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Satellite-Assisted Assessment Of The Effects Of Human Development On Coral Reefs, Roatan Honduras

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by

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ABSTRACT

Human intervention is degrading coastal marine environments on a global scale. Coral reefs are one of the most valuable ecosystems occupying coastal areas that provide a vast majority of consumable fish, recreational activities, and protection from extreme events, such as tropical cyclones. Unfortunately, as human development activities such as the construction of buildings, road infrastructure, and agriculture increase in coastal areas, so do the detrimental impacts on the health of coral reefs. Consequences of coastal development such as pollution, sedimentation, eutrophication, overfishing, and recreational activities have been linked to diminished coral health. A degraded coral ecosystem can often be identified by a reduction in coral cover and an increase in macroalgae. This change is associated with a phenomenon known as a phase shift that occurs in response to environmental stressors. This research investigates the relationships between increasing human development activities and coral and macroalgae extent on and surrounding the Caribbean island of Roatan, Honduras. The extent of land development and coral reefs are classified from a series of Landsat images from 1985 to 2015. Coral reef healthiness is evaluated based on coral and macroalgae covers and their temporal variations. A spatial gridding system is used to assess the relatedness between the development pressures and the health of coral reefs over the 30-year period. Out of the 36 sampled grids, seven showed a statistically significant relationship between urban area change and a change in either coral or macroalgae populations. This study

demonstrates the capability of remote sensing to monitor coral reefs responding to land development of Roatan, Honduras.

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CHAPTER 1

INTRODUCTION

Globally, coastal marine environments are being degraded through human intervention. Some of the main pressures on marine environments are from coastal development, such as the construction of buildings and road infrastructure and agricultural land uses. Research has revealed that development activities have had detrimental impacts on the health of marine ecosystems, mainly coral reefs (Hughes et al., 2010; Mora, 2008; Vanderstraete et al., 2006).

Coral reefs are considered one of the most diverse and productive ecosystems. They provide habitats for many marine species, economic values from a vast majority of consumable fish, recreational services, as well as protection from extreme events such as tropical cyclones (Chabanet et al., 2005). On a global extent, 19% of the original coral cover has been lost and 35% is considered threatened (Wilkinson, 2008) from pressures including overfishing (Cheal et al., 2010; McManus et al, 2004), pollution, sedimentation (Fabricius, 2009; Halpern et al., 2013; Yeemin et al., 2013), eutrophication (Littler et al., 2006; Rogers, 1990), ocean acidification (Hoegh-Guldberg et al., 2007; Jokiel et al., 2008; Pandolfi et al., 2011), and ocean warming (Baker et al., 2008; West and Salm, 2003), and recreational activities (Allison, 1996; Hasler and Ott, 2008). Many of these factors can be attributed to coastal development, which often includes the impacts from agricultural land uses and urban development.

These aforementioned factors are stressors to coral reef environments and can diminish coral abundance and diversity, and eventually result in a phase shift. Phase shifts are characterized by a drastic and often persistent change to a reef's composition and structure and are consistently associated with a change from a hard coral-dominated community to one dominated by macroalgae (Nystrom et al., 2009). This shift can occur rapidly if it is induced by a sudden and dramatic event, or can take decades under less extreme, but chronic pressures (Hughes et al., 2010; McClanahan et al., 2002; Nystrom et al., 2000). Coral reefs, along with many ecological systems, exist in different states of equilibrium. Shifting from one state to another is not immediate and occurs over time, which can vary (Graf, 1977). These shifts are typically unidirectional, meaning once a shift occurs to a less desirable state, there is minimal likelihood that the ecosystem will return to the previous state (Nystrom et al., 2009). For example, the presence of an over abundance of algae on reefs prevents the settlement of new corals by covering the substrate and reducing the photosynthetic abilities of the zooxanthellae (a mutualistic symbiont), which reduces the dispersal of coral larvae (Diaz-Pulido et al., 2009).

Monitoring reef environments is of the utmost importance as coastal anthropogenic pressures increase. Between 1970 and 2012, average coral cover for 88 global locations declined from 34.8% to 16.3% (Jackson et al., 2014). In addition, macroalgae increased from 7.0% to 23.6% between 1984 and 1998 (Nystrom et al., 2008). Estimates suggest that over half of the world's coral reefs could be lost in the next 30 years (Wilkinson, 2008). The need for urgent action to protect coral reefs through improved management and monitoring approaches has been acknowledged by the United Nations (United Nations, 2014).

Traditional coral reef monitoring is conducted through in-situ surveys, which is time-consuming, expensive and spatially limited (Dustan et al., 2001). Remote sensing technology permits large spatial-scale analysis of coral reefs while conserving time and expenses (Andréfouët et al., 2001; El-Askary et al., 2014).

One of the most common uses of remote sensing in coral reef studies is to classify benthic cover and estimate reef area. Mumby et al. (2004) used CASI (Compact Airborne Spectrographic Imager) images to assess coral cover at scales of $1m^2$ and $0.25m^2$ (Mumby et al., 2004), which demonstrates that coral reef environments can be observed at high resolution. Maeder et al. (2002) used a 2000 IKONOS image to map benthic cover in Roatan Honduras for Half Moon Bay and Tabyana Bay. Their study identified sand at multiple depths, seagrass at varying densities, hard bottom, and reef cover at two depth intervals. The accuracy assessment resulted in an overall accuracy of 90% and 89%, respectively, for the two sites (Maeder et al., 2002). Benthic cover classification in Roatan, Honduras from a 2001 IKONOS image was conducted by Mishra et al. (2005). Despite the high-resolution benefits of CASI and IKONOS, these sensors are limited by accessibility and spatial and temporal availability.

Owing to their free availability and long-term continuity, Landsat data have also been applied in reef studies. Joyce et al. (2004) utilized unsupervised classification of Landsat-7 ETM+ images and Reef Check in-situ surveys to map bottom types in the Capricorn-Bunker Group reef of the Great Barrier Reef in Australia. The resultant accuracies ranged from 12-74% for individual reefs and 40.9% for the overall reef (Joyce et al., 2004). Bouvet et al. (2003) used Landsat-7 ETM+ data from 1995 and unsupervised classification to assess bottom types in New Caledonia. They concluded the best results were shown in the seagrass bottom type. Unsupervised classification is not feasible for other bottom types due to the Landsat resolution, and that multiple bottom types often classified into a single class (Bouvet et al., 2003). Naseer and Hatcher (2004) used Landsat 7 ETM+ images to estimate reef area in the Maldivian archipelago by classifying and measuring seven reef morphological categories. At 30m resolution, their measurement was only 50.4% of the original estimation of reef area from traditional techniques (Naseer and Hatcher, 2004).

Time series of satellite images can be used to assess benthic cover over time using change detection techniques. Palandro et al. (2003) analyzed four images from 1984 to 2000 using Landsat 5 TM and 7 ETM+ to assess coral cover at Carysfort Reef in the Florida Keys. With the use of change detection techniques, the researchers observed a loss of coral cover by 88% over 16 years. The results were confirmed by in-situ surveys, which observed a 92% loss in coral cover over 18 years (Palandro et al., 2003). A longer-term study (1987 to 2013 using Landsat 5 TM, 7 ETM+, and 8 OLI (Operational Land Images) images) conducted by El-Askary et al. (2014) employed supervised classification and change detection techniques to quantify the changes in coral cover and macroalgae for reefs in the Red Sea near Hurghada, Egypt. The results from 1987 to 2000 showed an increase of macroalgae by 93% and a decrease in coral cover of 40% (El-Askary et al., 2014). From 2000 to 2013, they observed a 19% increase in macroalgae and a 46% decrease in coral cover (El-Askary et al., 2014). Dustan et al. (2001) used texture analysis and change detection to examine reef changes in Key Largo, Florida from 1982-1996. Over this time period, they observed a greening of the substrate suggesting an increase in macroaglae. This was confirmed by in-situ surveys (Dustan et al., 2001).

Many remote sensing-assisted coral reef studies use a combination of classification and change detection techniques to map benthic cover and quantify changes over time (Dustan et al., 2001; El-Askary et al., 2014; Palandro et al., 2003). When discussing these results, authors point to anthropogenic influences as a possible driver of this change, however, they do not attempt to empirically link these variables (Dustan et al., 2001; El-Askary et al., 2014; Vanderstraete et al., 2006). For example, Dustan et al. (2001) noted that reef areas near the coastline demonstrated greater change than reef areas further offshore. It was postulated that a possible cause of the change was anthropogenically driven; however it was not empirically demonstrated (Dustan et al., 2001). Similarly, while El-Askary et al. (2014) did not examine the level of human influence surrounding the Hurghada reefs, it was acknowledged that the observed changes were likely due to human influence and climate factors. Vanderstraete et al. (2006) linked a quantifiable change in human influence with a change in reef cover in the Red Sea near Hurghada, Egypt. Using Landsat imagery along with land use land cover (LULC) and coral reef classification the authors showed direct impacts from increased anthropogenic activity and a degraded reef ecosystem nearshore characterized by a loss of coral cover and an increase in macroalgae (Vanderstraete et al., 2006).

The goal of this research is to examine coral reef response to increasing human activities in Roatan, Honduras. Currently, there has been minimal research conducted on Roatan. The few studies investigating coral reefs using remote sensing studies in this region have been spatially and temporally limited (Maeder et al., 2002; Mishra et al., 2005). In addition, the extent of human development has not been previously quantified. Therefore, for this study, a series of Landsat images from 1985-2015 are used to assess the extent and changes of human development, coral, and macroalgae. Landsat imagery has a wide variety of uses including monitoring land use change and classification (Ellis et al., 2011; Jia et al., 2014; Li and Wang, 2009), detecting coastal bathymetry (Louchard et al, 2003; Pacheco et al., 2014), assessing suspended sediment concentration (Baban, 1995; Min et al., 2012), and analyzing changes in coral reef cover (Andréfouët et al., 2001; El-Askary et al., 2014; Palandro et al., 2008; Vanderstraete et al., 2006). Classification, change detection, and grid-based statistics are employed to assess the relationship between these changing variables.

CHAPTER 2

STUDY AREA AND DATA

2.1 Study Area

The study area encompasses the island and reef area of Roatan, Honduras, which is the largest of the seven Bay Islands off the coast of Honduras. Roatan is situated between $86^{\circ}\ 22'$ and $86^{\circ}\ 37'$ W and $16^{\circ}\ 15'$ and $16^{\circ}\ 25'$ N (Figure 2.1). The climate is tropical with a temperature range from 25 to 29 degrees Celsius (Harbourne et al., 2001). The annual rainfall exceeds 200 cm with a dry season from November to February (Harbourne et al., 2001). The Bay Islands are popular tourist destinations offering fishing and other reef related activities with Roatan serving as the primary destination (Harbourne et al., 2001; Maeder et al., 2002; Stonich et al., 1998). The coral reef environment illustrates high biodiversity with over 66 hard-coral species and 226 species of reef fish (Harbourne et al., 2001). With support from the World Bank, Inter-American Development Bank, United States Agency for International Development, and the United Nations, the Honduran Government signed a joint tourism promotions pact with other countries in Central America in 1988. A similar pact was signed in 1996 that designated the tourism industry as a principal economic growth strategy for Central America (Moreno, 2005). These developments served as an integral kick-start to the booming economic and tourism development on the island since 1988. From 1990 to 2000, the number of tourists rose from 10,000 to 90,000 and the full time resident population increased from 20,000 to 30,000 (Moreno, 2005). To accommodate the increase in population and tourism, hotels, infrastructure, and other ancillary services were constructed (Harbourne et al., 2001). This rapid increase in tourism and economic development puts increased pressure on the environment, especially coral reefs, which attract a majority of the tourists (Maeder et al., 2002; Stonich et al., 1998).

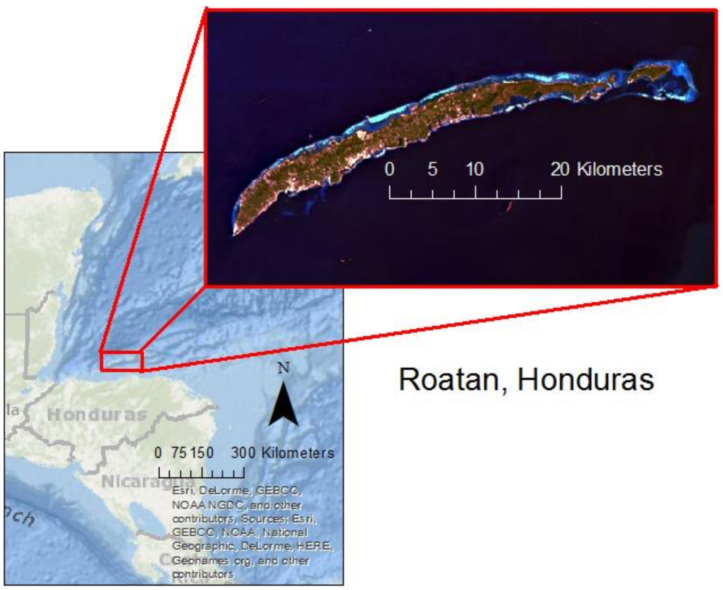

Figure 2.1: Location of Roatan, Honduras. The inset is an example true color Landsat image acquired on February 4, 2015.

2.2 Data

Six Landsat images were selected from 1985 to 2015 for path 017 and row 049 (Table 2.1). Landsat data is the most appropriate sensor to employ for this study due to its long-term image acquisition, applicability over land and sea and 30m spatial resolution (Min et al., 2012). Final image selection was based on acquiring images with little to no cloud cover during Roatan's dry season to reduce the impacts of precipitation, which would anomalously increase the amount of sedimentation and runoff (Harbourne et al., 2001).

A class map from a 4-m Ikonos image on April 1, 2001 (Mishra et al., 2005) is used as a training source to confirm bottom types. Collecting training data from a previously classified image is a tested method when local field data is unavailable (El-Askary et al., 2014). The classification scheme for this image is as follows: 1) dense seagrass; 2) mixed: seagrass/sand/algae; 3) mixed: coral/sand; 4) sand; 5) coral; and 6) deep water. At 4-m resolution, this map has a producer's accuracy of 84% when compared with actual benthic observations (Mishra et al., 2005). This class map and Google Earth imagery are also used for validation sources of the reef area and land cover, respectively.

Table 2.1: Data for image classification. OLI is Operational Land Imager

Satellite	Date	Spatial Resolution	Source
Landsat 4 TM	2/1/1985	30 m	USGS
Landsat 5 TM	3/22/1991	30 m	USGS
Landsat 5 TM	2/13/1995	30 m	USGS
Landsat 7 ETM+	4/10/2001	30 m	USGS
Landsat 5 TM	2/25/2011	30 m	USGS
Landsat 8 OLI	2/4/2015	30 m	USGS
Image for Training / Validation			
Satellite	**Date**	**Spatial Resolution**	**Source**
Ikonos	4/1/2001	4 m	Mishra et al., 2005 - Deepak Mishra personal communication

CHAPTER 3

METHODS

3.1 Image Preprocessing

Each Landsat image was atmospherically corrected using the FLAASH (Fast Line-of-Sight Atmospheric Analysis of Spectral Hypercubes) atmospheric correction algorithm in ENVI. FLAASH converts radiance to surface reflectance and removes the atmospheric noises that vary among images. The images were clipped to the subset of the Island of Roatan and the surrounding reefs. A land/marine mask was created by using an NDVI (Normalized Difference Vegetation Index) thresholding method (Maeder et al., 2001) to isolate the land area and reef area, which were later used for image classification.

3.2 Classification

In land areas, the following classes were identified: urban, forest, grass, upland herbaceous, and wetland. In this study, the urban class represents areas with anthropogenic manipulation, which include roads (paved and unpaved), buildings, and disturbed soil. The upland herbaceous class is representative of various and sparse vegetative cover that does not qualify as forest or grass. Ground truthing data was accessed through Google Earth Imagery synced to the viewer in ERDAS Imagine.

The land use land cover (LULC) classification was conducted using supervised and unsupervised techniques depending on the availability of ground truthing data. The supervised maximum likelihood classifier (MLC) was used on the 2011 and 2015 images. The unsupervised k-means clustering classifier was applied on the 1985, 1991, 1995, and 2001 images due to a lack of temporal-matching ground truthing images. Each image was classified into 100 clusters, which were later merged into the five classes based on the post-priori knowledge from the supervised images.

Classification of marine areas was trained with the 4m IKONOS image as outlined in Mishra et al. (2005) that temporally coincides well with the Landsat 7 ETM+ image from April 10, 2001. The IKONOS image was classified by Mishra et al. (2005). In this study, it was resampled to a 30m pixel size to spatially match the Landsat resolution, and was used to extract training data points from the class map by Mishra et al. (2005). The 2001 Landsat 7 ETM+ image was classified using a supervised method following the identical classification scheme from Mishra et al. (2005): 1) dense seagrass; 2) mixed: seagrass/sand/algae; 3) mixed: coral/sand; 4) sand; 5) coral; 6) deep water. Once the spectral signatures for the 2001 Landsat image were created, they were applied to the remaining images (after performing a histogram match). The same set of classification results in marine areas was thus extracted for all years.

3.3 Change Detection

The Union Matrix function in ERDAS Imagine, which is a pixel-by-pixel change detection method, was employed to assess how each individual pixel changed from one class to another over time. Change detection was performed on six image pairs to determine change in chronological steps as well as from 1985 to 2015 (Figure 3.1). The output images for the land and marine changes were recoded to only examine changes in the urban class and coral and mixed: sand/seagrass/macroalgae class (MSSM), respectively. The mixed: coral/sand (MCS) class was not used as an indicator of coral change because it would have over-represented the coral class by inviting portions of prevalent sand areas into the classification after resampling.

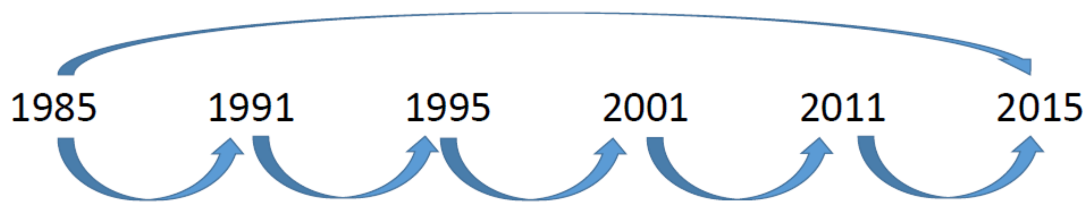

Figure 3.1: Demonstration of paired image comparison.

3.4 Pixel-based Hot Spot Analysis

For the land change maps, hot spot analysis was employed to identify areas with the greatest urban changes. This tool, available in ArcMap, calculates the Getis-Ord Gi* statistic for a feature, which outputs a z-score for each feature (ESRI, 2016). The more significant the clustering, the larger the z-score will be (ESRI, 2016). Pixels that changed to the urban class were coded as a "2" and those that did not were coded as "1". For a cluster to be identified as a hot spot, it has a value of "2" and is surrounded by other "2's". Statistically, a hot spot is identified with a high z-score (< -2.58 or > +2.58) at 99% confidence level ($p < 0.01$).

3.5 Grid Sampling

A spatial gridding system was generated. A grid size of 3000 m by 3000 m was overlaid on the change detection images (Figure 3.2). Only grids that contained land and marine areas were analyzed. Within these grids, the amount of urban pixels, coral change and MSSM change were extracted.

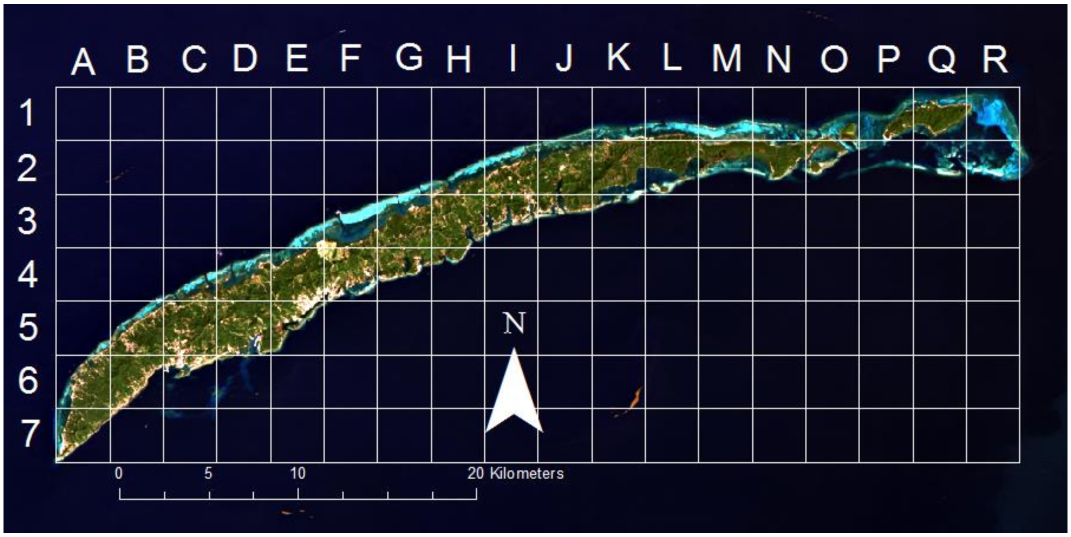

Figure 3.2: The 3000m x 3000m grid layout with alphanumeric naming system.

3.6 Correlation Analysis

Spearman's ranked-order correlation coefficient was used to determine if the changes in urban and coral and MSSM were statistically significant. Spearman's rank is a non-parametric bivariate measure of correlation between two variables (Sheskin, 2007):

$$\rho_s = 1 - \frac{6\sum d^2}{n[(n^2)-1]} \qquad \text{(Eqn. 1)}$$

where *n* is the sample size and *d* is the difference between ranks. The output of the Spearman's rank coefficient, ρ_s ranges from -1 to 1 indicating the degree to which a monotonic relationship exists between two variables. Positive and negative coefficients indicate increasing and decreasing relationships, respectively. Spearman's rank is used to assess the association of the changes between urban and MSSM and between urban and coral. Statistical significance is tested at 90% confidence level ($p < 0.10$).

CHAPTER 4

RESULTS

4.1 Image Classification

Tables 4.1, 4.2, and 4.3 show the accuracy assessments for the 2001, 2011, and 2015 image classifications in land areas. Kappa values greater than 0.80 represent strong agreement, whereas Kappa values between 0.4 and 0.8 represent modest agreement (Jensen, 2005). The overall classification accuracy for the entire image ranged from 82 to 87% with Kappa values between 0.77 and 0.83. The Conditional Kappa values for the urban class of all images in this study ranged from 0.64 to 0.87.

Table 4.1: Accuracy assessment of the 2015 LULC classification. The upland herbaceous class is denoted as "UH".

		Reference								
		Urban	Forest	Grass	UH	Wetland	Total	Kappa	User's	Producer's
Classified	Urban	44	3	1	8	4	60	0.64	73	94
	Forest	0	30	0	0	0	30	1.00	100	68
	Grass	2	8	18	2	0	30	0.55	60	95
	UH	1	3	0	26	0	30	0.83	87	72
	Wetland	0	0	0	0	30	30	1.00	100	88
	Total	47	44	19	36	34	180			
	Overall: 82% ; Overall K: 0.77									

Table 4.2: Accuracy assessment of the 2011 LULC classification. The upland herbaceous class is denoted as "UH".

		Reference								
		Urban	Forest	Grass	UH	Wetland	Total	Kappa	User's	Producer's
Classified	Urban	27	1	0	2	0	30	0.87	90	82
	Forest	1	28	0	1	0	30	0.91	93	80
	Grass	2	5	17	6	0	30	0.51	57	100
	UH	0	0	0	29	1	30	0.96	97	76
	Wetland	3	1	0	0	26	30	0.84	87	96
		33	35	17	38	27	150			
	Overall: 85%; Overall K: 0.81									

Table 4.3: Accuracy assessment of the 2001 LULC classification. The upland herbaceous class is denoted as “UH”.

		Reference								
		Urban	Forest	Grass	UH	Wetland	Total	Kappa	User's	Producer's
Classified	Urban	24	0	2	3	1	30	0.75	80	83
	Forest	0	29	0	1	0	30	0.96	97	88
	Grass	0	2	26	2	0	30	0.83	87	90
	UH	5	1	1	22	1	30	0.67	73	79
	Wetland	0	1	0	0	29	30	0.96	97	94
	Total	29	33	29	28	31	150			
	Overall: 87% ; Overall K: 0.83									

Table 4.4 shows the LULC percent covers for each Landsat image. From 1985 (Figure 4.1) to 2015 (Figure 4.2), the dominant land cover land use was forest, representing 61.0% of land area in 1985 and 51.5% in 2015. In 1985, the second most dominant cover type was upland herbaceous area with 13.4%, whereas the urban class covered 6.3% of land area. By 2015, the forest class remained as the dominant class, whereas the urban class rose to 24.6% (Table 4.4; Figure 4.2). Additional classified images are located in Appendix A.

Table 4.4: LULC classification from 1985 to 2015.

	Percent Cover					
	1985	1991	1995	2001	2011	2015
Urban	6.3	8.5	9.9	14.8	20.5	24.6
Forest	61.0	52.7	61.5	56.5	50.7	51.5
Grass	12.5	13.4	11.9	17.5	17.5	11.9
Upland Herbaceous	13.4	18.9	11.2	6.1	7.1	6.8
Wetland	6.8	6.5	5.5	5.1	4.2	5.2

The accuracy assessment of the marine classified 2001 Landsat image produced an overall accuracy of 86% and an overall Kappa of 0.83 (Table 4.5). Table 4.6 shows the marine classification results for the Landsat time series. In 1985, the dominant benthic class was sand (Figure 4.1). The coral and MSSM classes covered 13.7% and

2.8%, respectively. By 2015 (Figure 4.2), sand remained the dominant class at 31.0%, however the coral class increased to 14.9% and the MSSM class increased to 4.4%.

Table 4.5: Accuracy assessment of the 2001 marine classification.

		Reference									
		Sand	Coral	MSSM	MCS	Deep Water	Seagrass	Total	Kappa	User's	Producers
Classified	Sand	18	2	0	0	0	0	20	0.88	90	90
	Coral	2	17	0	1	0	0	20	0.82	85	81
	MSSM	0	0	14	0	0	6	20	0.66	70	100
	MCS	0	2	0	17	0	1	20	0.82	85	90
	Deep Water	0	0	0	0	20	0	20	1.00	100	91
	Seagrass	0	0	0	1	2	17	20	0.81	85	71
	Total	20	21	14	19	22	24	120			
	Overall: 86% ; Overall K: 0.83										

Table 4.6: Marine classification from 1985 to 2015

	Percent Cover					
	1985	1991	1995	2001	2011	2015
Sand	31.0	31.9	29.3	28.8	29.0	31.0
Coral	13.7	19.5	21.5	31.8	21.5	14.9
MSSM	2.8	2.8	2.2	4.7	3.3	4.4
MCS	6.6	7.9	6.6	10.5	5.0	7.7
Seagrass	45.9	37.9	40.4	24.2	41.2	42.0

4.2 Land and Marine Changes in 1985-2015

Figure 4.3 shows the extent of urban change from 1985 to 2015. Overall, there was an increase in urban area by 18.3%. Urban areas in 1985 were mainly limited to the southwestern coastal areas and sparse coverage inland. By 2015, nearly the entire coastline was dominated with the urban class, most noticeably in the western portion of the island. Figure 4.4 shows the changes in coral and MSSM from 1985 to 2015. Overall, there was a net increase in coral by 1.2% and a net increase of MSSM by 1.6%. A majority of the changes occur on the north and eastern end of the island.

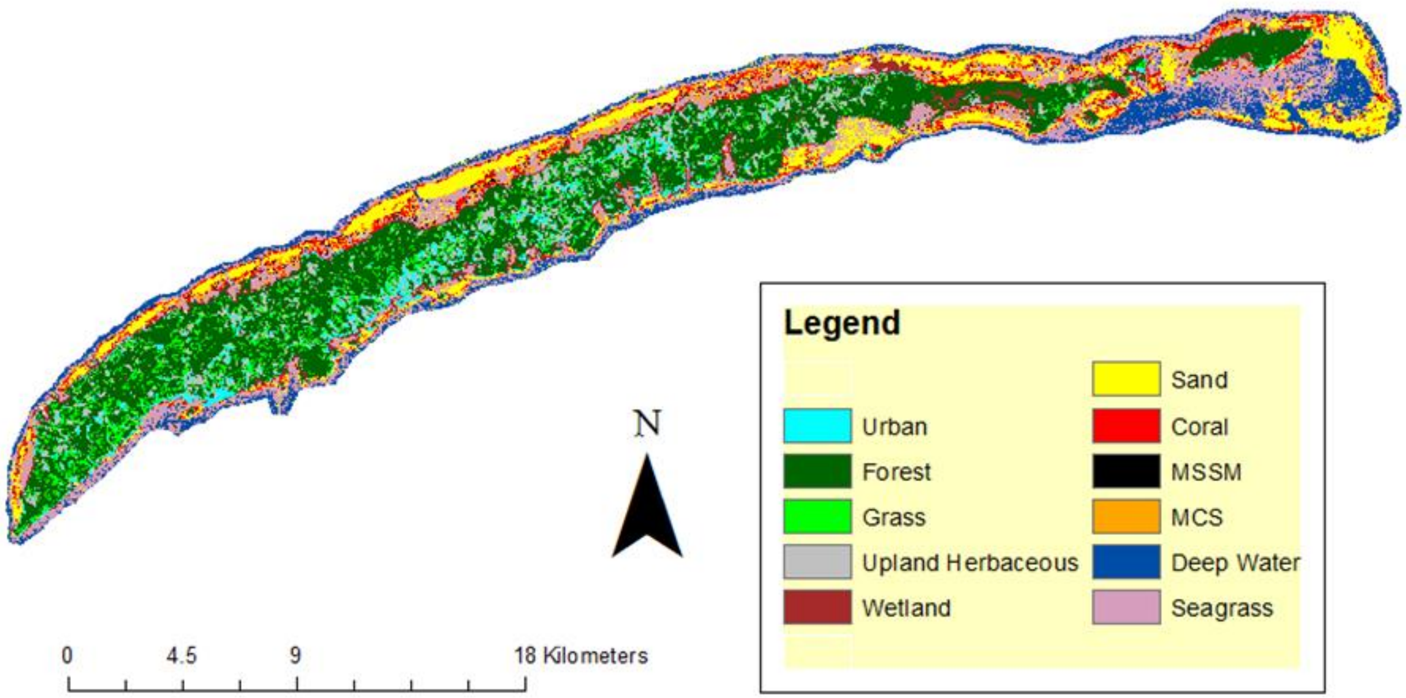

Figure 4.1: The 1985 classified results of Roatan and surrounding reef.

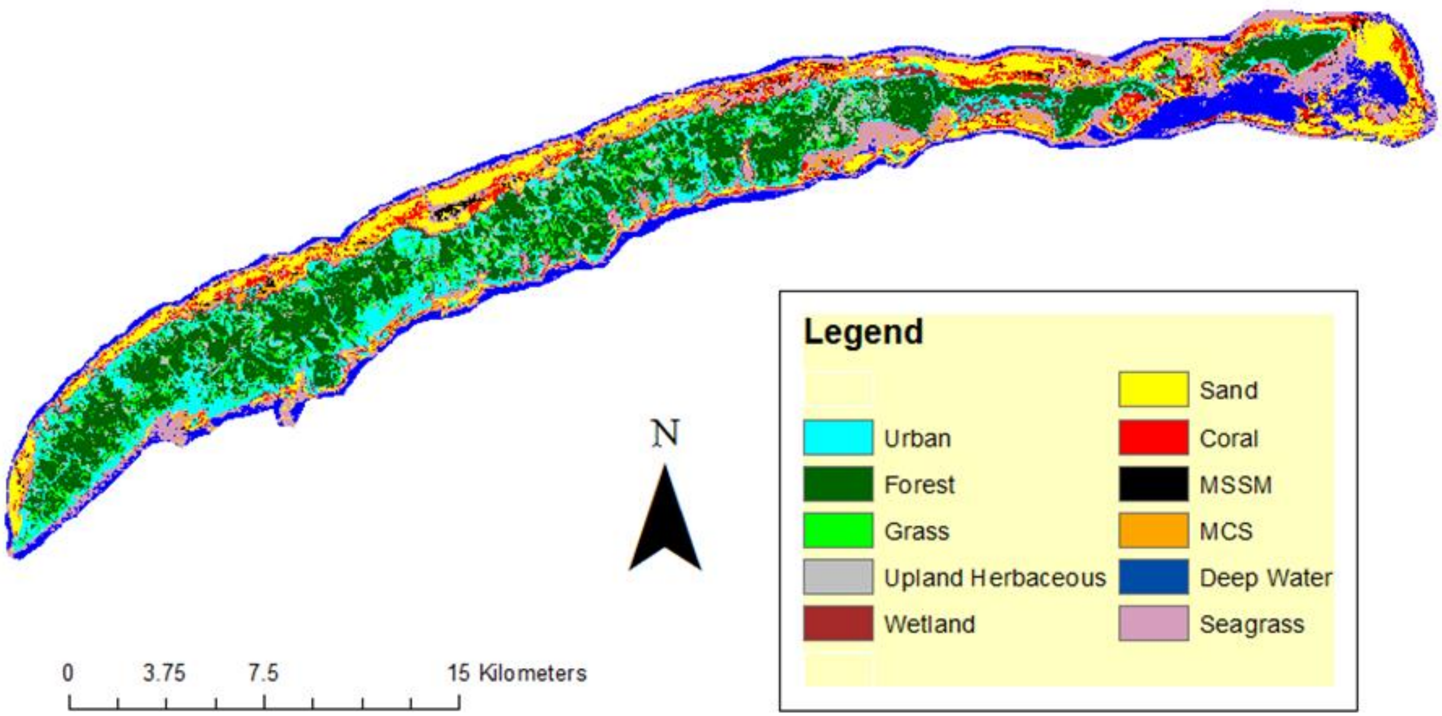

Figure 4.2: The 2015 classified results of Roatan and surrounding reef.

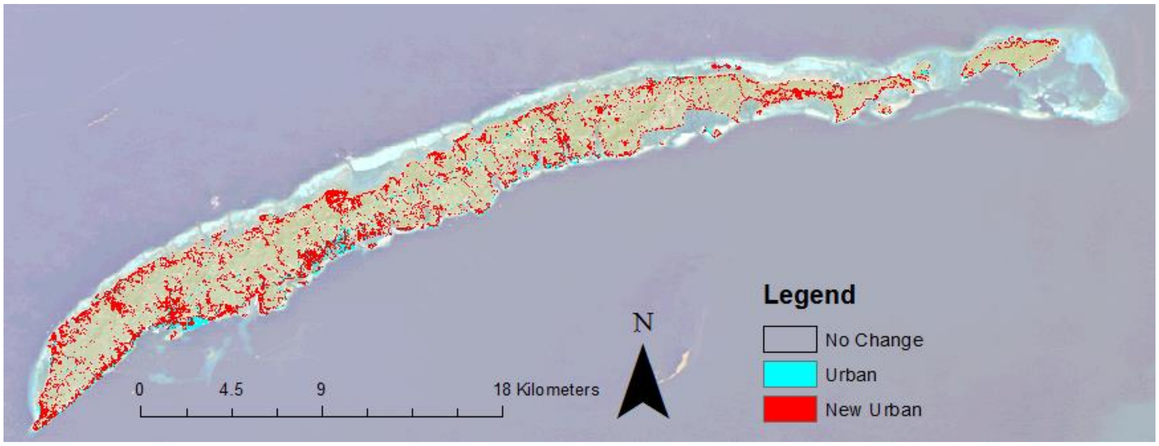

Figure 4.3: Distribution of urban change (1985 to 2015) of Roatan.

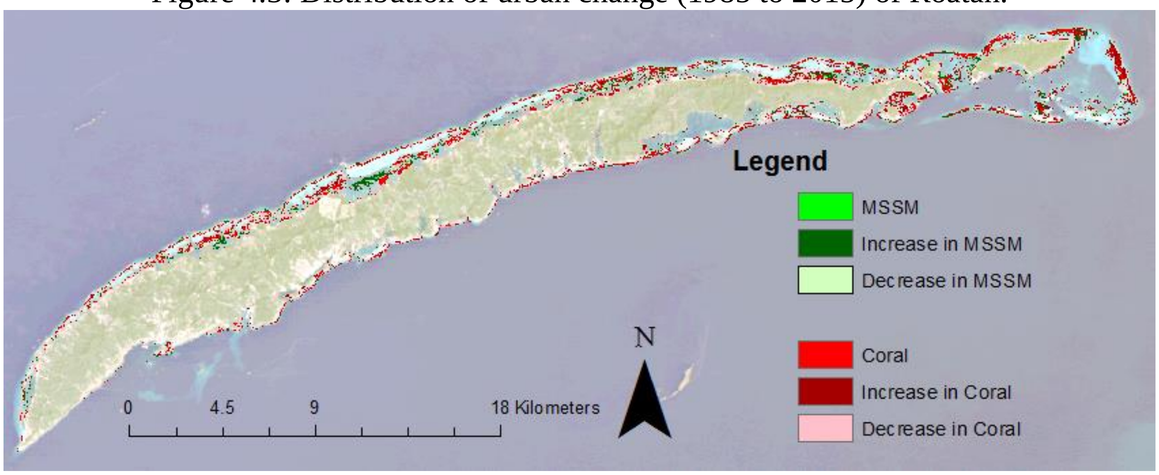

Figure 4.4: Distribution of macroalgae and coral changes (1985 to 2015) of Roatan.

4.3 Hot Spots of Land Development

Using Grid A6 as an example, Figure 4.5 demonstrates the outputs of hot spot analysis. Red areas in the figures indicate urban clusters that are statistically (hot spots). In grid analysis, if a hot spot is identified within a grid the entire grid is designated as a hot spot grid. This study identified 17 grids where the new urban change was statistically significant (Figure 4.6, red grids). The yellow grids in Figure 4.6 designate grids that were not hot spots, but contain both land and marine areas. Grids K1 and L1 contain both land and marine area but were excluded from this study due to the prevalence of natural sand, which erroneously classifies as urban. Table 4.7 shows the year each grid was identified as a hot spot.

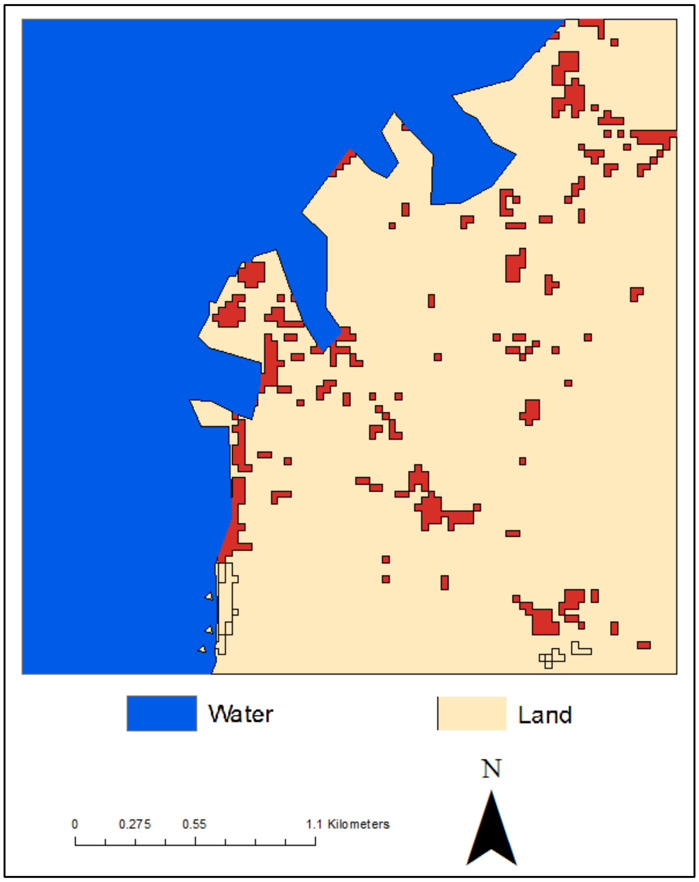

Figure 4.5: Example urban hot spots for grid A6 (1985-1991). The red area indicates a hot spot, therefore the entire grid is designated as a hot spot grid.

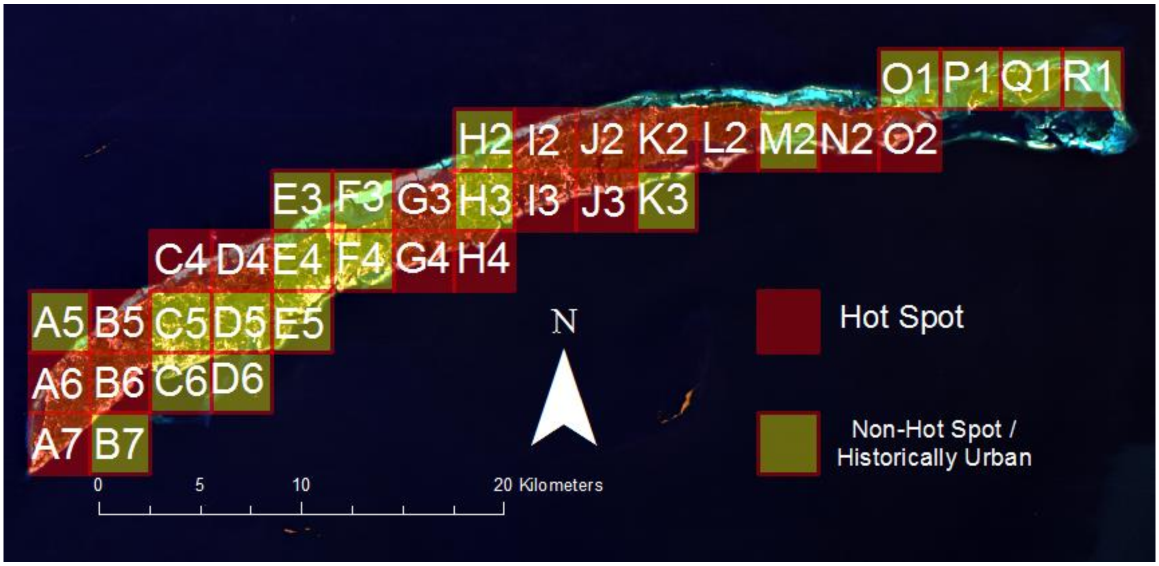

Figure 4.6: Location map of analyzed grids showing hot spot and non-hot spot grids.

Table 4.7: Hot spot grids and the corresponding

Hot Spot Grids				
1985-91	1991-95	1995-01	2001-11	2011-15
A6	D4	J2	L2	A7
B6	G3		N2	I2
B5	G4		O2	I3
	K2		C4	J3
	H4			

4.4 Relationships between land development and coral changes.

Spearman's rank-ordered correlation coefficient analysis found seven statistically significant relationships at 90% confidence (Table 4.8). Within the hot spot grids, two grids were statistically significant. Grid A6 showed a positive monotonic relationship between the urban and MSSM class with $\rho_s = 1$ and $p = 0.0167$ (Figure 4.6). Grid K2 showed a negative monotonic relationship between the urban and coral class with a $\rho_s = -0.9$ and $p = 0.0833$. Among the non-hot spots, five grids were statistically significant. Three of these grids (F3, A5, and R1) had a statistically significant positive monotonic relationship between urban and MSSM indicating an increase in macroalgae. However, grids C6 and D6 were a statistically significant and negatively monotonic between urban and MSSM suggesting that as urban increases, the MSSM class decreases. Grid C6 was the only location where a statistically significant decreasing monotonic relationship between urban and coral was found among the non-hot spots. If considering the entire study area for the urban and coral classifications, $\rho_s = -0.27$, which suggests a negative monotonic relationship. Additional grid results are located in Appendix B.

Spatially, four out of the seven statistically significant grids are located on the far western end of the island (Figure 4.7). Of those grids, the ones located on the northern coast were positively monotonic between urban and MSSM, whereas the grids along the southern coast were negatively monotonic.

Table 4.8: Results of Spearman's Correlation Coefficient.
Blue cells indicate statistically significant relationships ($p < 0.10$).

Results of Spearman's Correlation Coefficient					
	Grid	Urban and MSSM		Urban and Coral	
		ρ_s	P-value	ρ_s	P-Value
	Hot Spots				
1	A6	1.0000	0.0167	-0.5000	0.4500
2	B6	-0.4617	0.4333	-0.7000	0.2333
3	B5	0.5643	0.3667	-0.3000	0.6833
4	D4	0.3000	0.6833	-0.1000	0.9500
5	G3	0.3000	0.6833	-0.1000	0.9500
6	G4	-0.1000	0.9500	-0.3000	0.6833
7	H4	0.6000	0.3500	0.2000	0.7833
8	K2	0.2000	0.7833	-0.9000	0.0833
9	J2	0.6000	0.3500	-0.2000	0.7833
10	L2	-0.5000	0.4500	-0.7000	0.2333
11	N2	-0.2000	0.7833	-0.5000	0.4500
12	O2	0.5000	0.4500	-0.8000	0.1333
13	C4	0.8000	0.1333	-0.2000	0.7833
14	A7	0.6000	0.3500	-0.3000	0.6833
15	I2	0.1000	0.9500	0.4000	0.5167
16	I3	-0.2000	0.7833	-0.5000	0.4500
17	J3	0.5000	0.4500	-0.2000	0.7833
	Non-Hot Spots / Historically Urban				
18	C6	-0.9000	0.0833	-0.9000	0.0833
19	E5	-0.6000	0.3500	0.3000	0.6833
20	E4	0.3000	0.6833	-0.6000	0.3500
21	F3	0.9000	0.0833	-0.2000	0.7833
22	F4	0.1026	0.9000	-0.3000	0.6833
23	H2	0.7000	0.2333	-0.2000	0.7833
24	M2	-0.2000	0.7833	0.7000	0.2333
25	D5	0.3000	0.6833	-0.2000	0.7833
26	E3	0.6669	0.2667	0.6669	0.2667
27	A5	1.0000	0.0333	0.1539	0.8333
28	B7	0.6669	0.2667	-0.7000	0.2333
29	D6	-0.9000	0.0833	-0.7000	0.2333
30	K3	0.5000	0.4500	-0.8000	0.1333
31	H3	0.1000	0.9500	0.0000	1.0000
32	O1	-0.1000	0.9500	-0.7000	0.2333
33	P1	-0.2000	0.7833	0.2000	0.7833
34	Q1	-0.6000	0.3500	-0.6000	0.3500
35	R1	0.8721	0.1000	0.0000	1.0000
36	C5	-0.0513	1.0000	-0.6000	0.3500
	Overall				
		-0.0723	0.3418	-0.2741	0.00024

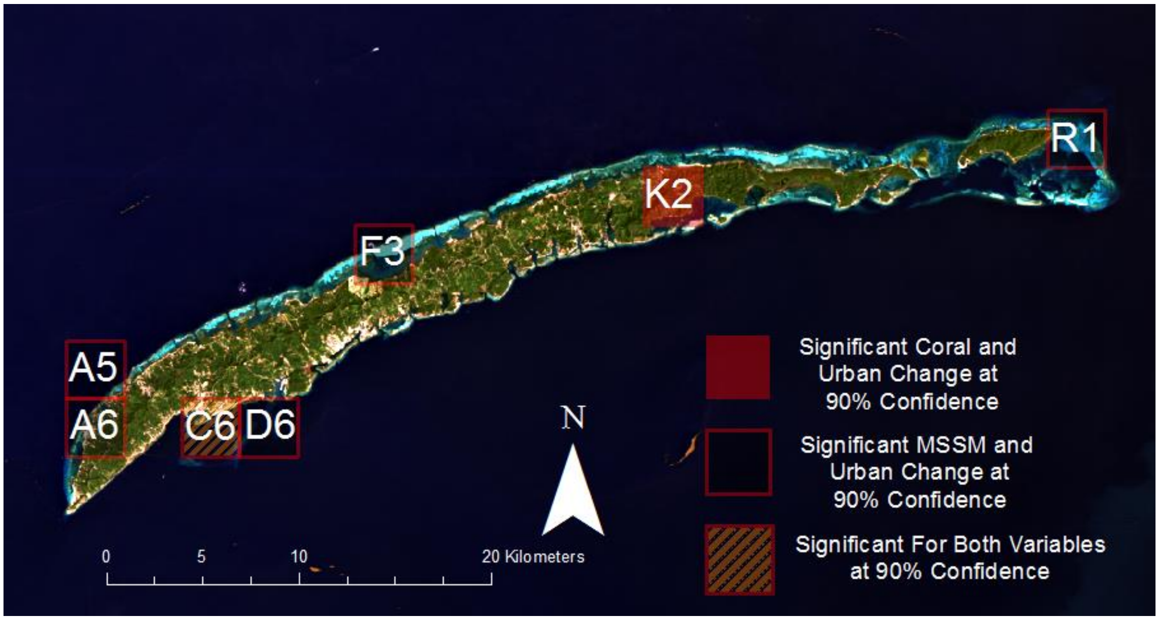

Figure 4.7: Locations of statistically significant grids.

CHAPTER 5

DISCUSSION

While decreasing coral cover and increasing macroalgae have been observed in coral reefs associated with anthropogenic impacts in previous studies (e.g., Dustan et al., 2001; El-Askary et al.,2004; Vanderstraete et al., 2006), these effects have not been quantitatively assessed. Vanderstraete et al. (2006) notes that the observed negative changes on the coral reef system in Hurghada, Egypt are likely linked to coastline changes, specifically landfill and dredging activities. However, there is no calculation of the LULC changes (Vanderstraete et al., 2006). El-Askary et al. (2004) also examined reef health near Hurghada, Egypt and concluded that the likely cause was anthropogenic, specifically a booming tourism industry and urbanization, but this coastal change was neither quantified nor linked to off-shore changes (El-Askary et al., 2004). Dustan et al. (2001) mentioned the various LULC practices taking place in Key Largo and concluded that the greatest reef degradation was occurring near areas of human manipulation, but did not statistically link nor map these relationships (Dustan et al., 2001).

This research quantitatively assesses the relationships between urbanization, coral, and microalgae and determined that there was not a consistent relationship, but they are spatially unique. Furthermore, these results suggest that macroalgae populations respond more readily to urban change compared to coral since twice as many grids showed a statistically significant relationship between urban and MSSM than between urban and coral.

In addition, grids A5 and A6, which were both positively monotonic between urban and MSSM are geographically adjacent, and located on the southwest tip of the island. Grids C6 and D6, also on the southwest portion of the island and adjacent to each other, were both negatively monotonic between urban and MSSM. These findings suggest that the spatial relationship between these variables may exceed the 3000m by 3000m scale of the current grids. Observing that all the significant grids were neighbored by at least one hot spot grid strengthens this point. Spatial dependencies in coral reef degradation responding to urban development deserves further investigation in the future.

Among the seven statistically significant grids, five of them were non-hot spots or historically urban areas. This suggests that a possible reason why the hot spot grids did not show as many significant relationships is that enough time has not passed. As mentioned in the phase shift literature (Graf, 1977; Hughes et al., 2010; McClanahan et al., 2002; Nystrom et al., 2000), a shift in states is not immediate and responds differently to different levels of disturbance.

Only limited grids were observed to reveal relationships between human development and coral/macroalgae change. This may be attributed to classification errors, applying a broad definition of an urban class, the pre-determined spatial scale in the grid analysis, as well as natural factors.

The classification phase of this research relied heavily upon 30m Landsat imagery, which was shown to be appropriate by many authors such as Palandro et al.

(2003), Bouvet et al. (2003), Dustan et al. (2001), El-Askary et al. (2014), and Vanderstraete et al. (2006), but is coarse compared to higher resolution imagers such as IKONOS or Quickbird. In this research, urban classes showed Kappa values ranging from 0.64 to 0.87. While acceptable, it allows room for improvement with better images and training data. The lower Kappa value for urban can mostly be attributed to a confusion with the upland herbaceous class in land classification. In addition, marine classification was based off of the Mishra et al. (2005) image, which is limited to the extreme western portion of the island thereby limiting the area available for training. Further, the MSSM class resulted in a lower Kappa value mainly due to confusion with the seagrass class.

The main LULC used in this study was urban. Urban comprised a wide range of cover types from high-density areas with buildings to sand roads . These different manifestations of 'urban' interact differently with the environment, for example, where a dense urban area with paved roads and buildings will have a large impervious area thus increasing possible runoff into the coastal environment. Alternatively, a sand road is more pervious and runoff would be limited. The process of urbanization might also be more damaging than the presence of urban cover. The initialization of development may cause a sizeable change in the amount of sediment entering the reef areas. However, since most of the statistically significant grids were non-hot spots and historically urban, the process of urbanization does not seem to be an important factor. Quantifying the level or type of urbanization was not considered here, but could be a factor in detecting phase shifts and an avenue for additional research.

The scale of analysis was based on the 3000m by 3000m grid system. Altering the spatial extent of the grids would produce different results especially since there is evidence that the effects of human development can extend beyond the current grid size. In addition, each grid is unique and contains a different land to water ratio. For example, a grid that contains mostly marine area with a small portion of land may actually be affected by urban land use, but it is not identified due to the constraints of the grid. Future research may involve combining adjacent grids to better capture the total effect of the surrounding urban land use on the reef environment.

Integrating additional natural and social variables, such as precipitation, sea surface temperature (SST), sea surface salinity (SSS), ocean acidification and population data, into this research would be valuable for future study. Being able to exclude a natural variable's influence would strengthen the argument that there is a correlation between local anthropogenic activity and degraded coral reef ecosystems. In addition, population data would be another representation of human influence and may also demonstrate a stronger statistically significant relationship for coral and macroalgae.

CHAPTER 6

CONCLUSIONS

To the best of our knowledge, this study was the first to utilize remote sensing data and techniques to quantitatively assess the relationship between the LULC and the impacts of human development on the reefs of Roatan, Honduras. Using a Landsat image time series from 1985 to 2015, this research demonstrated that the extent of urbanization increased by 18.3%. During that same time period, there was an overall increase in coral cover and MSSM of 1.2% and 1.6%, respectively. To better understand the spatial relationships of change, Roatan was segmented using a 3000m by 3000m grid and correlation analysis was conducted to compare the urban, coral, and macroalgae classes. This analysis demonstrated that seven of the 36 grids had a statistically significant relationship between human development, coral, and macroalgae. Identifying areas where these relationships are present is valuable for marine park managers and other coastal conservation groups for determining at-risk areas that are entering a phase shift. Additional research involves improving classification accuracies, altering the size of the grids as well as combining grids to determine at what scale these relationships are most evident, and including other environmental and social data. Coral reef monitoring should consist of a combination of remote sensing techniques as well as in-situ surveys. This research could be applied to other locations and be used to identify at-risk areas outside of Roatan.

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APPENDIX A – ADDITIONAL CLASSIFIED IMAGES

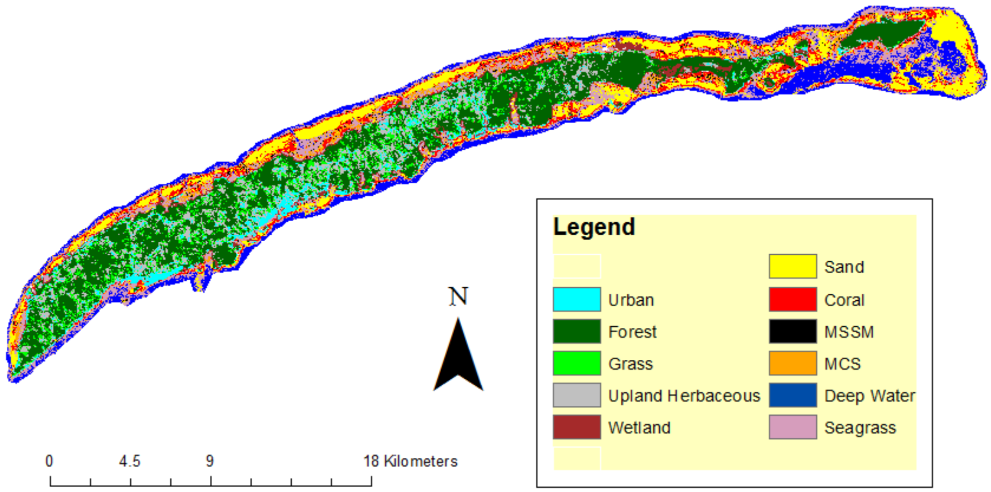

Figure A1: 1991 classified image

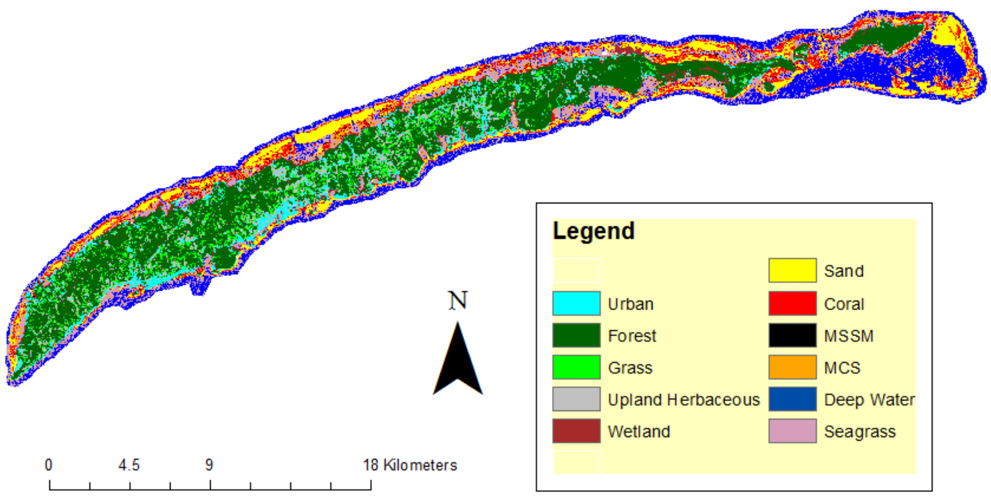

Figure A2: 1995 classified image

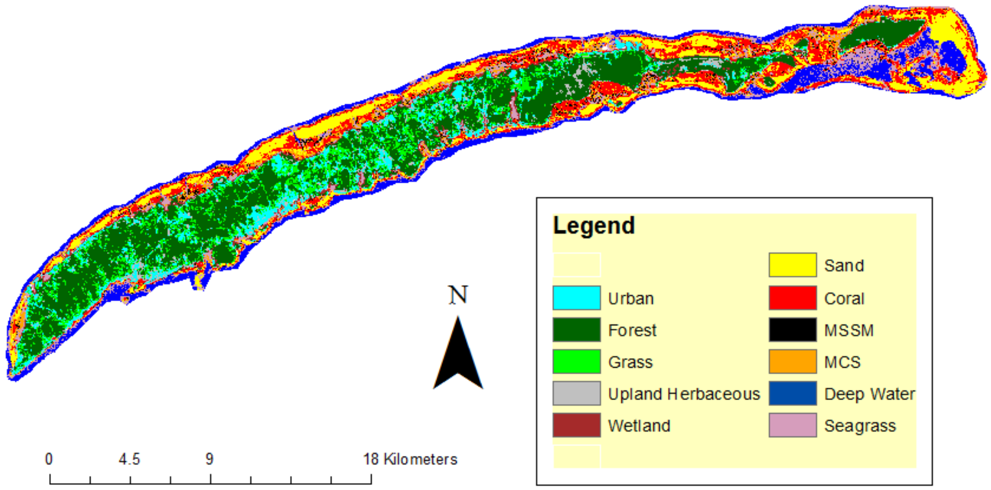

Figure A3: 2001 classified image

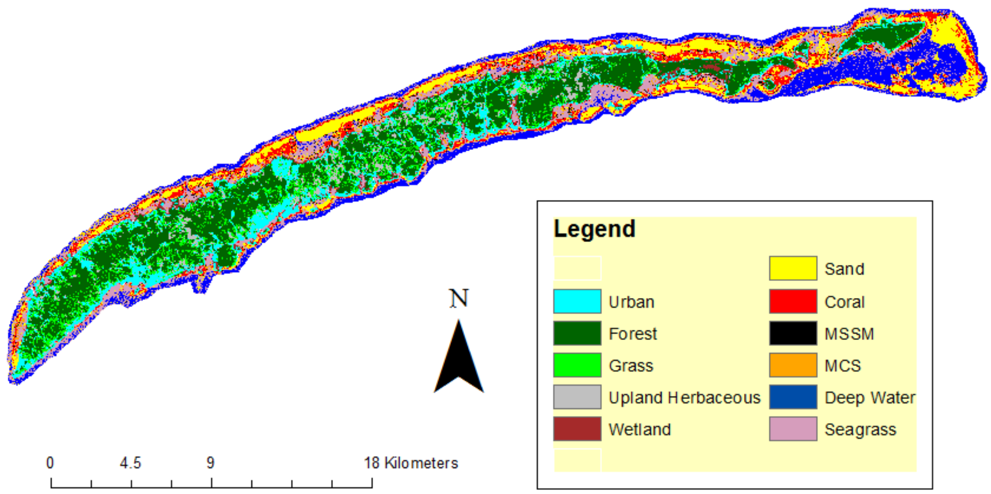

Figure A4: 2011 classified image

APPENDIX B – ADDITIONAL GRAPHS FOR SIGNIFICANT GRIDS

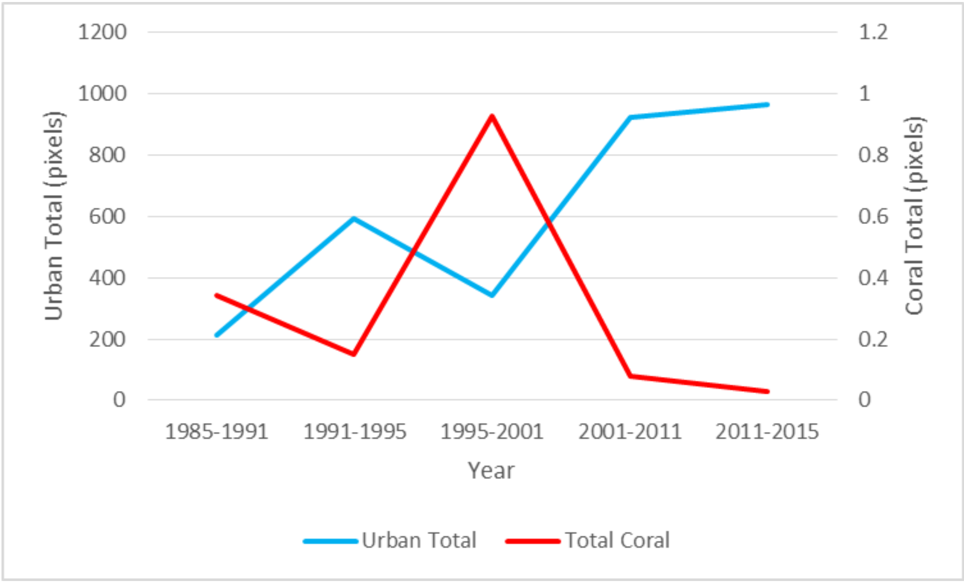

Figure B1: Urban change and coral change for hot spot grid K2

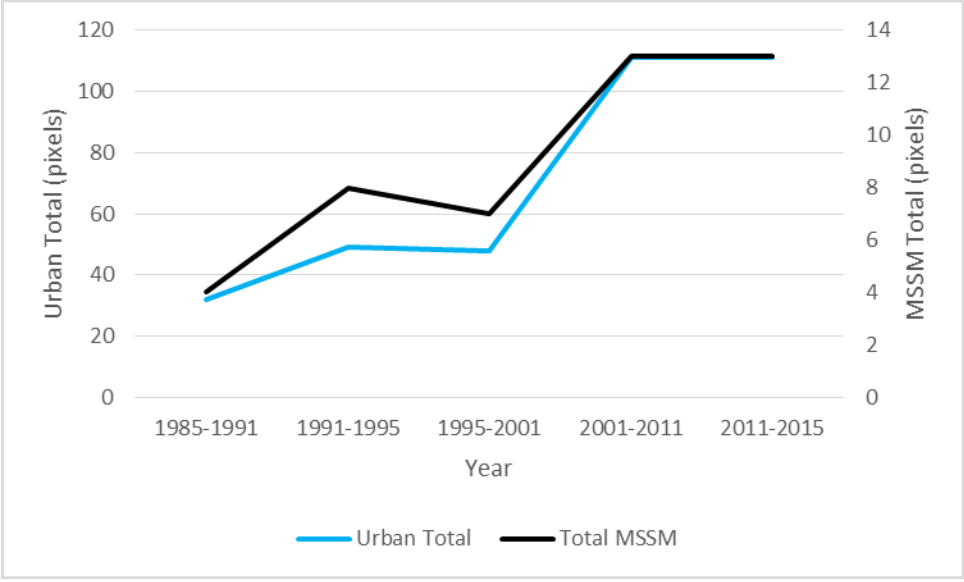

Figure B2: Urban change and MSSM change for grid A5

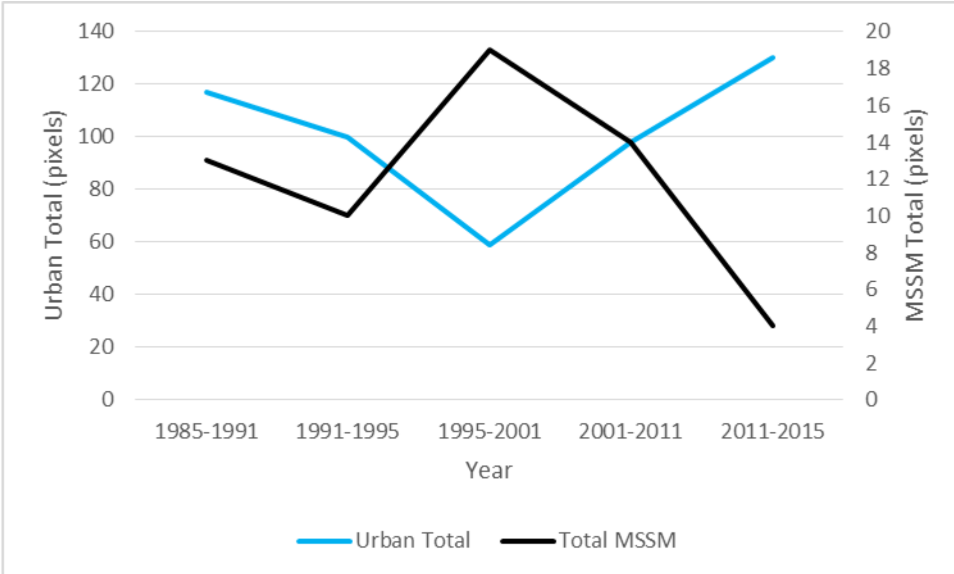

Figure B3: Urban change and MSSM change for hot spot grid D6

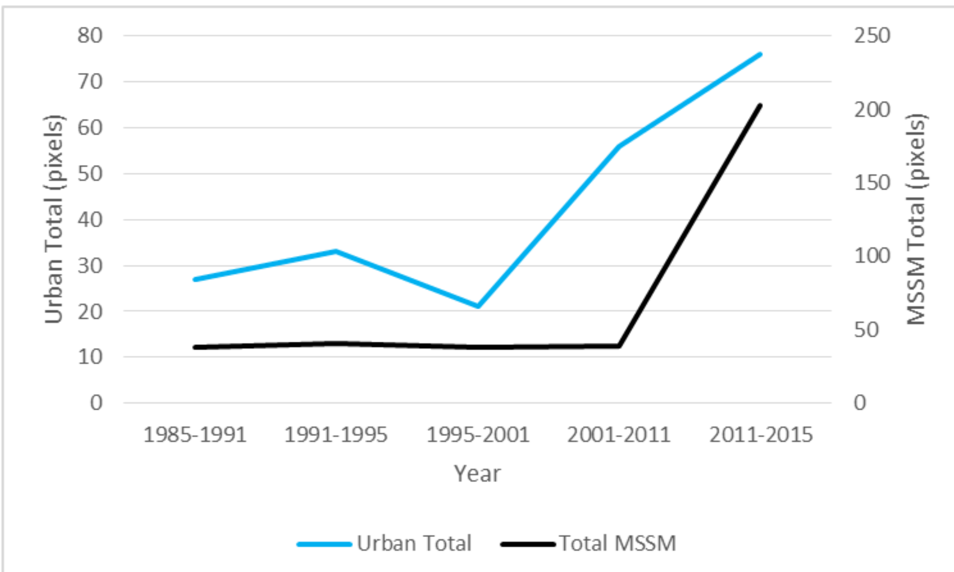

Figure B4: Urban change and MSSM change for hot spot grid R1