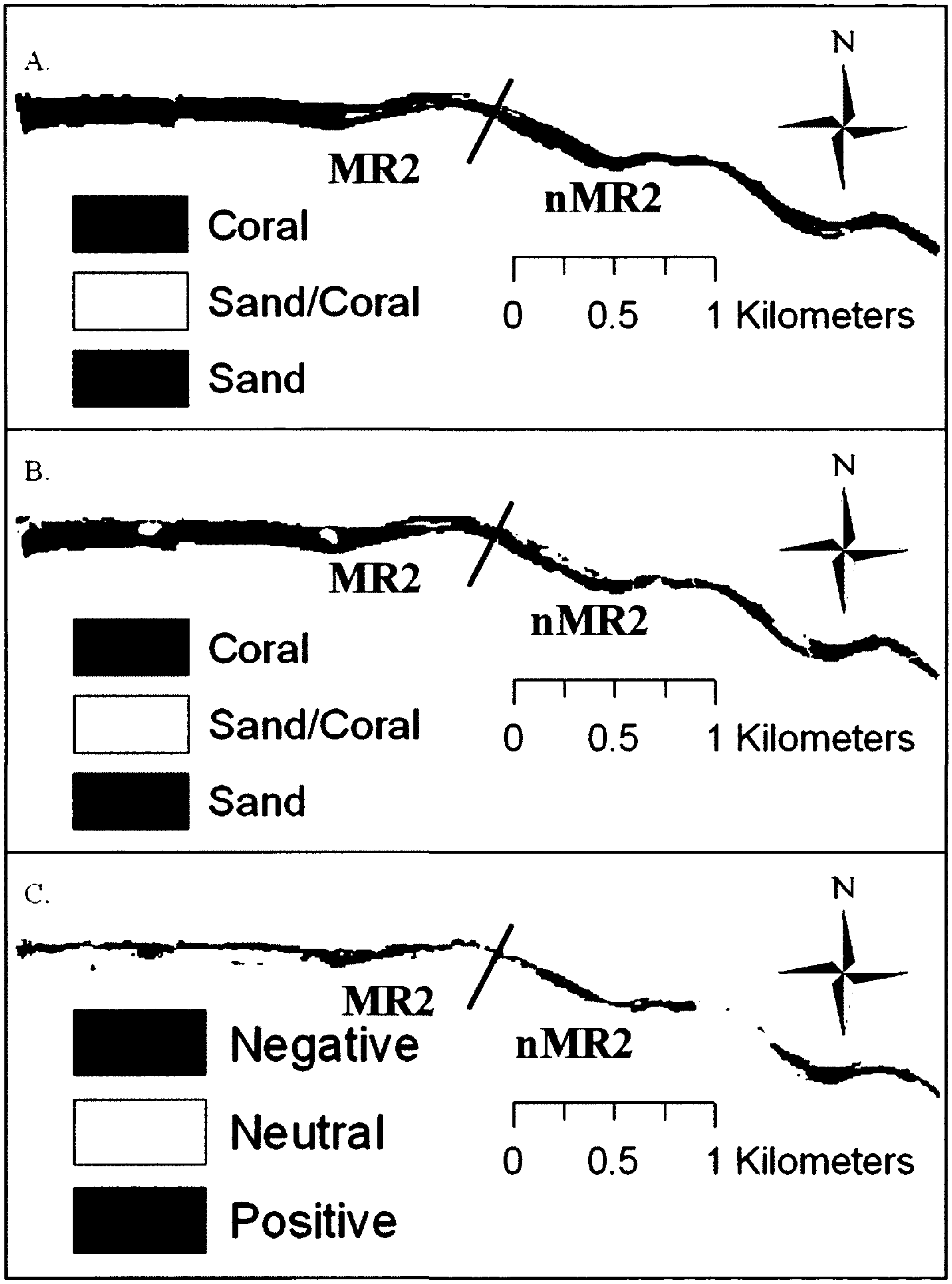
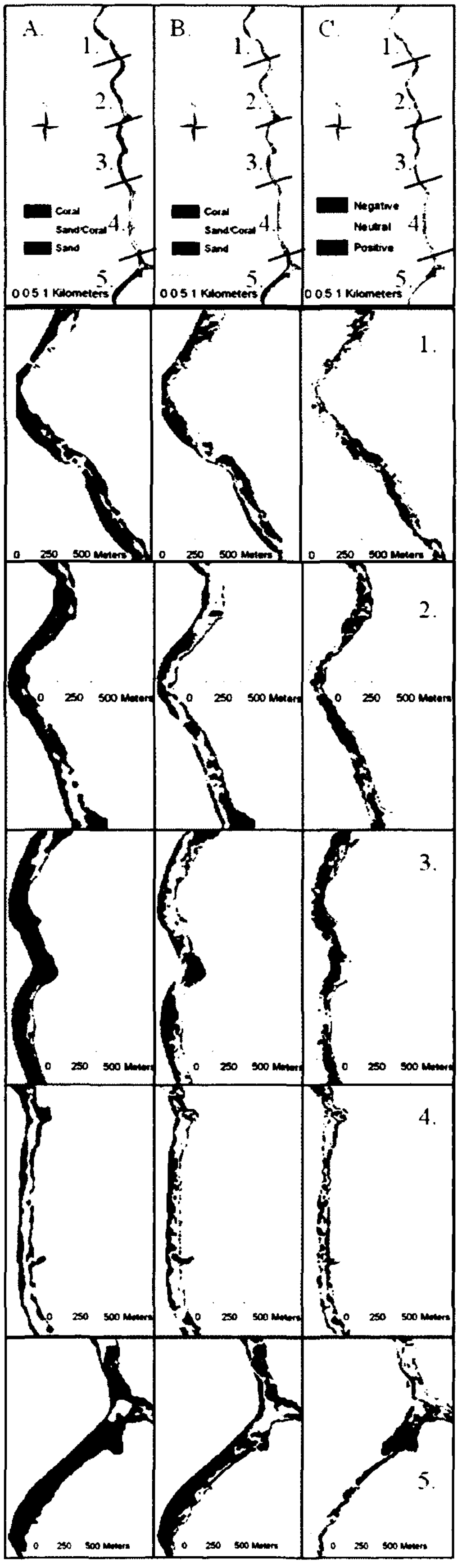


Figure 15. The portion of Bonaire's coastline that is oriented east to west, which includes one of the no-diving marine reserves (MR2) in the early 1980s (A.) and 2007-09 (B.). The negative, neutral and positive change values over the time period (C.).


A.
B.
C.
1.
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5.
Coral
Sand/Coral
Sand
Negative
Neutral
Positive
0 0.5 1 Kilometers
0 250 500 Meters

Figure 16. The northwest shore of Curacao in the early 1980s (a.) and 2007-09 (b.). The negative, neutral and positive change values over the time period (c.). The numbers 1-4 in the maps on the right indicate close-ups of areas as indicated on the maps on the left.

CONCLUSIONS

Further studies would be useful to determine how changes in habitat fragmentation and coral loss have affected reef denizens. The islands of Bonaire and Curaçao could potentially offer an ideal location for comparing the effects of coral habitat fragmentation with and without total coral loss because of their identical baseline data collected in the early 1980s on coral cover. This study and further research on the effects of habitat fragmentation in the absence of loss have implications for the establishment of marine protected areas (MPAs), potentially for determining a critical size at which an MPA would be most effective. Such a finding would be useful in an ecosystem-based management setting as size needs likely vary between invertebrates and fishes, as well as for preserving this living habitat, the underlying coral organism itself. Recent studies have shown that that the amount of coastline included in a reserve and its connectivity to neighboring reserves are more important for the success of an MPA than just its total area (Kaplan et al. 2006, Moffitt et al. 2009). Bonin et al.'s (2011) work suggests that habitat loss, and not fragmentation, is the primary cause of declines in reef fish abundance and diversity, making habitat loss prevention a top priority for the conservation of biodiversity. While very little work has been done on the size and diversity of reef fishes on Bonaire, reef fish populations on Curaçao have been degraded, particularly near population centers (Woodley et al. 1997). In places where habitat fragmentation is the biggest issue, facilitating dispersal and protecting connectivity should be made a priority; if habitat loss is the greatest threat, protection of large areas of reef habitat should be of the most importance (Lindenmayer and Fischer 2007).

Studies such as this demonstrate the applicability of satellite remote sensing methods in rapidly creating time-sensitive maps of coral reef cover, which are particularly useful before and after disturbances like tropical storms and hurricanes, both of which are increasing in frequency. This study also proves the usefulness of such satellite-derived maps for comparing to older, ancillary maps, the data for which was collected with more traditional methods, such as aerial photography and SCUBA collected by van Duyl (1985). However, these techniques are limited to shallow water (< 20 m) and are only as reliable as the initial data collected and used in ground truthing.

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CHAPTER 5

Cost-benefit analysis of a tiered-pay system for the Bonaire National Marine Park

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Abstract

The island of Bonaire, formerly part of the Netherlands Antilles, which was recently dissolved, and now a municipality of the Netherlands, has a long-established marine protected area, the Bonaire National Marine Park (BNMP). The BNMP is funded by mandatory nature fees and dive tags purchased by island residents and visitors. Past increases in these fees have met with mixed success. The goal of the park's management is not revenue maximization, but rather, coral reef conservation. However, operating expenses must be met by the fee system to provide enforcement of park regulations and education to the visitors and local population. Overuse by divers results in environmental degradation of coral reefs at other locations, and usage statistics suggest that Bonaire is exceeding the critical dive threshold of 4,000-6,000 annual dives at many of its most popular dive sites. Our analysis shows that an increase in the nature fee, accompanied by a tiered-pricing system for the most popular dive sites, would increase revenue while deterring some underwater visitors, likely resulting in reduced degradation.

Introduction

The Bonaire National Marine Park (BNMP), which surrounds the island of Bonaire, (capital Kralendijk 12° 15′ N, 68° 28′ W), is a unique example of marine policy because it is aimed specifically at the protection of the coral reef ecosystem. Bonaire has a long history of marine preservation, beginning with turtle protection in 1961, the prohibition of spear fishing in 1971, and protection for coral, dead or alive in 1975. The establishment of the Bonaire Marine Park (BMP) followed in 1979, with the park extending from the shoreline out to the 60 m depth contour around the entire island [1]. From the late 1970s into the early 1980s, 40 permanent moorings were placed along the leeward coast for use by dive boat operators to avoid reef destruction associated with anchor dropping. Shortly after, the park ran into funding issues, which eventually led to problems in enforcement of the park rules and maintenance of the extensive mooring system. In 1990, the problem of funding was resolved by the establishment of a mandatory annual park user-fee of 10 USD per scuba diver. The fee system included a dive tag to be worn on each diver's buoyancy compensator while diving in the park. In 1991, in addition to a new marine park manager, four park rangers were hired and two no-diving marine reserves were established along the northwest coast. In 1999, the BMP became the BNMP, a national park of the Netherlands Antilles. The Stichting Nationale Parken Bonaire (STINAPA) [2], a non-governmental, not-for-profit foundation was commissioned by the island government to conserve Bonaire's natural and historical heritage. In 2003, STINAPA Bonaire was restructured to include management of the BNMP in the water, and the Washington Slagbaai National Park (WSNP) on land, under a single director. In 2005, the park user fee was renamed the nature fee, which stipulated that all users, not exclusively divers, were subject to paying for park usage. Scuba divers are now required to pay 25 USD for an annual pass or 10 USD for a single day. Other users of the park, swimmers, sailors, snorkelers, etc., pay 10 USD for an annual pass. Purchase of the 25 USD dive tag includes entrance to WSNP, but non-divers who pay only the 10 USD nature fee are required to pay an additional 15 USD for entrance to WSNP. Residents of Bonaire, Curaçao, Saba, Sint Maarten, Sint Eustatius, and Aruba are exempt from paying the non-diver nature fee, but must pay for entrance to WSNP or for a 25 USD dive tag to dive in the BNMP. Proceeds go directly to STINAPA and are used entirely for the management of the two parks under its care [2]. These user fees provide a "sustainable financing mechanism" for the marine park [3].

Bonaire's largest economy is scuba diving tourism, and a 1997 report by UNESCO reported that Bonaire's tourism is growing annually by 7-10% [4]. More current data shows that tourist arrivals in Bonaire are still increasing, however at an average rate of only 6% between 2002 and 2008 [5]. The number of divers has also increased in that time period, with an average annual increase of 4% in dive tag sales [6]. Nearly 70% of Bonaire's tourists were divers in 2008 [6] and the average diver spends 2800 USD per trip [7] compared to 1500 USD for tourists in general in the Caribbean [8]; both amounts are adjusted for inflation based on the value of the 2009 USD. The 2005 dive tag price increase resulted in a nearly 8% decrease in dive tag sales from 2004 to 2005 (Figure 1), but a 130% increase in total revenue, due to the 150% increase in dive tag cost [6]. The following year dive tag sales experienced a near-full recovery, with an 8% increase in sales [6].

Numerous studies have shown that scuba divers look for high-quality coral reef habitats, as indicated by live coral coverage, coral and fish diversity, and water clarity [9, 10]. For this reason, the residents of Bonaire are highly invested in the protection and preservation of the coral reef structure that dominates the island's leeward coast. A number of studies have been conducted on Bonaire to determine BNMP user preferences, including willingness-to-pay (WTP) for reef preservation. A 2005 study found that tourists in Bonaire enjoyed the island largely because of its marine wildlife, notably the extensive, healthy corals and abundant, diverse fish fauna [11]. A significant proportion, 80%, of the respondents in the study indicated they would be unwilling to revisit for the same price if coral reefs were negatively affected by climate change [11]. While it is difficult to determine whether a tourist will *actually* not return to the island following reef degradation, a study in the Philippines found a reduction in diving tourism following mass bleaching of its reefs in 1998 [12]. While climate change impacts on the reef will not respect the boundaries of marine protected areas (MPAs), it has been suggested that MPAs closed to fishing and other detrimental activities could potentially buffer the effects of climate change and contribute to the reef's resiliency [13], its ability to bounce back and recover to its previous state following a disturbance.

Scuba diving is generally considered a non-consumptive use of a reef because one diver's use of the resource does not prohibit use by future users, and it is therefore generally considered non-destructive. However, recent research has shown the effects of recreational scuba divers to be detrimental to the coral reef structure [14-16]. A critical dive threshold of between 4000 and 6000 dives annually per site has been posited as the point when reef quality begins to suffer [17]. Given this critical number of dives, in

1993, sites receiving more than 3% of the total annual number of dives on Bonaire would begin to show negative impacts as a result of diving [17]. Data collected in 2001 found that although Bonaire is home to 86 dive sites, approximately 35% of the dives take place in only 10 of the sites, with 10% of all dives taking place in the top two dive sites [18]. These findings indicate a wide disparity in use among the dive sites on Bonaire, which is likely due, in part, to the disparity in live coral cover and diversity. It was also found that coral cover and diversity are highest, on the main island of Bonaire, at sites closest to the marine reserves, and that cover and diversity both drop off with increasing distance from the marine reserves [19]. Four of the top 10 most popular dive sites lie less than 6 km from one of the two marine reserves. Other reasons for preference of the remaining six top sites could be related to accessibility or some special feature of interest, e.g., the shipwreck Hilma Hooker, the number one most popular site [18].

In 1991, using a contingent valuation (CV) survey of divers in Bonaire, an overwhelming 92% of divers were willing to pay a nature fee [17]. The survey yielded an average willingness-to-pay (WTP) of 27.4 USD annually [17]; adjusting for inflation since 1991, this WTP works out to just over 42 USD in 2009. More recently, a study based on data collected in 2002 found a substantially greater WTP of between 74 and 145 USD annually [18], adjusted for inflation from 2002 to 2009 USD. Most recently, diver WTP has declined since both the 1991 and 2002 studies to an average of 38.0 USD annually adjusted from 2008 to 2009 USD [20]. A panel commissioned by the National Oceanic and Atmospheric Administration (NOAA) endorsed CV as a technique for the measurement of passive use values of natural resources in 1993 [21]. In fact, as a result of the 2002 study [18] the BNMP management ultimately changed their fee structure in

2005 [2] to 25 USD annually for scuba divers and 10 USD annually for non-diving users of the park. Based on variable WTPs, and the disparity of use in dive sites, a tiered pricing system, where divers would be required to pay additional fees for more pristine and/or overused dive sites, is proposed.

Theory

Given that most willingness-to-pay (WTP) estimates for dive tags in Bonaire exceed the current dive tag rate, a change in pricing of the nature fee to a tiered pricing system is suggested. While the current rate of 25 USD annually for scuba divers and 10 USD annually for other marine park users, generated nearly one million USD in 2007 [6], a tiered system would likely result in increased revenue, while discouraging overuse of popular dive sites, which may be on the brink of environmental degradation based on the critical dive threshold [17]. A tiered system, as laid out in Table 1, would increase the annual cost of diving from 25 USD to 30 USD and increase the annual cost for non-divers from 10 USD to 15 USD. The nature fee for non-divers would continue to exclude residents of the former Netherlands Antilles, as in the current system. Single-day rates would no longer be made available, as most island visitors stay for longer than a day [22], so this rate has been deemed unnecessary. These new fees would exclude use of the top 10 most visited dive sites [18]. To visit these top 10 premium dive sites, divers and non-divers would have to pay an additional fee of 20 and 10 USD, respectively. Previously, it was found that adjusting the dive tag price from 25 to 50 USD could result in a 7.9% to

40.0%[1] decrease in diver visitation to the island and concluded that while a flat 50 USD dive tag would maximize the revenue stream from dive tag sales this may not be in the best interest of the island as a whole [18]. Bonaire's economy is dependent on tourism, which largely consists of divers. The potential damage to the island's economy would far outweigh the increased revenue from dive tag sales. A tiered system such as the one suggested above could mitigate this impact on the economy by keeping the majority of dive sites near their original price. Protection of dive sites and prevention of reef degradation can be achieved through an increase in fees for those specific sites that are overused, while not discouraging divers from coming to Bonaire altogether.

[1] Two of the three mechanisms used by Thur (2003) to estimate willingness to pay included dive tags prices of 25and 50 USD while the third mechanism used 20 and 50 USD. This third method resulted in the 40.0% decline in diver visitation. It is included here to represent the upper bound of potential revenue outcomes.

Table 1. Existing and Proposed Fee System*

Visitor Type:	Existing System	Proposed System
Non-diver[1]	$10	$15
Non-diver Premium[1,2]		$10
Diver	$25	$30
Diver Premium[2]		$20

*All monetary values are reported in USD.

[1]Non-divers include swimmers, snorkelers, sailors, and other users of the park.

[2]Premium fees are charged for the visitation of the 10 most popular locations as identified by Thur (2003); this fee is in addition to the basic fee.

Results

The proposed changes in fees may result in a range of outcomes with respect to diver visitation as well as revenue. Of the 51,988 individuals that visited the Bonaire National Marine Park in 2008, 36,219 were divers and 15,769 were not [6]. Naturally, any change in price will cause some consumers to reconsider their choices; unfortunately, information on how the non-diving visitors would react to a price change is unavailable. However, the 2002 study was conducted while the dive tag price was 10 USD and thus showed how divers would react to an increase of dive tag prices to 20 or 25 USD [18]. In the case of a change in price from 10 to 20 USD, a 6.3% decrease in diver visitation was predicted. In the case of an increase from 10 to 25 USD, the results varied between 0.0% and a 14.3% decrease in diver visitation [18]. In 2005 the price of the dive tag actually *did* increase from 10 to 25 USD, accompanied by a nearly 8.0% decrease in dive tag sales from 2004 to 2005, thereby validating Thur's prediction [6] (Figure 1). In the current case, an increase in non-diver entrance fees from 10 to 15 USD is discussed as well. While the demography of non-divers is relatively unknown, the popularity of Bonaire for diving lends one to assume a fair portion of non-divers are likely traveling with divers, such as family members. As such, their decisions are likely made as a group (or household); therefore, similar decreases in visitation by non-divers as divers was assumed and one can expect the proposed increase in entrance fees to 15 USD to result in less than a 14.3% decrease[2]. A 14.3% decrease in visitation by non-divers to the BNMP, with the increase in tag cost, will result in an increase in revenue of 45,044 USD. Revenue would

[2] While an 8% decrease in diver visitation was experienced in 2005 as a result of the increased dive tag fee, Thur's (2003) estimation of 0.0 to 14.3% decrease will be used. This is due in large part because translating these numbers to non-diver fees will necessarily have some margin of error.

increase 78,845 USD if there is no decline in visitation. This results in an expected 28.6 to 50.0% increase in revenue from non-diver fees alone.

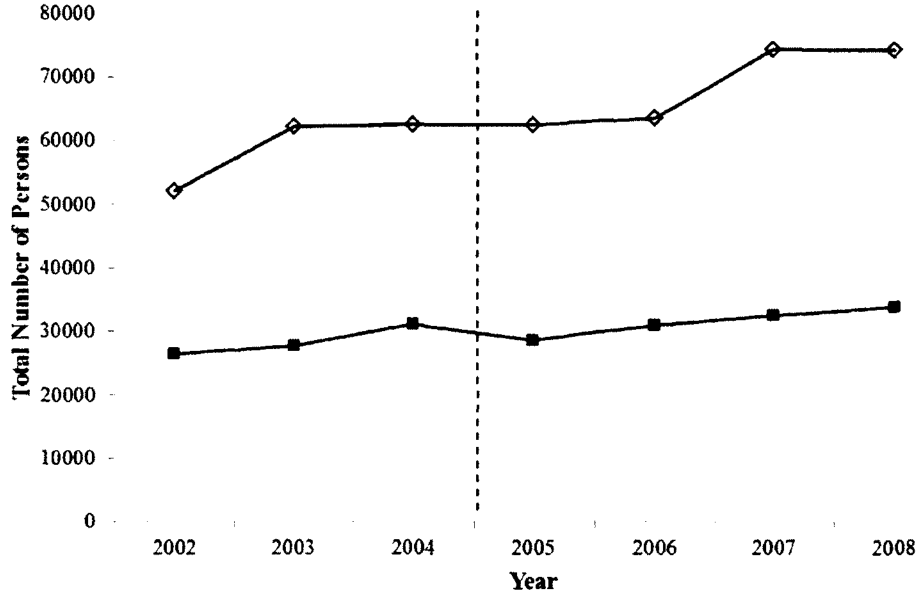

Figure 1. Annual visitors to Bonaire (open diamond) and annual dive tag sales (closed square) from 2002-2008. Dotted line represents the increase in dive tag cost from 10 to 25 USD in 2005.

Tag fees for scuba divers will increase under the proposed plan from 25 to 30 USD. Only the reaction of divers to a fee change of 20 to 30 USD was estimated, which resulted in a 21.0% decrease in visitation [18]. Forgoing a more sophisticated model due to data limitations, it seems reasonable to assume that half or more of the 21.0% decline in diver visitation will occur in the top half of the 20 to 30 USD interval. This means at a minimum, a 10.5% decrease in visitation is expected for a change in the fee from 25 to 30 USD. This decline in visitation will still result in a 7.4% increase in revenue. Using the number of divers from 2008 this would be an increase in revenue of 66,788 USD. The upper bound of a decrease in visitation would be no more than 21.0% and this would result in a decrease in revenue from general dive tag sales of 5.2%. Note that while the basic dive tag fee increase could result in a revenue loss, the proposed system as a whole will not.

As stated previously, a decrease in diver visitation ranging from 0.0-14.3% was predicted when dive tag fees rose from 10 to 25 USD and 7.9- 40.0% when dive tag fees rose from 25 to 50 USD [18]. These estimates assumed a flat increase in tag prices. The changes proposed here only require the top 10 dive sites to increase from 25 to 50 USD for divers and 10 to 25 USD for non-divers. The 10 most popular sites were identified as receiving 34.67% of the total dives conducted [18]. Assuming that the sub-population of divers frequenting these 10 sites carry the same distribution of values with regard to the dive tag fee, one can expect an increase in the fee from 25 to 50 USD to result in a reduction in visitation of 7.9-40.0% at these sites in particular. Here again it is assumed that non-divers act similarly to fee changes as divers. Thus the non-diver fee increase from 10 to 25 USD will result in a 0.0% to 14.3% decrease in non-divers at these 10 sites.

The benefit of this tiered system of fees is this reduction in visitation will only affect the 10 most popular sites. Presumably, the majority of these individuals will begin visiting elsewhere around the island as opposed to going to a completely different island. From an environmental standpoint this could assist greatly in lowering the impact of visitors at these most popular sites without completely destabilizing the island's economy.

Combining the increase in fees with the premium for visiting the more popular dive sites will result in a greater amount of revenue overall. Using the range of expected decreases in visitation associated with the general dive tag and the premium fee, it is expected that the revenue from dive tag sales will increase by 11.4% to 32.9%. The 11.4% increase in revenue is associated with a 21.0% decrease in overall divers and a 40.0% shift away from the premium sites. The 32.9% increase in diver revenue is associated with a 10.5% decrease in overall divers and a 7.9% shift away from premium sites. When considering just the non-divers, the potential increase in revenue ranges from 58.3% to 84.7%. The 58.3% increase in revenue is found using the 14.3% reduction in non-diver visitation, while the 84.7% increase in revenue is based on a 0.0% decrease in non-diver behavior.

Finally, examining the entire proposed fee structure and its effects on revenue shows an increase in revenue of 18.4-40.6%. The 18.4% increase in revenue assumes a 21.0% decrease in overall divers, and a 40.0% shift away from premium sites as well as a 14.3% reduction in non-diver visitation. The 40.6% increase in revenue uses the 10.5% decrease in overall divers, 7.9% shift away from premium sites and assumes no change in non-diver visitation. Table 2 summarizes the potential visitation and revenue effects of the proposed tiered system (Figure 2).

Table 2. Visitation and Revenue Effects of the Proposed Fee System*

Visitor Type:	Existing System: Visitors	Existing System: Estimated Revenue	Scenario A: % Change in Visitors	Scenario A: Remaining Visitors	Scenario A: New Revenue	Scenario A: % Change in Revenue	Scenario B: % Change in Visitors	Scenario B: Remaining Visitors	Scenario B: New Revenue	Scenario B: % Change in Revenue
Non-diver	15,769	157,690	-14.3%	13,561	202,734	28.6%	0.0%	15,769	236,535	50.0%
Non-diver Premium[1]	5,467	0**	-14.3%	4,686	46,859	NA[2]	0.0%	5,467	54,671	NA[2]
				Subtotal	249,593	58.3%		Subtotal	291,206	84.7%
Diver	36,219	905,475	-21.0%	28,602	858,064	-5.2%	-10.5%	32,409	972,263	7.4%
Diver Premium[1]	12,557	0**	-40.0%	7,534	150,686	NA[2]	-7.9%	11,566	231,327	NA[2]
				Subtotal	1,008,750	11.4%		Subtotal	1,203,590	32.9%
Total	51,988	1,063,165	-19.0%	42,118	1,258,343	18.4%	-7.3%	48,178	1,494,796	40.6%

*All monetary values are reported in USD.

**The existing system is not tiered thus there is currently no additional revenue from visitors at the premium sites.

[1]Both Premium lines are strictly valuing the added fee for visiting a premium site. For Non-divers this is an additional $10; for Divers this is an additional $20.
[2]Because the existing system is not tiered it is not useful to calculate a percent change in revenue for the premium fee.

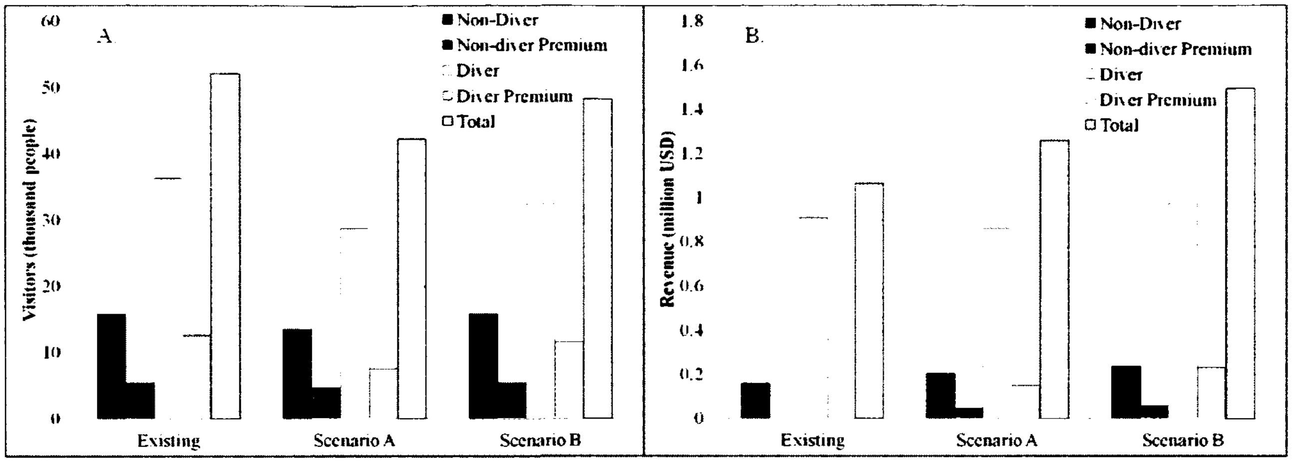

Figure 2. Changes in the non-diver, non-diver premium, diver, diver premium and total numbers of visitors (A.) to the BNMP and estimated revenue given the existing system and scenarios A and B.

Discussion

While an increase in user fees on Bonaire would result in increased park revenue, the goal of the Bonaire National Marine Park is not maximization of revenue, but rather to preserve the coral reef ecosystem and protect the organisms that call it home. The introduction of a tiered pricing system could potentially result in significant changes in revenue, but, more importantly, deter some divers from diving at the most heavily-used dive sites, which are at increased risk for environmental degradation. Deterrence of 40.0% of divers away from the most popular dive sites could potentially bring the number of dives at those sites under the critical threshold for reef damage [17]. However, given increases in diving tourism in Bonaire, from 16,798 dive tags sold in 1992 to 33,939 in 2008 [20], it is likely that annual dives in Bonaire already surpass the carrying capacity of a total of 190-200 thousand dives per year for the BNMP suggested [17]. The actual number of dives on Bonaire and at each site can be estimated using data from [6] and [18]. Given that 33,939 dive tags were sold in 2008 [22], and the majority of divers make between 11 and 15 dives during their trip to Bonaire [18], between 373,329 and 509,085 dives take place annually on the island, greatly exceeding the 190-200 thousand dive carrying capacity [17]. A critical threshold level of 4-6 thousand dives per site before reef quality suffers has also been proposed [17]; given that 5.2% of dives take place at the most popular dive site, the Hilma Hooker shipwreck [18], this site experiences approximately 19,413 dives at the low end and 26,472 at the high end on an annual basis. Only dive sites outside the top 25 most frequented could hope to fall into Dixon's suggested threshold of 4-6 thousand dives per site. These calculations of annual total dives and dives per site are conservative given that only tourist divers with American addresses were surveyed [18], excluding island residents who presumably dive more frequently given their proximity to the reef and the amount of time they spend on the island.

Although the primary goal of the fee system is to reduce dive stress on the reef, the increased revenue can provide benefits in the form of funding additional education, as well as informative materials. The 2008 study found that although the majority (88%) of tourists are aware of the BNMP prior to arrival, many are unaware of what the current fee is used for and how it supports park maintenance [20]. Under the current system, upon purchasing a tag, divers have to watch a short video on the rules of the marine park and how they serve to protect the reef ecosystem and its inhabitants. Good buoyancy control and not touching the reef are emphasized. In addition to these important points, the problems posed by overuse of a dive site and inadvertent destruction by divers could be imparted to users, particularly those purchasing the expanded-use tag. It should also be made clear to users how the fee is used by STINAPA to support its goal to conserve Bonaire's natural and historical heritage through the sustainable use of its resources [2]. A restriction on dive site use will also likely increase the need for policing and enforcement to ensure that users of the preferred sites have indeed paid the increased fee. In addition to recommended dive thresholds, it was suggested that the apparent threshold stress level could be adjusted by increased park management and/or diver education [17]. Given both increased education of divers *and* increases in park management, both facilitated by increased revenue in this proposal, the stress threshold could move from 200 thousand dives annually to 400 thousand dives annually [17], which brings it much closer to the number of dives we can assume are taking place on the island given the increase in dive tag sales and the average number of dives per visit.

Conclusions

In summary, a 5 USD increase in nature fee for divers and non-diving users of the Bonaire National Marine Park is proposed, which would exclude use of the top 10 most frequented dive sites, here referred to as premium sites. In order to visit the premium sites, divers and non-divers would be required to pay an additional fee of 20 and 10 USD, respectively. This would equal a total cost of 50 USD for divers and 25 USD for non-divers on an annual basis. While previous work concluded that a flat rate of 50 USD would maximize revenue for STINAPA [18], a tiered system avoids the potential economic damages of decreased tourism, while increasing revenue and redistributing diver visitation away from overused sites, mitigating reef degradation as a result of diver visitation. The fact that the most recent WTP in 2008, 38.0 USD [20], is lower than this proposed premium dive rate of 50 USD, further suggests that diver visitation to overused sites will be reduced. There will still be some decrease in tourism, thus further study of the impact on Bonaire's economy is warranted.

Future research would be beneficial to reevaluate diver preferences and habits [18], since that data was collected a decade ago. It would also be beneficial to expand the survey. Including non-American tourists and island residents in particular, whose WTP and use of the reef would be expected to differ from tourist divers, would facilitate a better characterization of divers' values. Given the addition of the 10 USD nature fee in

2005, sending a similar survey to purchasers of the nature fee would also be beneficial. Should this not be possible, expanding the diver survey to inquire about non-diving members of their travel party may be useful. Evidence of positive management can increase user acceptance of higher fees [23], and user attitudes toward fee changes should be continually evaluated [20]. It would also be useful to revise the critical dive threshold [17], as Bonaire is clearly exceeding the number of dives per site, as well as for the island as a whole, yet still maintains relatively high coral cover [24].

Glossary

Marine Protected Area (MPA): An umbrella term for an area in the marine environment that is protected in some way from human activity with the goal of protecting living, non-living, cultural, and/or historic resources.

Willingness-to-pay (WTP): A technique used in economics to calculate the maximum amount a person would be willing to pay, sacrifice or exchange to receive a good or service.

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DISSERTATION CONCLUSIONS

This study created modern benthic habitat maps using relatively new satellite remote sensing techniques. New techniques for removing the effects of atmospheric and water column particulates on the scattering of light were successfully employed to account for these factors, creating more reliable maps of the benthos. These maps were then compared to a historical data source, *Atlas of the Living Reefs of Curaçao and Bonaire (Netherlands Antilles)* (van Duyl 1985) to examine how coral cover has changed between the two time periods. The major conclusions of this study are:

- Coral cover on the island of Bonaire, Dutch Caribbean, declined over the comparison time period from 62% to 30% in the shallow reef (< 10 m).
- Declines in coral cover were greatest in the shallowest part of the reef (< 5 m) and owed mostly to the loss of branching acroporids, whereas van Duyl's head coral group, which included *Diploria* spp., *Montastrea* spp., *Siderastrea* spp., *Dichocoenia*, *Colpophyllia*, *Porites astreoides*, *Meandrina meandrites* and *Stephanocoenia michelinii*, actually increased over the time period.
- Coral cover was highest on Bonaire along the area adjacent to the capital city of Kralendijk, where most of the population resides and most development has taken place.
- Only the no-diving marine reserve that was located along the sheltered part of the coastline did better in terms of increased coral cover over the time period, as well as fewer and larger patches of coral.

- The sheltered half of the accompanying island of Klein Bonaire also faired well over the time period, declining in coral cover, but exhibiting fewer and larger coral patches in 2008-09.
- Coral cover on the island of Curaçao was initially lower than on Bonaire, 34% in the early 1980s and remained the same in 2007-09, but became increasingly patchy.
- The Curaçao Underwater Park declined slightly in coral cover from 38% to 34% between the two time periods, but the adjacent area of unprotected reef actually increased in total coral cover from 20% in the early 1980s to 33% in 2007-09. Both protected and unprotected areas became increasingly patchy and less contiguous.
- The Bonaire National Marine Park is charging visitors less than what divers have stated they are willing to pay for diving within a marine protected area (MPA), and its most popular dive sites are being overused based on the critical dive threshold established by Dixon et al. (1993).
- An increase in snorkeler and diver fees, accompanied by a tiered pay system, would likely result in decreased dive stress on the most-frequented sites, as well as increased revenue for park management and education efforts.
- Satellite remote sensing techniques are a powerful method for coarsely mapping (3-4 habitat classes) large areas of coral reefs, more quickly than traditional methods such as video transects conducted by SCUBA. Satellite-derived maps are useful for assessing damage, particularly following disturbances (e.g.,

hurricanes) or over time, and for identifying areas for management concern and conservation.

APPENDIX A

Bounding coordinates of the four corners of the areas MR1, nMR1, MR2 and nMR2 in latitude and longitude, degrees, minutes, seconds.

Site	Location	Latitude	Longitude
MR1	North, offshore corner	12° 15’ 48”	-68° 25’ 6”
	North, coastal corner	12° 15’ 46”	-68° 25’ 5”
	South, offshore corner	12° 14’ 55”	-68° 25’ 1”
	South, coastal corner	12° 14’ 57”	-68° 24’ 59”
nMR1	North, offshore corner	12° 16’ 21”	-68° 25’ 00”
	North, coastal corner	12° 16’ 23”	-68° 24’ 56”
	South, offshore corner	12° 15’ 48”	-68° 25’ 6”
	South, coastal corner	12° 15’ 47”	-68° 25’ 4”
MR2	West, offshore corner	12° 13’ 8”	-68° 22’ 33”
	West, coastal corner	12° 13’ 15”	-68° 22’ 35”
	East, offshore corner	12° 13’ 10”	-68° 21’ 15”
	East, coastal corner	12° 13’ 12”	-68° 21’ 15”
nMR2	Northwest, offshore corner	12° 13’ 10”	-68° 21’ 15”
	Northwest, coastal corner	12° 13’ 12”	-68° 21’ 15”
	Southeast, offshore corner	12° 12’ 47”	-68° 20’ 3”

	Southeast, coastal corner	12 ° 12’ 49”	-68 ° 20’ 3”

APPENDIX B

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Atmospheric correction procedure

In a single scattering approach, the radiance received by a space borne sensor at the top of the atmosphere (TOA) in a spectral band centered at a wavelength, λ_i, $L_t(\lambda_i)$, can be divided into the following components [24],[25]:

$$L_t(\lambda_i) = L_r(\lambda_i) + L_a(\lambda_i) + T(\lambda_i)\, L_g(\lambda_i) + t(\lambda_i)\, L_w(\lambda_i) \tag{1}$$

where $L_r(\lambda_i)$ and $L_a(\lambda_i)$ are radiance contributions associated with air molecules (Rayleigh scattering) and aerosols (including Rayleigh-aerosol interactions) respectively, T is the direct atmospheric transmittance, $L_g(\lambda_i)$ is the sun-glint component, t is the diffuse atmospheric transmittance, and $L_w(\lambda_i)$ is the desired water leaving radiance. Sun-glint is usually avoided through tilting of the sensor. Hence, $T(\lambda_i)\, L_g(\lambda_i)$ may be ignored, and consequently, Eq. (1) can be written as:

$$L_t(\lambda_i) = L_r(\lambda_i) + L_a(\lambda_i) + t(\lambda_i)\, L_w(\lambda_i) \tag{2}$$

First, contribution due to ozone absorption was removed from the TOA radiance as given
by Hu et al. [18]:

$$L_t^*(\lambda_i) = L_t(\lambda_i)\ e^{[\tau oz(\lambda i) * ((1/\cos\theta o) + (1/\cos\theta v))]} \tag{3}$$

where $L_t^*(\lambda_i)$ is TOA radiance measured by the satellite in the absence of ozone, θ_v is satellite viewing zenith angle, θ_0 is solar zenith angle, and $\tau_{oz}(\lambda_i)$ is ozone optical depth, which was computed as [25], [26]:

$$\tau_{oz}(\lambda i) = k_{oz}(\lambda_i) * (DU/100) \tag{4}$$

where $k_{oz}(\lambda_i)$ is ozone absorption coefficient taken from Gregg and Carder [27] and DU is ozone concentration in Dobson units obtained from the TOMS website.

i) *Computation of Rayleigh Path Radiance* ($L_r(\lambda_i)$): Rayleigh path radiance is the contribution of Rayleigh scattering by air molecules to TOA radiance. It was computed as given by Gordon [26]:

$$L_r(\lambda_i) = (F_0'(\lambda_i) * \omega_{0r} * \tau_r(\lambda_i) * P_r) / (4\pi * \cos\theta_v) \tag{5}$$

where $\tau_r(\lambda_i)$ is Rayleigh optical thickness, P_r is Rayleigh scattering phase function, ω_{0r} is single scattering albedo (equal to 1), and $F_0'(\lambda_i)$ is instantaneous extraterrestrial solar irradiance adjusted for the Sun-Earth distance as [18]:

$$F_0'(\lambda_i) = F_0(\lambda_i) \;/\; [1.00014 - 0.01671*\cos(2\pi(0.9856002831*\text{julianday}-3.4532868)/360) - 0.00014*\cos(4\pi(0.9856002831*\text{julianday}-3.4532868)/360.0)^2] \tag{6}$$

where $F_0(\lambda_i)$ is extraterrestrial solar irradiance. The $F_0(\lambda_i)$ values were adopted from Nickel and Lab [28]..

1. *Computation of Rayleigh optical thickness* ($\tau_r(\lambda_i)$): The value of Rayleigh optical thickness, $\tau_r(\lambda_i)$ at any atmospheric pressure P was calculated as given by Hansen and Travis [29]:

$$\tau_r(\lambda_i) = P/P_0\,[0.008569\lambda_i^{-4}\,(1 + 0.0113\lambda_i^{-2} + 0.00013\,\lambda_i^{-4}) \tag{7}$$

where λ_i is wavelength in μm, and P_0 is standard atmospheric pressure of 1013.25 millibars.

2. *Computation of Rayleigh scattering phase function* ($P_r(\theta_\pm)$): The computations of Rayleigh scattering phase function involved the direct scattered light and the scattered light which is specularly reflected at the air-sea interface. It was computed as given by Doerffer [30]:

$$P_r(\theta_\pm) = \tfrac{3}{4}\,(1 + \cos^2\theta_\pm) \qquad (8)$$

where $\theta_\pm$ represents the scattering angles. The – and + subscripts indicate the direct scattered light and direct scattered plus the specularly reflected light at the air-sea interface, respectively. The scattering angles in the direction to the sensor and in direction to the sensor via the air-sea interface is given by,

$$\cos\theta_\pm = \pm\,(\cos\theta_0\cos\theta_v - \sin\theta_0\sin\theta_v\cos(\Delta\Phi) \qquad (9)$$

where $\Delta\Phi$ represents the relative azimuth angle. Note that by definition the relative azimuth angle is the absolute difference between the satellite azimuth and the solar azimuth angles. In this definition the sun vector is considered in the down direction (sun to surface). However, it is a common practice (also instituted in SeaDAS) to define both the sun vector and the sensor vector in the upward direction and thus the relative azimuth angle was obtained as,

$$\Delta\Phi = \Phi_v - 180 - \Phi_0 \qquad (10)$$

where Φ_0 and Φ_v are solar and satellite azimuth angles, respectively. Further, to keep the values between ±180°, 360° was added or subtracted when the relative azimuth angles were less than -180° and greater than 180°, respectively. At the air-sea interface, another phenomenon occurs with the specularly reflected light that should be accounted for in the computation of Rayleigh phase function is Fresnel reflection. It is the reflection that occurs when light propagates through media with different refractive indices. As none of the relevant media (air or water) were magnetic, when the light was polarized with the electric field of the light perpendicular to the incident light (s-polarized), the Fresnel reflection coefficient is calculated as:

$$R(\theta_i)_s = \sin^2(\theta_i - \theta_j) / \sin^2(\theta_i + \theta_j) \tag{11}$$

where θ_i is solar zenith angle (θ_0) for R (θ_0) and satellite viewing zenith angle (θ_v) for R (θ_v)calculation, and θ_j is determined through Snell's law as:

$$\sin(\theta_i) / \sin(\theta_j) = \eta = 1.333 = \text{refractive index of water} \tag{12}$$

When the light is polarized in the same plane as the incident light (p-polarized), the Fresnel reflection coefficient is calculated by:

$$R(\theta_i)_p = \tan^2(\theta_i - \theta_j) / \tan^2(\theta_i + \theta_j) \tag{13}$$

Assuming the incident light contains an equal mix of s- and p-polarizations, the Fresnel reflection coefficient was computed as [17]:

$$R(\theta_i) = 0.5[\, R(\theta_i)_s + R(\theta_i)_p \,] \tag{14}$$

The total Rayleigh scattering phase function was computed as given by Doerffer [30] and Gordon and Wang [31]:

$$P_r(\theta_\pm) = P_r(\theta_-) + [\, R(\theta_v) + R(\theta_0) \,]\, P_r(\theta_+) \tag{15}$$

where is $P_r(\theta_\pm)$ total Rayleigh scattering phase function, $P_r(\theta_-)$ is Rayleigh scattering phase function when solar radiation is directly backscattered to the sensor, and $P_r(\theta_+)$ is Rayleigh scattering phase function due to the specularly reflected light at the air/sea interface in addition to the direct backscattered light.

ii) Computation of Aerosol Path radiance ($L_a(\lambda_i)$): Aerosol path radiance is the contribution of scattering by particles similar to or larger than the wavelength of light such as dust, pollen, smoke or water vapor in the atmosphere to the TOA radiance. Unlike L_r, which can be computed fairly accurately, L_a is difficult to determine since the aerosol scattering is highly variable and many times there is no a priori information on their optical properties and size distributions. By using the sensor radiances above 700 nm, it is possible to determine L_a indirectly [31]. Over Case 1 clear-waters, water-leaving radiance is negligible in the NIR bands because of strong NIR absorption by water, thus, the radiance measured is essentially the contributions from the atmosphere. Therefore L_a, could be estimated after removing L_r from the TOA radiance at the NIR bands. To estimate L_a in the visible wavelengths, one NIR band is required for assessing the magnitude of aerosol's contribution and another is required for assessing its dependence on wavelength. Gordon and Wang [31] atmospheric correction algorithm uses the SeaWiFS NIR band centered at 865 nm to estimate the aerosol scattering and 765 nm band together with 865 nm band to extrapolate into visible. [For Quickbird® multispectral satellite imagery the NIR band centered at 830 nm, and the 660 nm band together with 830 nm to extrapolate into visible.]

$Ln[\, L_a(\lambda) \,/\, F_0'(\lambda)$ for QuickBird® NIR bands centered at 660 and 830 nm were plotted against λ, and ε was determined as the negative of the slope of the straight line as:

$$[\, Ln\{ L_a(\lambda_{830}) / F_0'(\lambda_{830}) \} - Ln\{ L_a(\lambda_{660}) / F_0'(\lambda_{660}) \} \,] / (\lambda_{830} - \lambda_{660}) = -\varepsilon \tag{16}$$

Once ε was known, the for the wavelengths below 700 nm were determined as:

$$L_a(\lambda_i < 700\text{nm}) = L_a(\lambda_{830})(F_0' / F_0'(\lambda_{830}))\, e^{[-\varepsilon(\lambda i/\lambda 830)]} \tag{17}$$

iii) Computation of Diffuse Transmittance (t (λ_i)): Diffuse transmittance from the water surface to the satellite was computed as [18]:

$$t(\lambda_i) = e^{[-(\tau r(\lambda i)/2) * (1 / \cos \theta v)]} \tag{18}$$

iv) Computation of water-leaving radiance (L_w(λ_i)): The desirable water-leaving radiance at a specific wavelength was computed by rewriting the Eq. (2) as:

$$L_w(\lambda_i) = (L_t{}^*(\lambda_i) - L_r(\lambda_i) - L_a(\lambda_i)) / t(\lambda_i) \tag{19}$$

v) Computation of normalized water-leaving radiance (${}_nL_w$(λ_i)): The ${}_nL_w$ is approximately the radiance that would exit the ocean in the absence of atmosphere with sun at the zenith at mean earth-sun distance (1 AU), and was computed as given by Gordon and Voss [33]:

$${}_nL_w(\lambda_i) = L_w(\lambda_i) / d^2 * \cos \theta_0 * e^{\{-(\tau r(\lambda i)/2) * (1 / \cos \theta o)\}} \tag{20}$$

where d is earth-sun distance in astronomical unit (AU).

vi) Computation of remote sensing reflectance (R_{rs}(λ_i)): The R_{rs} associated with ${}_nL_w$ was computed as given by Gordon and Voss [33]:

$$R_{rs}(\lambda_i) = {}_nL_w(\lambda_i) / F_0(\lambda_i) \tag{21}$$

This atmospheric correction procedure does not include out-of-band correction, whitecap correction, surface roughness influences and contribution of L_a to diffuse transmittance.

However, these corrections will not significantly change the overall accuracy of the procedure particularly for small lakes or estuaries on low wind speed days when whitecap and surface roughness terms are minimal.

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