USE OF LANDSAT FOR MANAGING COASTAL ECOSYSTEMS OF THE USVI

Dissertation

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ABSTRACT

Over the past several decades, benthic and terrestrial ecosystems of the United States Virgin Islands have been impacted by changes in land use. The increased amounts of road and home construction have contributed to soil erosion, resulting in significant land based runoff during heavy rains. While it is well documented that excessive levels of land based sediments impact coral reefs and seagrasses negatively, the challenge remains for policy makes to enact legislation that strictly regulates land use activities throughout the watershed.

The objective of this study is to determine whether Landsat may be used to demonstrate a correlation between terrestrial change and benthic habitat change for the USVI. Based on this, Landsat may provide a planning tool in management and policy decisions regarding the protection, regulation, monitoring, or in-situ data analysis of coastal regions.

Five Landsat satellite images of the USVI that spanned a total of 16 years were atmospherically corrected, orthorectified and coregistered. Unsupervised classification, accuracy assessment, and change detection techniques were applied to the terrestrial and benthic habitats of each image. The changes in

habitats were evaluated to determine whether a correlation could be found between the types of change observed on land and those in the benthic habitats.

The data indicated a shift from live coral to dead, algae-covered coral in watersheds where development had occurred. The data were more conclusive for St. Croix than for St. Thomas and St. John, which were smaller in size and had a greater amount of cloud cover than St. Croix. It was concluded that Landsat satellite data may be used as a tool by coastal policymakers and managers in making decisions, such as identifying watersheds where financial resources should be expended for further analysis or where expenditure on strict enforcement of land use regulations may be most effective for benthic habitat protection or restoration.

To my brother Deepuk,
who dove with sharks, worked in the Bering Sea, danced bhangra.
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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Motivation for Research

Coastal areas throughout the globe are increasingly threatened by habitat destruction, eutrophication, sedimentation, over fishing and other forms of disturbance. For several decades, the Caribbean Sea and its watersheds have exhibited signs of environmental stress, including species extinction, mass mortality of coral organisms, and reduction of marine resource productivity (Rivera-Monroy, et al. 2004).

Studies suggest a combination of natural and anthropogenic factors are the cause for the decline in benthic ecosystem health (Gardner et al., 2003). Rural and urban development have been shown to contribute to higher levels of land-based runoff (Hundecha and Bardossy, 2004), a significant form of nonpoint source pollution in the Caribbean. Sediment deposition may lead to eutrophication and smother corals or may exacerbate the effects of natural bleaching events. Nemeth and Nowlis (2001) found the peak of the 1998 bleaching event in the United States Virgin Islands (USVI) occurred during a period of higher than average deposition of land-based sediments to coastal habitats.

Coastal Policy and Management in the USVI

The primary nonpoint source pollutants impairing USVI coastal waters and degrading USVI coral reef systems are sediment and partially-treated sewage. The majority of the construction taking place in the USVI does not employ effective erosion or sediment control practices. Studies have indicated that in St. John's Hawksnest Bay, increased soil erosion from the watershed led to a gradual reduction in coral growth (Rogers, 1990).

In 2002, the USVI Department of Planning and Natural Resources (DPNR) received federal approval for its Coastal Nonpoint Pollution Program. The program designates terrestrial areas nearer the seashore as Tier 1 and lands further from the coast as Tier 2. While construction activities in Tier 1 must meet specific regulatory criteria, development in Tier 2 is subject to little to no regulation. Because a significant amount of development occurs in Tier 2 areas, many land-based pollutants are unregulated, leading to significant sediment deposition at the base of coastal watersheds following heavy rains. Many reefs, marine meadows, salt ponds, and mangroves are located at the base of watersheds with steep hills where road and home construction has increased steadily over the last two decades.

The Virgin Islands Coastal Zone Management Act, Section 903(b), states the need to preserve the function and integrity of coral reef ecosystems. USVI Marine reserves, which include the Virgin Islands National Park and the East End Marine Park, have been established to protect reef ecosystems and to allow

recovery of reef fish (USGS, 2006). Studies found the size and abundance of several fisheries in the Virgin Islands National Park to be similar within and outside park boundaries, indicating that the reserve may not be effective in serving as a refuge for fish. The control of erosion, sedimentation runoff and sewage discharge from all parts of the USVI have veen noted as integral to protecting marine reserves and improving the overall quality of USVI coastal waters.

Research indicates that unregulated development and the associated decrease in vegetation on land with steep terrain and easily eroded soils contribute to significant amounts of terrestrial runoff. It is also well documented that excessive amounts of terrestrial runoff impact both coral and seagrass habitats negatively. Although terrigenous sources have often been hypothesized as the cause for coral decline, a quantitative link between landscape processes on contiguous drainage basins and coral growth has not been established (Miller and Cruise, 1995).

Coastal managers in the USVI have indicated that the formulation of policies that more effectively protect benthic habitats from land-based pollutants requires evidence that indicates benthic ecosystem are impacted by land use activities from throughout the watershed (Hodge, pers. comm., 2004).

Research Objective

The objectives of this study are to determine whether Landsat may be used to describe change over a sixteen year period in both terrestrial and benthic habitats of the USVI, to identify a correlation between benthic habitat change and

terrestrial habitat change, and to provide information that may be used by coastal policy makers and planners in their decision making regarding the expenditure of resources for monitoring, analysis, or regulation of coastal habitats.

Studies have demonstrated that satellite and airborne remote sensing technology provides a cost effective and objective monitoring tool for determining the long-term impacts of land use change (Dekker et al., 2005) over large spatial scales. Using five sets of Landsat images between the years 1985 and 2001, the extent and type of change on land and in the benthic habitats of the USVI will be evaluated. Based on research that indicates terrestrial runoff impacts both coral ands seagrass habitats negatively, land use change in pristine and developed watersheds will be compared to determine whether the maps derived from the Landsat imagery reveal a correlation between land use change and benthic habitat change.

Landsat imagery alone cannot be used to prove that terrestrial runoff from a watershed causes decline in benthic ecosystem health at the base of the watershed. However, Landsat data may provide a decision making tool to determine where to expend resources for further study. Used in conjunction with in situ water quality analysis and other field studies, Landsat data may be used to form a causal connection between land use activity at the watershed level and benthic habitat change in coastal areas located at the base of the watershed.

By helping to identify the extent to which land use activities within a watershed impact benthic habitats at the watershed base, the methodology may

provide policy makers in the USVI with information that enhances the formulation and implementation of measures that more rigorously protect coastal areas from the impacts of both direct and indirect sources of land-based pollution.

Literature Review

Impact of Development on Coastal Watersheds

Coastal margins, the interface between the land and the sea, are complex, dynamic environments where a vast array of biological, chemical, geological, and physical processes occur over multiple time and space scales (Miller et al., 2005). Nearshore ecosystems are vulnerable to changes in currents, sedimentation, temperature and salinity, as well as to anthropogenic activities that lead to habitat alteration or destruction (Cuevas-Jimenez and Ardisson, 2002). Coastal areas, highly valued for their natural harbors and as sites for resorts, ports, industry, and residence, have experienced tremendous growth in population density. The population of U.S. coastal counties increased by 69% between 1950 and 1980, making population densities in these areas five times greater than areas further inland (Cicin-Sain and Knecht, 1998). In 2010, U.S. coastal populations are expected to reach 127 million, rising 14% from their 1990 level (Cicin-Sain and Knecht, 2000).

Changes in terrestrial landscapes may be related to changes in the water quality of a watershed (Klemas, 2001). Increasingly impacted by industrial and residential development, coastal waters receive sewage, sediments, nutrients, pesticides, pathogens (Barale and Folving, 1996) and other pollutants from

watersheds draining into the coastal environment. Activities, such as the clearing of forests for agricultural or urban development, can negatively impact near-shore fisheries, coral ecosystems, and water quality by altering sediment loads, chemical loads, and watershed hydrology (Basnyat et al., 2000). Some diseases of corals and other benthic organisms have coincided with an increase in anthropogenic inputs (Aronson and Precht, 2001) . The rapid wide-ranging changes in land cover in the tropics have received particular attention due to their potential impact on erosion and increased runoff and flooding (Mas, 1999).

Nonpoint Source Pollution

Land-based runoff containing chemicals, sediments, degrading organic materials or other particulate or dissolved substances (Barale, 1996) may be categorized as either point source pollutants or nonpoint source pollutants. While point source pollutants originate from an identifiable source, nonpoint source pollutants, such as those from most agricultural and forestry activities, are not traceable to any discrete facility or site (Ice et al., 2004).

Historically, U.S. pollution policy has provided a focus on water quality protection efforts for particular industries, facilities, or sources of pollutants. Despite a significant reduction in the amount of pollutants originating from point sources, the nation's water quality has remained at risk from nonpoint sources of pollution (Jacobsen et al., 2006), signaling the need to protect water quality through a watershed-based approach.

Nonpoint sources are often induced by natural processes, such as runoff resulting from rain or snowmelt (Ice et al., 2004). Runoff may be described as the net water supplied to channels following evaporation, interception, surface retention, and infiltration to underlying aquifers (Sharma, et al., 2001). Watershed characteristics impacting surface runoff include slope length, upstream catchment area, volume and velocity of runoff water, nature of eroding sediments, and vegetation type and density (Klemas, 2001). Higher rates of soil erosion are correlated with lower soil shear strength, larger particle diameter, and lower clay content (Fan and Wu, 2001).

Agricultural runoff is a form of nonpoint source pollution that may contain nitrogen, phosphorous, sediment, animal waste, pesticides, and salt (US EPA Office of Wetlands, Oceans, and Watersheds, 1993). The major pollutants in urban runoff include sediments, nutrients, oxygen-demanding substances, road salts, heavy metals, petroleum hydrocarbons, pathogenic bacteria, and viruses (US EPA Office of Water, 1999). Many urban pollutants enter coastal waters as a result of stormwater overflow following heavy rains. Solid pollutant loads in urban stormwater runoff have been reported to be significantly higher than loads found in treated sewage (Tsihrintzis and Hamid, 1997).

A number of studies have identified residential and urban built-up areas as the strongest contributors of increased nitrate levels downstream (Basnyat et al., 2000). A study of land use change in Louisiana's Upper Barataria Basin Estuary from 1972 to 1992 found that increased urbanization was correlated with higher

levels of nitrates downstream (Nelson et al., 2002). A conceptual rainfall runoff model in the Rhine River Basin suggested that while urban growth led to an increase in runoff following storms, intensive reforestation considerably reduced runoff volume (Hundecha and Bardossy, 2004).

Impact of Terrigenous Input on the Caribbean's Benthic Habitat

The tropical islands of the Caribbean are characterized by diverse ecosystems, increasing population density, and a growing number of tourist facilities. Only 10% of plant life in Caribbean watersheds is primary or original growth (Meyer et al., 2000), due in part to development spurred on by tourism. The Caribbean travel industry, which had gross earnings of $17 billion in 1999, is projected to earn $78.4 billion by 2013 (Rivera -Monroy et al., 2004). Caribbean waters receive significant levels of nonpoint source pollution in the form of sewage, solid-waste leachates from landfills, industrial and agricultural runoff, petroleum products and sediment runoff (Villasol et al., 1998).

Historically, disturbance of the Caribbean's benthic habitats has resulted from storms, coral disease, and intensive fishing practices (Rogers and Beets, 2001). More recently, an increase in sedimentation, due in part to the transport of eroded soils to the sea, has become a major form of nonpoint source pollution in many of the Caribbean's bays and coastal waters (Villasol et al., 1998). Heavy precipitation causes soil to erode more rapidly from the islands' steep slopes (Fan and Wu, 2001), particularly from construction sites and other areas with unexposed

soil. In Mayaguez Bay, Puerto Rico, a strong positive correlation (r = 0.92) was found between sediment yield into the bay and fresh water influx (Miller and Cruise, 1995).

Estimates indicate that 29% of Caribbean reef areas (Rivera-Monroy, et al. 2004) are considered high risk due to runoff and the resultant sediment deposition caused by deforestation, poorly regulated construction, nutrient contributions from hotel and shipping wastes, reef mining, bleaching, dredging and over fishing (Szmant, 2001; Hughes, 1994). Evaluations suggest that human population expansion and associated anthropogenic inputs to the coastal oceans are now the most serious causes for seagrass habitat loss (Short and Wyllie-Echeverria, 1996).

In the US Virgin Islands, the rapid, unplanned growth that began in the 1960s has resulted in sediment run-off, dredging and filling of coral ecosystems, discharge of sewage effluents into the bays, and unregulated disposal of solid wastes. Zeiman (1975) found that in Christiansted Harbor, St. Croix, sewage pollution and dredging led to a 66% reduction in seagrass cover over a 17 year period. Dredging and filling of benthic areas near St. Croix's Southshore Industrial Complex operations also led to loss of seagrass beds and coral cover.

Rogers (1982) found that benthic habitat in Brewer's Bay, located in southwest St. Thomas, had been significantly impacted by development. Damage to corals and reduction in seagrass density was attributed to sand extraction, dredging, solid waste dumping, watershed development, and sewage effluents. The study also found that an airport runway expansion in St. Thomas

Bay required filling parts of the bay, which led to burial of seagrass and corals. A study by Zeiman (1975) of Lindbergh Bay, St. Thomas, found that dredging led to decreased depth distribution and loss of seagrass biomass.

Construction and other land use practices have contributed to deteriorating water quality in the USVI's nearshore areas (USVI CZMP Handbook, 1988). Agriculture constitutes 6% of the USVI economy and 1% of the labor force (UNEP, 1998), suggesting that a majority of soil runoff results from other forms of landscape alterations, such as the construction of homes and roads. At present, the majority of land-based pollution in the USVI is nonpoint source pollution. It is composed primarily of nutrients from septic tanks and animal manure and sediments from soil erosion (Sayeli, pers. comm., 2006).

Seagrass

Ecosystem health may be defined as the condition in which an ecosystem is active, maintains organization and autonomy over time, and is resilient to stress (Costanza, 1992). Seagrass are a valuable structural and functional component of coastal ecosystem health. They stabilize sediments, cycle nutrients, provide valuable nursery habitats for fish, and supply detritus to nearshore marine food webs (Dekker et al., 2005; Daby et al., 2002; Kjerfve, 1998). Natural disturbances to seagrass include hurricanes, grazing, sediment bioturbation, disease, and land based runoff caused by heavy or prolonged rainfall (Short and Wyllie-Echeverria, 1996).

Seagrass, light, and nutrients

In the Caribbean's oligotrophic waters, seagrass depend on the sediments in which they grow for nutrients (Ogden and Zeiman, 1977). Seagrass density is highest in shallow, sandy, nearshore areas, where the finer-grained sediments provide a high concentration of nutrients (Daby, 2003). Seagrass develop along a nutrient gradient based on the availability of soluble reactive phosphorous (SRP) and dissolved inorganic nitrogen (DIN), the primary limiting nutrients in coastal ecosystems (McGlathery et al., 1992; Lapointe and Clark, 1992).

In the Caribbean's characteristically coarse carbonate sands, phosphate is the limiting nutrient for seagrass growth (Rivera-Monroy, et al., 2004; Tussenbroek, et al. 1996). Benthic areas with the lowest levels of limiting nutrients support benthic algae, the simple chlorophyll-containing organisms that live attached to a benthic surface. Areas with higher levels of SRP and DIN are inhabited by *Halodule wrightii*, a seagrass that grows in relatively low light and nutrient conditions.

Halodule wrightii and *Syringodium filiforme* occupy an important role as pioneer species in the Caribbean Sea and colonize denuded sediments following perturbation. They are eventually replaced by the climax species *Thalassia Testudinum* in the successional sequence of Caribbean seagrass (Gallegos et al., 1994; Den-Hartog, 1970).

Light controls the depth to which seagrass grows (Longstaff and Dennison, 1999). *Thalassia testudinum* has a high tolerance for low light conditions and can

survive for up to 5 months in light conditions below their minimum light requirement (Lee and Dunton, 1997). Loss of *Thalassia* seagrass indicate a significant reduction in light availability. On average, the light requirement of seagrass is 11% of surface light (Duarte, 1991). Increases in seasonal rainfall can increase turbidity and limit the depth of seagrass distribution, as has been documented in the Florida Indian River Lagoon, in San Francisco Bay, and in Guadalupe Estuary, Texas (Short and Wyllie-Echeverria, 1996).

Various studies have found runoff from land to be the major pathway of nutrient input (Johannson and Lewis, 1992). Natural eutrophication, which occurs when coastal waters receive small amounts of land-based nutrients, can enhance seagrass productivity but higher levels of nutrient input may result in widespread mortality of seagrass (Daby, 2003). The most productive stands of seagrass occur where a balance exists between high water clarity and land-based nutrient enhancement.

Greater levels of nutrient input lead to growth of phytoplankton in the water column and epiphytic algae on seagrass leaves (Vermaat, 1996; Tomasko and LaPointe, 1991), which decrease the amount of light available for photosynthesis by seagrass. Over time, phytoplankton, which is less structurally complex than seagrass, continues to out compete seagrass in low light conditions (Forqueran and Zeiman, 1991). The subsequent break-down of dead phytoplankton by organisms in the water column leads to depletion of oxygen. Agricultural runoff, stormwater discharge, or untreated sewage, which often accelerate the process of

eutrophication, have been associated with excessive epiphyte growth on seagrass (LaPointe and Clark 1992), toxic algal blooms, fish kills, and loss of corals (Carpenter et al., 1998).

In a Florida Keys study, critically low levels of dissolved oxygen and high levels of nutrients were observed following the discharge of untreated sewage into coastal waters (Lapointe and Matzie, 1996). Cumulative wastewater impacts have resulted in long-term loss of *Thalassia Testudinum* and replacement by phytoplankton, macroalgae, and *Halodule wrightii* (Tomasko and Lapointe, 1991). The impact of nutrient loading on the seagrass of Hillsborough Bay, Florida, was indicated by the dramatic recovery of seagrass beds after a reduction in anthropogenic nutrient inputs (Johansson and Lewis, 1992).

The complete loss of seagrass in Pensacola Bay, Florida between 1949 and 1979 was reported to result from urbanization, particularly the discharge of waste water and pulp mill effluents (Zimmerman and Livingston, 1979). In Ninigret Pond, Rhode Island, *Z. marina* distribution declined in direct relationship to housing development and associated nutrient loading (Short and Wyllie-Echeverria, 1996).

Seagrass and Sediment Deposition

The strong coupling between seagrass and sediment dynamics resembles that of terrestrial vegetation and sand dunes on land (Marba et al., 1994a). Seagrass are able to grow on unconsolidated sediments due to an extensive root-rhizome system that maintains the plants securely in place (Fonseca and Kenworthy, 1987). This root system also stabilizes sediments with stems that

extend horizontally under the sea bottom. Horizontal growth rates range from 1 to 10 m/year (Vermaat, 1996). Higher horizontal growth rates can aid seagrass in growing away from an area that has experienced a high rate of sedimentation. Seagrass plants exposed to higher than average sediment accretion rates also grow to greater lengths vertically (Marba, et al. 1994a) than those exposed to lower sedimation accretion rates.

Excessive rates of sedimentation may prove harmful to seagrass Turbidity, induced by turbulent waves, can smother seagrass completely (Short and Wyllie-Echeverria, 1996). Increased rainfall between 1981 and 1987 reduced the cover of *Z. marina* beds in Baja California, Mexico, most likely due to burial by eroded sediments (Ibarra-Obando and Escofet, 1987). A correlation was found between a reduction in seagrass biomass and increased siltation in Southeast Asian seagrass meadows (Terrados et al., 1999).

Nutrients and Corals

Caribbean corals develop to their greatest extent in clear, oligotrophic seas (Dustan et al., 2001; Miller and Cruise, 1995), largely due to the symbiotic relationship between the coral organism and zooxanthellae (Heikoop et al., 1998), which fix inorganic carbon and nitrogen into organic compounds that are translocated to the coral host. During the past several decades, coral reefs in the Greater Caribbean have experienced a decrease in live coral cover and biological diversity, (Gardner et al., 2003) and an overall phase shift away from corals and

towards macroalgae (Lapointe, 1997). In the Florida Keys, stormwater discharge led to elevated concentrations of DIN and chl-a, resulting in eutrophication-induced coral death (Lapointe and Matzie, 1996).

Sediment and Corals

The deposition of sediments on reefs reduces the amount of illumination required by the coral's zooxanthellae for photosynthesis and physically interferes with respiration and feeding (Miller and Cruise, 1995). The weakened reefs become more susceptible to disease or bleaching. Increased deposition of terrigenous sediments to Caribbean coastal waters constitutes one of the primary threats to coral reefs in the Caribbean (Rogers, 1990). The spatial extent of reefs in Mayaguez Bay, Puerto Rico, has declined significantly over the past few decades, hypothesized to be the result of siltation and high turbidity (Miller and Cruise, 1995). Careless clearing of land combined with destruction of coastal mangrove forests have had the most damaging influence on coral reefs in the Caribbean (Kjerfve et al., 1998).

The impact of land use practices on reefs is heightened during periods of heavy precipitation. Up to 83% of annual sediment yield in Puerto Rico (Miller and Cruise, 1995) is observed from September through November, the months with the greatest rainfall. Concentrations of suspended solids at Caribbean reefs were up to 19 times greater during high river discharge than during low discharge (Miller and Cruise, 1995).

Benthic Ecosystem Interactions

In Caribbean waters less than 10 m deep, corals are conspicuously linked with seagrass beds, predominantly *Thalassia testudinum* (Ogden and Zeiman, 1977). Seagrass prevent sediments and excess nutrients from disturbing coral function. They buffer the effects of runoff on corals by attenuating the wave energy (Ward et al., 1984) that resuspends sediments into the water column (Kjerfve et al., 1998), by stabilizing sediments with their extensive root system, and by trapping sediments with their leaves and stems. The broad leaves of *Thalassia* act as a very large filter, removing fine particles from the water and depositing them as sediment (Koch, 1999).

Caribbean seagrass are unique due to the number of consumers that feed on them, their epiphytes, and macroalgae (Ogden and Zeiman, 1977). Seagrass beds are an important food source for herbivorous reef fish (Randall, 1963) and provide nursery and breeding areas for fish and invertebrates (Vermaat, 1996; Short and Wyllie-Echeverria 1996; Thayer, et. al. 1975). Settlement of fish larvae occurs more likely in seagrass than corals, since competition for food and shelter is less (Sweatman and Robertson, 1994). Herbivorous reef fish occupy seagrass beds by night and seek shelter in nearby reefs during the day (Alevizon, 2002).

The excretory and fecal products from fish are a significant source of nitrogen and phosphorous for corals in the nutrient poor waters of the Caribbean.

Enhanced coral growth has been linked to nutrients released on St. Croix reefs by French and white grunts (*H. flavolineatum and H. plumieri*) (Meyer et. al, 1983).

Caribbean seagrass also benefit from the nutrient-rich excretory products released by fish (Alevizon, 2002). In USVI patch reefs, a positive correlation was found between the number of resident fish, such as *Haemulidae*, and nearby seagrass density (Meyer et al., 1983; Gladfelter et al., 1980).

An increase in nutrient or sediment levels or a decrease in light availability impact both corals and seagrass negatively. The linkages that exist between coral and seagrass ecosystems magnify this impact. Severe disturbances in nutrient, light, or sediment levels may limit or prevent seagrass growth and transform corals into dead hard bottom, usually dominated by algae. Because corals rely on seagrass to buffer them from excessive land-based nutrients and sediments, a decline in seagrass impacts corals negatively. The destruction of coastal estuaries, seagrass beds, and mangrove forests that act as sediment traps are among the most damaging influences on coral reefs around the world (Ogden and Gladfelter, 1986).

Coastal Nonpoint Source Pollution Policy

In the United States and U. S. territories, nonpoint source pollution policy in coastal watersheds derives from the Coastal Zone Management Act (CZMA) and the Clean Water Act (CWA). The CZMA was enacted in 1972 to provide federal grants to states to develop and implement programs for the management, use, protection, and development of their respective coastal zones, defined as the

region from the shoreline to three miles offshore (Public Law 101-508). To receive federal grants, states must develop a federally approved coastal management program and be able to provide matching grants.

Growing public awareness and concern for controlling water pollution led to the enactment of the Federal Water Pollution Control Act Amendments of 1972 (Public Law 107-303, 2002) commonly known as the Clean Water Act (CWA). The Clean Water Act, which established guidelines for managing and monitoring water quality in the nation's surface and ground waters, provided federal guidance for managing point sources of pollution to waterways. Neither CZMA nor CWA addressed the management of nonpoint source pollution, an issue that had not yet begun to receive significant attention from managers or the public.

Amendments to the CZMA and CWA

In the early 1980s, a National Oceanic and Atmospheric Administration (NOAA) study was initiated to quantify the status and temporal and spatial trends of concentrations of key contaminants and to assess the nation's coastal and estuarine environments with biological indicators and other water quality parameters (Cantillo et al., 1984). The consequences of policies that had focused on controlling point source pollutants were addressed. It was suggested that national water policies be restructured to address nonpoint sources of pollution (Duda et al., 1988).

In the late 1980s, the USEPA Office of Water recognized that 45% of water quality impairment in estuaries was due to nonpoint sources of pollution (US Congress, 1988). In 1987, to recognize the need for greater federal guidance in state and local nonpoint source efforts, Congress amended the Clean Water Act (CWA) to include the Section 319 Nonpoint Source Management Program.

Section 319 was the first federal legislation to require states to prepare management plans for the regulation of nonpoint source pollution. The USEPA recommended that states develop a watershed plan for each impaired body of water, identifying the maximum amount of point and nonpoint sources of pollution the water body could receive and still maintain water quality standards as specified by the EPA (Santhi et al., 2001). The proposed program guidance, which provided states the discretion to implement management measures (Federal Register, Vol. 58, No. 200), lacked a clear and workable means for ensuring that water was protected from nonpoint source pollution (US Congress, House of Representatives: September 6, 1988). By the late 1980s, nonpoint source pollution management in coastal zones had evolved into a patchwork of education and assistance approaches with little regulation.

In 1990, the CZMA was reauthorized as the Coastal Zone Act Reauthorization Amendment (CZARA). Section 6217 of CZARA was the first federal legislation to significantly address the problem of nonpoint source pollution in coastal areas, mandating that all state and U.S. territories with a federally

approved coastal zone management plan develop programs to control nonpoint source pollution affecting coastal areas (Federal Register, 1991).

Under CZARA Section 6217, NOAA and USEPA were required to jointly provide technical guidance to coastal states and local governments in identifying techniques to monitor changes in coastal habitats, as well as to determine compliance with coastal nonpoint source regulations. States were asked to identify the pollutants that were to be regulated, quantify estimates of the effects of pollution reduction, and develop any necessary monitoring techniques that would help determine the effectiveness of pollution reduction measures (US Code, 1994). The formulation and implementation of a successful coastal nonpoint pollution program required close coordination between state-level agencies that until then had worked separately to address water quality and coastal zone management issues.

Because most coastal nonpoint source pollution results from land use activities that fall under state or local jurisdiction, CZARA cannot mandate the regulation of land use activities. However, CZARA does indicate that if a state fails to submit an acceptable coastal nonpoint pollution management program to the Department of Commerce within 30 months of issuance of the federal guidance, portions of their CZMA grants and Water Pollution Control Act grants will be withheld until such a program is submitted (Public Law 101-508, 1990). As a prerequisite to CZM program approval, states are also required, based on conservation needs or ecological and esthetic value, to identify coastal areas that

require specific attention. Each state's coastal zone management program must implement procedures that preserve or restore these Areas of Particular Concern (APCs) (Devine, 2001).

Executive Order 13089

Coastal areas in the US's Caribbean territories contain coral ecosystems, which are both high in biodiversity and highly susceptible to land-based sources of pollution. Over the past decade, the protection of these areas has received increasing attention from U.S. policymakers. In 1998, Executive Order 13089 established the Interagency U.S. Coral Reef Task Force, co-chaired by the Secretary of the Interior and the Secretary of Commerce through the Administrator of NOAA. The U.S. Coral Reef Task Force is charged with developing and implementing a comprehensive program of research and mapping to inventory, monitor, and identify the major causes and consequences of degradation to coral reef ecosystems in U.S. states and territories (Rogers and Miller, 2001).

Coastal Policy in the United States Virgin Islands

Management of nonpoint source pollution in the USVI is set forth in the Virgin Islands Coastal Zone Management Act, Section 903 (b), which indicates that a primary goal in the area of environmental protection is to maintain or increase coastal water quality through control of erosion, sedimentation, runoff, siltation, and sewage discharge. In May, 2002, the USVI's Coastal Pollution Control Program

won approval for federal funding from NOAA and EPA. Nonpoint source pollution management measures have focused on reducing soil erosion from construction sites.

Coastal policy in the USVI is based on a two-tier system. The outer boundary of Tier 1 is located three miles out from the USVI coastline. The landward boundary, which is based upon convenience or political factors, may be at the shoreline or in some terrestrial portion of a watershed. Development in Tier 1 is managed by the USVI Coastal Zone Management Program (CZMP) and subject to stringent erosion control. As stated in the NOAA-EPA Final Decision Document for the Virgin Islands Coastal Nonpoint Program (2002), all development in Tier 1 must be consistent with the goals, policies, and permit conditions of the VICZMA and Environmental Protection Law. Development includes all activities that alter terrestrial areas, shorelines, and submerged lands. Tier 2, which includes all terrestrial lands not within Tier 1, is regulated by the USVI Department of Environmental Protection (DEP). Though construction in Tier 2 is subject to erosion control, enforcement is minimal and fines are small.

Some of the USVI's Marine Protected Areas (MPAs) have been ineffective in protecting fish populations due to uncontrollable factors, such as hurricanes, and due to limited enforcement of regulations that restrict fishing practices in the MPAs. In situ assessment of the benthic habitats of two marine protected areas in the USVI suggest that these areas may also be impacted negatively due to

current land use policy in the USVI (Rogers and Beets, 2001), which strictly enforces land use regulations near the coastline, but provides little regulation of land use in the upper regions of watersheds.

Remote Sensing for Coastal Management and Policy

The present scheme for managing nonpoint source pollution has not been successful for a number of reasons. These include complexities in the relationship between the EPA 319 and the NOAA 6217 programs, lack of adequate funding, and lack of political will to support voluntary measures with enforceable ones (Cicin-Sain and Knecht, 2000). Responding to the mounting pressures on the nation's coastal and marine environment, the Oceans Act of 2000 established a 16 member commission to review federal ocean policies and to make recommendations to Congress.

One of the primary aims of this act is to protect the coastal and marine environments from pollution. The final report of the Commission requires an assessment of existing and planned facilities associated with ocean and coastal activities, including the use of remote sensing through satellites and other appropriate platform technologies (106th Congress, 2nd session, S. 2327).

Remote sensing is the science of obtaining information about an object or area through the analysis of data acquired from an airborne or satellite sensor (Lillesand and Keifer, 2000). Satellite images provide data over large areas in a digital format, facilitating image processing and exchange of data with Geographic Information Systems (GIS) (Ferguson and Korfmacher, 1997). The combination of

remote sensing and GIS has been shown to provide quantitative estimates of coastal and estuarine habitat conditions (Muttitaton and Tripathi, 2004; Klemas, 2001) that may be used to arrive at cost-effective plans for watershed-level conservation and management (Sharma et al., 2001).

Satellite sensors are used to assess sea surface color, temperature, roughness and elevation, and to assist in studies of coastal land use, land cover, hydrology, ecology, water quality, sediment transport, coastal runoff, and circulation, the evolution of plankton communities, carbon cycling, and climatology (Barale and Folving, 1996).

A 1995 National Research Council study on science, policy, and the coast noted that with increasingly sensitive remote sensing equipment, it will be possible to integrate monitoring with research on a scale that takes into account the spatial and temporal aspects of nearshore processes. Coastal researchers and managers are increasingly using remote sensing technology to research, monitor and protect coastal areas (Cuevas-Jimenez and Ardisson, 2002; Klemas, 2001) because remote sensing techniques provide synoptic, spatially referenced information over large spatial scales (Barale and Folving, 1996).

These techniques may be used to manage and monitor coastal water quality (Klemas, 2001). When used in conjunction with data on circulation patterns and on sediment transport from watersheds, remote sensing may be used to demonstrate a direct coupling between land and ocean processes (Miller and Cruise, 1995).

Mapping Terrestrial Habitat Cover and Change with Landsat Sensors

Landsat satellites have been in orbit since 1972, providing over 30 years of temporal coverage that has been applied to studies of coastal land cover change and processes (Miller and Cruise, 1995). The reflectance of healthy green vegetation dramatically increases in the near infrared (NIR) range due to the internal structure of the plant leaf, which reflects 40% to 50% of incident energy from 0.7 to 1.3 µm (Lillesand and Keifer, 2000). Beyond the NIR range, drops in the energy reflected by vegetation occur at 1.4, 1.9 and 2.7 µm because water in the leaf absorbs energy at those wavelengths. Reflectance peaks beyond NIR occur at 1.6 and 2.2 µm (Lillesand and Keifer, 2000).

Landsat satellite sensors are able to differentiate the unique spectral signature of ground features based on the radiometric response of these features in the visible (0.4 to 0.7 µm), near infrared (0.7 to 1.3 µm), and middle infrared (1.3 to 3 µm) portions of the electromagnetic spectrum (Lillesand and Keifer, 2000). The Landsat MSS sensor, carried onboard the first series of Landsat satellites, has a spatial resolution of 80 m^2 and four spectral bands, two each in the visible and NIR portions of the electromagnetic spectrum. This sensor became secondary with the 1984 launch of Landsat-4. Landsat-4 carries the Landsat Thematic Mapper (TM), which has a spatial resolution of 30 m^2 in the visible (1, 2, and 3) and infrared (4, 5, and 7) bands.

The Landsat-7 satellite's Enhanced Thematic Mapper Plus (ETM+) 8-band multispectral scanner, which has been in orbit since 1999, has lower spatial noise

(Masek et al., 2001) than TM, a panchromatic band (band 8) with 15 m^2 resolution, and a thermal infrared band (band 6) with 60 m^2 resolution (Klemas, 2001). The geographic coverage of each TM and ETM+ scene is an area 170 x 183 km in size (Lillesand and Keifer, 2000).

Because the products of Landsat-5 and 7 are very similar following radiometric normalization, data from the two sensors may be used together to measure and monitor changes in landscape phenomena over time (Vogelmann et al., 2001). The cost of monitoring large coastal areas has greatly decreased, making Landsat imagery attractive (Nelson et al., 2002; Klemas, 2001) to map and analyze changes that can be used as inputs to land management and policy decisions (Yuan et al., 2005). A multitemporal study of remotely sensed data is a widely used method for assessing changes in biophysical systems over time (Miller and Yool, 2002). Important landscape level environmental indicators can be extracted from Landsat data, including change in the size, configuration, and health of coastal habitats and vegetation cover (Klemas, 2001).

Automated detection of land cover change in satellite imagery is complicated by many temporal factors, including differences in spectral bands and spatial resolution, spatial misregistration, variation in radiometric response of the sensors, differences in distribution of cloud and cloud shadow, variations in solar irradiance and solar angles, variations in atmospheric scattering and absorption, and differences in phenology (Yuan and Elvidge, 1998). Radiometric, geometric,

and atmospheric corrections and coregistration are required to normalize data from various dates used in change detection studies (Miller and Yool, 2002).

Postclassification comparison methods use classifications of images acquired on different dates to produce “from - to” change maps (Jensen, 1996). A postclassification analysis of TM and ETM+ scenes of Twin Cities, Minnesota demonstrated the effectiveness of utilizing Landsat to produce accurate landscape change maps (Yuan et al., 2005). Analysis of change patterns in these maps assisted in addressing questions related to policy, transportation, and population growth. A multitemporal postclassification study of land use in the coastal zone of northwest Mexico utilized Landsat MSS and Landsat TM images from 1973 to 1997. A change detection matrix and Kappa coefficient showed an accuracy of 61% to 68%, enough to determine the general trends of change in the system, which indicated a loss of natural cover and an increase in vegetation fragmentation (Berlanga-Robles and Ruiz-Luna, 2002).

To detect land cover change in Louisiana, Landsat MSS images from 1972, 1985, and 1992 were each coregistered with a digital elevation model (DEM) and transformed to Universal Transverse Mercator (UTM) coordinates. The postclassification analysis indicated that from 1972 to 1992, urban land cover in the state had increased from 25% to 50%, as a result of conversion of agricultural land to densely urban land (Nelson, et al., 2002). A three-class unsupervised classification and change detection using Landsat bands 4 and 7 to determine the

impact of a forest fire was found to be more accurate than maps created from photointerpretation (Miller and Yool, 2002).

Image differencing and write function memory insertion can be used in a preclassification process for use in change detection. These techniques locate changes, but do not provide information on the nature of change (Yuan et al., 2005). Write-function memory insertion involves the use of color additive theory for visual change detection. Data from two or three dates are placed chronologically in the red, green, and blue computer display write functions. The resulting color combinations are used to visually interpret the direction and magnitude of change.

Terrestrial biomass is a commonly used and accepted parameter for evaluating an ecosystem condition (Klemas, 2001). Write function memory insertion may be used to ascertain the degree of change in terrestrial biomass using the Normalized Difference Vegetation Index (NDVI). The NDVI, developed to determine vegetation health and vigor, is found to be highly correlated with crown closure, leaf area index, and other vegetation parameters (Hayes and Sader, 2001). The NDVI incorporates information from the red and NIR bands (Lillesand and Keifer, 2000) and can be defined as:

$$NDVI = (Red - NIR)/ (Red + NIR) \quad (2.1)$$

where R = red wavelength band

NIR = near infrared wavelength band

The NDVI is applied widely to detect change in biomass. Using Landsat TM data of Guatemalan forests to determine a change in forest cover, the write-function technique was found to have both high accuracy and ease of interpretation (Hayes and Sader, 2001). NDVI composites of Landsat scenes were used to assess the magnitude and direction of change in biomass of Ban Don Bay, Thailand (Muttitaton and Tripathi, 2004).

Image differencing is a widely used change detection technique (Hayes and Sader, 2001) that has been shown to provide high quality results when used with Landsat data (Vogelmann, et al., 2001). The brightness value of pixels from one date are subtracted from the brightness values of the same pixels for another date (Klemas, 2001). Standard deviations from the mean brightness value are used to differentiate change from non-change pixels. The subtracted channels may have infiormation from one band or multiple bands, as occurs when using NDVI data.

In a multitemporal study of the Nile Delta and adjacent coastal regions, NDVI features derived from 10 Landsat TM images dating from 1984 to 1993 were used to assess the status of agricultural lands in the Nile Delta, and the adjacent Western Desert and coastal region. The change maps, which had an accuracy of 96%, indicated that between 1986 and 1993, the amount of cultivated land in these regions increased by 43.3% (Lenney et al., 1996).

Mapping Benthic Habitat Cover and Change with Landsat

Large-scale quantitative assessments of coral reefs and seagrass are needed to establish quickly and accurately the status of these ecosystems, to

monitor general trends, and to discriminate between anthropogenic and natural influences (Dekker et al., 2005; Bouvet et al., 2003; Hughes, 1994). Space-based observations offer a cost-effective way to map reefs and seagrass in remote locations and to observe the responses of coral reef ecosystems to disturbances on a geographic scale that until recently was not possible (Dustan et al., 2001; Mumby et al., 1997).

Spectral information of benthic features at the water's surface is the result of electromagnetic radiation, or radiant flux, that enters the ocean, propagates to the bottom, is modified by absorption and scattering properties of the water column and benthic substrate, and propagates back toward the surface. Wavelengths beyond 680 nm are significantly attenuated by pure water (Dekker et al., 2005). Light is further restricted from being reflected up through coastal waters due to absorption and scattering by phytoplankton suspended organic and inorganic matter, and dissolved organic substances (Dekker et al, 2005). Parameters that influence the amount of radiant flux leaving the water include depth, substrate type, water clarity, water column features, and atmospheric conditions.

The radiance received at a satellite sensor from benthic features consists of the water-leaving radiance plus the radiance due to atmospheric effects. Benthic habitats can be mapped from space when sufficient radiant flux can be detected by the satellite sensor (Green, et al., 2000). In the coastal region,

remote sensing technology has most commonly been used for producing habitat maps and monitoring the impacts of development on coastal habitats (Green, et al., 2000).

The ideal conditions for mapping benthic habitats in nearshore environments are low depth, high transparency, and stability of the water column (Cuevas-Jimenez and Ardisson, 2002). Coasts in the relatively undeveloped areas of the Caribbean are characterized by clear waters and a substrate of carbonate sand bottom, both of which increase the amount of radiant flux leaving the water (Cuevas-Jimenez and Ardisson, 2002).

Mapping Coral Reefs with TM and ETM+

Landsat TM and ETM+ provide an operational remote sensing platform for locating reef ecosystems and mapping their general zonation patterns and dominant benthic communities (Dustan et al., 2001), including coral, seagrass, algae, and mangroves (Mumby et al., 1999). Landsat-based maps of Red Sea reefs were created successfully despite variable coral cover and depth, significant currents, and lack of in situ radiance measurements (Purkis et al., 2002).

Landsat-7 ETM+ images of New Caledonia in the South Pacific were used to distinguish benthic habitats in areas less than 5 m in depth and to discriminate coral reef geomorphology and seagrass bed structure (Bouvet et al., 2003). Comparing satellite data for mapping the geomorphologic detail of

reef habitat, Landsat-5 TM had an accuracy of 73%, SPOT XS had 67% accuracy, and SPOT Pan and Landsat MSS had accuracies less than 60% (Mumby et al., 1997).

Water column attenuation at wavelengths longer than 600 nm and obscuration by atmospheric scattering at wavelengths less than 500 nm, prevents the intrinsic spectral signature of various reef components from being discerned (Lubin et al., 2001). Furthermore, because Landsat radiometric sensors were designed to distinguish cloud-free terrestrial areas, they are not ideal for discrimination of the spectral features that distinguish coral reef species from one another (Lubin et al., 2001). To successfully map nine or more reef habitat categories in relatively shallow, clear waters, the high resolution airborne sensor CASI (Compact Airborne Spectrographic Imagery) achieved an 81% accuracy, whereas aerial photography was 67% accurate, and Landsat and SPOT sensors were 37% accurate (Mumby et al., 1997).

Landsat TM images contain information that can be used to distinguish areas of no change from areas of change in coral habitats without the necessity of accurate target identification (Dustan et al., 2001). A change analysis of Florida's Carysfort Reef, the first study to utilize Landsat imagery to detect change in a coral reef ecosystem, indicated that living coral cover had been reduced from 40% in 1975 to 5% in 2000 (Dustan et al., 2001). These findings were supported by in situ data, which indicated that the Carysfort Reef underwent a 90% decrease in coral

cover from 1982 to 2000. The study demonstrated that Landsat images could be used to identify the extent, but not the type of change.

A second time series analysis of Carysfort Reef used four Landsat-5 images and one Landsat-7 image spanning 1984-2002, in conjunction with high resolution IKONOS imagery and aerial photography, to demonstrate that the loss in Carysfort Reef's coral cover could be detected to a high degree of accuracy (Palandro et al., 2003). Landsat 7 ETM+ images of Florida and Hawaii reef tracts indicated that the assessment of rates of change in reefs was most effective when general identification categories, such as sand, reef background (pavement, rubble, and heavily grazed dead coral), and reef foreground (living corals and macroalgae) were used (Andrefouet et al., 2001).

Mapping Seagrass with TM and ETM+

A change analysis of seagrass using Landsat scenes from 1988 to 2002 indicated a decrease in seagrass cover with an accuracy of 76% or greater (Dekker et al., 2005). Landsat ETM+ data were found feasible for monitoring changes in shallow seagrass beds over large areas of New Caledonia (Bouvet et al., 2003). Since little ground truth data existed for this region, an unsupervised classification and radiometric criteria were used to aggregate Landsat data into habitat classes. Due to insufficient spectral resolution, Landsat cannot be used to distinguish seagrass from epiphytes (Bouvet et al., 2003), algae or other benthic vegetation.

Thematic Map Accuracy

Most commonly, the accuracy of a thematic map is assessed with an error matrix, which compares classes in the thematic map with the classification of randomly selected samples of reference data (Powell, et al., 2004). In addition to assessing the degree of agreement between the reference data and map product, the error matrix also incorporates error resulting from incorrect georegistration of multiple images or change in land cover between the time the reference data were collected and when the satellite data were recorded (Powell, et al., 2004).

The Kappa statistic is a conservative measure of the differences between the actual agreement of reference data and an automated classifier and the chance agreement of reference data and an automated classifier (Miller and Yool, 2002). The Kappa statistic is used in conjunction with the error matrix because it incorporates all elements of the confusion matrix (Mas, 1999).

The minimum level of interpretation accuracy in the identification of land use and land cover categories from remote sensor data should be at least 85% (Rosenfield et al., 1982). Landsat studies using minimum mapping units of about 1 ha have yielded 85% or better classification accuracies (Klemas, 2001), when class assignment is based on aerial photographs and other ancillary data of higher spatial resolution than the Landsat data (Nelson et al., 2002). Landsat TM images and aerial photographs were used to assess the accuracy of MSS images in a land

cover change analysis of Louisiana (Nelson et al., 2002). Aerial photographs are used routinely to assess the accuracy of coral classification in Landsat scenes (Bouvet et al., 2003).

The accuracy of a change map is equivalent to the product of the classification accuracies of the thematic maps used to create the change map (Klemas, 2001; Mas, 1999). A more rigorous approach for determining the accuracy of change maps is to randomly sample areas classified as change and no change and use reference data to determine whether the change was correctly classified (Yuan et al., 2005).

Summary

Effective management of the impact of point and nonpoint source pollutants on coastal benthic habitats requires technologies that effectively quantify the impact of point and nonpoint source pollutants at the watershed level. Though designed for mapping terrestrial areas, Landsat TM and ETM have shown to be effective in the mapping and change detection of general coral reef classes. Because Landsat TM was launched in 1984, the TM and ETM+ sensors provide over 20 years of data for the analysis of terrestrial and benthic habitat change. These change maps may be used in conjunction with higher resolution aerial or satellite data to better interpret the impact of land use change on the benthic habitats. Not only may this remote sensing methodology prove useful to coastal

managers for monitoring watershed-level impacts of land use on benthic habitats, but this methodology may provide information to develop policies that limit activities that are harmful to coastal ecosystems.

CHAPTER 2

STUDY AREA

The United States Virgin Islands (USVI) is a United States territory located in the eastern extreme of the Greater Antilles at approximately 18°N latitude and 64° W longitude. The USVI is comprised of three main islands, St. Croix, St. Thomas and St. John (Figure 2.1) and 54 small islands and cays (Boulon, 1986). St. Thomas and St. John are volcanic in origin and situated 3 mi (4.8 km) apart on a contiguous shelf shared with Puerto Rico and the British Virgin Islands. St. Croix, located 40 mi (64 km) to the south of St. John, consists of volcanic rock and limestone formed from former coral reefs.

St. Thomas, St. John, and St. Croix are 28 mi^2 (73 km^2), 20 mi^2 (52 km^2) and 84 mi^2 (218 km^2) in size, respectively. Over 70% of St. Thomas has hillsides with slopes over 20% (USVI CZMP, 1988) (Figure 2.2). The highest point of elevation in St. Thomas is 1556 ft (474 m) above sea level. The island has very little flat land suitable for extensive construction. St. John has a steeper terrain,

with 80% of the island having slopes of 20% to 30% (Boulon, 1986). By contrast, over 50% of St. Croix has land with a slope under 10% (USVI CZMP, 1988).

The USVI have a subtropical climate. On average, the year-round temperature is 80°F (26.7 ^{o}C). The islands are relatively low in elevation and have little rainfall when compared to larger Caribbean Islands to the west. Average rainfall is approximately 38.36 in on St. Thomas and 43.31 in on St. John (Figure 2.3). St. Croix is drier than much of the Greater Antilles due to its low elevation and the significant amount of land clearing that has occurred throughout the island. Average rainfall is 40 in (102 cm) in the west and 30 in (76 cm) in the east. The dry season is from January to May and the rainy season is September to November (CIA World Factbook, 2006).

The warm temperatures and drying easterly trade winds contribute to rapid evaporation of moisture. During periods of intensive rainfall, up to 6 in (15 cm) of rain may fall in a 24 hour period, leading to flooding, increased sedimentary runoff, and a lowering of coastal water quality. The USVI has no natural fresh water bodies. However, following heavy rains, many active freshwater streams, called guts, form on the island. Animal and plant populations change seasonally with factors related to the amount of rainfall. Hurricanes pose a threat during the rainy season. In 1989, Hurricane Hugo caused significant damage to St. Croix's terrestrial and coral reef environments.

Terrestrial Vegetation

The vegetation of the Virgin Islands may be characterized as subtropical due to the unique conditions created by climate, soil, wind, elevation and aspect. The original virgin forests were lush and diverse, consisting of more than 500 tree species. Cutting and clearing for sugarcane production and the introduction of nonindigenous plants has dramatically altered the landscape.

Numerous types of vegetation characterize USVI's terrestrial landscape. Its moist evergreen forests may be divided into three categories. Basin moist forests develop in coastal watershed basins where runoff from large upland watersheds collects. Gallery moist forests are forests that grow along the margins of drainage guts, carrying runoff from large upland watersheds.

Upland moist forests occur at elevations high enough to experience rainfall totals above 120 cm (47 in) per year. A dry period of 2 to 4 months is common, during which time some foliage reduction can be observed. Recent hurricane disturbance has temporarily increased vine, herb and small shrub growth in moist forests. Due to flash floods, gallery moist forests are characterized by younger trees and less stratification than the upland type. Basin forest types are well represented on all the Virgin Islands. The semi-deciduous forest is the dominant forest cover throughout the USVI.

Over 50% of St. John Island is semi-deciduous forest. A large percentage of the remaining undeveloped land on St. Thomas and a majority of the small quantity of remaining forest on the northwestern hill slopes of St. Croix are also semi-

deciduous. In most cases these forests may be found on north-facing hillsides of all the main islands, upper southern facing elevations below 250 m (820 ft), and in basins without large watersheds. The gallery semi-deciduous forest is limited to the smaller riparian corridors, such as ravines, guts and intermittent streams found within dry forest watersheds. The taller dry forest species find their maximum height in these locations. The semi-evergreen forest is found above 900 m (2951 ft) in elevation on northwest facing slopes below moist forest levels. This cover type is found to a greater degree on the west end of St. Thomas. Drought deciduous forests are found on all three islands, but may be difficult to delineate other than during the annual dry season from January to May. Structure and species composition is much the same as semi-deciduous forests, but this type is characterized by greater than 75% deciduous species. The formation is found mainly at lower elevations below 250 m (820 ft), on south and southwest facing slopes and in south and west aspects along the north shores.

Woodlands are characterized by an open tree canopy where the crowns are generally separated and cover is approximately 25% to 60%. They are more common to St. Croix than to St. Thomas and St. John. Height of the canopy varies greatly and may be from 8 to 20 m (26 to 99 ft). The gallery semi-deciduous woodland is found only in St. Croix. It is primarily found in guts and ravines at low elevation and appears as strips of open canopy woodland where the guts pass through cleared areas. The semi deciduous woodland is found primarily on abandoned agricultural land on St. Croix, in central and eastern St. Thomas, and

as a coastal woodland habitat along the shore seaward of salt ponds and flats. These areas are typically influenced by hurricane winds, which reduce canopy cover temporarily. The drought deciduous woodland occurs most commonly on St. Croix's drier eastern hills and in patchy areas across the island.

Shrub lands may be found in dry locations at low elevations on all islands and offshore cays. The thin soils, strong winds and low moisture limit vegetation height to a maximum of 5 m (16 ft). Gallery scrublands are found most commonly in small guts and ravines, where additional catchments allow soil moisture to prevail. Thickets and shrubs are quite common on all three islands and cays. This cover is characterized by thorny shrub communities, which generally form a very dense, closed cover community. The height of the vegetation averages approximately 4 m (13 ft) with occasional emergent trees. The mixed dry scrubland cover is common to the drier parts of all three islands, east and south shores, and low elevation locations.

Cacti and agaves are common though scattered, while vegetation height can range from 1 to 10 m (3.3 to 33 ft). The shorter forms are common to very exposed locations, such as the east sides of headlands on the south shores. Coastal hedge communities are shaped by wind shear and salt spray. These dense patch communities vary from 1 to 3 m (3.3 to 9.8 ft) in height. They are generally found on east, southeast or northeast coastal areas with exposure to prevailing winds. The formation may occur on the berms of beaches, seaward of salt ponds and flats, or above rocky coasts or pavement.

Herbaceous communities occur in areas of very low rainfall, along the coast, or in areas that have been subject to disturbance by human activities associated with agriculture and grazing of livestock. Many herbaceous communities may have a small percentage of shrub species and broad-leaved evergreen and semi-deciduous trees as a component. These ecosystems, dominated by grasses, are maintained by grazing and fire and are generally located on nearly flat or moderate slopes. Pastures are primarily grasslands with a very low incidence of shrub and tree species occurring as part of the community.

Mangrove forests occur at the shoreward extreme of tidally flooded areas. On St. Croix, mangroves exist in Salt River Canyon on the north coast and at Great Pond on the south coast. Very few places in St. Thomas or St. John have mangroves due to the steep terrain and lack of rivers or streams. On St. John, mangroves are located in Leinster Bay and Newfound Bay on the north coast. Other areas in St. John with mangrove stands are Mary's Creek, Hurricane Hole, and Coral Bay (Tetratech, 1991). Mixed swamp areas are semi-permanent and are tidally flooded communities composed of a mixture of mangrove and wetland tree and shrub species.

Salt ponds are semi-permanently flooded coastal ponds. These coastal features are formed as a result of coral growth across an indented bay or shoreline feature. Storm deposited materials eventually form a berm that separates the pond from the sea. As these features age and mature, salt flats are formed landward of

the berm, creating very unusual habitat types that also capture sediment from upland areas, thereby protecting fragile reef and seagrass beds from the impacts of sedimentation (Boulon, 1986).

Benthic Environment

The low level of dissolved nutrients in the USVI's oligotrophic waters can be attributed to year-round steady temperatures, which result in a permanent thermocline that restricts nutrients to the lower portion of the water column. Additional factors contributing to low nutrient levels include high rates of organic decomposition, which prevents the accumulation of nutrients in the upper portions of the water column.

The following definitions of the benthic categories are from the NOAA NOS Characterization of Benthic Habitats of Puerto Rico and the USVI (2000). The best formed *coral reefs* occur in shallow waters that receive minimal terrigenous sediments (Figure 2.4). The USVI, with almost no river systems, have coastal areas ideal for the development of corals. Most of the coral reefs surrounding the Virgin Islands are shallow fringing reefs that parallel the islands' coastline. More than 40 coral species grow on true reefs, as well as on hard pavement, submerged boulders, and rock ridges (Devine, 2001). Bank reefs and spur and groove formations occur in deeper waters.

Caribbean reefs have a typical zonation pattern. The reef crest is the flattened, shallow and often emergent portion of the reef, usually colonized by wave resistant invertebrates and emergent coral. The back reef is landward of the

barrier or fringing reef crest. Behind the back reef, shallow lagoons may form, creating a protected area. The floor of the lagoon is usually colonized by various coral species, as well as patches of seagrass, algae or sand. Dead, unstable coral rubble often occurs landward of well-developed reef formations and may be categorized as a back reef feature. It is often colonized with filamentous and other forms of macroalgae.

The upper seaward portion of the reef, known as the fore reef, is comprised predominantly of highly branching *Acripora palmata*, which provides a daytime refuge for many species of fish and is very well adapted to strong wave action (Bak, 1985). The lower fore reef is colonized with a high percentage of various species of live coral (Devine, 2001). The following description of fringing coral reefs is based on surveys taken by the USVI Department of Planning and Natural Resources (DPNR). St. Thomas has a submerged barrier reef that extends from Inner Bass Island to the western end of St. Thomas. St. John has extensive, well-developed reefs towards the eastern edge of the island, along East End and from Johnson Bay to Ram Head. In 1989, Hurricane Hugo caused significant damage to St. Croix's reefs. Prior to Hugo's impact, bank barrier coral reefs were well developed and formed long undeveloped ridges, sheltering the coasts from heavy wave action and forming lagoons that were covered with extensive seagrass beds (UNESCO, 1982).

Carbonate pavement is characterized by a flat, hard, carbonate substrate that supports the growth of seagrass and has a low percentage of live coral cover. This habitat type may be found in shallow bays, such as Cinnamon Bay, St. John, between upper and lower fore reefs and at the lower fore reef where the reef slope is gradual and has low live coral cover. Raised portions may be colonized by algal turf.

A patch reef is commonly a small reef isolated from shore and other reef systems by sand, seagrass or hard bottom. Patch reefs can range in size from less than 3 m (9.8 ft) in diameter to tens of meters in diameter. When located within seagrass beds or algal plains, they are surrounded by a halo of sand, formed as a result of night time foraging by the fish that live within the reef.

Gorgonian coral, a horny coral that forms into upright branching colonies, can be found in areas of high current velocities, such as between small islands. Because the gorgonian structure prevents light from reaching the substrate, these areas have little to no algal growth. Significant portions of the U.S. Virgin Islands coastlines at depths from 0 to 3 m are fringed by exposed bedrock. The bedrock, which may be covered by macroalgae, hard coral, or gorgonians, can be found along most rocky shores, especially at land formations ending in points.

Sand bottom is comprised of very fine to coarse carbonate sand with few living sessile organisms. These areas are typically found exposed to currents or wave energy. Examples of this habitat type are found off north shore beaches where winter swells prevent establishment of seagrass beds. Seagrass, which

grow in carbonate sands, occur throughout the USVI in association with coral reefs and mangrove areas (Alvezion, 2002). They are well-developed along shallow sandy shelves and are often found in protected waters between rocky headlands (UNESCO, 1982). When found near coral reefs *Thalassia* tend to grow in carbonate sediments on the floor of lagoons and at the base of the outer reef crest. *Thalassia testudinum* is the area's predominant as well as climax species of seagrass. Other seagrass species include *syringodium, Halodule* and *Halophila.*

Prior to Hurricane Hugo, extensive seagrass could be found in St. Croix's lagoons and along the deeper sandy portions of shelf areas (UNESCO, 1982). Most seagrass demonstrate seasonal variation, with greater density during the wet season and minimal growth in the dry winter months (Daby, 2003). Seagrass in the USVI have historically been heavily grazed by green turtles. However, the numbers of these marine animals have declined drastically. In many areas, hurricane waves have caused extensive blowouts of *Thalassia* beds (Marba et al., 1994a). Other disturbances have resulted from intensive recreational use, land use practices, dredge and fill operations, and marina construction (Devine, 2001).

In 1982, seagrass could be found in St. Thomas at Saba Island, Perseverence Bay, Brewers Bay, Red Point, and Lindberg Bay. In 1988, seagrass in St. John could be found growing in Hawksnest Bay, Trunk Bay, Cinammon Bay, Maho Bay, and Francis Bay. Along St. Croix, seagrass occurred in Manning Bay in 1985, along the South Shore Industrial Complex in 1981 and 1987, at Tague Bay Lagoon in 1987 and at Christiansted Harbor in 1975.

Algal ridges are characterized by coralline algae growing on dead *Millepora* and *A. palmata* colonies. The algae can be found growing between the mean low tide levels to about 17 cm (6.7 in) above that level. In spring, the maximum range for algal growth in St. Croix is approximately 35 cm (14 in). Algal plains are one of the predominant habitat types covering insular shelf regions at depths exceeding 20 m (66 ft) (Beets et al., 1986). Numerous species of red, green and brown macro algae make up this community, which can have a canopy height from a few centimeters to over a meter. Associated with these algal communities is a substrate composed of carbonate algal nodules, which range in size from less than 1 cm to greater than 10 cm (0.39 to 3.9 in) (Beets et al., 1986).

Major USVI fisheries are, from largest to smallest, reef fish, lobster, pelagic fish, and conch (UNESCO, 1982). By the late 1980s (USVI CZMP, 1988) many fish were being harvested at maximum sustainable yield. Currently, demand outstrips supply of these fisheries.

Development Trends

Since 1960 the USVI population has increased from 32,000 to approximately 110,000 (CIA World Factbook, 2006). Over the past 20 years, St. Thomas has experienced the greatest growth in population and urban development. Because coastal topography hinders the construction of homes, most residential development occurs inland. The construction of roads and homes has altered the existing land cover and led to an increase in sediment runoff, the effects of which are very noticeable in the bays and harbors at the base of

developed watersheds. After heavy rains, large plumes of sediments can be seen extending from drainage guts into island bays and harbors. The lack of natural freshwater resources in the face of rising population size is a concern.

Tourism is the primary economic activity, accounting for 80% of Gross Domestic Product (GDP) and employment (CIA Factbook, 2006). In 1960, tourist accommodations in the US Virgin Islands consisted of 1400 rooms. By 1975, the number had grown to 4000 with 66% on beachfront property (USVI CZMP, 1988). Currently, due largely to St. Thomas Harbor, one of the best natural deepwater harbors in the Caribbean, the islands normally host 2 million visitors a year (CIA World Factbook, 2006).

The manufacturing sector consists of petroleum refining, textiles, electronics, pharmaceuticals, and watch assembly. One of the world's largest petroleum refineries is located on the southern coast of Saint Croix. Due to its gradual slopes, about 85% of the USVI land suitable for farming is on St. Croix. However, very few farming operations exist (USVI CZMP, 1988) and most food is imported to the islands.

Protected Regions and Areas of Particular Concern

The 6,623 ha (66 km^2) of protected areas in the US Virgin Islands is equivalent to 19% of the Island's landmass. Protected terrestrial areas include the Virgin Islands National Park (5,308 ha or 53 km^2) and the Virgin Islands Biosphere Reserve, both located on St. John (Figure 2.5). The Virgin Islands National Park was established in 1976 and encompasses 54% of St. John (Boulon 1986).

Most of St. John is surrounded by a fringing reef that extends a quarter mile from shore and may be up to 200 m in width in parts. Marine reserves aid in the preservation or recovery of fish and the reefs and seagrass beds upon which they depend. However, monitoring reef fisheries in the Virgin Islands National Park indicates the abundance and sizes of several important fisheries species are similar inside and outside park boundaries, indicating that the park is not providing a refuge (USGS, 2006) for these fisheries.

Protected marine regions on St. Croix include the East End Marine Park in St. Croix and Buck Island Reef National Monument. Areas of Particular Concern (APC) on St. Croix include Christiansted Waterfront Sandy Point, Southgate Pond at Chenay Bay, Frederiksted Waterfront, Sugar Bay at Salt River, and Point East End-Point Udall Southshore Industrial Area complex.

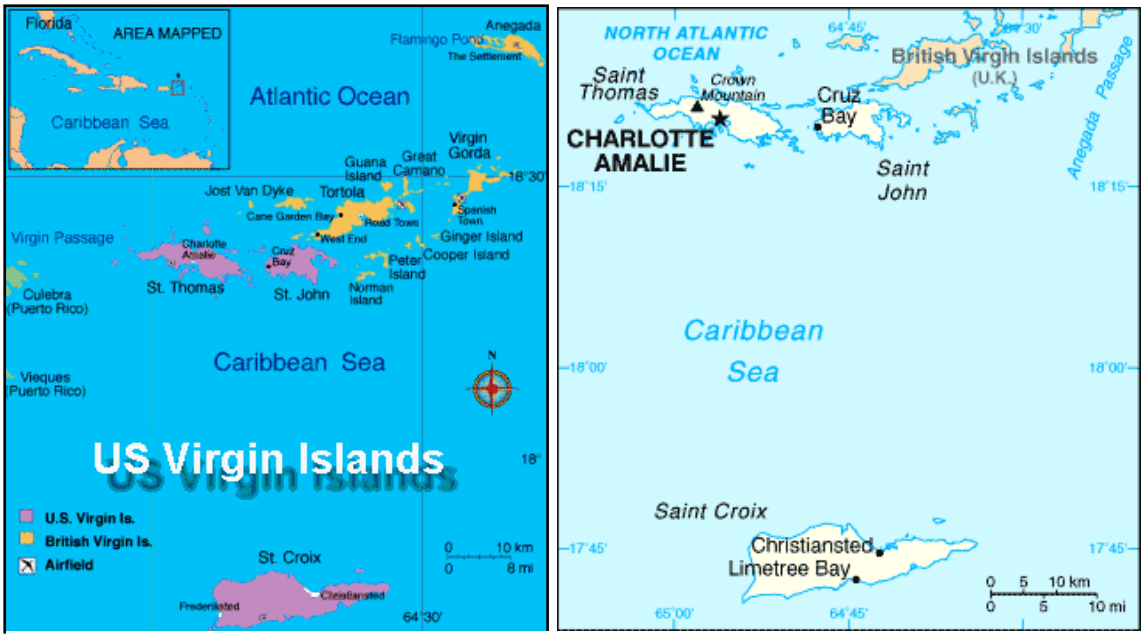

Figure 2.1 : Location maps showing the US Virgin Islands (sources: Magellan Geographic (left) and CIA World Factbook, 2006) (right)).

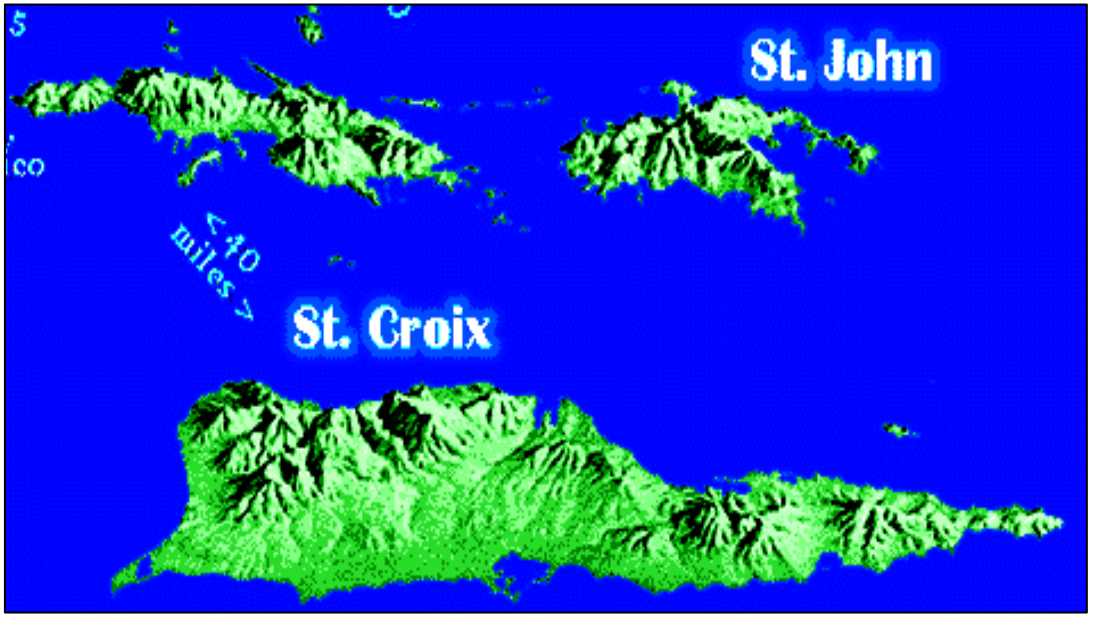

Figure 2.2: USVI Topography; source: COBEX, USVI.net

Average Annual Rainfall	
Island	**Average Annual Rainfall**
St. Thomas & Water Island	38.36"
St. John	45.31"
St. Croix	40.30"

Storm Activity from 1819-2001											
Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sept**	**Oct**	**Nov**	**De**
							34	38			

Figure 2.3: Rainfall and Storm Averages for the USVI

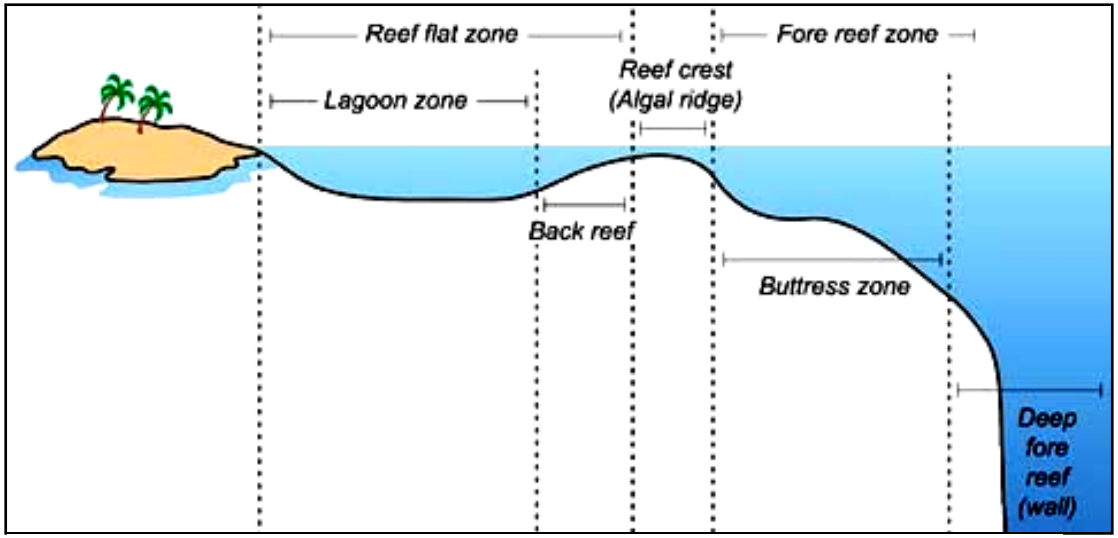

Figure 2.4: Coral Reef Zones. (NOAA National Ocean Service, 2006)

Figure 2.5: Protected Regions of USVI (US National Park Service, 2004).

CHAPTER 3

METHODOLOGY

Introduction

A time-series analysis of change in the terrestrial and benthic habitats of the USVI was conducted using Landsat scenes and ancillary data ranging from 1985 to 2001. The process involved a number of steps similar to those used in previous Landsat change detection studies, including those published by Yuan et al. (2005), Muttitanon and Tripathi (2004), Miller and Yool (2002), Nelson et al. (2002), and Mas (1999). These steps included radiometric, atmospheric and geometric correction of the Landsat images, image coregistration, masking of clouds and cloud shadows, image classification, accuracy assessment, and change detection. The benthic and terrestrial portions of each image were classified separately using an unsupervised classification and the accuracy of each scene was determined using aerial photographs and other ancillary data for reference data.

Two techniques, write function memory and postclassification analysis, were used to detect changes in terrestrial and benthic habitats. The type and

degree of change in pristine and developed watersheds were compared to determine whether terrestrial development is correlated with benthic habitat change. For watersheds where change on land is correlated with change in the benthic habitat, other variables that could have contributed to changes in the benthic habitat were examined to assess whether terrestrial runoff was a major force in causing benthic habitat change.

Data Sets

Image processing and analysis were carried out on USVI Landsat images from five dates that spanned a total of 16 years (Figure 3.1). The Landsat scenes were purchased from the United States Geological Survey EROS Data Center. All scenes had Level 1G radiometric and systematic geometric correction to 250 m based on Landsat payload correction data files and radiometric correction based on the pre-launch calibration of the sensor's reflective band detectors (Williams et al., 2000). The image data were in rescaled 8-bit unsigned integer digital number (DN) values. Because the USVI fall on two Landsat scenes, a total of ten TM and ETM+ images were acquired, two for each of the five years chosen for analysis. (Figs. 3.2, 3.3)

For optimal change detection results, Landsat scenes should be near anniversary dates with minimal clouds, haze, or humidity (Klemas, 2001). The portions of the Landsat scene over water should have minimal sun glint. Obtaining near anniversary images of areas frequented by clouds is difficult (Mas, 1999). Obtaining multi-date imagery of two adjoining Landsat scenes that were low in

cloud cover and sun glint (Table 1) was even more difficult to achieve. Several of the final images that were selected had substantial cloud cover and showed seasonal variation.

Ground control points (GCPs) were used to register the image data to a ground reference system. Thirty GCPs were acquired throughout the Landsat images (path 4/row 47 and path 4/ row 48) using a Garmin GPS unit. Points were collected from St. Thomas, St. John, and Eastern Puerto Rico. Additional GCPs were obtained from 1999 aerial photomosaics prepared by NOAA's National Ocean Service (NOAA NOS, 2000). The aerial photos had a scale of 1:9600. The GCPs were used in conjunction with a 7.5-minute digital elevation model (DEM) to orthorectify the scenes.

Ground truth data collected on or near the date of each Landsat scene was not possible due to the historical nature of the study (Nelson, et al., 2002). Larger-scale changes in habitat coverage are best described by aircraft overflights (Mumby, et. al. 1997), which may be used to assess the map accuracy of Landsat data when sufficient ground data are not available (Miller and Yool, 2002). The 1999 photomosaic of the islands and aerial photos from other years served as ground truth for class aggregation and accuracy assessment of land. A NOAA NOS benthic habitat map of the USVI from 1999, and the 1999 aerial photomosaic were used as ground truth for benthic portions of the 1999 scene. Spectral characteristics of the mapped areas were evaluated to determine the land cover categories.

Ancillary data included vector files of watersheds and subwatersheds, bathymetry, tide levels, rainfall levels, and wind speed and direction for the scene dates. All image processing and analysis were conducted with the PCI Geomatica software.

Methods

Radiometric Correction

The radiance, or amount of light detected by the a satellite sensor, is usually not equivalent to the actual reflectance from ground features due to a number of factors that alter the reflectance, including solar illumination angle, atmospheric conditions, viewing geometry, and calibration factors such as gain and offset (Lillesand and Keifer, 2000). Because the study required analysis of the same area acquired in different years, it was necessary to normalize these variables with radiometric correction techniques.

Scenes were purchased with level 1G correction, indicating that differences in brightness values due to sun angle and earth-sun distance had been normalized (Klemas, 2001). The solar zenith correction normalizes the sun elevation angle by dividing each pixel value in a scene by the cosine of the sun zenith angle. The earth-sun distance correction normalizes differences in earth-sun distance resulting from seasonal variability. The algorithm used to standardize both the solar zenith angle and the earth-sun distance on the irradiance incident on the earth's surface was defined by (Lillesand and Keifer, 2000):

$$E = [E_0 \cos\theta_0] / d^2 \qquad (3.1)$$

where:

E = normalized solar irradiance

E_0 = solar irradiance at mean earth-sun distance

θ_0 = sun's angle from the zenith

d = mean earth-sun distance

The following equation was used to convert the digital radiometric numbers in each band to absolute units of spectral radiance (Lillesand and Keifer, 2000):

$$L = [(LMAX - LMIN)/255]DN + LMIN \qquad (3.2)$$

where:

L = spectral radiance measured

LMAX = radiance at which the channel saturates

LMIN = offset of the response function, which represents the lowest digital number for the channel

DN = digital number value recorded

Atmospheric correction

The composite radiance signal at any given pixel location is expressed by the following equation (Lillesand and Keifer, 2000).

$$L_{tot} = pET + L_p \quad (3.3)$$

where:

L_{tot} = total spectral radiance measured by a sensor

p = reflectance of object

E = irradiance of object

T = transmission of atmosphere

L_p = path radiance

Reflected radiance is affected by the path length through the atmosphere. Shorter wavelengths are scattered by smaller atmospheric particles in a process known as Rayleigh scattering. Rayleigh scattering contributes to a loss in radiance contrast (Mumby and Edwards, 2002). Mie scattering occurs when the reflected energy of longer wavelengths is scattered by water particles and other relatively large particles in the atmosphere. Water particles may absorb or reflect energy from the visible and NIR wavelengths, obscuring the true spectral signal of ground features. Algorithms by Rabe and Stumpf (1997) were applied to each scene to remove the effects of Mie and Rayleigh scattering.

Additional atmospheric and radiometric correction are needed for analysis of areas under water. A thin layer of haze over water can alter spectral signatures in satellite images enough to create the false impression of spectral change between two dates (Klemas, 2001). Because deep water absorbs all NIR energy, any radiance in the NIR band over water can be assumed to be caused by haze. To

correct for haze the path radiance of the NIR band was calculated over cloud-free deep water. This NIR value was then subtracted from all pixels in each of the wavelengths.

Any radiance caused by reflectance of sunlight on the water's surface will obscure the spectral signature of benthic features. Radiance caused by reflectance of sunlight from the water's surface was determined by finding the digital number (DN) of the blue, green, and red wavelengths in cloud shadows of haze-corrected pixels. Because the DN value should be zero in cloud shadows that have been corrected for haze, the DN value of any remaining reflectance represents surface water reflectance. The surface water reflectance values of the blue, green, and red wavelengths were subtracted from all water pixels in the blue, green, and red channels, respectively. Because longer wavelengths are absorbed near the water's surface, surface reflectance was not calculated for the NIR and MIR wavelength bands.

Geometric Correction and Scene Coregistration

Geometric correction, which accounts for the earth's rotation and variation in sensor viewing angle, is required because raw satellite images usually contain geometric distortions. These distortions must be corrected before the satellite scenes can be used as a geographically accurate map base. The sources of these distortions include variations in altitude, sensor platform velocity, panoramic

distortions, earth curvature, atmospheric refraction, relief displacement, and nonlinearities in the sweep of the sensor's instantaneous field of view (Lillesand and Keifer, 2000).

Systematic distortions, which are predictable, are corrected prior to addressing random, or unpredictable, distortions. An example of a systematic distortion in satellite imagery is the eastward rotation of the earth beneath the satellite during image acquisition. This causes each optical sweep of the scanner to cover an area slightly to the west of the previous sweep (Lillesand and Keifer, 2000). To correct this skew distortion, each successive scan line must be offset slightly to the west, which results in the parallelogram appearance of the resulting image. Systematic corrections had been performed by the USGS prior to data purchase.

Random distortions were corrected with geometric correction and orthorectification. Separate mathematical models were required for orthorectification of the TM and ETM images. GCPs were incorporated into the TM model to geometrically correct the 1992 scene. The appropriate projection and coordinate systems were applied. With an image to image correction algorithm, the 1992 TM image served as the georeferenced base map to orthorectify the 1985 and 1987 TM scenes, as described in Klemas (2001). A geometric correction of the 1999 ETM+ scene was performed with a separate mathematical model. The 2001 ETM+ scene was geometrically corrected by performing an image to image correction, using the geometrically corrected 1999 scene for georeferencing. The

image to image correction process provided for the coregistration of scenes, a necessary step in change detection for preventing errors caused by misregistration of scenes (Bouvet et al., 2003).

Unsupervised Classification of Land Use and Land Cover

Because water and land have spectral characteristics very different from one another, the benthic and terrestrial portions of each satellite image were classified separately (Swaya et al., 2003). The land portion of the USVI was subsetted from the Landsat scene using a geometrically corrected outline of the islands. A cloud mask was created for each scene by digitizing polygons around the clouds and shadows and converting the polygon to a bitmap.

An unsupervised classification was performed on each scene, whereby the image was automatically segmented into spectrally similar unknown classes (Cuevas-Jimenez and Ardisson, 2002). All six Landsat bands and the cloud mask were chosen as inputs to the unsupervised classification algorithm, from which 40 output categories were created. The Iterative Self-Organizing Data Analysis Technique (ISODATA) was used to create clusters in the unsupervised classification.

Aerial photographs and spectral signatures representative of each class served as reference data to assign, or aggregate, each class to a known category. Classes were assigned based on the assumption that the reference data set is a statistically valid sample of the mapped area, that the reference samples are accurately coregistered with the map, that each map pixel corresponds to a single

land cover type, and that if time has elapsed between map acquisition and date of reference data, that land cover has not changed in the interim (Powell et al., 2004). Cluster busting, a technique used to isolate and perform a second or third unsupervised classification of clusters with indistinct or mixed spectral signatures, was performed.

Land cover classes represent segments of a continuum where the boundaries between classes may not be spectrally distinct (Powell, et al., 2004). Categories of urban land use may be categorized as residential, commercial, industrial, and institutional, or more generally, into high density, medium density, and low density development subclasses (Klemas, 2001). Based on the spatial and spectral limitations of Landsat sensors and the available ground truth data, the 40 original classes were aggregated into the following terrestrial categories: null class, dense urban, rural, cultivated land, and dense vegetation.

It is difficult to distinguish grassland versus cultivated lands using the existing TM bands (Klemas, 2001) . The rural and cultivated land classes were aggregated into the rural class, which improved the classification accuracy by 20 percent. The new rural category included low vegetation and low development, such as roads and houses in rural areas

Unsupervised Classification of Benthic Habitats

While the red, green and blue regions of the electromagnetic spectrum are able to penetrate shallow portions of the water column and be reflected back to the satellite sensor, the greatest water penetration is located at approximately in the

blue-green portion of the spectrum, at 0.47 um (Cuevas-Jimenez and Ardisson, 2002). In comparison, water absorbs a great deal of the visible red, near IR, and middle IR wavelengths.

Prior to performing an unsupervised classification of the benthic areas of each scene, it was necessary to create a single mask that incorporated the land and the clouds over the water. This was accomplished by digitizing a polygon around the clouds, adding the polygons to the land outline vector layer, and creating a bitmap from the combined vectors. Areas greater than 8 to 10 m in depth were also masked, as that is generally the depth at which Landsat data can no longer be used to clearly discern the spectral signatures of major benthic classes. The blue, green and red bands were used as input channels and 40 classes were selected for output. The ISODATA algorithm was chosen for unsupervised class formation.

The 1999 aerial photomosaic and benthic habitat maps served as reference data for class aggregation of the 1999 scenes. The final five classes were the null class, dense linear coral, coral, low seagrass, dense seagrass, and sand. Because no benthic maps or cloud-free aerial photographs were available for the other four dates, classes were assigned based on the methodology used to classify the 1999 scenes (see Dekker et al., 2005). This methodology used detailed habitat maps and aerial photographs of the USVI's benthic areas to

identify the texture, shape, location, and color of benthic features. The characteristic shape, texture, location, and albedo of benthic features was used to assign spectral clusters to benthic classes.

Information about the characteristic location of benthic habitats was also used to assign spectral clusters to the correct class. In situ studies by Gladfelter et al. (1980) and Patriquin (1975) indicated that Caribbean seagrass are most abundant in shelf lagoons behind barrier reefs. Bouvet et al. (2003) noted that in satellite images or aerial photographs, the extensive seagrass beds found in large reef lagoons and semi-protected bays may frequently be confused with outer reef slopes.

The characteristic halo around patches in dark areas was indicative of patch reefs within seagrass beds. The halo is caused by the herbivorous reef fish that feed on *Thalassia* at night and live within the reefs during the day (Alevizon, 2002; Sweatman and Robertson, 1994; Ogden and Zeiman, 1977; Randall, 1963).

To decrease the number of cloud pixels in the scenes, the unsupervised classification results of the 1985 and 1987 images were merged to create a 1986 composite, which had significantly less cloud cover than the 1985 and 1987 images. The unsupervised classifications of the 1999 and 2001 scenes were merged to create a composite 2000 scene. This procedure was also applied to created NDVI composites for 1986 and 2000.

Accuracy Assessment

The size of the random sample used for accuracy assessment was based on the following calculation (and Fitzpatrick-Lins,1986).

$$N = (z\ z\ p\ q)/(E\ E) \tag{3.4}$$

where:

N = approximate sample size

p = expected % accuracy

q = 100 – p

E = allowable error

z= standard normal deviation for a 90% (or 95%) two-sided confidence interval

Narrow limits of allowable error (E = 2%) are required when the reference data involves little to no field work, to offset any procedural errors.

An accuracy assessment of the terrestrial classifications was performed on all scenes for which aerial photography was available. When reference data did not cover the whole scene, the Landsat image was subsetted to match the spatial extent of the reference data and random points were generated over the subsetted scene, as described in Nelson, et al. (2002). The accuracy reflected the subsetted portion of the scene.

The sample size, n, was determined by:

$$n = B\ (X) \qquad (3.5)$$

where:

B = the number of points used for a sample that covered the whole scene

X = the percentage of the scene covered by the subsetted portion

(Nelson et al., 2002).

An error matrix, which compares counts of agreement between reference and map data by class (Powell et al., 2004) for points in a random sample, provided the accuracy statistics. The off-diagonal elements in the matrix represent the misclassified points and the diagonal elements represent the correctly classified points (Yuan et al., 2005). Overall accuracy levels greater than 85% were deemed acceptable (Rosenfield et al.,1982). User and producer accuracies were also calculated. A Kappa statistic, which indicates the percentage of classification accuracy that is greater than that resulting from chance (Miller and Yool, 2002), was calculated for each map. Using the 1999 benthic maps and aerial photomosaics as reference data, the accuracy of the benthic portions of the 1999 scenes was calculated.

Change Detection

Change in the terrestrial and benthic habitats was determined with two techniques: 1) post classification analysis of change between two or more dates

and 2) write function memory insertion. To detect change on land using write function memory insertion, an NDVI was calculated for each scene using the following equation:

$$NDVI = (NIR - R)/ (NIR + R)* 100 \quad (3.6)$$

where

NIR = near infrared wavelength band

R = red wavelength band

The NDVI data from three dates were inserted chronologically into the blue, green, and red channels. Major changes in NDVI between dates appear in combinations of the primary (red, green, blue) or complementary (yellow, magenta, cyan) colors (Hayes and Sader, 2001). The magnitude and direction of biomass change was determined visually based on knowledge of the degree and magnitude of change associated with the different color combinations.

To detect change in benthic habitats with write function memory, DN values of the blue and green wavelengths were compared to determine which band had the greatest depth penetration. The blue band was found to have the greater depth penetration than the green band. The blue band, which had earlier been corrected for haze and surface reflectance, was referred to as the blue water reflectance channel. The blue water reflectance channels of three scenes were inserted chronologically into an RGB viewing composite to determine the magnitude and direction of change, based on the resulting color combinations.

In addition to providing base maps of the USVI for the five dates, the unsupervised classification results were used to determine the change in both terrestrial and benthic categories over time. The second change detection technique, post classification analysis, required creating change maps that depicted the change in class that occurred in the time period between two scenes. The expected accuracy of each change map was the product of the accuracies of the “from” and “to” maps (Yuan et al., 2005). Change map accuracy was also determined by noting the change category of randomly selected pixels and seeing whether the maps of unsupervised classifications verified this change (Yuan et al., 2005). 29, 27-40.

Landsat Scene	Cloud Cover **low : < 30%,** **moderate: 30% ≤ 60%** **high: > 60%**	Sun Glint	Season	Sensor
St. Thomas and St. John Landsat Scene ID: Path 4/ Row 47				
February 26, 2001	Low	None	Dry	ETM+
September 17, 1999	Moderate	Some	Wet	ETM+
August 12, 1992	Low	Some	Wet	TM
October 10, 1987	Low	Some	Wet	TM
March 10, 1985	Low	Some	Dry	TM
St. Croix Landsat Scene ID: Path 4/row 48				
January 25, 2001	Moderate	None	Dry	ETM+
September 17, 1999	Low	Some	Wet	ETM+
August 12, 1992	High	None	Wet	TM
November 19, 1987	Moderate	Some	Wet	TM
February 22, 1985	Low	None	Dry	TM

Figure 3.1: Landsat scenes selection criteria

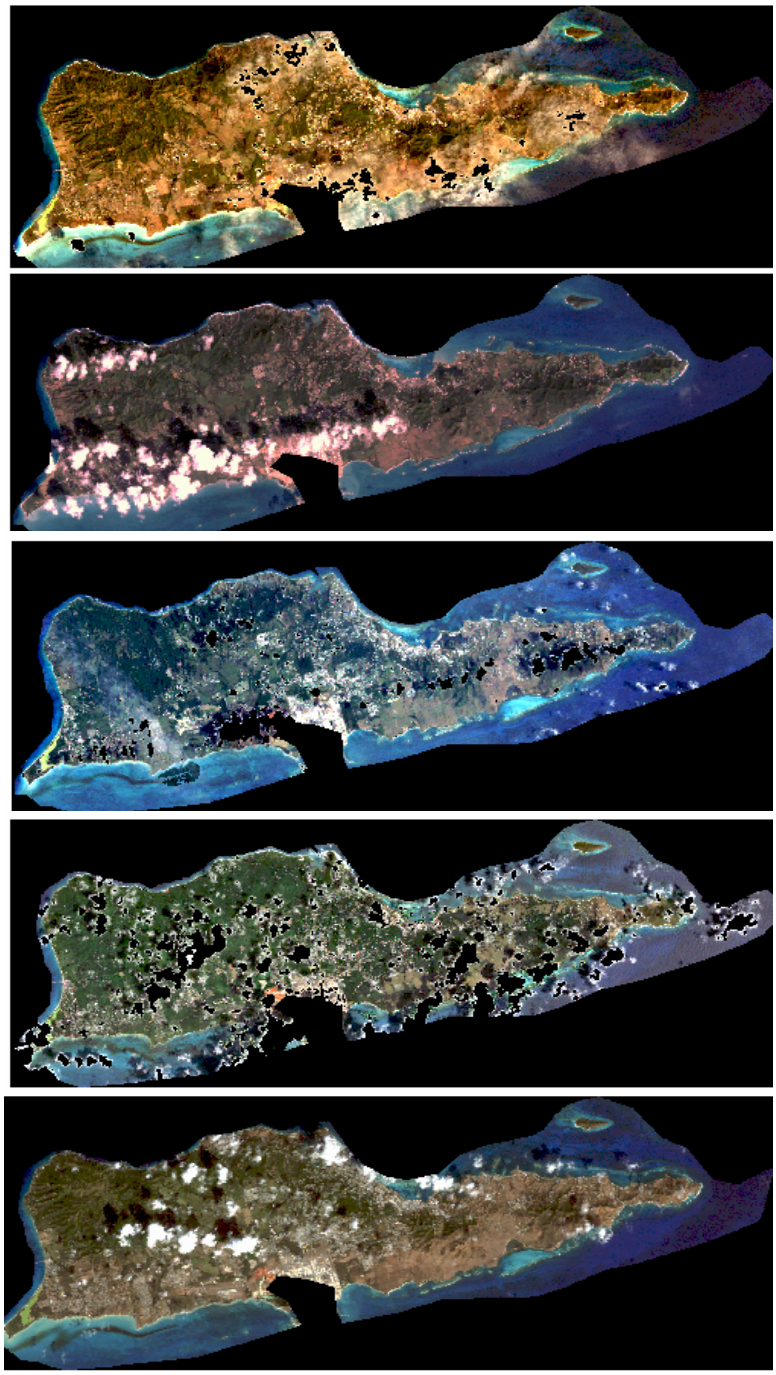

Figure 3.2 : Landsat scenes of St. Croix. Dates of these scenes from top to bottom: 1985, 1987, 1992, 1999, and 2001.

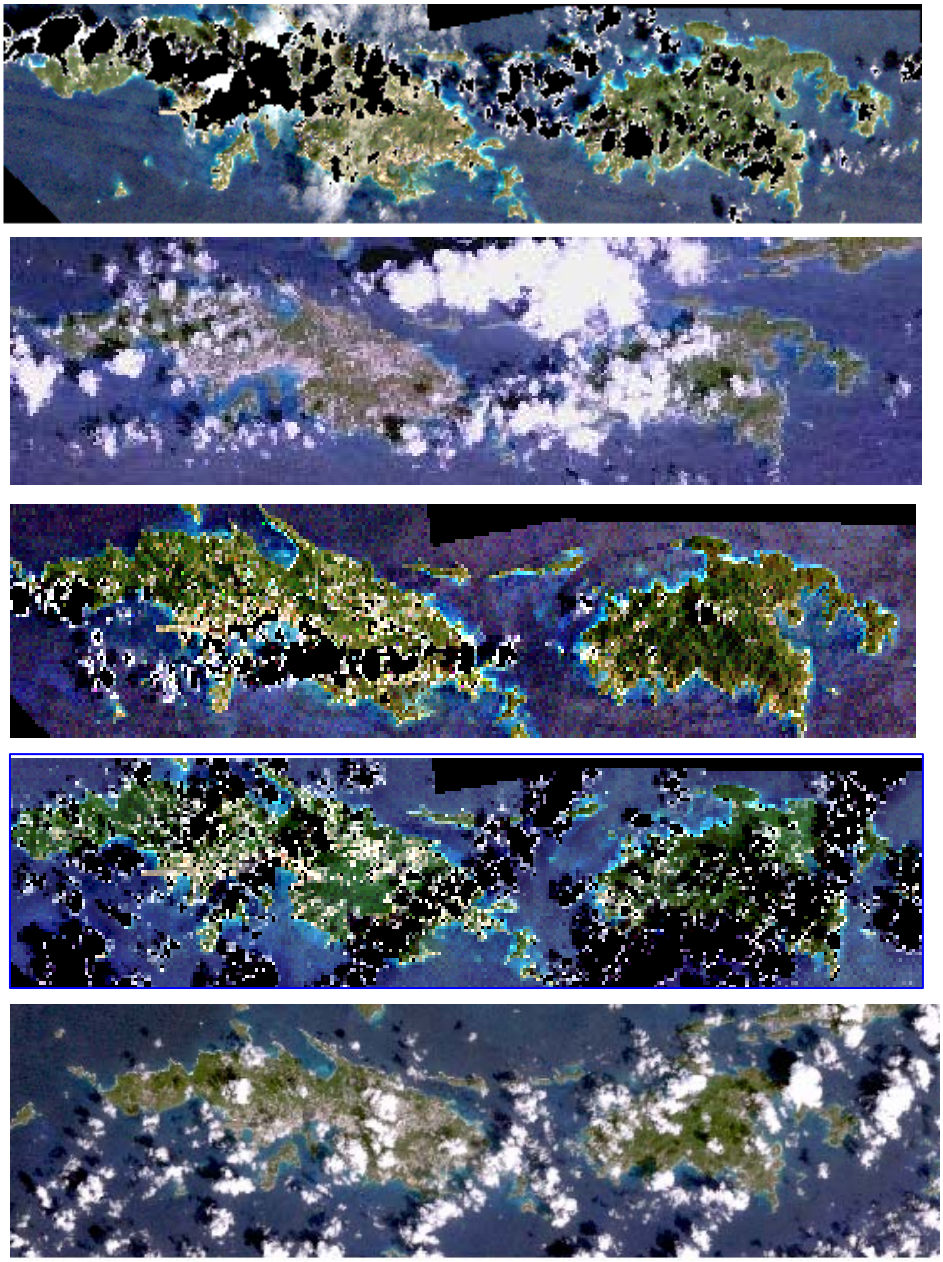

Figure 3.3: Landsat scenes of St. Thomas and St. John.
Dates of these scenes, from top to bottom:
1985, 1987, 1992, 1999, and 2001.

CHAPTER 4

RESULTS

St. Croix Data Results

Unsupervised Classification of Terrestrial Areas

Using aerial photos and spectral curves as reference, the initial 40 spectral classes in the terrestrial portion of the unsupervised classification was separated into five classes, 1) densely vegetated forest, 2) farms, 3) rural areas with unpaved roads, low vegetation and sparse construction or development, 4) dense urban development and 5) the null category, which represented all unclassified pixels, including all water and cloud-covered areas (Figure 4.1).

Accuracy assessments of this classification resulted in a low accuracy, ranging from 20% to 50%. Analysis of each sample point in the accuracy assessment process indicated that over 80% of the error was due to both errors of commission and omission in the farm class. This was consistent with findings by Klemas (2001) that demonstrated the difficulty in distinguishing grassland from cultivated lands in Landsat TM or ETM scenes (Klemas, 2001). Therefore, the cultivated land class was aggregated into the rural class (Figure 4.2). As a result,

rather than five classes, four classes were used to describe the land cover mapped from the Landsat scenes . The overall classification accuracy was higher than with the previous classification. The accuracy level for the 1992 classification increased from 30.4% to 90.78% . Although the new thematic maps contained less information than with the five classes, improving the classification accuracy made the map more useful for land management and policy analysis (Congalton and Green, 1999).

The 1985, 1987, 1999, and 2001 Landsat scenes of St. Croix each had 40% or greater cloud cover over land. The spectral signal from the earth was distorted by dense clouds and by the haze or diffuse cloud cover near or around the cloud edges. It was necessary to mask all pixels totally or partially covered by clouds or haze. To increase the number of cloud-free pixels for classification, the 1985 and 1987 classifications were merged to create a 1986 composite, which had less than 15% cloud cover. The unsupervised classifications of the 1999 and 2001 scenes were merged to create a composite 2000 scene. The original 1992 scene had less than 10% cloud cover and was used in its entirety. The classifications of the 1992, 1986, and 2000 scenes are shown in Figure 4.5.

From 1986 to 2000, the urban and rural areas increased in size (Figure 4.3), while the number of vegetated pixels decreased. The areal coverage of each class was determined with the following equation:

$$a = n\,(900\ m^2)\,(.000001) \qquad (4.1)$$

where n = number of pixels in the class

900 m^2 = areal size of one Landsat pixel in m^2

a = class size in km^2

Change Detection of Terrestrial Areas

Change detection of terrestrial areas was conducted with the postclassification maps and by NDVI change analysis. In postclassification change analysis, the unsupervised classification of one date was subtracted from the unsupervised classification of a second date. The unsupervised classes were coded so that the difference of each pair of land cover classes had a unique value. Postclassification change maps were created for the following pairs of scenes: 1986 to 1992, 1992 to 2000, and 1986 to 2000. The resulting change maps had six “from and to” classes that indicated the type of change that had occurred. Areas that had not changed were incorporated into the null class. Change analysis was performed only on pixels that were cloud-free in both the before and after scenes. The postclassification change maps for St. Croix are shown in Figure 4.7.

Figure 4.6 depicts the areal extent for each change category in the terrestrial change maps for St. Croix. In both the 1986 to 2000 and 1986 to 1992 change maps, the greatest change in land cover was from the vegetated to rural class. In the 1986 to 2000 map, the vegetated to rural class was three times greater than the other categories of change. In the 1986 to 1992 map, the vegetated to rural class was over seven times the area of most of the other change

categories. From 1986 to 2000, the vegetated to urban and rural to urban classes were the second and third largest change categories. These results indicate that the urban and rural classes increased in areal extent for St. Croix, while the vegetated pixels decreased in areal extent.

Use of NDVI for Determining Land Cover and Change

The Normalized Difference Vegetation Index (NDVI) quantifies the degree of vegetation cover on the Earth's surface and is based on the measured intensity of the red (R) and near infrared (NIR) portions of the electromagnetic spectrum.

$$NDVI = (NIR - red) / (NIR + red)$$

NDVI maps of St. Croix, calculated using the red (3) and NIR (4) Landsat bands are shown in Figure 4.8. NDVI values fall within a range from 0 to 1. Higher NDVI values appear lighter and indicate areas with greater vegetation density, while the lower NDVI values appear darker and represent dense urban areas. Intermediate grey values represent rural areas. The NDVI values for urban areas occurred between 0 and 0.35, densely vegetated areas occurred in NDVI vales from 0.65 to 1. The NDVI values supported the results of the unsupervised classification results for 1986, 1992, and 2000.

In addition to determining terrestrial change with postclassification analysis, a second change detection method employed was write function memory. The NDVI results for the 1986, 1992, and 2000 maps were placed chronologically into the red, green, and blue color displays. The resulting map, shown in Figure 4.9,

displays a range of colors that indicate the direction and magnitude of change in vegetation cover based on the NDVI values of the three input maps.

For example, a dark pink or orange pixel indicated a change from high vegetation to low vegetation. This was based on the NDVI values of that pixel in 1986, 1992, and 2000, which showed a progressive decrease in NDVI value. Yellow pixels indicated a change from vegetated to exposed soil or urban land cover and blue indicated an increase in vegetation cover. A greater intensity of color indicated a greater magnitude of change in vegetation. The type of change seen in the write function map supported the categories of change displayed in the postclassification change maps for change from 1986 to 1992, 1992 to 2000, and 1986 to 2000.

Unsupervised Classification of Benthic Areas

For the unsupervised classification of benthic areas, the 1999 aerial photos and the NOAA Benthic Habitat Maps were used as background information to separate the benthic portion of each scene into appropriate categories. The 50 initial spectral classes were aggregated into six classes: dense linear coral, aggregated coral, low density seagrass (areas with up to 30% seagrass cover), dense seagrass, sand, and null. The null class included cloud-covered, land, and deep water pixels. Figure 4.10 shows an example of the resulting thematic map.

An initial accuracy assessment of the classification indicated confusion between the low density seagrass and dense seagrass classes, which is most likely due to the difficulty in discriminating the optically similar signatures of the two

classes with Landsat (Bouvet et al., 2003). These classes were merged into one class, which improved overall accuracy. Due to confusion in distinguishing the dense linear and aggregated coral pixels, these two classes were merged into one class, which also improved the overall accuracy.

The unsupervised classification of benthic habitats did not capture the differences in reef types and species, but the classification did successfully differentiate sand, seagrass, and coral. Andrefouet et al. (2001), who used Landsat 7 ETM+ to map change in reefs in Florida and Hawaii, found that the Landsat data was most effective when general identification categories, such as sand, reef background (pavement, rubble, and heavily grazed dead coral), and reef foreground (living corals and macroalgae) were used.

The final classification had four classes: null class, coral, seagrass, and sand. The classification accuracy was further improved by removing all areas greater than approximately 9 m in depth from the classification. This was due to the fact that, in clear water, Landsat TM and ETM+ sensors are unable to discriminate spectral signatures of objects greater than approximately 9 m in depth (Mumby et al., 1997). Figure 4.11 shows the 1987 unsupervised classification with four classes.

The number of cloud pixels in the benthic classification were minimized by combining the cloud-free benthic pixels in the 1985 and 1987 scenes and the 1999

and 2001 scenes. The unsupervised classification of the 1986 composite, 2000 composite, and 1999 scenes are shown in Figure 4.12. Figure 4.13 shows the characteristics of each of these benthic classes

An accuracy assessment was conducted for the 1999 Landsat scene alone, since benthic ground truth data were available only for that year. Given the overall accuracy of the 1999 benthic scene and because identical contextual rules were used to assign classes for all five dates, benthic classification accuracies for the other scenes were deemed acceptable, as previously demonstrated by Nelson, et al. (2002). The overall accuracy of the 1999 benthic habitat classification was 92.1%. Overall accuracy of 1999 benthic habitat classification was 92.1%. Figure 4.14 shows the areal extent of the St. Croix benthic classes for the three Landsat image dates.

Change Detection of Benthic Areas

Change detection of the benthic areas was conducted with the final classification maps. The results of this analysis for St. Croix are shown in Figure 4.16. The areal extent of each category in the change maps is shown in Table 4.15. Changes in color and spatial patterns were used to distinguish types of benthic habitat change. While a transition in color from bright blue to dark green indicated a shift from sand to seagrass, a transition from dark blue to dark green represents a shift from deep sand to seagrass.

A second method used to detect change in benthic areas was with the write function procedure for the blue water channels. The basis for this method is

the spectral intensity of benthic pixels in the blue band, which was found to penetrate the water column to a greater extent than the other Landsat bands. A decrease in spectral intensity of the blue band indicates a change from coral to algae or from sand to seagrass because both types of change indicate a decrease in bottom albedo or brightness. Figure 4.17 shows the composite of the haze-corrected blue bands for the 1985, 1992, and 1999 scenes, which were placed chronologically in the red, green, and blue color guns.

St. John and St. Thomas Data Results

Because St. Thomas and St. John are contained in one Landsat scene, the accuracy assessments for the terrestrial and benthic portions of each image were performed on the entire scene. The classification and change detection results for St. Thomas and St. John were analyzed separately to account for the differences in development trends for the two islands. Terrestrial and benthic change on St. John and St. Thomas were determined with postclassification change analysis.

St. John

The unsupervised classification results for the terrestrial portions of the St. John Landsat scenes are shown in Figure 4.19. From 1986 to 2000, while the size of the urban and rural classes increased, the extent of the vegetated class decreased (Figure 4.18). The results of postclassification change detection for the terrestrial areas of St. John is shown in Figure 4.20. The sizes of the associated change categories are shown in Figure 4.21. The results of the

unsupervised classification and postclassification change detection for the benthic areas of St. John are shown in Figure 4.22 through 4.25.

St. Thomas

From 1986 to 2000, the number of urban and rural pixels on St. Thomas increased in size, while the number of vegetated pixels decreased. Terrestrial and benthic change detection on St. John and St. Thomas was determined with postclassification change analysis. The unsupervised and postclassification change maps for the terrestrial and benthic portions of St. Thomas are shown in Figures 4.26 through 4.33 .

Accuracy Assessment

An accuracy assessment was performed on the scenes for which aerial photos, which served as reference data, were available. An unsupervised classification with an accuracy below 85%, which is the minimal level of acceptable accuracy for a thematic map (Rosenfield et al., 1982) was not accepted. The classified and reference data for each point in the accuracy assessment were compared to ascertain the source of the error. A new unsupervised classification was conducted. This process was repeated for each classification result until an accuracy of 85% or greater was achieved. The accuracy of each postclassification change map was the product of the accuracies of the “from” and “to” maps.

Figures 4.34 through 4.37 show the error matrices, user accuracies, and producer accuracies for the terrestrial and benthic habitats of St. Croix for 1985,

1992, and 1999. Figures 4.38 and 4. 39 show the accuracy results for the terrestrial and benthic habitats of St. Thomas and St. John for 1999.

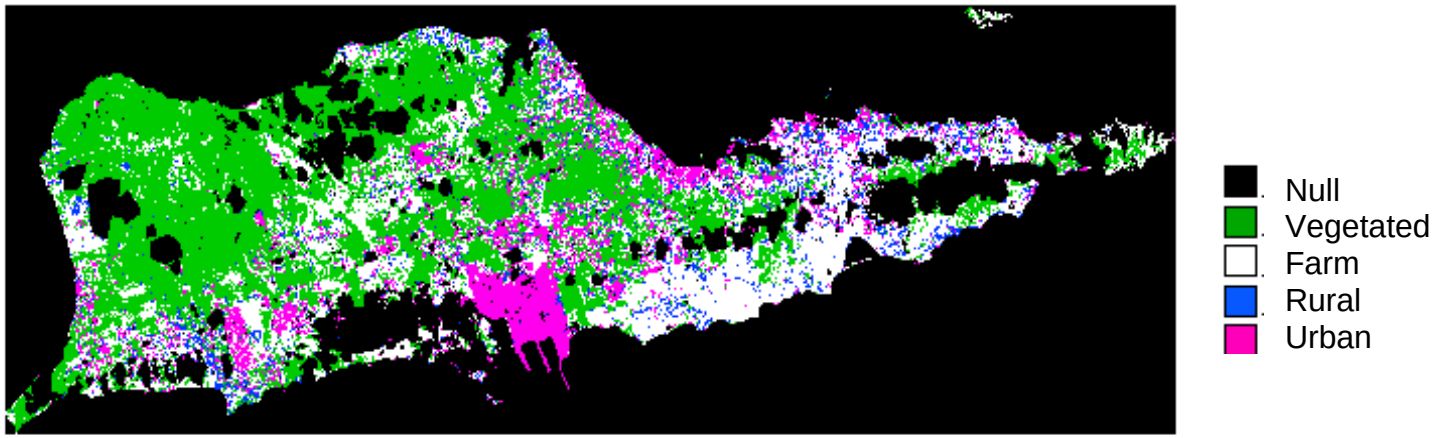

Figure 4.1 : Unsupervised Classification With 5 Classes for 1992 Landsat Scene of St. Croix

Land Cover Class	Description
Vegetated	Primary and Secondary Growth Forest, Shrubs, Orchards
Rural	Farms, Bare Fields, Bare Soil, Dirt Roads, Rural Housing, Planted Grass, Pasture
Urban	Paved Roads, Dense Urban Development, Residential, Industrial, Mixed Urban or Built-up land

Figure 4.2: Description of Terrestrial Unsupervised Classification Classes

Total Area: 118 km^2			
Year	**class**	**km^2**	**%**
1986	vegetated	144.67	60.28
	rural	51.99	21.66
	urban	23.31	9.71
	clouds	20.04	8.35
1992	vegetated	91.98	38.33
	rural	78.04	32.51
	urban	23.57	9.82
	clouds	46.42	19.34
2000	vegetated	93.56	38.98
	rural	75.69	31.54
	urban	42.05	17.52
	clouds	28.70	11.96

Figure 4.3: Areal Extent of Terrestrial St. Croix Land Use Classes

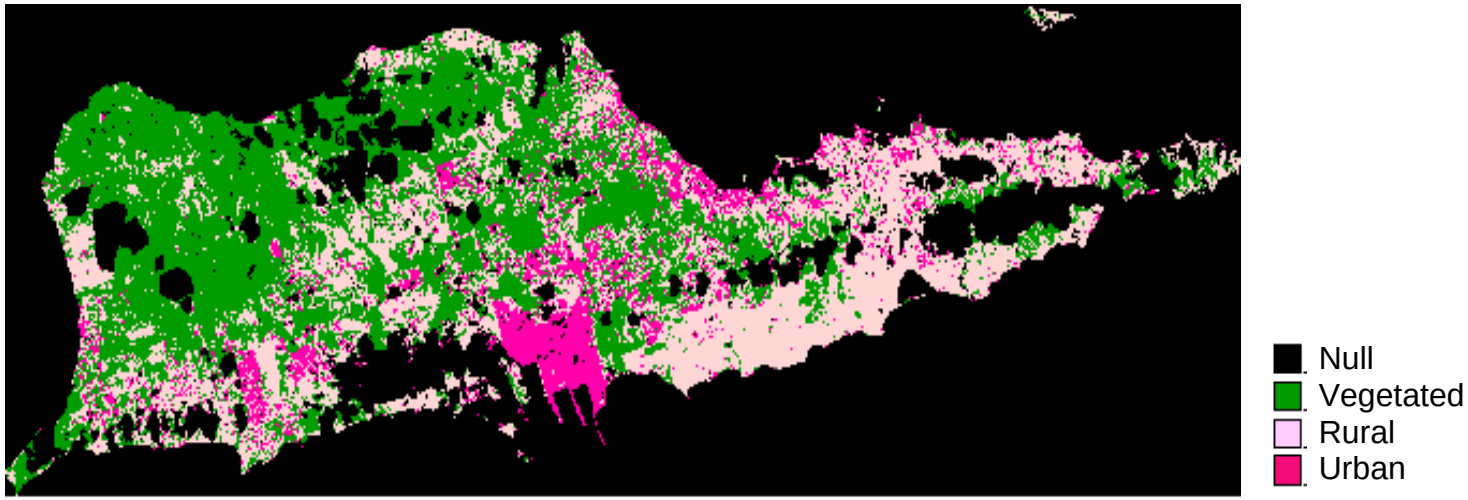

Figure 4.4 : Unsupervised Classification With Four Classes for 1992 Landsat Scene of St. Croix

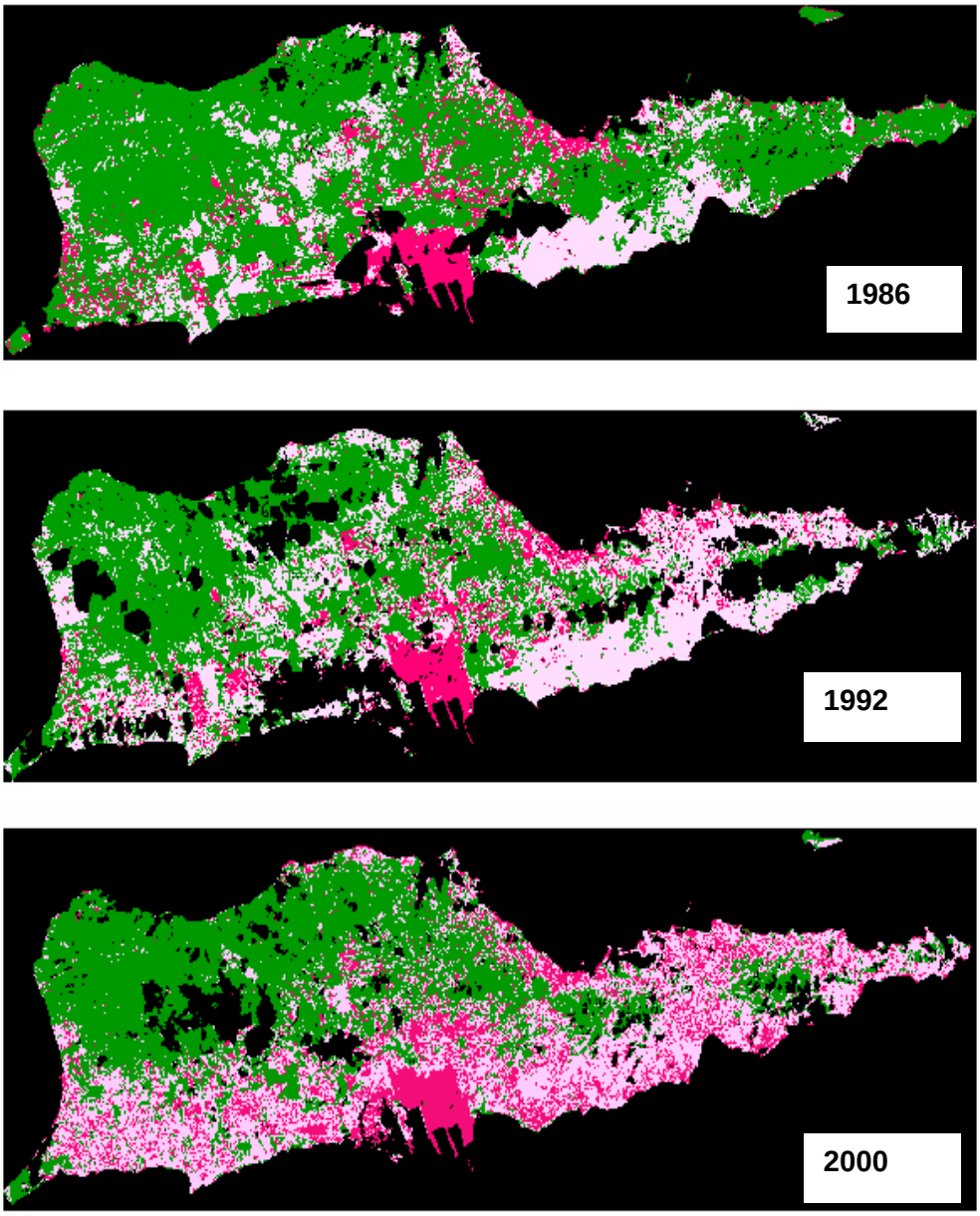

Figure 4.5: Terrestrial Unsupervised Classification of Landsat Scenes for St. Croix

	Vegetated to Urban km^2 (%)	Vegetated to Rural km^2 (%)	Rural to Urban km^2 (%)	No Change km^2 (%)	Urban to Rural km^2 (%)	Rural to Vegetated km^2 (%)	Urban to Vegetated km^2 (%)	Cloud Cover km^2 (%)
1986 to 2000	3.04 (5.43)	36.74 (15.31)	12.22 (5.09)	117.24 (48.85)	6.6 (2.75)	9.74 (4.06)	2.43 (1.01)	42 (17.5)
1986 to 1992	5.43 (2.26)	35.02 (14.59)	5.03 (2.09)	119.76 (49.9)	5.67 (2.36)	8.64 (3.6)	2.48 (1.03)	58 (24.17)
1992 to 2000	0.99 (2.08)	16.51 (6.88)	16.27 (6.78)	111 (46.25)	7.42 (3.09)	16.64 (6.94)	1.69 (0.7)	65.47 (27.28)

Figure 4.6 : Areal Extent of Terrestrial Change Classes for St.Croix

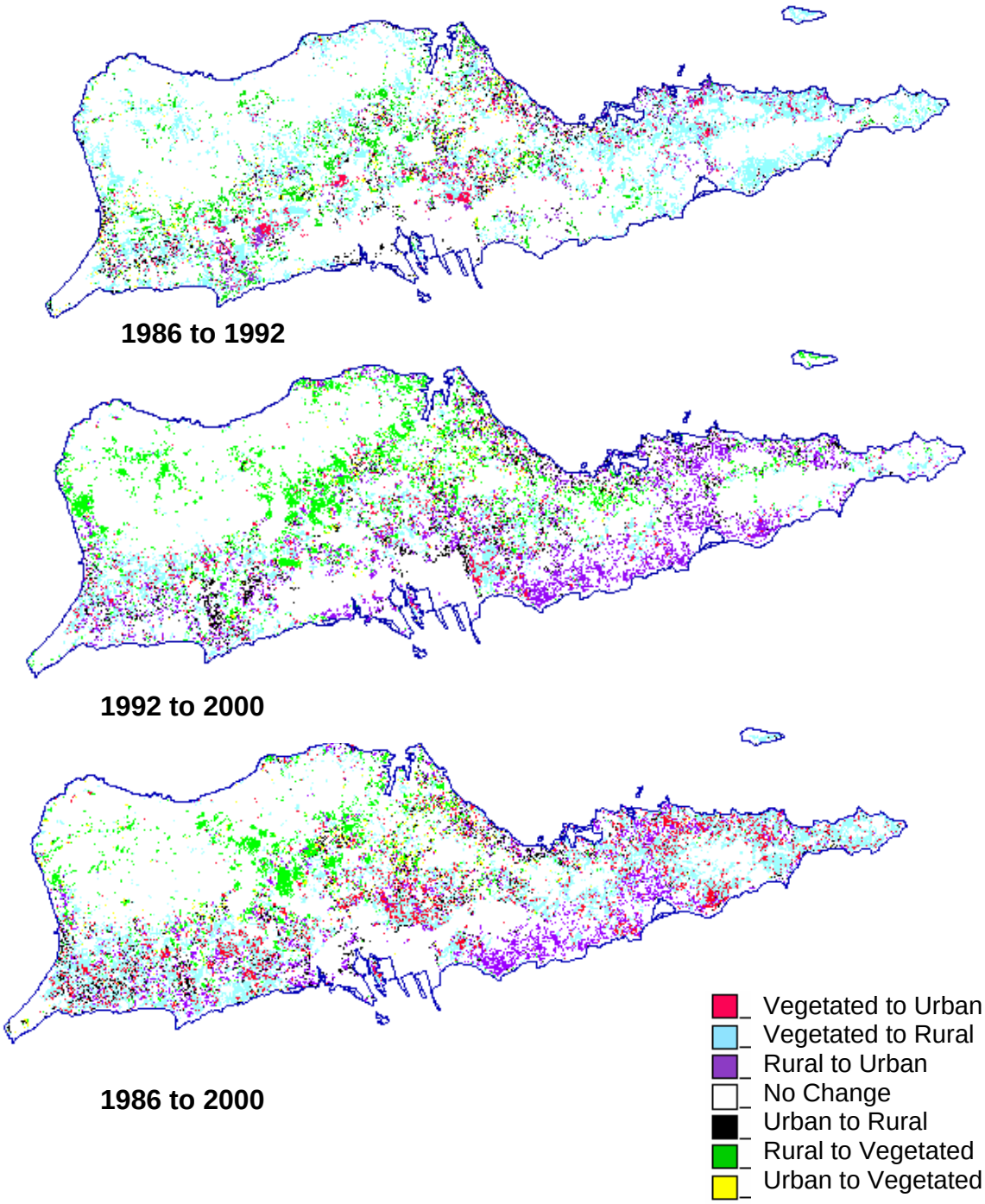

Figure 4.7: Postclassification Change Maps for Terrestrial Areas of St.Croix

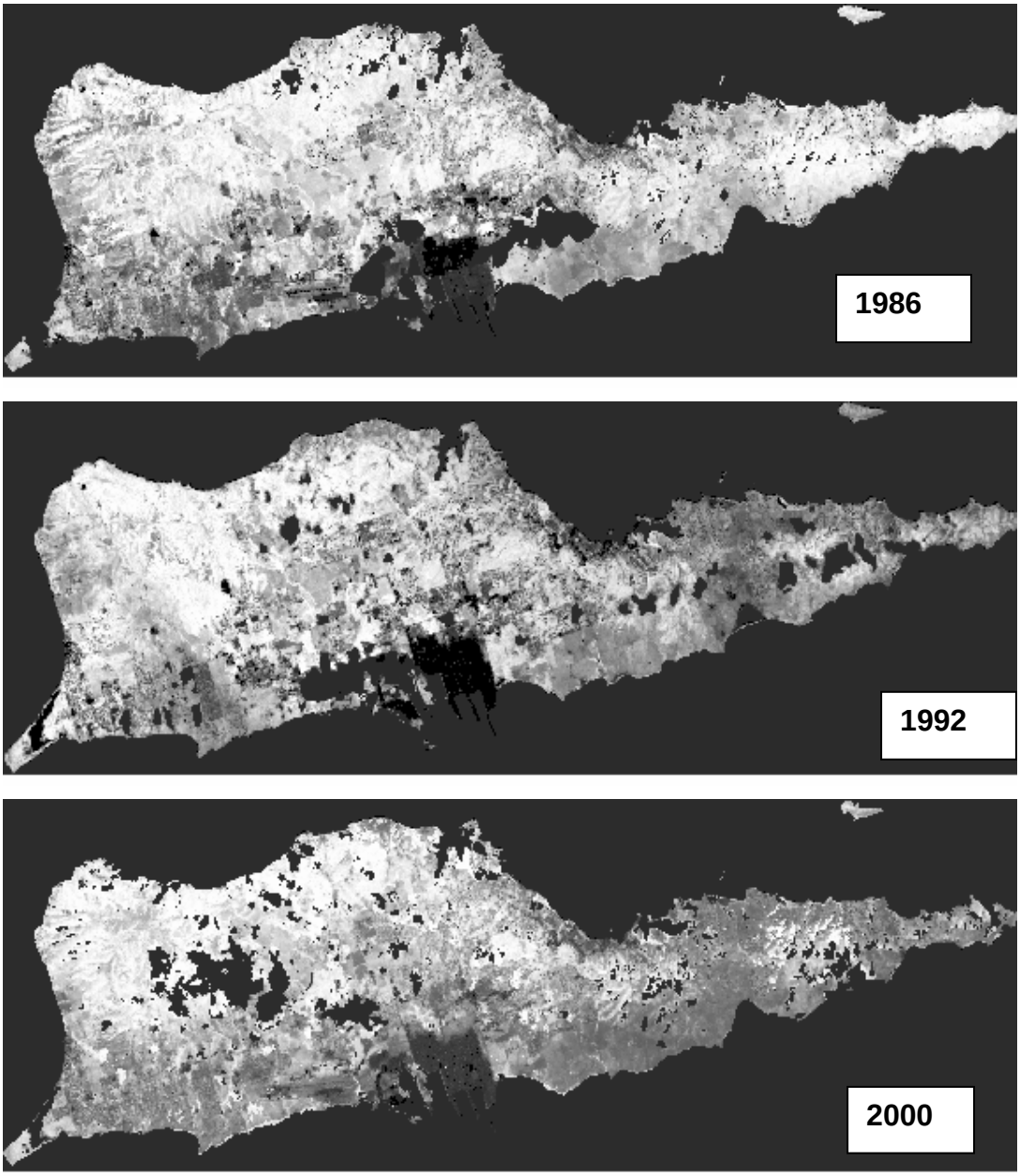

Figure 4.8: NDVI of Landsat St. Croix Scenes

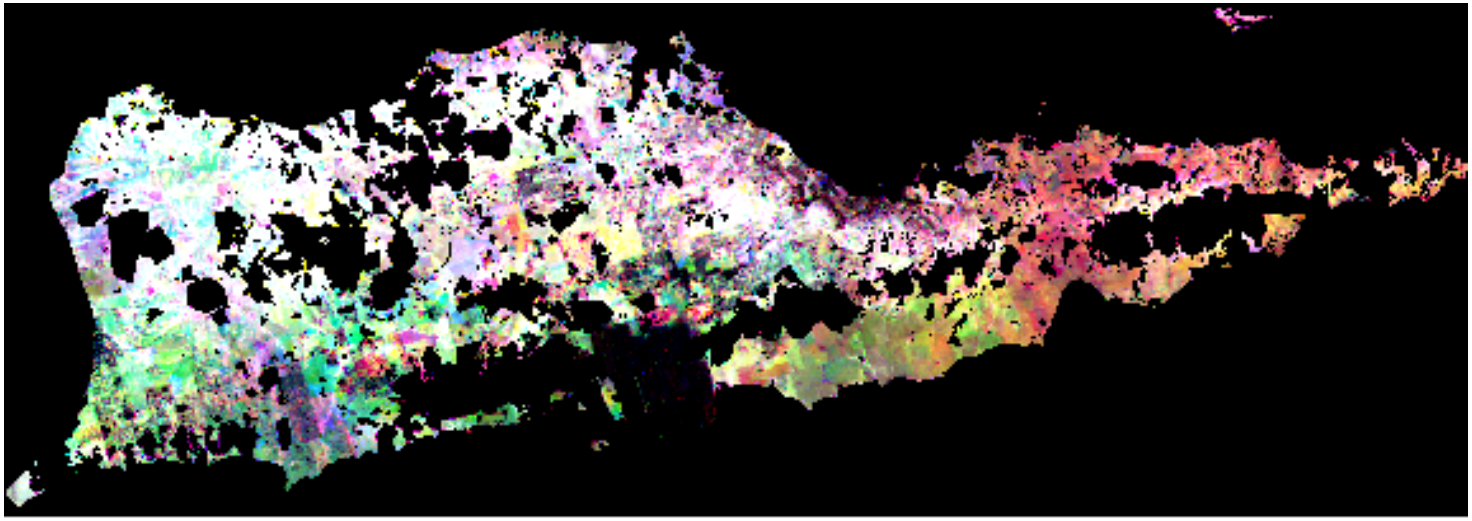

Figure 4.9: Change Detection of St. Croix for 1986 to 2000 using the write function technique. NDVI layers for 1986, 1992, and 2000 are placed in the red, green, and blue color layers, respectively. The composite of the NDVI values in the three layers gives a unique color which represents the magnitude and direction of change in land cover. Areas in dark pink or orange indicate a significant decrease in NDVI values over the 1986-2000 period, which signifies a shift in land cover from vegetated to rural or urban. Light pink and yellow areas are the result of a small decrease in NDVI. Areas of bright blue indicate an increase in NDVI, which may be seasonal.

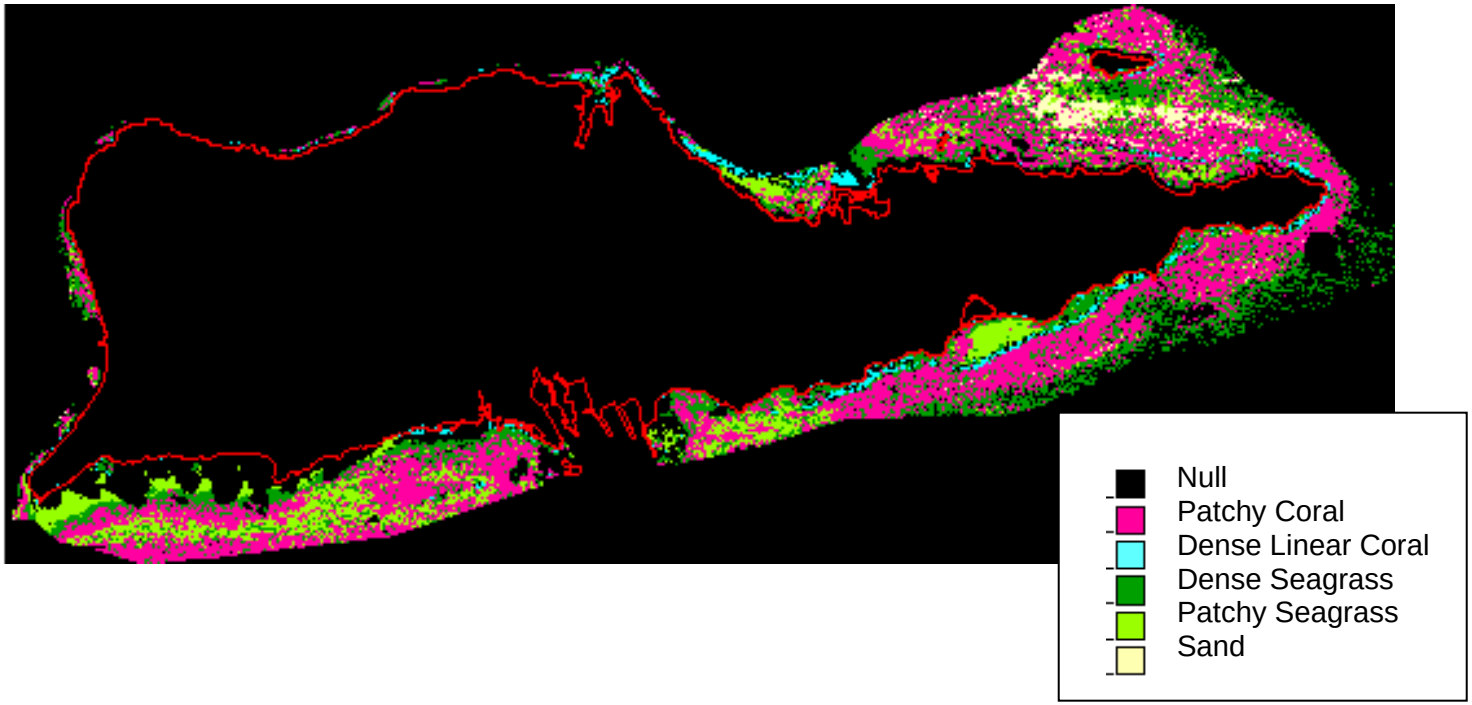

Figure 4.10: Unsupervised Classification of Benthic Areas of St. Croix with Six Classes

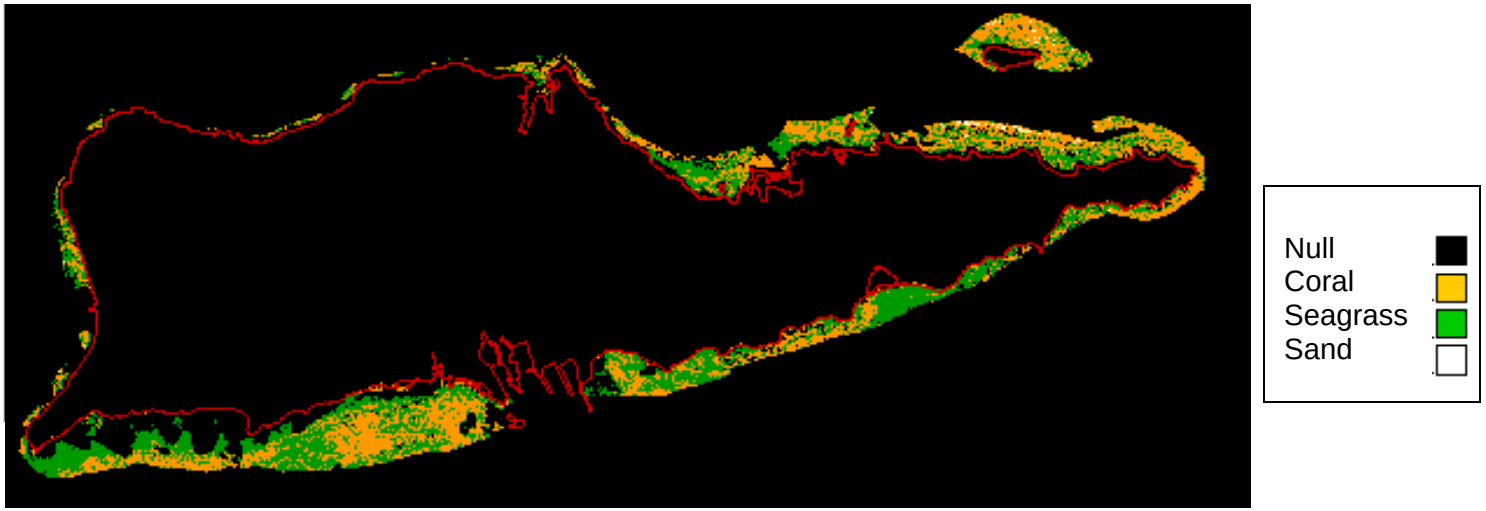

Figure 4.11 : Unsupervised Classification Showing Four Classes of Benthic Habitats for St. Croix.

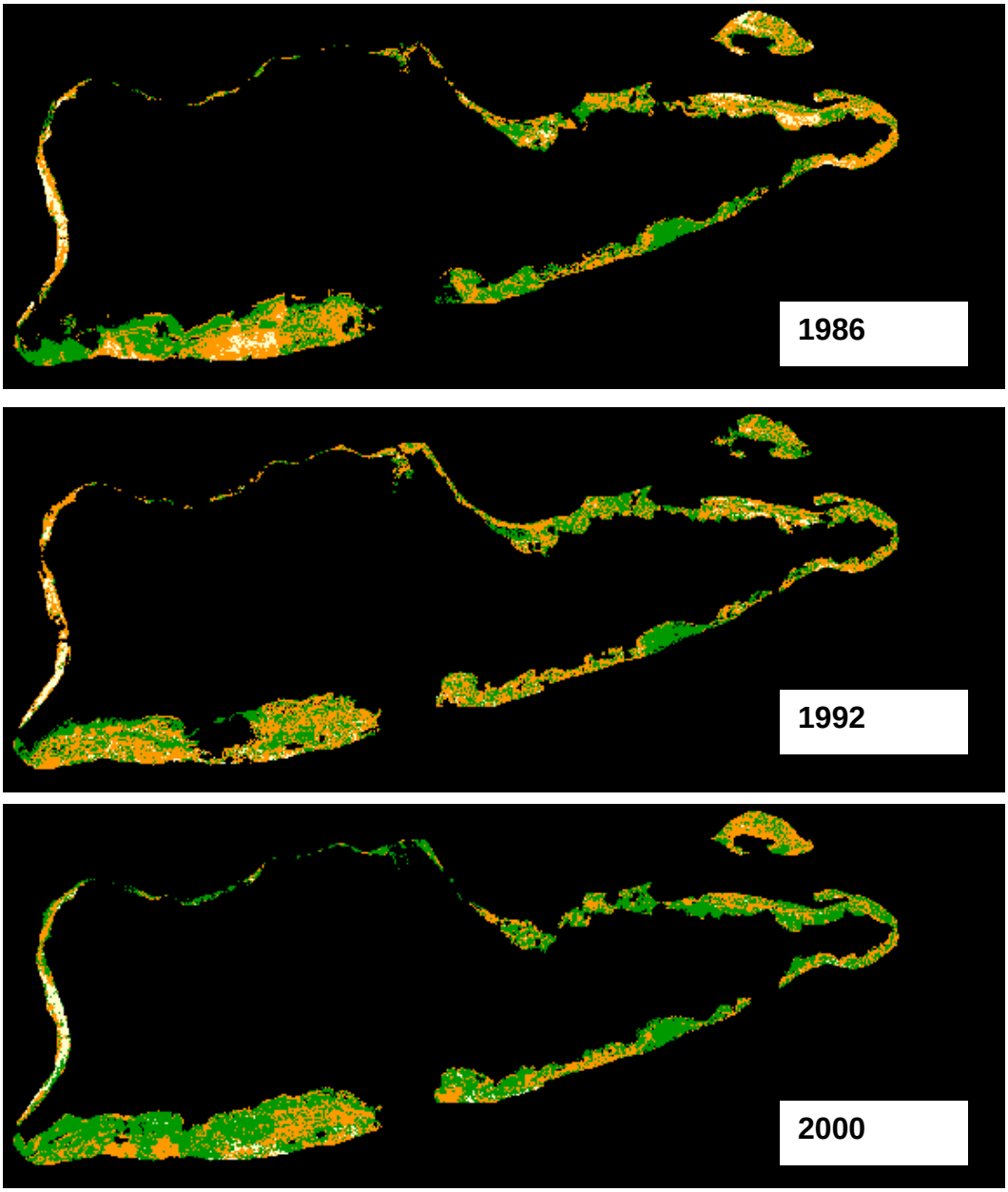

Figure 4.12 : Unsupervised Classification for Benthic Habitats of St. Croix.

Land Cover Class	Description
Coral	Linear Reef, Patch Reef, Spur and Groove Reef, Colonized Pavement
Seagrass	Continuous Seagrass Cover (60 to 100%), Partial Seagrass Cover (30 – 60%), Macroalgae (Continuous and patchy)
Sand	Calcium Carbonate Sand, low seagrass cover (less than 20%)

Figure 4.13 : Characteristics of benthic unsupervised classification categories

Total area: 68 km2			
Year	**Class**	**Area (km^2)**	**%**
1986	coral	30.83	45.33
	seagrass	25.21	37.06
	sand	4.54	6.68
	clouds	7.43	10.93
1992	coral	32.76	48.16
	seagrass	25.01	36.76
	sand	3.3	4.85
	clouds	6.96	10.23
2000	coral	26.71	39.26
	seagrass	38.08	55.98
	sand	2.10	3.09
	clouds	1.14	1.67

Figure 4.14: Areal Extent of St. Croix Benthic Classes

	Coral to Sand km^2 (%)	Seagrass to Sand km^2 (%)	Coral to Algae km^2 (%)	No Change km^2 (%)	Seagrass to Coral km^2 (%)	Sand to Seagrass km^2 (%)	Sand to Coral km^2 (%)	Clouds km^2 (%)
1986 to 1992	1.03 (1.52)	0.28 (0.40)	14.03 (20.63)	29 (42.63)	7.83 (11.52)	1.66 (2.44)	2.06 (3.03)	12.12 (17.83)
1992 to 2000	1.64 (2.42)	0.53 (0.78)	8.62 (12.68)	24 (35.28)	9.98 (14.66)	0.88 (1.29)	2.26 (3.32)	20.11 (29.57)
1986 to 2000	0.97 (1.42)	0.39 (0.58)	15.63 (22.98	28 (41.16)	7.54 (11.08)	1.1 (1.62)	1.62 (2.38)	12.77 (18.78)

Figure 4.15: Areal Extent of Benthic Postclassification Categories for St. Croix in km^2 with percentage value in parentheses.

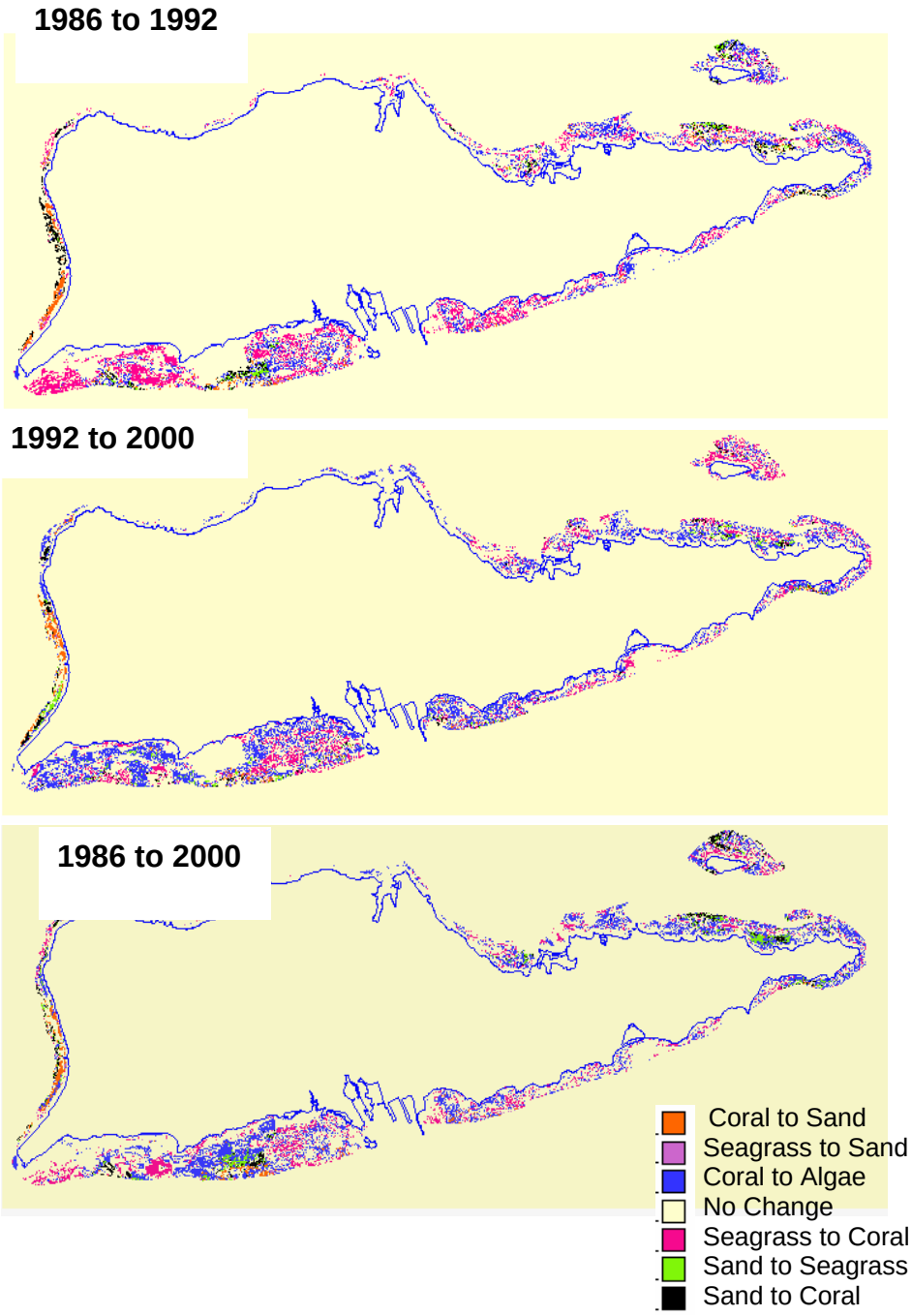

Figure 4.16: Postclassification Change Analysis of Benthic Habitats for St. Croix

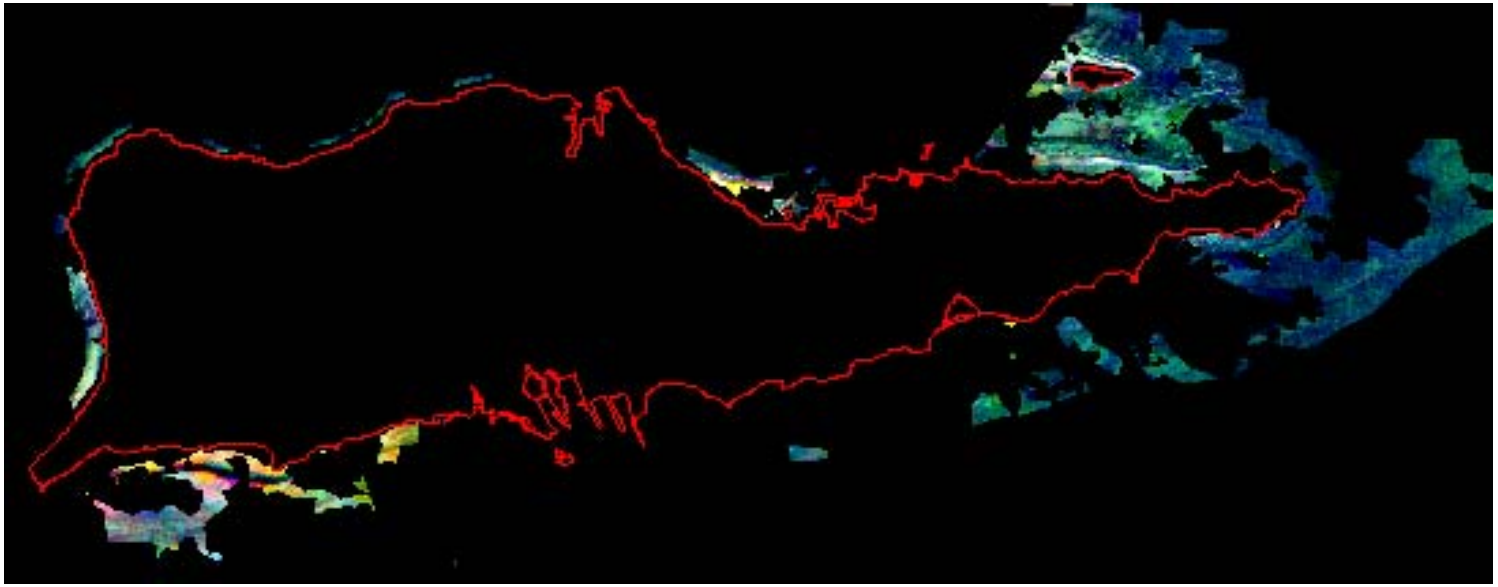

Figure 4.17: Write Function for Benthic Change Analysis of the 1985, 1992, and 1999 Landsat Scenes for St. Croix The blue water reflectance channels for the three years are placed chronologically in the red, green and blue layers. Areas in yellow, pink or red indicate a significant decrease in blue water reflectance, which signifies a change from sand to seagrass or from coral to algae. Areas in light blue result from an increase in blue water reflectance values, indicating a change from seagrass to sand or from dense to sparse seagrass cover.

Total Area: 52 km^2			
Year	**Class**	**Area (km^2)**	**%**
1986	vegetated	33.36	64.15
	rural	0.53	1.02
	urban	2.58	4.97
	clouds	15.68	30.16
1992	vegetated	37.95	72.99
	rural	2.82	5.43
	urban	7.21	13.87
	clouds	4.0	7.71
2000	vegetated	27.40	52.70
	rural	3.9	7.5
	urban	3.26	6.27
	clouds	17.44	33.53

Figure 4.18: Areal Extent of Terrestrial Categories for St. John

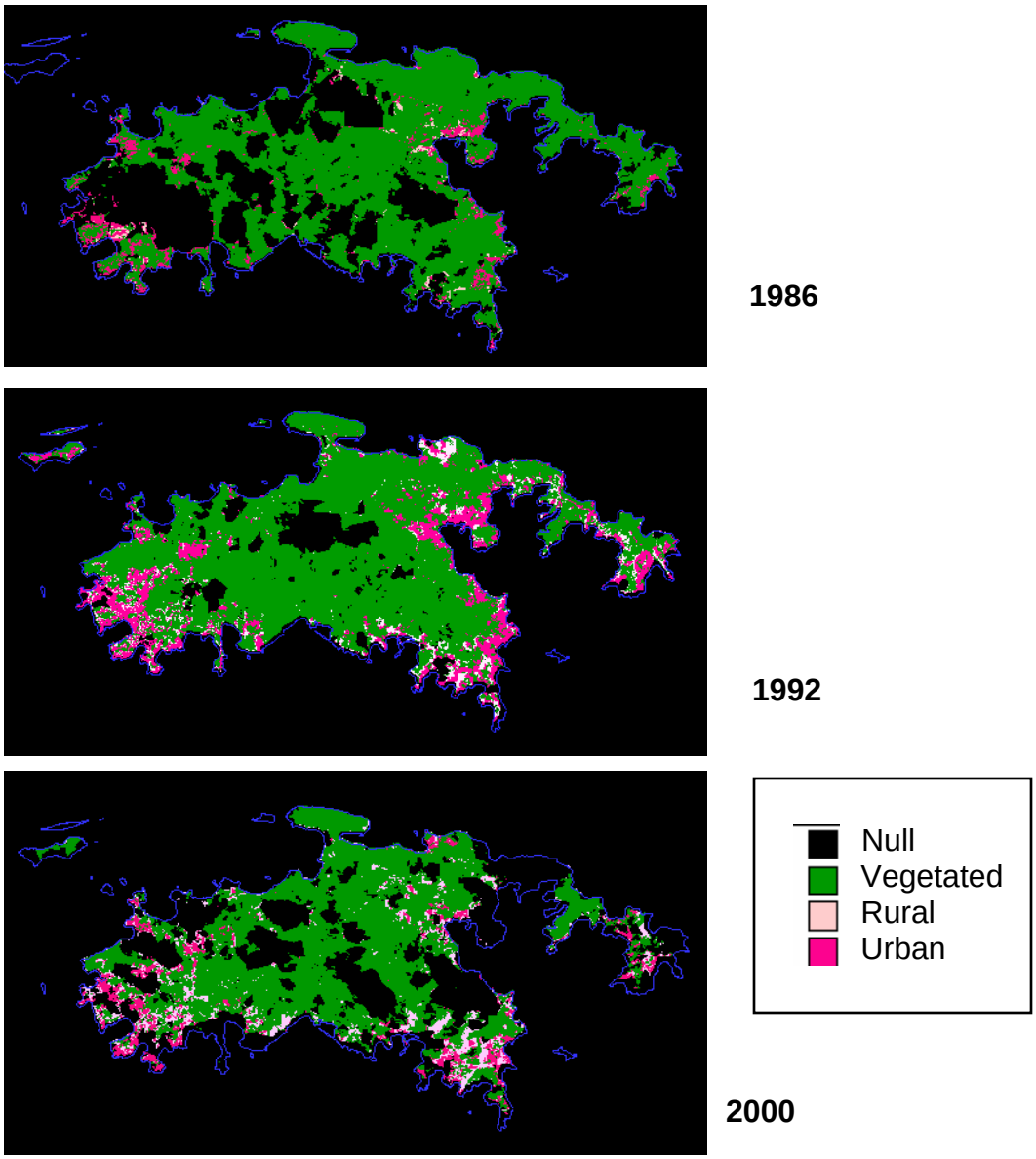

Figure 4.19: Unsupervised Classification of the Terrestrial Portion of St. John

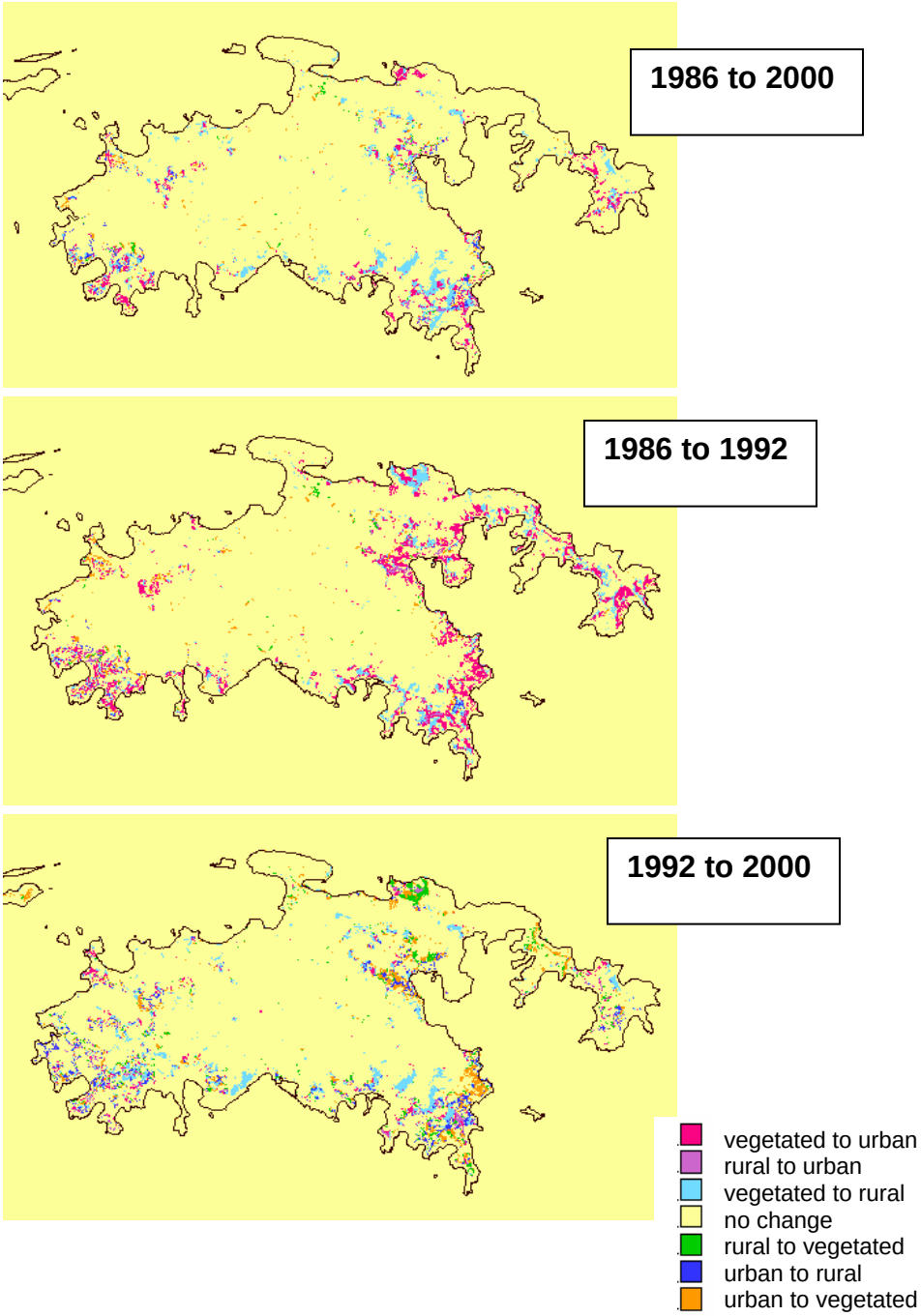

Figure 4.20: Postclassification change for St. John

	Vegetated to Urban km^2 (%)	Vegetated to Rural km^2 (%)	Rural to Urban km^2 (%)	No Change km^2 (%)	Urban to Rural km^2 (%)	Rural to Vegetated km^2 (%)	Urban to vegetated km^2 (%)	Clouds km^2 (%)
1986 to 2000	1.48 (2.85)	2.23 (4.29)	0.13 (0.26)	18.0 (34.61)	0.39 (0.74)	0.16 (0.3)	0.47 (0.91)	29.14 (56.04)
1986 to 1992	3.78 (7.27)	1.83 (3.52)	0.28 (0.53)	26.0 (50.0)	0.23 (0.44)	0.15 (0.29)	0.68 (1.31)	19.05 (36.64)
1992 to 2000	0.72 (1.38)	2.09 (4.03)	0.47 (0.91)	24.0 (46.15)	1.12 (2.15)	0.94 (1.81)	1.58 (3.04)	21.23 (40.83)

Figure 4.21: Areal Extent of Terrestrial Postclassification Categories for St. John, in km^2 with percentage value in parentheses.

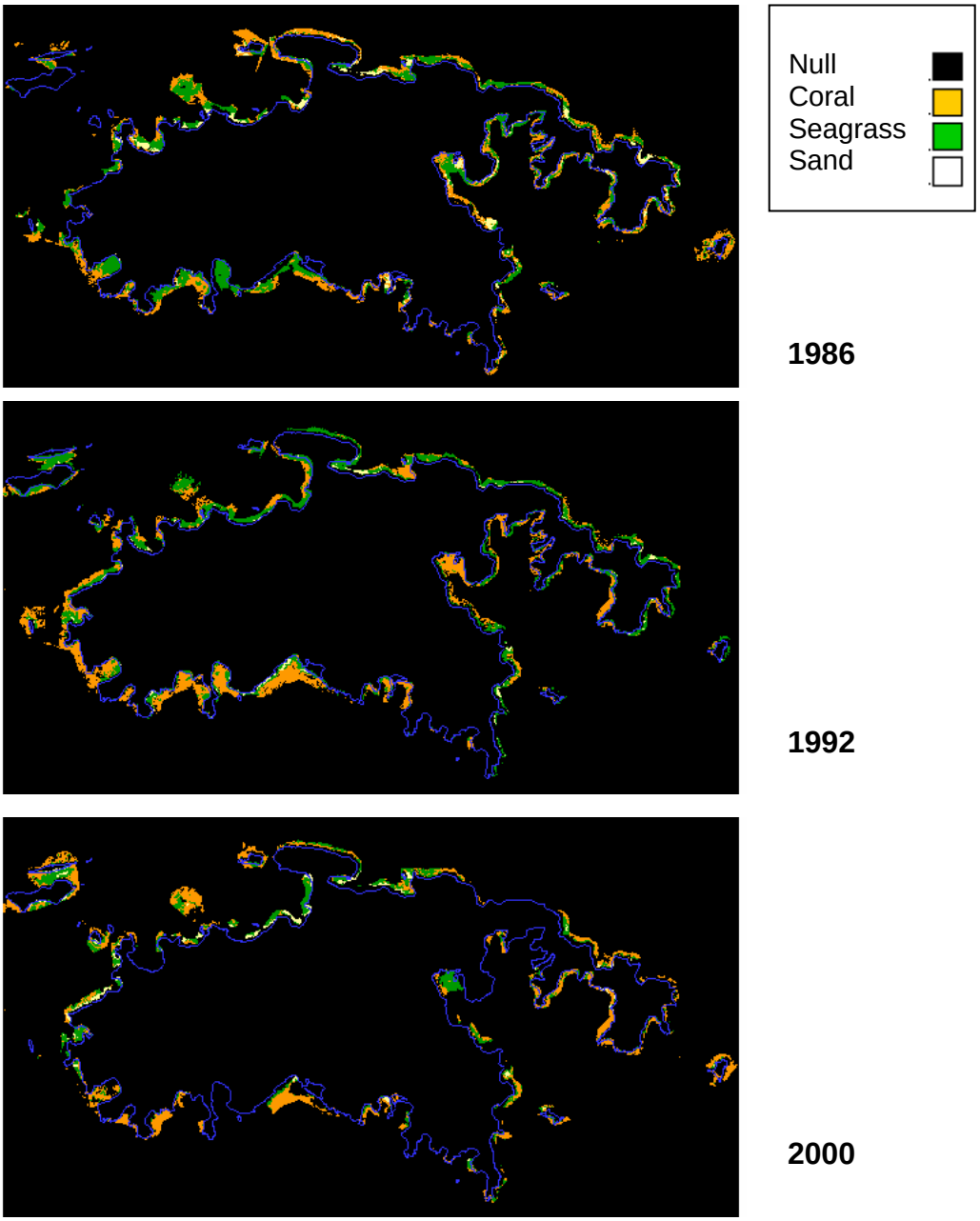

Figure 4.22 : Unsupervised Classification of Benthic Areas of St. John

Total area: 12 km^2			
Yea	**Class**	**Area (km^2)**	**%**
1986	coral	4.52	37.62
	seagrass	4.37	36.38
	sand	0.99	8.25
	clouds	2.13	17.75
1992	coral	4.77	39.70
	seagrass	4.4	36.65
	sand	0.36	2.99
	clouds	2.48	20.66
2000	coral	4.44	36.98
	seagrass	1.9	15.81
	sand	0.62	5.13
	clouds	5.05	42.08

Figure 4.23: Areal extent of Benthic Categories for St. John

	Coral to Sand km^2 (%)	Seagrass to Sand km^2 (%)	Coral to Algae km^2 (%)	No Change km^2 (%)	Seagrass to Coral km^2 (%)	Sand to Seagrass km^2 (%)	Sand to Coral km^2 (%)	Clouds km^2 (%)
1986 to 2000	0.09 (0.73)	0.24 (1.99)	0.51 (4.26)	2.3 (19.16)	1.35 (11.24)	0.12 (1.03)	0.17 (1.38)	7.22 (60.21)
1986 to 1992	0.03 (0.28)	0.04 (0.31)	0.90 (7.52)	3.3 (27.49)	2.04 (16.97)	0.56 (4.66)	0.18 (1.54)	4.95 (41.32)
1992 to 2000	0.09 (0.73)	0.51 (4.21)	0.83 (6.87)	2.5 (20.83)	1.14 (9.46)	0.04 (0.31)	(0.02) (0.13)	6.9 (57.46)

Figure 4.24: Areal extent of Benthic Postclassification Change Classes for St. John in km^2 , with percentage value in parentheses

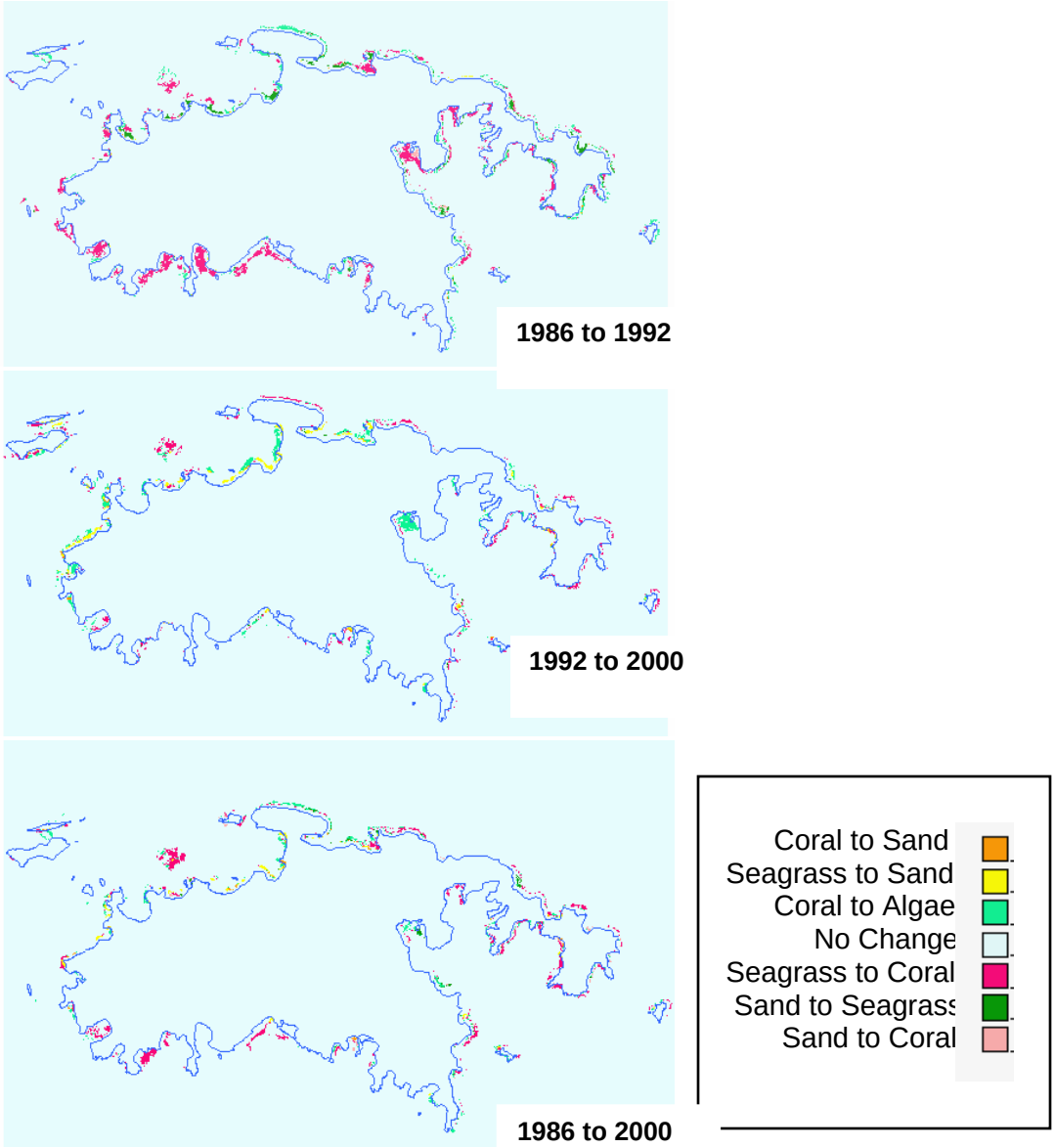

Figure 4.25: Benthic Postclassification Change Maps for St. John

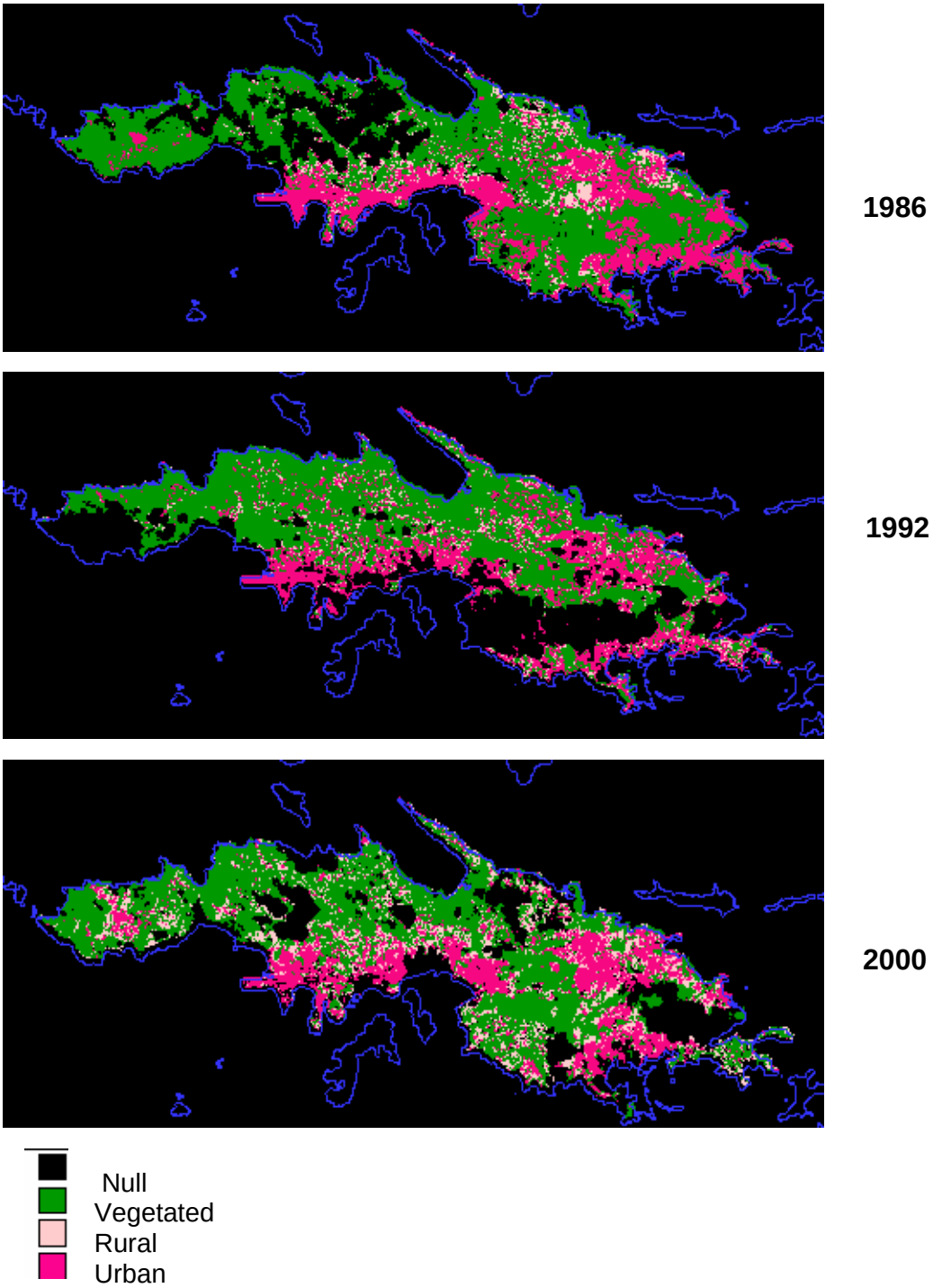

Figure 4.26: Unsupervised Classification of the Terrestrial areas for St. Thomas

Total Area: 73 km^2				
Year	**Class**		**Area (km^2)**	**%**
1986	vegetated		40.61	55.62
	rural		4.86	6.66
	urban		18.31	25.09
	clouds		9.96	13.65
1992	vegetated		36.25	49.66
	rural		3.92	5.37
	urban		16.54	22.31
	clouds		16.29	21.92
2000	vegetated		34.17	46.81
	rural		10.33	14.15
	urban		15.06	20.63
	clouds		13.44	18.41

Figure 4.27 : Areal extent of Terrestrial Categories for St. Thomas

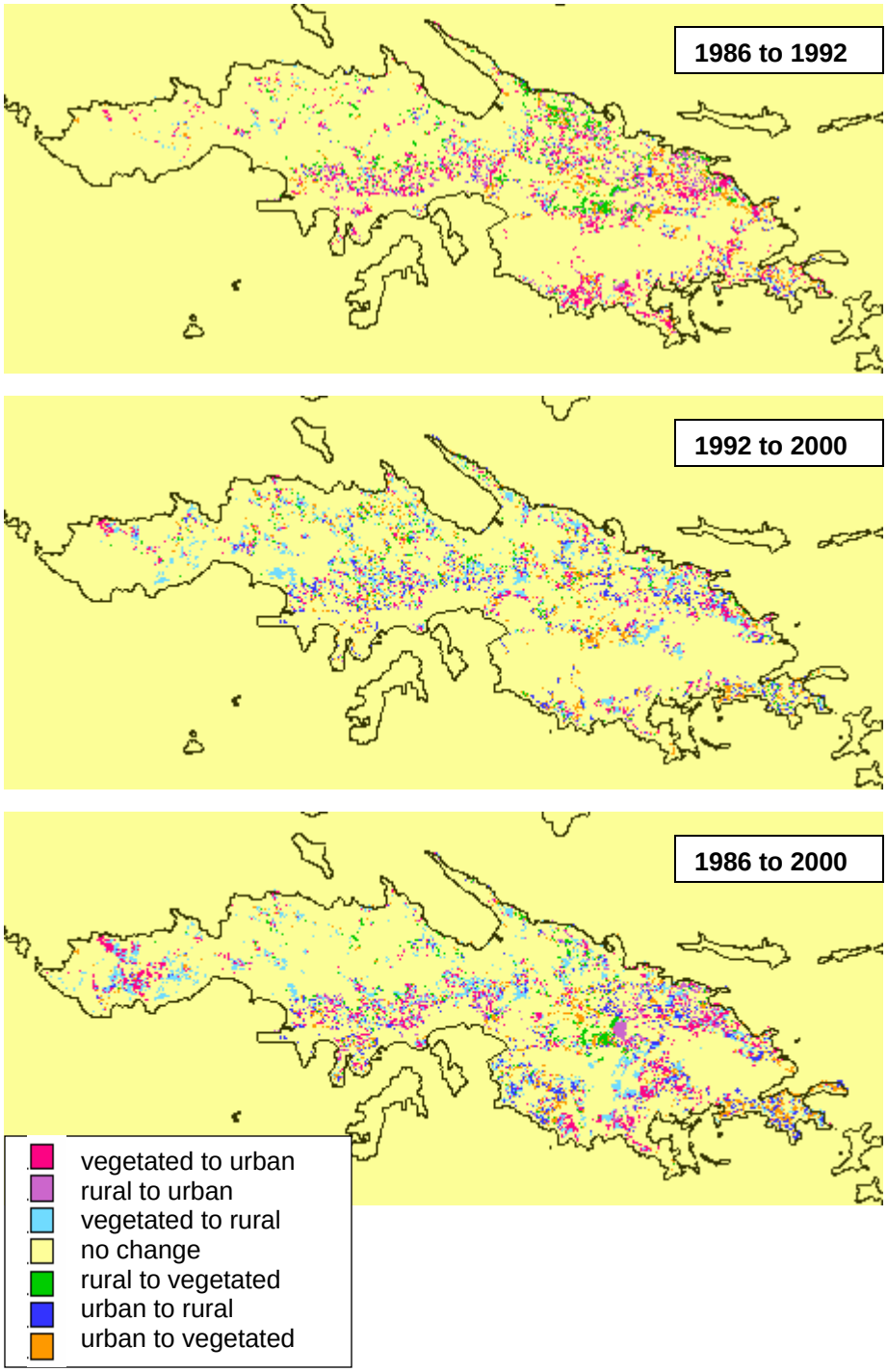

Figure 4.28: Terrestrial Postclassification Change Maps for St. Thomas

	Vegetated to Urban km^2 (%)	Vegetated to Rural km^2 (%)	Rural to Urban km^2 (%)	No Change km^2 (%)	Urban to Rural km^2 (%)	Rural to Vegetated km^2 (%)	Urban to Vegetated km^2 (%)	Clouds km^2 (%)
986 to 2000	3.87 (5.31)	.07 6.94)	.73 2.37)	33 (45.21)	2.62 (3.58)	1.30 (1.78)	2.59 (3.54)	23.82 (31.27)
986 to 1992	.26 5.84)	1.6 (2.21)	.95 (2.67)	32 (43.84)	1.02 (1.40)	1.85 (2.53)	2.14 (2.93)	0.3 (1.51)
992 to 2000	2.17 (2.97)	4.26 (5.84)	0.94 (1.28)	31 (42.47)	2.61 (3.58)	1.27 (1.73)	2.69 (3.69)	8.06 (38.44)

Figure 4.29: Areal extent of Terrestrial Postclassification Change Classes for St. Thomas, in km^2 , with percentage value in parentheses.

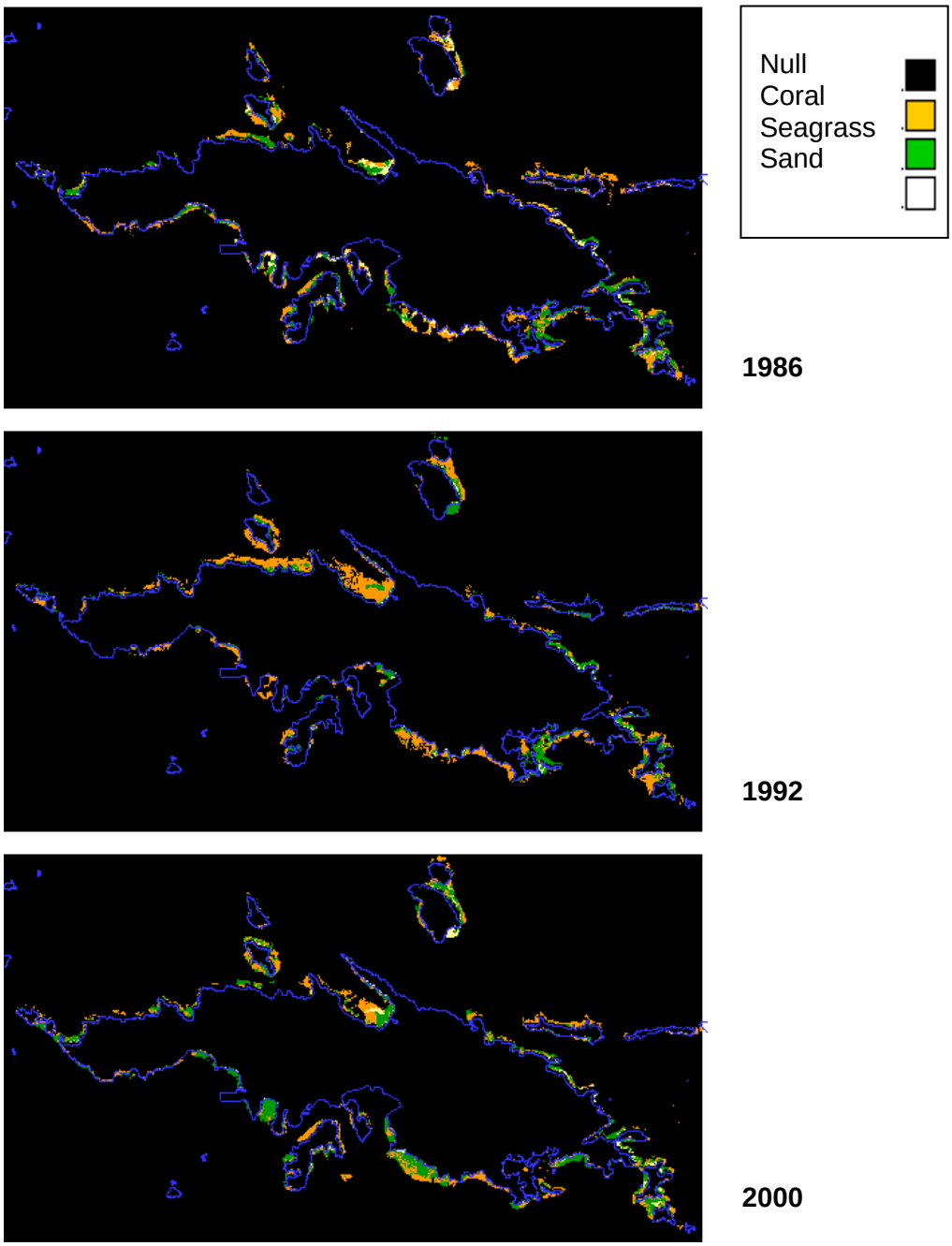

Figure 4.30: Unsupervised Classification of Benthic Areas for St. Thomas

Total area: 17 km^2				
	class		**km^2**	**%**
1986	Coral		7.4	43.61
	seagrass		4.97	29.2
	sand		1.87	10.97
	Clouds		2.76	16.22
1992	coral		8.10	47.64
	Seagrass		4.71	27.71
	sand		0.42	2.48
	Clouds		3.77	22.17
2000	Coral		8.84	51.99
	Seagrass		4.37	25.69
	sand		1.21	7.13
	Clouds		2.58	15.19

Figure 4.31 : Areal extent of Benthic Categories for St. Thomas

	Coral to Sand km^2 (%)	Seagrass to Sand km^2 (%)	Coral to Algae km^2 (%)	No Change km^2 (%)	Seagrass to Coral km^2 (%)	Sand to Seagrass km^2 (%)	Sand to Coral km^2 (%)	Clouds km^2 (%)
1986 to 2000	0.28 (1.65)	0.42 22.48)	1.32 (7.79)	4.5 (26.46)	2 (11.75)	0.24 (1.41)	0.89 (5.21)	7.35 (43.25)
1986 to 1992	0.03 (0.20)	0.05 (0.29)	1.09 (6.39)	4.4 (25.87)	1.65 (9.72)	0.54 (3.16)	0.83 (4.89)	8.5 (49.48)
1992 to 2000	0.18 (1.07)	0.85 (5.0)	1.78 (10.45)	5 (29.4)	1.1 (6.46)	0.06 (0.35)	0.04 (0.21)	8 (47.06)

Figure 4.32: Areal Extent and Percentage of Benthic Postclassification Change Categories for St. Thomas

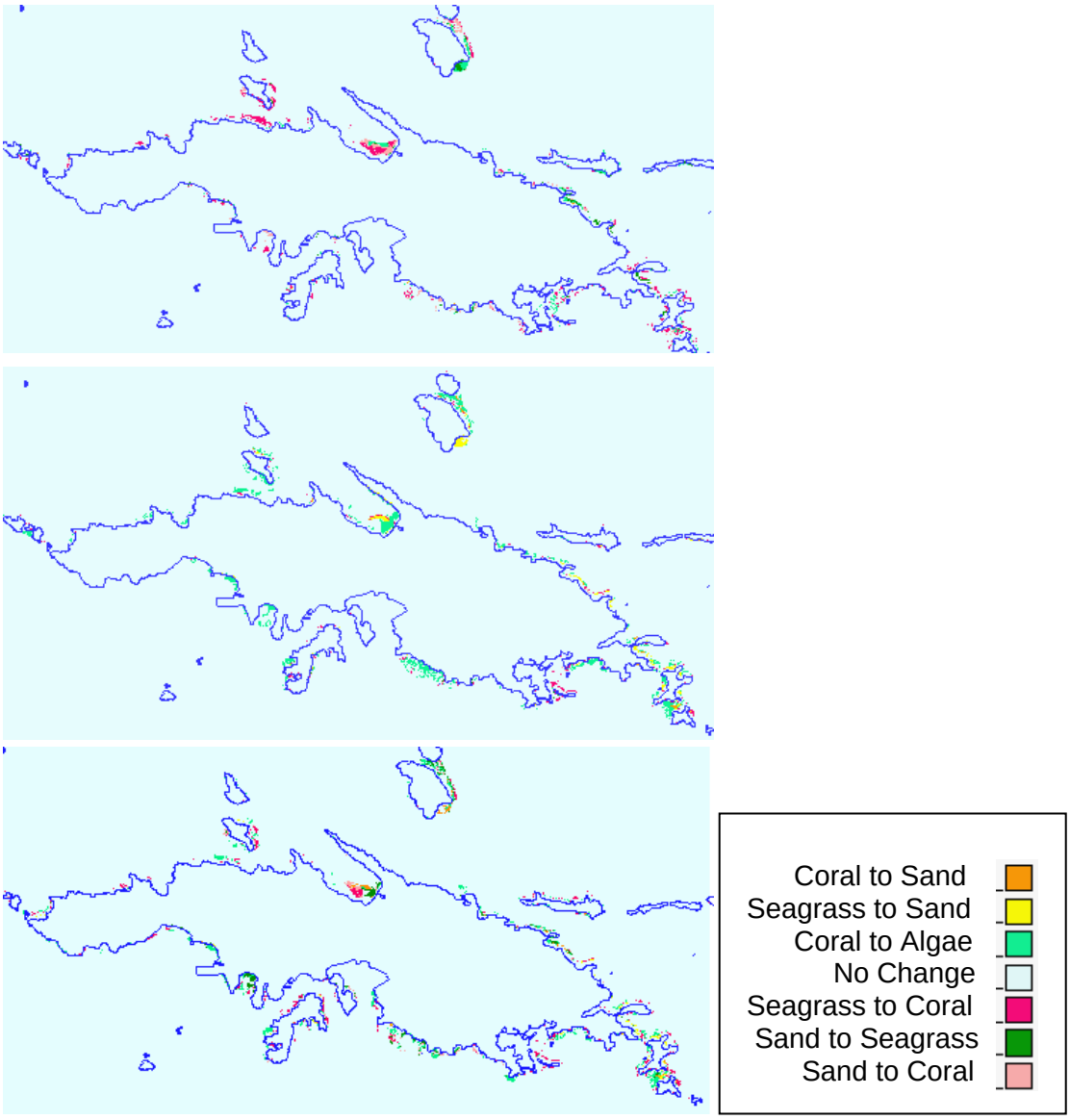

Figure 4.33: Benthic Postclassification for Categories for St. Thomas

Error Matrix of the Classification Map for the 1985 Landsat Scene of St. Croix Terrestrial Habitat

	Reference Data			
Classified Data	Vegetated	Rural	Urban	Totals
Vegetated	**22**	1	1	24
Rural	0	**7**	0	7
Urban	0	4	**30**	34
Totals	22	12	31	**65**

Overall Accuracy: 90.77%

Class	Producers Accuracy (%) (omission error)	Users Accuracy (%) (commission error)
Vegetated	100.0 (0)	91.67 (8.33)
Rural	58.33 (41.67)	100.0 (0)
Urban	96.77 (3.23)	88.24 (11.76)

Overall Kappa Statistic: 0.85

Figure 4.34: Error Matrix and Accuracy of the Classification Map for the 1985 Landsat Scene of St. Croix Terrestrial Habitat

Error Matrix of the Classification Map for the 1992 Landsat Scene of St. Croix Terrestrial Habitat

Overall Accuracy: 90.77%

	Reference Data				
Classified Data	Null	Vegetated	Rural	Urban	Totals
Vegetated	0	**12**	0	0	12
Rural	2	0	**21**	0	23
Urban	2	0	2	**26**	30
Totals	4	12	23	26	**65**

Overall Kappa Statistic: 0.86

Class	Producers Accuracy (%) (omission errors)	Users Accuracy (%) (commission errors)
Vegetated	100 (0)	100.0 (0)
Rural	91.3 (8.7)	91.3 (8.7)
Urban	100.0 (0)	86.66 (13.34)

Figure 4.35: Error Matrix and Accuracy of the Classification Map for the 1992 Landsat Scene of St. Croix Terrestrial Habitat

Error Matrix of the Classification Map for the 1999 Landsat Scene of St. Croix Terrestrial Habitat

Overall Accuracy: 91.67%

	Reference Data				
Classified Data	Null	Vegetated	Rural	Urban	Totals
Vegetated	1	**76**	7	1	85
Rural	0	0	**56**	0	56
Urban	1	1	6	**55**	63
Totals	2	77	69	56	**204**

Overall Kappa Statistic: 0.88

Class	Producers Accuracy (%) (Omission Errors)	Users Accuracy (%) (Commission Errors)
Vegetated	98.70 (1.3)	89.41 (10.59)
Rural	81.16 (18.84)	100.0 (0)
Urban	98.21 (1.79)	87.30 (12.7)

Figure 4.36: Error Matrix and Accuracy of the Classification Map for the 1999 Landsat Scene of St. Croix Terrestrial Habitat

Error Matrix of the Classification Map for the 1999 Landsat Scene of St. Croix Benthic Habitat

	Reference Data			
Classified Data	Coral	Seagrass	Sand	Totals
Coral	**29**	1	0	30
Seagrass	3	**16**	0	19
Sand	1	1	**25**	27
Totals	33	18	25	**76**

Overall Accuracy: 92.11%

Class	Producers Accuracy (%) (Omission Error)	Users Accuracy (%) (Commission Error)
Coral	87.88 (12.12)	96.67 (3.33)
Seagrass	88.89 (11.11)	84.21 (15.79)
Sand	100.0 (0)	92.59 (7.41)

Overall Kappa Statistic: 0.88

Figure 4.37: Error Matrix and Accuracy of the Classification Map for the 1999 Landsat Scene of St. Croix Benthic Habitat

Error Matrix of the Classification Map for the 1999 Landsat Scene with St. Thomas and St. John Terrestrial Habitat

	Reference Data			
Classified Data	Vegetated	Rural	Urban	Totals
Vegetated	**14**	1	0	15
Rural	0	**14**	2	16
Urban	0	0	**19**	19
Totals	14	15	21	**50**

Overall Accuracy: 94.0%

Class	Producers Accuracy (95% Confidence Interval)	Users Accuracy (95% Confidence Interval)
Vegetated	100 % (0)	93.33% (6.67)
Rural	93.3% (6.7)	87.50% (12.5)
Urban	90.48% (9.52)	100 % (0)

Overall Kappa Statistic: 0.91%

Figure 4.38: Error Matrix and Accuracy of the Classification Map for the 1999 Landsat Scene with St. Thomas and St. John Terrestrial Habitat

Error Matrix of the Classification Map for the 1999 Landsat Scene with St. Thomas and St. John Benthic Habitat

	Reference Data			
Classified Data	Coral	Seagrass	Sand	Totals
Coral	**21**	2	0	23
Seagrass	2	**10**	1	13
Sand	1	0	**13**	14
Totals	24	12	14	**50**

Overall Accuracy 88.0%

Class	Producers Accuracy (Omission Error)	Users Accuracy (Commission Error)
Coral	87.50% (12.5)	91.30% (8.7)
Seagrass	83.33% (16.67)	76.92% (23.08)
Sand	92.86% (7.14)	92.86% (7.14)

Overall Kappa Statistic: 0.81%

Figure 4.39: Error Matrix and Accuracy of the Classification Map for the 1999 Landsat Scene with St. Thomas and St. John Benthic Habitat

CHAPTER 5

DISCUSSION

Introduction

Due to significant differences in development trends for St. Thomas, St. John, and St. Croix, the data results for each island will initially be discussed separately. A comparison will be made of the data results for the three islands, each of which possesses unique characteristics relevant to the impact of terrestrial change on benthic habitats. It will then be determined whether the data results indicate that terrestrial changes on the three islands may have impacted changes in their respective benthic habitats.

Thematic maps and postclassification change maps were created for St. Croix, St. John, and St. Thomas. To reduce the amount of cloud cover for the Landsat imagery, composites were formed for all three islands from the 1985 and 1987 unsupervised classification results and from the 1999 and 2001 unsupervised classification results. To create the 1986 composite, the non-cloud pixels from the 1987 map were used to replace cloud pixels in the 1985 map.

The 2000 composite was created by replacing cloud pixels from the 2001 map with cloud-free pixels from the 1999 map. While creating composite maps significantly reduced the number of cloud pixels for St. Croix, a substantial amount of cloud cover remained in the St. Thomas and St. John composites. Although this prevented classification of considerable portions of St. Thomas and St. John, it was possible to discern overall trends for both the terrestrial and benthic portions of these islands.

Increased rainfall during the USVI's wet season contributes to an increase in vegetation in both terrestrial and benthic areas. Seasonal variation may contribute to significant differences in soil moisture and vegetation cover between scenes (Mas, 1999), making it difficult to ascertain actual change from change due to seasonal variation. The November 1987, August 1992, and September 1999 scenes were acquired during the USVI's rainy season; the February 1985 and January 2001 scenes were acquired during the dry season. Because the 1986 and 2000 scenes were each composites of one wet and one dry scene, the impact of seasonal variation on change in vegetation may have been reduced.

For the 1992 scene, which was not a composite, increased rainfall during the wet season may have played a significant role in change in land cover. To differentiate actual change from seasonal change, it would have been necessary to compare land cover for two scenes from the same year, one from the wet season and one from the dry season. Because this was not done, it was difficult to differentiate actual change in land cover from change caused by seasonal differences.

For terrestrial habitats of St. Croix, in addition to creating thematic and change maps, an NDVI was calculated. The purpose of the NDVI calculation was to demonstrate its value when used in conjunction with thematic maps.

St. Croix Terrestrial Habitat

The unsupervised classification results for the 1986 and 2000 scenes of St. Croix indicated that from 1986 to 2000, the areal extent of vegetated areas had decreased and the areal extent of rural and urban areas had increased. The vegetated class decreased in size from 144.7 km^2 in 1986 to 93.6 km^2 in 2000 (Figure 5.1). During the same period, the rural class increased in size from 51.99 km^2 to 75.69 km^2 and the urban class increased from 23.31 km^2 to 42.05 km^2. Cloud cover, at 9.71% in the 1986 scene and 11.96% in the 2000 scene, was relatively low. The three largest categories in the 1986 to 2000 postclassification change map were “vegetated to rural” (36.74 km^2), “vegetated to urban” (13.04 km^2), and “rural to urban” (12.22 km^2) (Figure 5.2).

Of the three US Virgin Islands, St. Croix has had the greatest extent of terrain suitable for construction. While most of St. Thomas had been developed by the early 1980s, significant portions of St. Croix remained undeveloped. A significant amount of St. Croix was developed over the study period, as evidenced by the size of the “vegetated to rural” (36.74 km^2) and “vegetated to urban” (13.04 km^2) change categories. In northwest St. Croix, where slopes are steepest, the decrease in vegetation from 1986 to 2000 was small compared to other parts of the island.

Prior to the 1980s, St. Croix had a thriving tourist sector. Hurricane Hugo, a category 4 hurricane, caused extensive damage to St. Croix's terrestrial and coral habitats in 1989, drastically impacting St. Croix's economy. Development would most likely have been greater following 1989 had Hugo not struck the island. Because most land on St. Thomas and St. John has been developed, undeveloped areas on St. Croix are projected to be converted to residential and urban land in the coming decade (Rohring, B., pers comm., 2006).

St. Croix Benthic Habitat

Light Attenuation

The water leaving spectral signature of benthic features is a function of the water column, as well as the ocean floor. As water depth increases, light is exponentially attenuated, obscuring the spectral signature of benthic features. To accurately interpret benthic albedo, benthic habitat characterization was limited to areas less than 9 m in depth.

The spectral signature of benthic features in waters less than 9 m in depth may be obscured by a temporary increase in depth caused by tidal fluctuation. For mapping benthic habitats in most regions, mean low tide or lower is preferred (Klemas, 2001) since the spectral signal of benthic features may be severely distorted during high tide (Dustan et al, 2001). The difference in depth between high and low tide for the Landsat scene dates was less than 0.5 m (NOAA National Center for Coastal Ocean Science, 2006), indicating that the spectral signatures of the benthic features in the unsupervised classification were not obscured by tidal level fluctuation.

As light travels through the water column, it may be reflected, absorbed, or reemitted as fluorescence (Ackelson, 2003) by chlorophyll-a, sediments (Baban, 1997), or other materials. The optical signature of benthic features in the characteristically clear waters of the USVI may be obscured by the addition of sediments or other materials in the water column.

To correctly interpret change in benthic habitats, it was necessary to account for factors that may have obscured the optical signature of benthic features and resulted in misclassification. Higher than average wind velocity may resuspend benthic sediments and cause turbidity. The mean wind speed for St. Croix for the five dates was between 3.5 to 5.5 m/sec (NOAA National Oceanographic Data Center, 2006), indicating that the water column clarity had not been impacted by resuspension of benthic sediments.

Seasonal Variation

Higher amounts of rainfall during the Caribbean's wet season contribute to a greater level of land-based sediment and nutrient input to coastal waters. In watersheds that do not have excessive amounts of soil erosion, maximum seagrass volume and algal growth are associated with the rainy season, which occurs from September through November (Daby et al., 2003). Because the 1986 and 2000 scenes were each composites of one wet and one dry scene, the portion of change that resulted from increased rainfall could not be determined. The 1992 scene, not a composite, was acquired during the USVI's wet season. An increase in terrestrial vegetation could be presumed to result from the greater rainfall at this time of year.

A portion of the increase in benthic vegetation from 1986 to 1992 may also be attributed to the higher level of rainfall during the wet season, which would cause a greater amount of nutrient input from terrestrial to benthic areas. However, because no maps were created for the dry season for 1992, it was not possible to ascertain the portion of change in terrestrial or benthic vegetation that was seasonal.

Algal Growth

Although spectral similarity prevents clear differentiation of algae from seagrass in Landsat images (Armstrong, 1993), their unique morphologies may be used to distinguish one from the other. While seagrasses are vascular rooted plants that require a sand substrate for growth, algae are simple-celled organisms capable of growing on most surfaces, including dead coral. Seagrasses are not able to grow on hard substrate such as coral. Therefore, it may be surmised that a change from coral cover to benthic vegetation indicates a change from live coral to algae-covered dead coral.

Algae and seagrass may also be differentiated based on changes in spatial patterns. The change from live coral to dead hard bottom dominated by algae alters both spatial patterning and variability of pixel brightness (Dustan et al., 2001). An increase in the pixel heterogeneity of the benthic portions of St. Croix suggested a greater amount of patchiness in seagrass or coral cover, as live coral was replaced by dead coral and algae. Because information from

Landsat alone cannot be used to differentiate seagrass from algae, the "seagrass" category in the unsupervised classification includes both seagrass and algae.

The unsupervised classification results of St. Croix's benthic habitats indicated an increase in benthic vegetation from 25.21 km^2 in 1986 to 38.08 km^2 in 2000 (Figure 5.3). The postclassification change map for the 1986 to 2000 period showed that 15.63 km^2 (Figure 5.4) of benthic areas had changed from coral to algae, indicating that the increase in benthic vegetation was more a result of algal than seagrass growth. The "coral to algae" category was the largest category of change for all three benthic change maps. The increase in algal cover may have resulted from sediment and nutrient input from land-based runoff, from a decrease in the number or algal grazing organisms, or from damage caused by hurricanes or recreational sports such as boating and diving.

The second largest benthic change category for the three change maps for St. Croix was "seagrass to coral". The "seagrass to coral" category indicates an increase in benthic albedo or brightness, which over the short time period for the study (1985 to 2001) cannot be attributed to replacement of seagrass by live coral. The increase in albedo is more likely the result of a decrease in seagrass density. The decrease in seagrass density would leave a greater amount of carbonate sand exposed to sunlight, increasing the amount of albedo recorded by the satellite sensor. Decease in seagrass density may be attributed to damage from hurricanes, boat propellers and anchors, or from smothering by excessive sediments.

For the 1986 to 2000 period, while 7.54 km^2 of land changed from seagrass to coral, 15.63 km^2 changed from coral to algae (Figure 5.4). The "coral to algae" class was more than twice the size of the "seagrass to coral" class, indicating coral had been impacted more than seagrass over the study period. These findings are supported by studies that indicate corals are more sensitive to decreases in light and an increase in nutrients than are seagrasses. While *Thalassia testudinum* has a high tolerance for low light conditions and can survive for up to five months in light conditions below their minimum light requirement (Lee and Dunton, 1997), corals require clear, oligotrophic waters to develop and function.

The RGB composite of blue water reflectance channels for the 1985, 1992, and 1999 Landsat scenes (Figure 4. 17) also indicated that benthic vegetation had increased in southwest St. Croix, and in areas north of St. Croix, near both Buck Island and Christiansted (Figure 5.5).

The unsupervised classification of the five pre-composite Landsat scenes of St. Croix (1985, 1987, 1992, 1999, and 2001) indicated that the areal extent of benthic vegetation was greatest in the 2001 scene. The 2001 scene was taken during the dry season, suggesting that the increase in algal cover for this scene was large enough to offset any decrease in benthic vegetation that may have resulted from lower amounts of rainfall during the dry season.

These findings are supported by in-situ studies in the USVI that indicate a decrease in the areal extent of living coral and an increase in macroalgae cover (Rogers and Beets, 2001) over the past two decades. Patterns of reef

reflectance from TM data of the Florida Keys suggested that the coral reef substrate had become greener over time and that reef albedo had decreased (Dustan et al., 2001). Such shifts, which were also observed in the Landsat scenes of the USVI, are consistent with the observation that algal populations are increasing in size.

A shift from live coral to dead, algae-dominated coral may be due to a number of factors, including a decrease in the number of fish, black sea urchins, or others herbivorous organisms that feed primarily on algae. Rogers and Miller (2001) found that the algal turfs characteristic of many Caribbean reef habitats appear to represent herbivore-tolerant assemblages, persisting under high grazing intensity and growing rapidly in response to a decrease in the number of herbivorous fish in tropical reef habitats. Gardner et al. (2003) and Dustan et al. (2001) found that algal populations flourished and overgrew much of the Caribbean benthic habitat, resulting in a significant decline in coral cover.

Storms, which may significantly alter benthic habitats, are more likely to occur in the rainy season. Daby (2003) noted that the threat to Caribbean coral reefs was heightened during the rainy season, when hurricanes or floods are likely to occur. Rogers and Beets (2001) found that while algal growth has increased in the USVI, seagrass densities have decreased due to damage from storms. In 1989, Hugo, a category 4 hurricane, was the largest to impact the Virgin Islands in over a century.

Hugo caused extensive damage to both the terrestrial and benthic portions of St. Croix. The Tetratech Group (1991) found that in the St. Croix Salt

River Canyon (Figure 5.5), *Halophila* seagrass cover was reduced as a result of storm-induced turbidity. The large amount of cloud cover, difference in Landsat scene dates from the dates for in-situ studies, and the spatial resolution of Landsat TM data prevented ascertaining whether these specific changes were visible in the classified maps.

Patriquin (1975) noted that seagrass blow-outs, which result from storm waves that uproot seagrass beds, may be over 30 m in size in the USVI. Eroded areas are more vulnerable to blowouts, which migrate seaward over time in shallow areas of intense wave action. Change in the areal extent of seagrass cover may also result from seagrass migration, which involves the generation of vast drifting mats of seagrasses. These migrations may be observed in nearshore and offshore waters throughout the Gulf of Mexico and the Caribbean, as noted by Fonseca and Kenworthy (1987).

Benthic ecosystems have also been heavily impacted by recreational activities, such as boating and diving. Damage to coral and seagrass ecosystems has been caused by boat anchors, water crafts running aground into coral and seagrass ecosystems, and breakage of corals by divers.

St. Thomas and St. John

Data results for St. Thomas and St. John, which are located on one Landsat scene, were less definitive than were the results for St. Croix. Clouds covered 33.53% of the terrestrial portion and 42.08% of the benthic portion of the 2000 composite scene for St. Croix. For St. Thomas, while clouds covered less

than 22% of the terrestrial portions of the unsupervised classifications, they covered 41.51% of the terrestrial change map and 43.25% of the benthic change map for the 1986 to 2000.

The portion of St. Thomas and St. John covered by clouds may be higher than that for St. Croix because St. Thomas and St. John receive higher rainfall and are smaller in size than St. Croix. Despite the large percentage of cloud pixels over the St. Thomas and St. John Landsat scene, it was possible to ascertain the types of change that occurred during the 1986-2000 period.

St. John Terrestrial Habitat

Data results for St. John's terrestrial areas revealed an increase in rural and urban cover and a decrease in vegetation for the 1986 to 2000 period. From 1986 to 2000, vegetated land cover decreased from 33.36 km^2 to 27.4 km^2, urban land increased from 2.58 km^2 to 3.26 km^2 and rural land increased form 0.53 km^2 to 3.9 km^2. (Figure 5.6) Over 60% of St. John has been designated a national park since the 1960s. Though development is not permitted within most of the park bounds, some private land ownership is permitted within the park. As expected, the greatest change from "vegetated to rural" or "vegetated to urban" occurred outside the St. John National Park boundaries. The 2000 Landsat scene shows that within park boundaries, urban and rural land cover was small in comparison to areas outside park boundaries (Figure 5.7).

From 1986 to 2000, the amount of land that changed from vegetated to urban was 1.48 km^2, while the amount of land that changed from vegetated to rural was 2.23 $km^{2,}$ (Figure 5.8). This suggests that the change from vegetated

to developed land was due less to intensive urban development and more due to rural development, such as the construction of homes in rural areas. These results are supported by the fact that of the limited terrestrial development allowed on St. John, most consists of construction of homes in rural areas.

Incongruous results, such as the increase in vegetation from 1986 to 1992 (33.36 km^2 to 37.95 km^2) and decrease in urban land from 1992 to 2000 (7.21 km^2 to 3.26 km^2) are most likely due to the high percentage of cloud-free pixels in both the 1986 and 2000 scenes, which prevented mapping 30% to 35% of the terrestrial habitat for those years.

St. John Benthic Habitat

Wind speed for St. Thomas and St. John at the water surface during the dates of Landsat scene acquisition was between 1.5 and 2.5 m/sec (NOAA National Oceanographic Data Center, 2006), indicating that resuspension of benthic sediments did not obscure benthic spectral signatures. The difference in water depth at high versus low tide was less than 1 ft (3 m) (NOAA NCCOS, 2006), signifying that the spectral signatures of the benthic features in the unsupervised classification were not obscured by tidal level fluctuation.

Over the 1986 to 2000 period, little change was observed for the coral class, which decreased from 4.52 km^2 in 1986 to 4.44 km^2 in 2000 (Figure 5.9). The seagrass category decreased significantly, from 4.37 km^2 in 1986 to 1.9 km^2 in 2000. These results may be attributed to the significant cloud cover over the 2000 scene (42.08%) as well as to the small portion of benthic habitat (12 km^2) that could potentially be mapped with Landsat.

The “seagrass to coral” class, which indicated a decrease in seagrass density, was the largest change category (in all the postclassification change maps (Figure 5.10). The “coral to algae” class, which indicated a shift from live coral to dead, algae-dominated coral, was the second largest category of change for all of the change maps (Figure 5.10). The study results, which indicate “seagrass to coral” and “coral to algae” as the largest categories of change for St. John, are supported by in situ studies. Muehelstein (1991) and Short and Wyllie-Echeverria (1996) found that aerial photographs indicated significant seagrass loss on St. Thomas and St. John following Hurricane Hugo. They observed that coral reefs in Great Lameshur Bay (Figure 5.11) that had been impacted by Hugo had not recovered as of 2001.

The postclassification change map (Figure 4.25) for the 1986 to 1992 period, which encompasses the occurrence of Hurricane Hugo, indicated that the largest change category for Lameshur Bay was “seagrass to coral”, which indicates a decrease in seagrass density. Decrease in seagrass cover may occur as a result of extensive storm damage (Daby, 2003; Patrinquin, 1975), as occurred during Hurricane Hugo. The decrease in seagrass density may also result from excessive amounts of land-based run-off. From the 1986 to 1992 period, a significant amount of development occurred on the lands adjacent to Great Lameshur Bay. The soil erosion caused by the construction of roads and homes may have led to a significant input of sediments to Great Lameshur Bay during the 1986 to 1992 period.

The 1992 scene was acquired during St. John's rainy season. The land-based runoff would have been significantly greater during the rainy season (Daby et al., 2003), impacting seagrasses and corals negatively due to a decrease in light availability and possible smothering by sediments. From 1992 to 2000, portions of Lameshur Bay (Figure 4.25) showed a change from coral to algae. This may be the result of land-based runoff from the conversion of vegetated to rural and urban land from 1992 to 2000 (Figure 4.19).

Lenny, a Category 3 hurricane, passed 80 km to the south of St. Thomas and St. John on November 19, 1999. Rogers and Miller (2001) noted that although hurricane damage was conspicuous at Newfound Reef, St. John (Figure 5.11), a change in coral cover did not appear to be significant. Newfound Reef is located outside the VI National Park bounds. The unsupervised classification of the benthic portion of St. John (Figure 4.22) indicated that from 1986 to 1992, there was a change from sand to seagrass. The increase in seagrass growth indicates a healthy benthic environment. This observation is supported by the fact that the land adjacent to Newfound Reef remained undeveloped from 1986 to 1992 (Figure 4.19).

Haulover Reef (Figure 5.11), located off St. John at the northeastern boundary of Virgin Islands National Park, is composed of dense coral cover. Miller et al. (2003) noted an outbreak of plague type II coral disease at this reef between 1997 and 2001. The data results indicate that from 1992 to 2000 and from 1986 to 2000, the largest category of change at Haulover Reef is "seagrass to coral", which indicates a decrease in seagrass density. Lands adjacent to this

reef changed from vegetated to urban from 1986 to 2000 (Figure 4.19). It appears that the decline in seagrass density may result from a combination of the impacts of land-based runoff and the plague type II outbreak.

St. Thomas Terrestrial Habitat

St. Thomas, one of the Caribbean's major centers for tourism, has experienced the impacts of the growing tourism industry both on the island and in its benthic habitats. The study results indicated that little land changed from vegetated to developed over the study period. These findings were supported by the fact that most land that was not too steep for construction of roads and homes had been developed prior to the years of the study period.

While rural land cover increased from 4.9 km^2 in 1986 to 10.3 km^2 in 2000, vegetated land cover decreased from 40.6 km^2 to 34.2 km^2 (Figure 5.12). The decrease in urban land from 18.3 km^2 in 1986 to 15.1 km^2 in 2000 may be attributed to clouds, which covered 18.41% of the terrestrial portion of St. Thomas in 2000. In 2000, the portion of land on St. Thomas that remained undeveloped (34.17 km^2) consisted of terrain too steep for construction activity.

The largest change categories in the 1986 to 2000 change map were the "vegetated to urban" class (3.87 km^2) and the "vegetated to rural" class (5.07 km^2) (Figure 5.13). While "vegetated to urban" (4.26 km^2) was the largest change category for the 1986 to 1992 period, "vegetated to rural" (4.26 km^2) was the largest change category for the 1992 to 2000 period. This finding is supported by the development trends for St. Thomas, which indicate that due to the extensive pressure for development by the tourism sector, the intensity of

development would have been significant throughout the study period. St. Thomas's tourist sector is based on the enormous cruise ships that dock several times a week at St. Thomas's vast harbor in Charlotte Amalie. Although tourism did not decline after 1992, most land near the urban centers had been developed. As a result, more development occurred in rural than in urban areas.

St. Thomas Benthic Habitat

The study results indicate that benthic habitat in St. Thomas did not change significantly over the study period. The coral class changed in size from 7.4 km^2 in 1986 to 8.84 km^2 in 2000. The seagrass category decreased in size from 4.97 km^2 in 1986 to 4.37 km^2 in 2000 (Figure 5.14). The decrease in seagrass density was indicated in the 1986 to 2000 change map by the "seagrass to coral" category, which increased 2.0 km^2. The seagrass to coral category was the largest category of change (Figure 5.15) for the 1986 to 2000 period, as well as for the 1986 to 1992 period (1.65 km^2). "Coral to algae" was the largest change category (1.78 km^2) from 1992 to 2000 and the second largest category for 1986 to 2000 (1.32 km^2) and for 1986 to 1992 (1.09 km^2).

Although the decrease in live coral cover on St. Thomas reefs is due to a combination of natural and anthropogenic factors, the large role played by tourism in the St. Thomas economy would suggest that anthropogenic factors have played a greater role in benthic ecosystem decline for the study period. These factors may include destruction to corals and seagrasses by boat

propellers, anchors, and divers, as well as a decline in coral and seagrass function and health as a result of sediment and nutrient input from land-based runoff.

NDVI Analysis for Terrestrial Vegetation Cover

In addition to performing an unsupervised classification of St. Croix, the extent of vegetation on land was determined with an NDVI analysis. The NDVI indicates the amount of terrestrial vegetation cover at each pixel based on a ratio of the pixel's red and near infrared (NIR) band values. Pixels with high NDVI values indicate densely vegetated land while lower values indicate urban land and other areas with little to no vegetation cover.

The individual (Figure 4.8) and write function (Figure 4.9) maps of the NDVI analyses for the 1986, 1992, and 2000 St. Croix Landsat scenes corroborated the findings in the unsupervised classification and postclassification change maps. For example, areas classified as vegetated in the unsupervised classification coincided with high NDVI values in the NDVI maps. Areas on the postclassification change maps that indicated a change from vegetated to rural or urban coincided with colors in the RGB composite that indicated a shift from high to low NDVI values.

No NDVIs were calculated for St. Thomas and St. John, as the NDVI map for St. Croix demonstrated sufficiently the value of an NDVI analysis in a land use and land cover study.

Summary

For the study period, the overall trend in the terrestrial habitats of St. Croix, St. Thomas, and St. John was a shift from vegetated to developed land. From 1986 to 2000, a significant portion of the benthic habitats of St. Croix and St. Thomas showed a shift from live coral to algae. The presence of algae on corals indicates that the corals had died or were in decline. The decline in live coral cover may result from a number of anthropogenic factors, including an increase in sediments and nutrients from land-based runoff, or destruction by divers or boat anchors and propellers. Though natural factors such as hurricanes may also contribute to a shift from coral to algae, the level of rural and urban development on St. Thomas and St. Croix suggests anthropogenic factors play a more significant role in benthic ecosystem decline than natural factors.

The type of benthic change greatest in magnitude for St. John was a decrease in seagrass density, indicated by the “seagrass to coral” class. The decrease in seagrass density may have resulted from hurricanes. Over the study period, St. John was impacted by several hurricanes, including Hurricane Hugo in 1989, Hurricane Georges in September, 1998, and Hurricane Jose, in October, 1999.

The second largest benthic change category for St. John was “coral to algae”. The fact that this is not the largest change category, as it is for St. Thomas and St. Croix, indicates that St. John’s benthic habitats may have been less impacted by terrigenous input than the other two islands. This is supported

by the fact that terrigenous inputs from land based runoff would have been significantly smaller on St. John, where development is restricted to lands outside the National Park boundaries, than for St. Thomas and St. Croix, where development is allowed to occur in most areas.

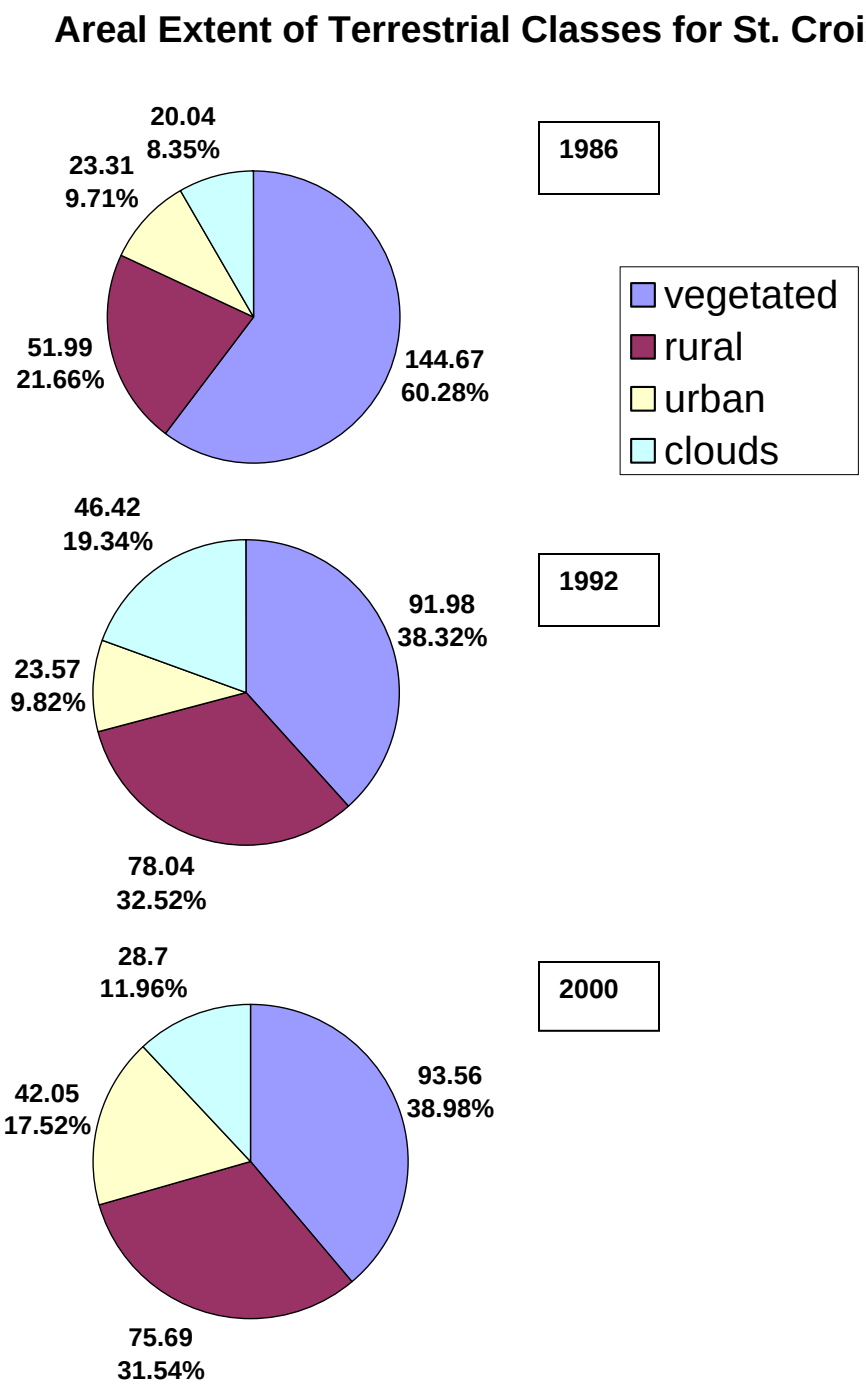

Figure 5.1. Areal Extent of Terrestrial Classes for St. Croix. The value of each pie slice is given in km^2 (above) and percent (below).

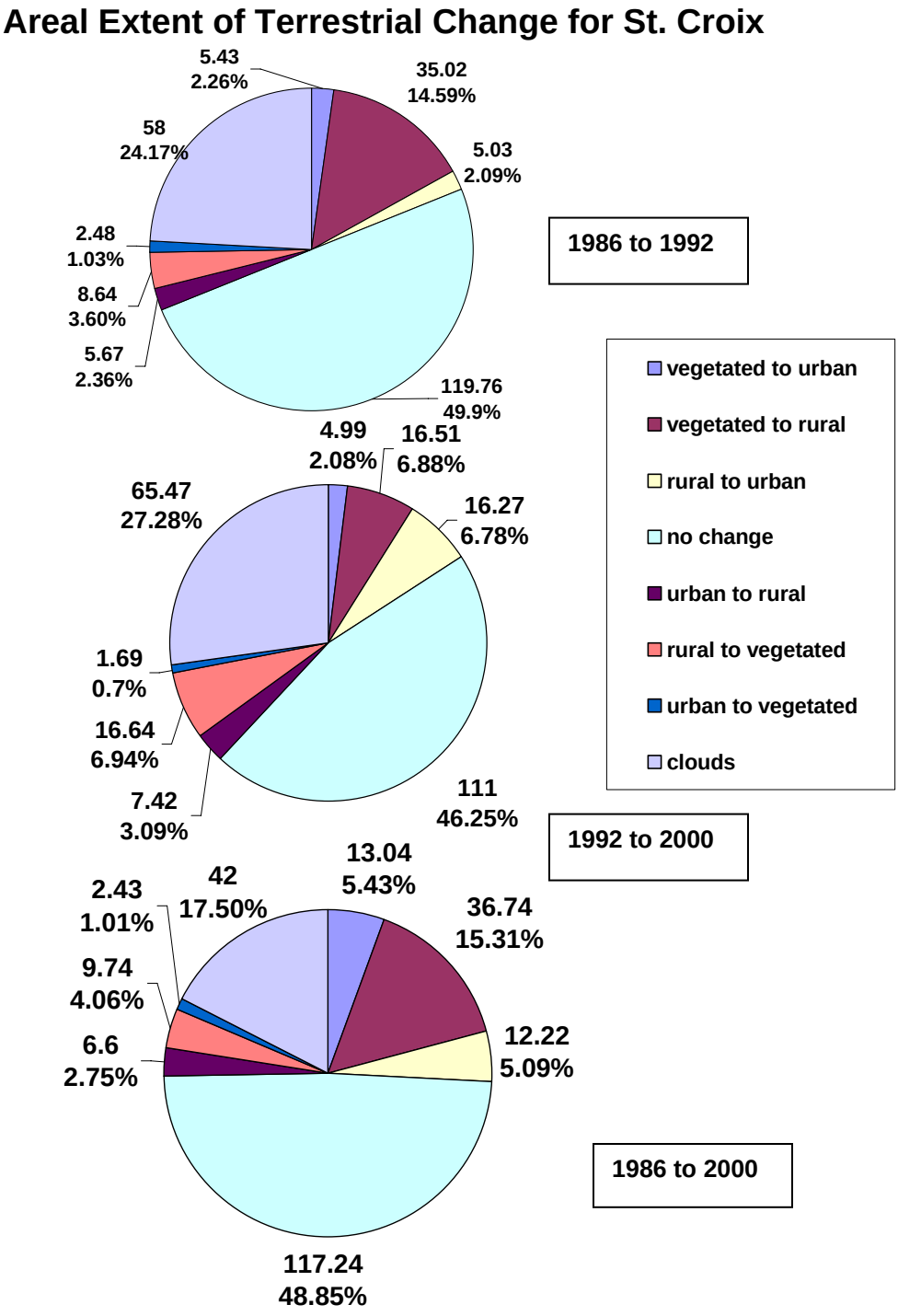

Figure 5.2: Areal Extent of Terrestrial Change for St. Croix. The value of each pie slice is given in km^2 (above) and percent (below).

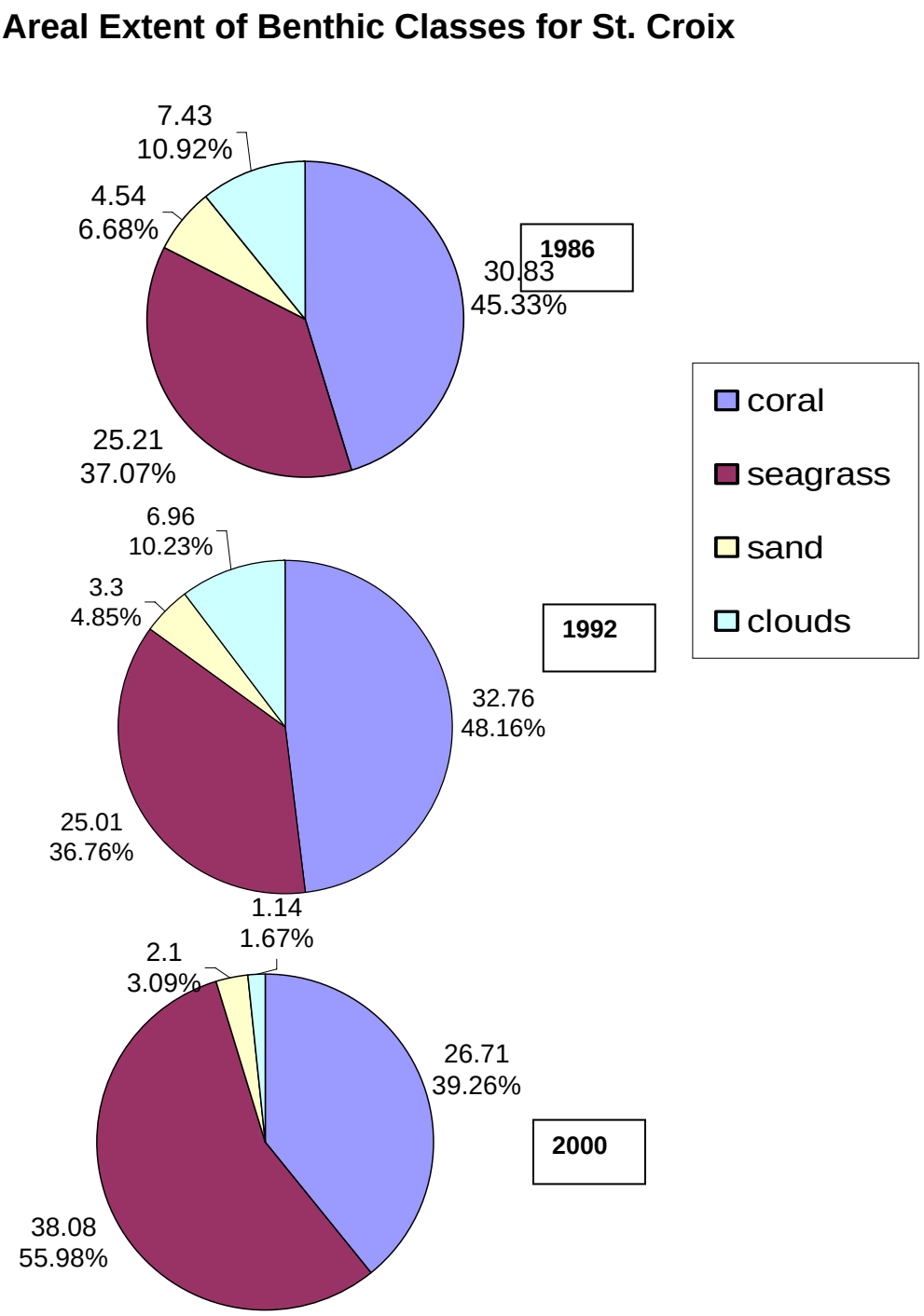

Figure 5.3. Areal Extent of Benthic Classes for St. Croix. The value of each pie slice is given in km^2 (above) and percent (below).

Areal Extent of Benthic Change for St. Croix

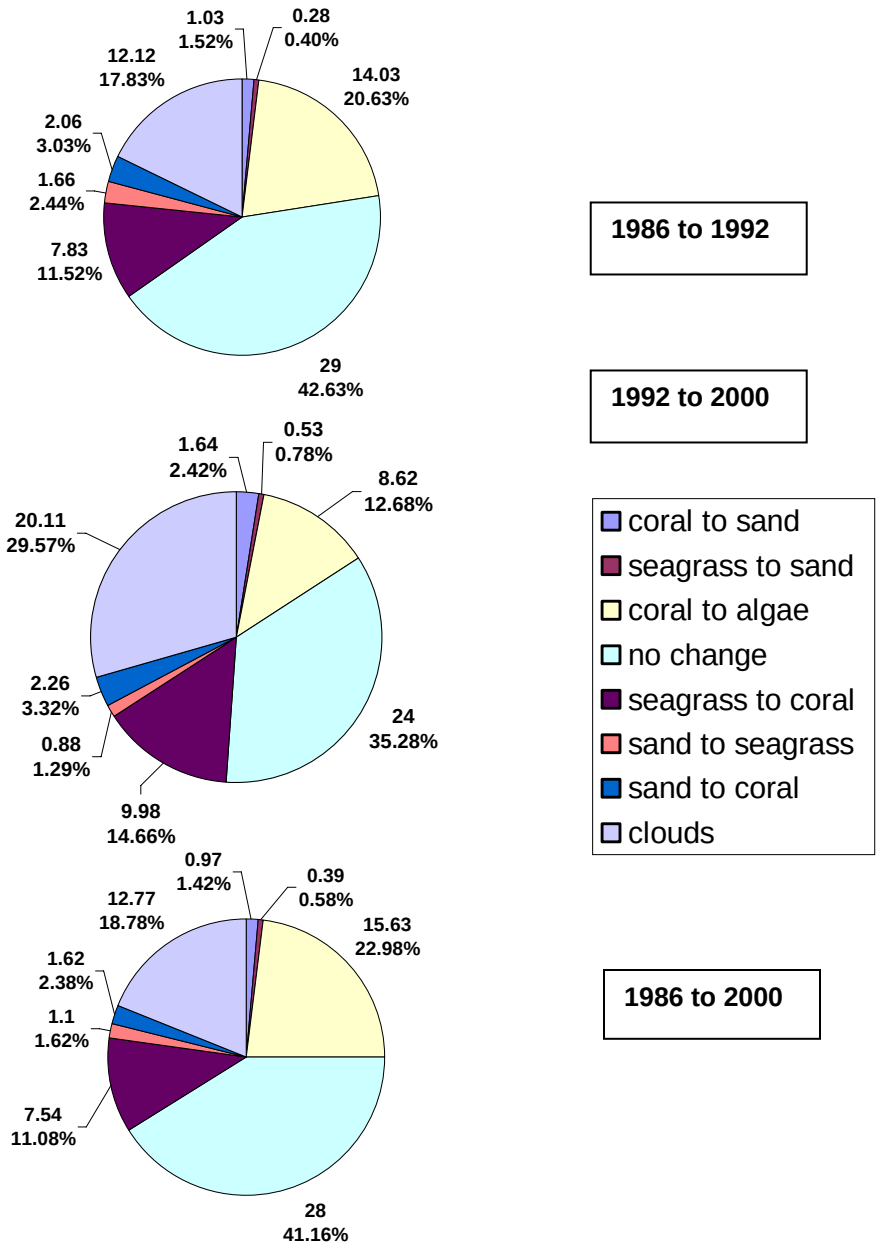

Figure 5.4 . Areal Extent of Benthic Change for St. Croix. The value of each pie slice is given in km^2 (above) and percent (below).

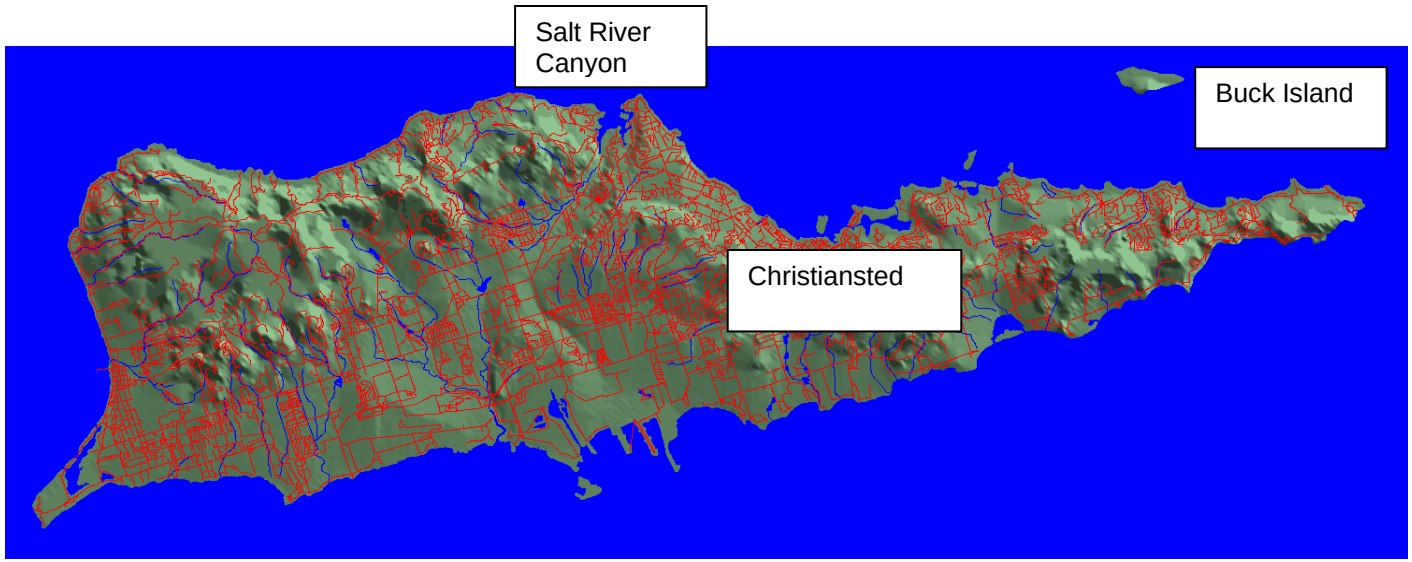

Figure 5.5: Areas on St. Croix that were mentioned in text. (UVI Caribbean Data Center, 2006)

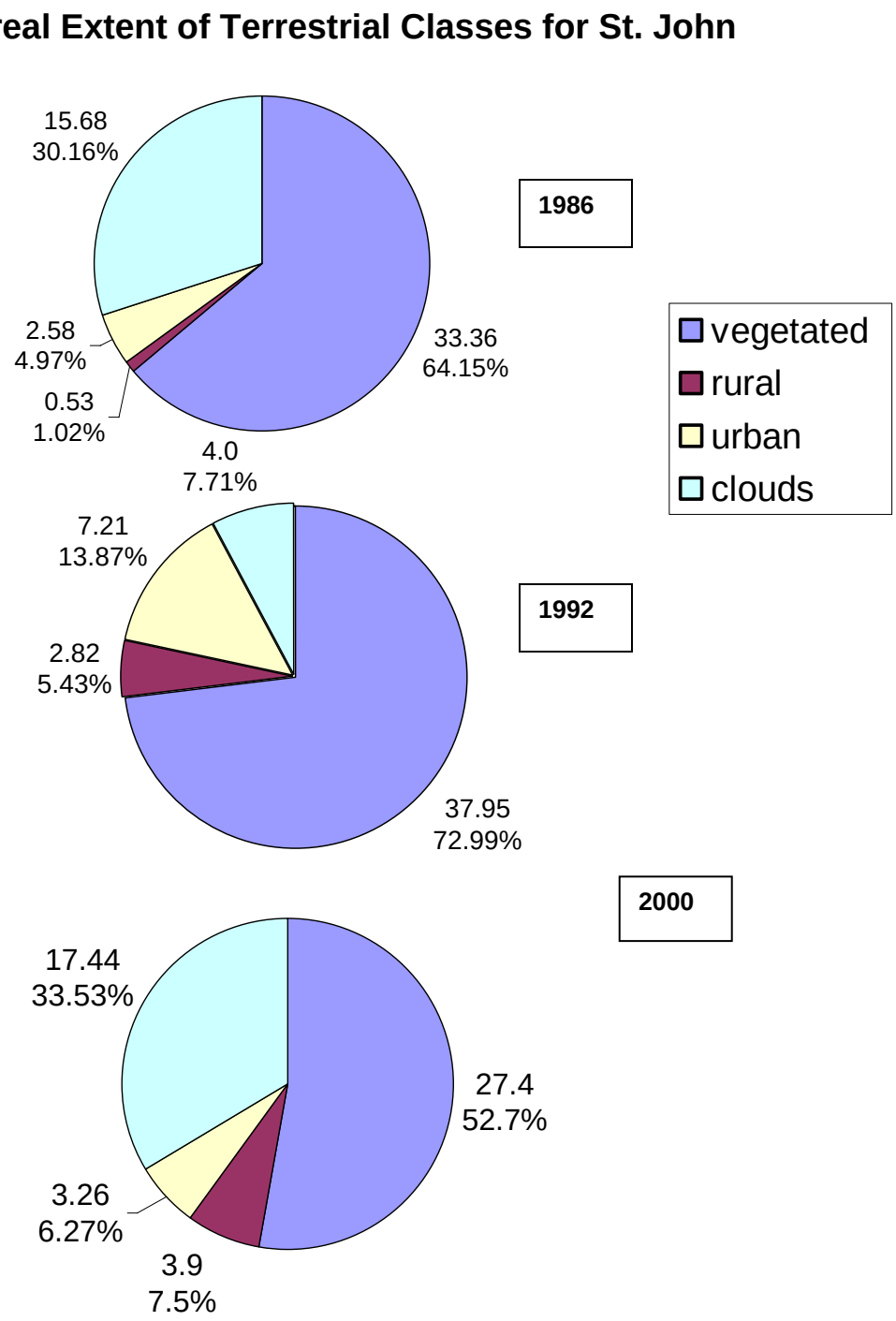

Figure 5.6. Areal Extent of Terrestrial Classes for St. John. The value of each pie slice is given in km^2 (above) and percent (below).

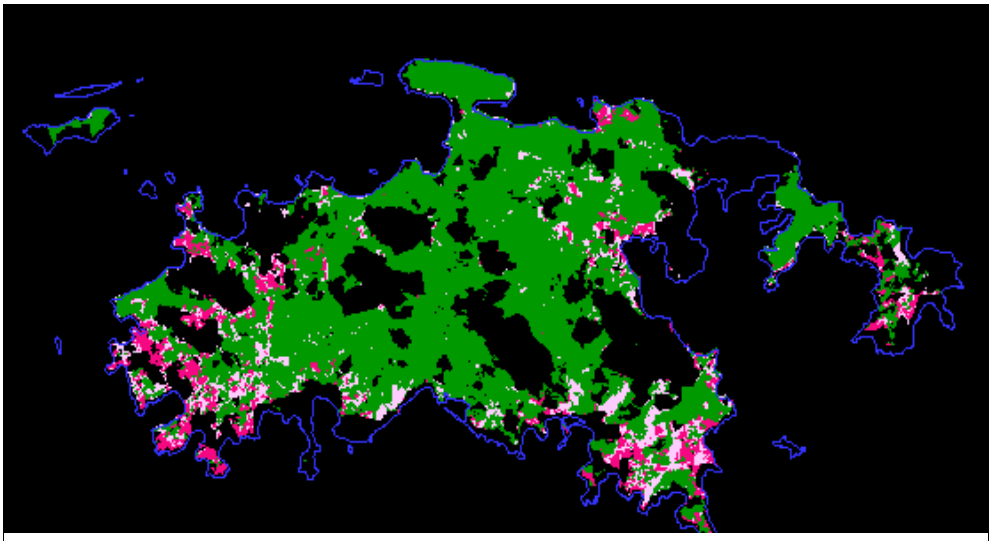

Figure 5.7 : Top: St. John National Park boundaries in green (U.S. National Park Service, 2004). Below: 2000 Unsupervised Classification of St. John Terrestrial Habitat shows that the majority of development has occurred outside park bounds.

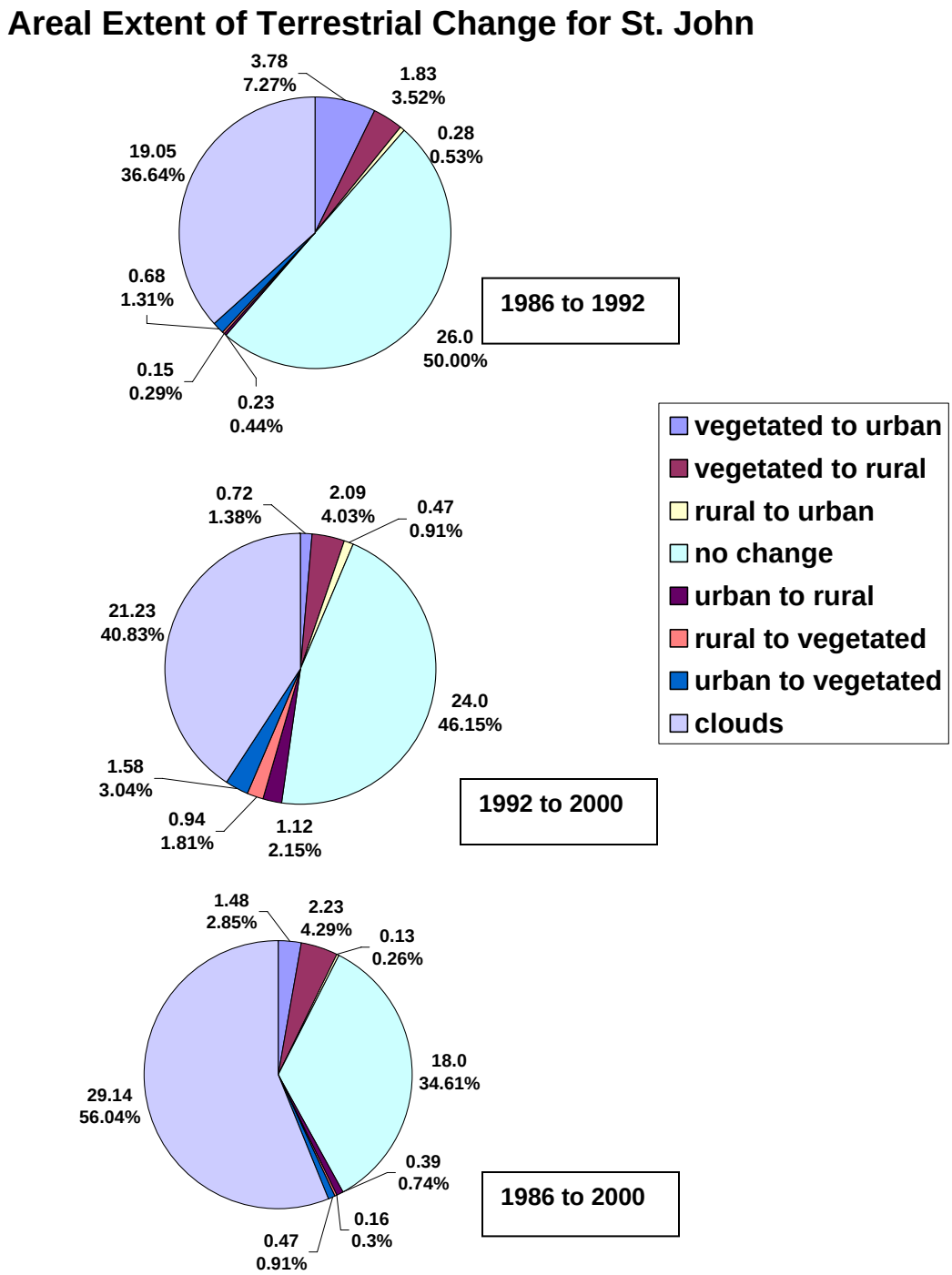

Figure 5.8. Areal Extent of Terrestrial Change for St. John. The value of each pie slice is given in km^2 (above) and percent (below).

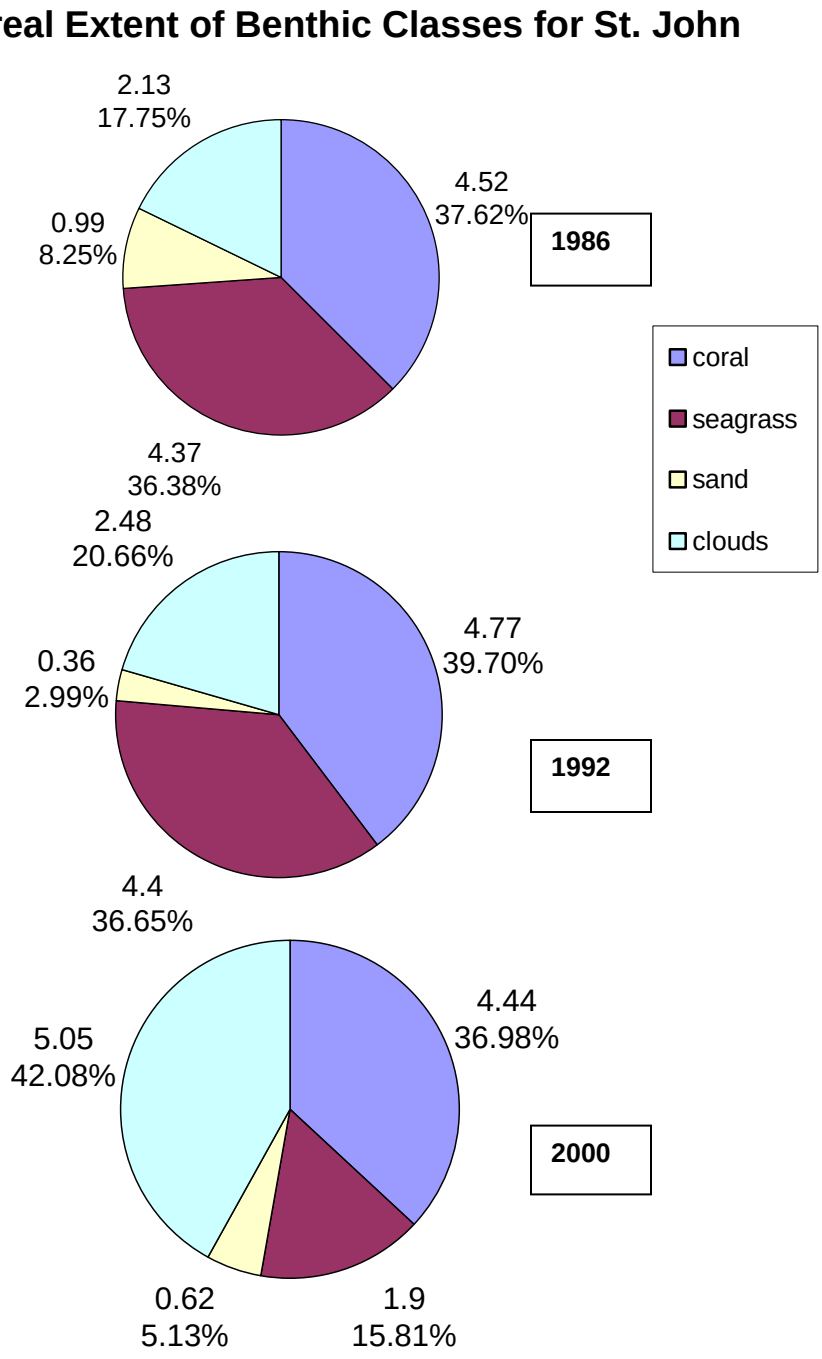

Figure 5.9. Areal Extent of Benthic Classes for St. John. The value of each pie slice is given in km^2 (above) and percent (below).

Areal Extent of Benthic Change for St. John

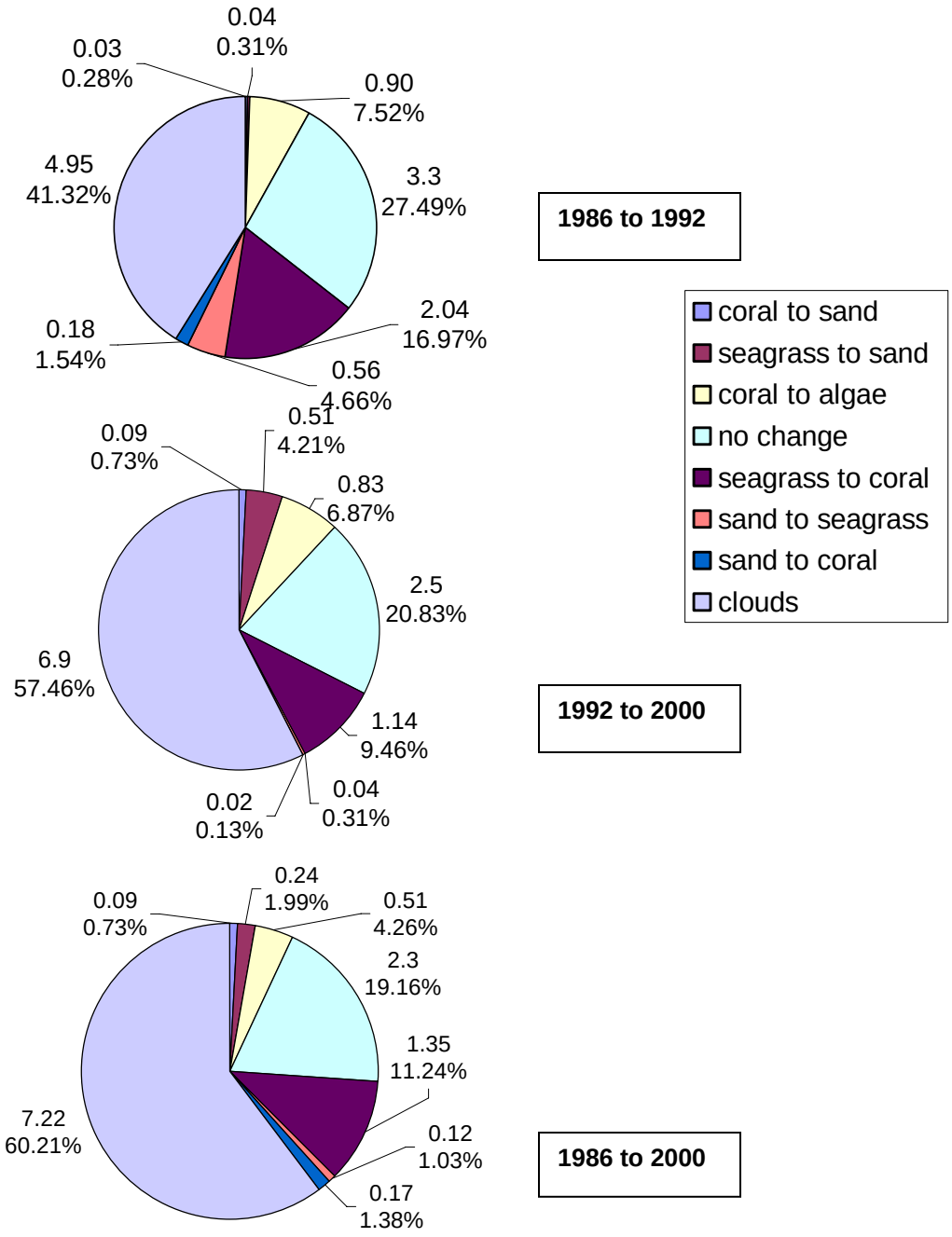

Figure 5.10. Areal Extent of Benthic Change for St. John. The value of each pie slice is given in km^2 (above) and percent (below).

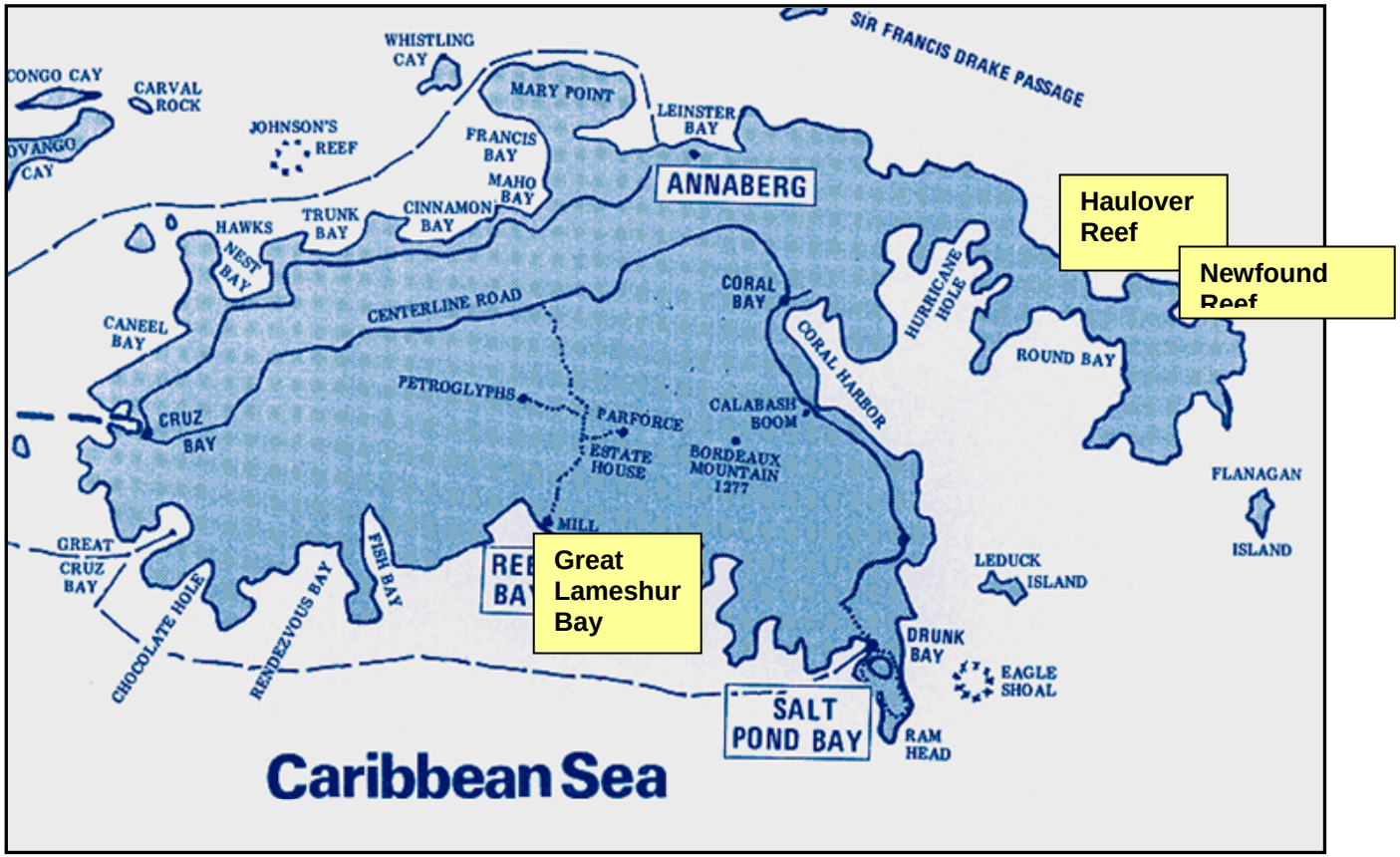

Figure 5.11 : Location of St. John sites mentioned in text.
(St. Thomas Graphics, 2006)

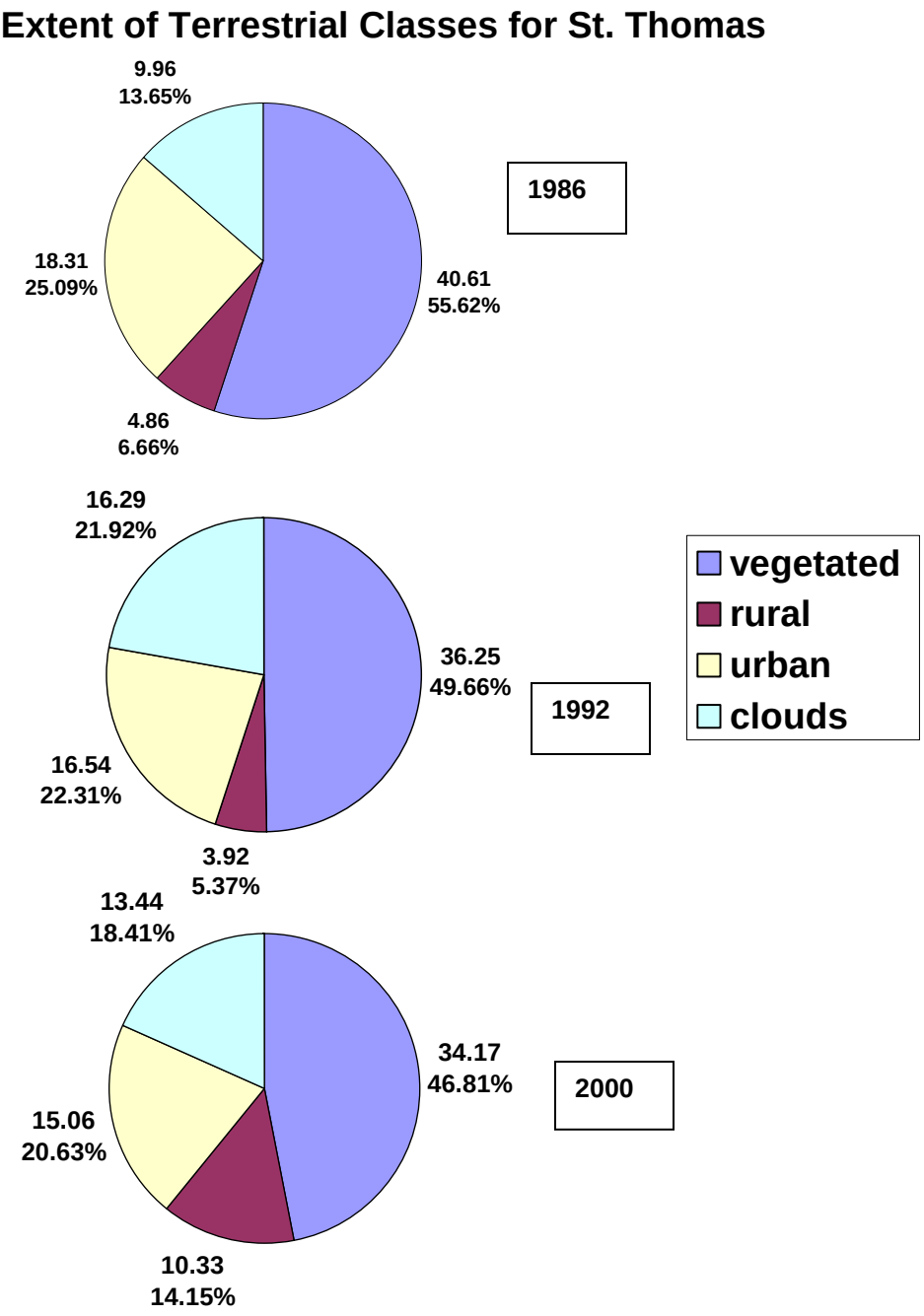

Figure 5.12. Areal Extent of Terrestrial Classes for St. Thomas. The value of each pie slice is given in km^2 (above) and percent (below).

Areal Extent of Terrestrial Change for St. Thomas

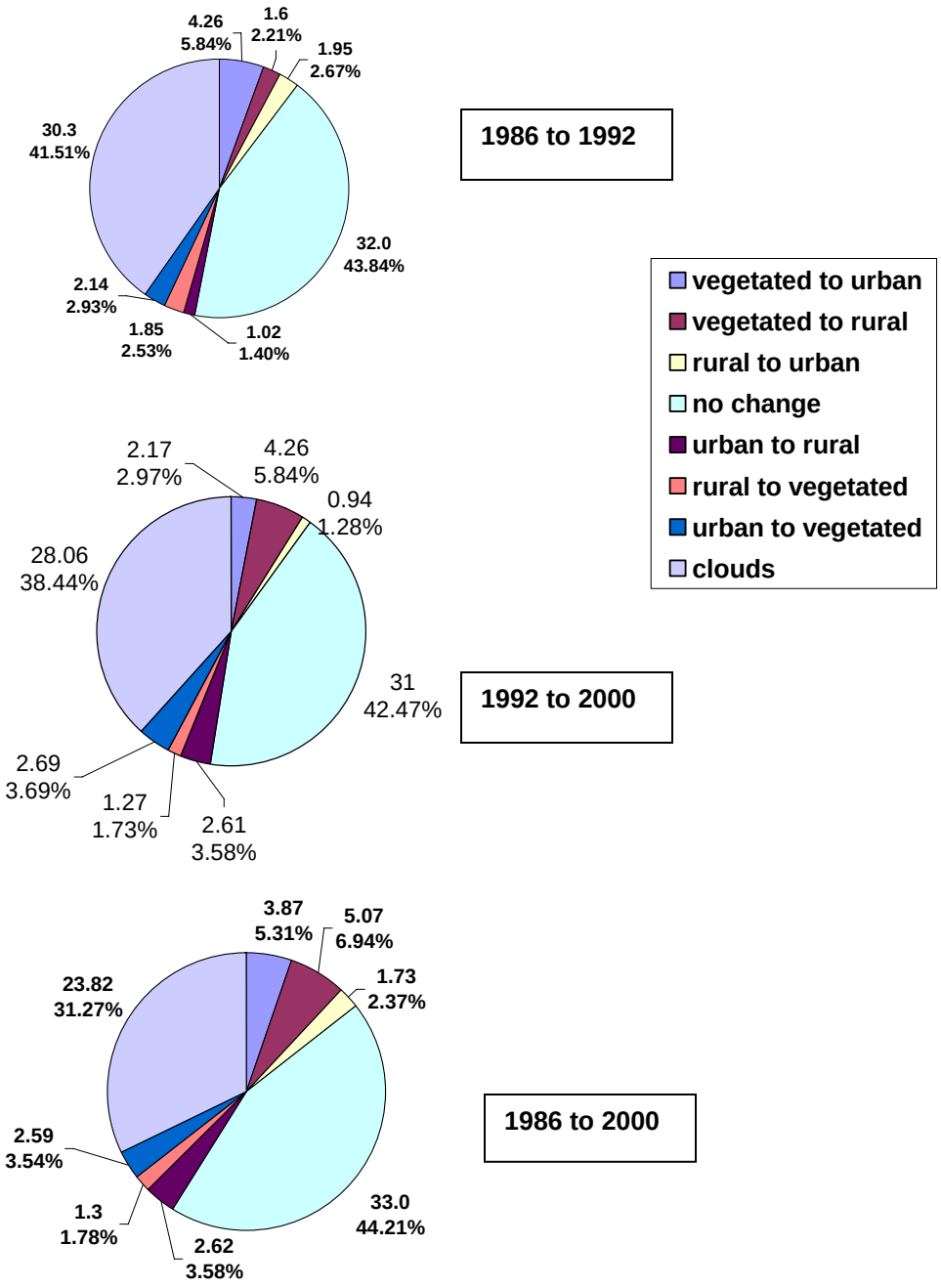

Figure 5.13. Areal Extent of Terrestrial Change for St Thomas. The value of each pie slice is given in km^2 (above) and percent (below).

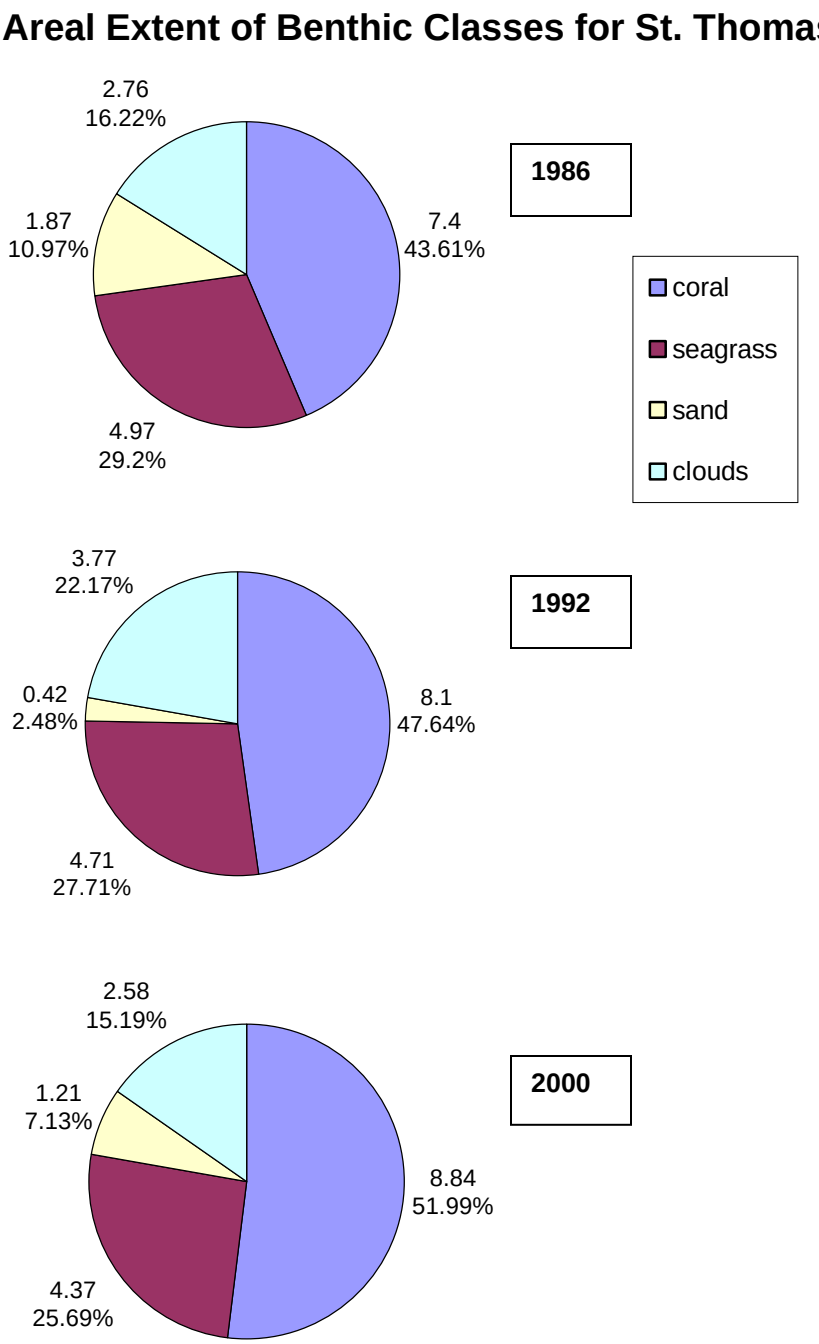

Figure 5.14. Areal Extent of Benthic Classes for St. Thomas. The value of each pie slice is given in km^2 (above) and percent (below).

Areal Extent of Benthic Change for St. Thomas

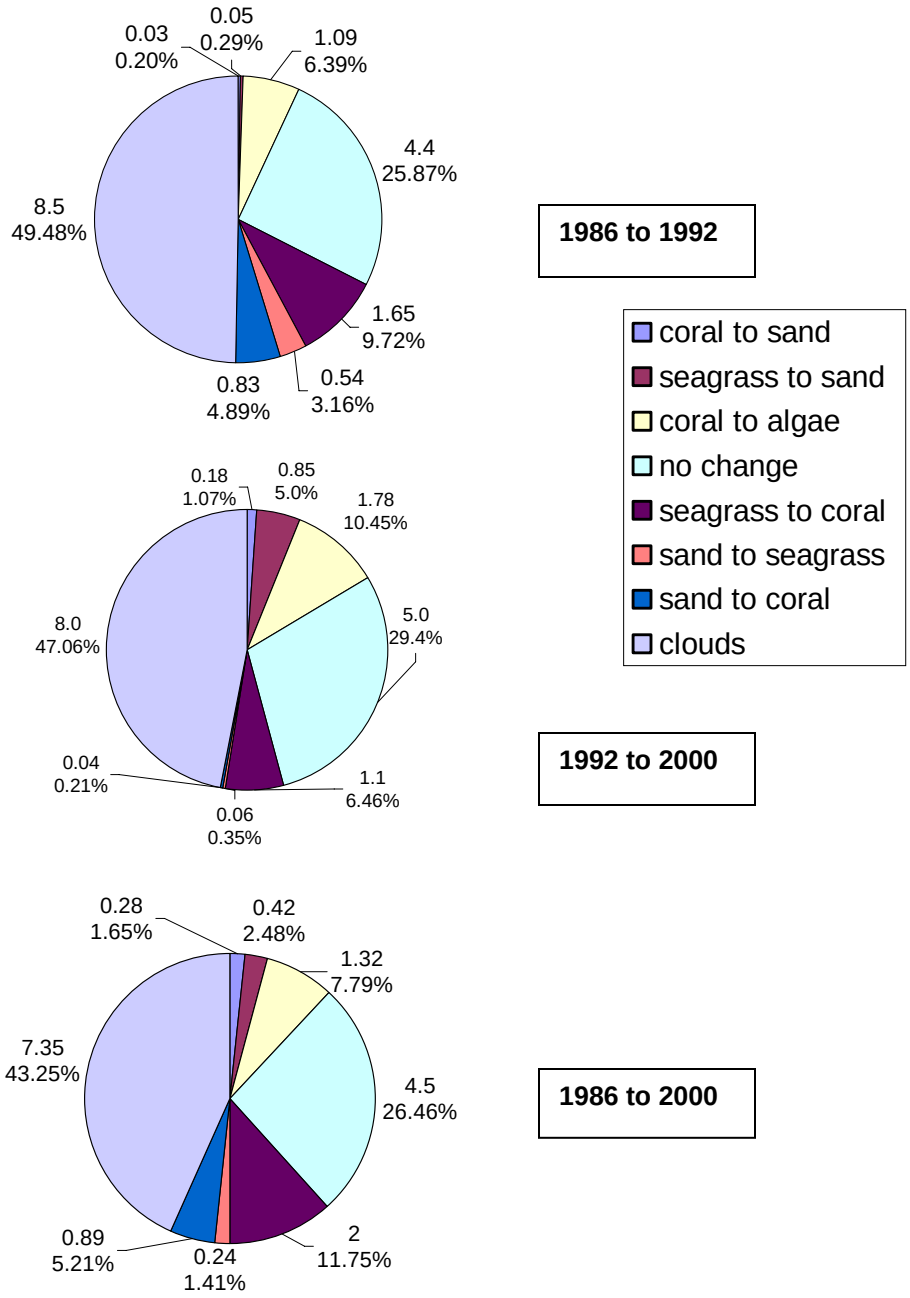

Figure 5.15. Areal Extent of Benthic Change for St. Thomas. The value of each pie slice is given in km^2 (above) and percent (below).

CHAPTER 6

USE OF LANDSAT DATA FOR MANAGEMENT AND POLICY APPLICATION

Optimal Classification and Change Detection

Landsat scenes offer a snapshot of the terrestrial and benthic portions of entire territories or regions, since the imagery covers large portions of the earth (185 × 185 km). In order to accurately classify an area and detect change in land cover with Landsat data, a number of steps must be executed. These steps, if performed efficiently, can be completed in a relatively short amount of time. However, if the analyst is not familiar with the nuances of each step, a great deal of time may be wasted. To avoid the various pitfalls encountered when processing the Landsat data for this project, the following information may be useful for future studies of coastal areas.

Scene Selection

The main obstacle to change analysis, which necessitates obtaining several Landsat scenes of the same area on different dates, was to find cloud-free images. To work with Landsat scenes that have a minimal amount of cloud cover, a number of steps can be taken. If specific areas in a country or region are identified for the Landsat analysis, scenes that are cloud-free in those

specific areas should be selected. By reducing the geographic area of interest, the likelihood of attaining cloud free scenes may be increased.

Mapping an area of interest with Landsat often requires two or more Landsat scenes. For example, St. Thomas and St. John are in one Landsat scene and St. Croix is in a different scene. Scenes were selected so that the dates for the St. Croix scene matched as closely as possible the dates of the St. Thomas and St. John scene. Because this greatly reduced the number of choices, several of the selected scenes had significant cloud cover.

Using the above example, a manager may avoid this pitfall by selecting scenes for St. Croix independently of the scene selection for St. Thomas and St. John. While the dates for the two scenes may differ by several weeks or months, the greater amount of cloud-free pixels in each scene will allow for a better classification and a more thorough change analysis.

When no cloud-free images for a given scene are available, a composite may be formed from several scenes acquired during the same season in a given year. Due to significant cloud cover in many of the scenes selected for this study, a 1986 composite was formed from the cloud free pixels in the 1985 and 1987 scenes and a 2000 composite was formed from the 1999 and 2001 scenes. Because each composite was formed from one wet and one dry year, it was difficult to differentiate seasonal change in vegetation from actual change in the image.

To distinguish actual change in land cover from variation caused by change in season, scenes from both the wet and rainy seasons from the same

year should be classified. A year with average rainfall should be selected to ensure the most accurate representation of seasonal change. The difference in the amount of vegetation in the wet and dry seasons may be used to estimate the amount of variation in change analyses that may be attributed to the seasonal component.

While a period of one to three years between scene dates would provide a more definitive and smoother change analysis (Dustan, 2001), the cost and time incurred in processing the additional images would be greater. An average period of four to five years between dates for Landsat change analyses is appropriate for both terrestrial (Yuan et al., 2005; Nelson et al., 2002) and benthic (Dekker et al., 2005; Bouvet et al., 2003; Dustan et al., 2001) areas. For this study area, five sets of Landsat scenes were used to document land cover changes over a 16-year time frame. Based on availability of cloud-free images the scenes were selected from 1985, 1987, 1992, 1999, and 2001. Due to significant cloud cover, the small gap in time for the 1985 and 1987 scenes allowed us to create a 1986 composite, while a 2000 composite was formed from the 1999 and 2001 scenes.

Pre-classification

Because we were initially unfamiliar with the software (PCI Geomatica) that was used to orthorectify, co-register, and atmospherically correct and classify each scene, a significant amount of time was expended in

orthorectification and atmospheric correction. Coastal managers or planners may save time by purchasing Landsat products that have been orthorectified, coregistered, and atmospherically corrected.

Classification

To decrease time involved in classifying imagery, the unsupervised classification should be performed on the same-season Landsat composite, not on the pre-composite scenes. An accuracy assessment should be performed on a classification scheme prior to performing other combinations of aggregations as well as prior to applying the classification scheme to other scenes. A classification scheme should be discarded if the classification yields less than 85% accuracy.

Because we initially were not aware that Landsat cannot accurately record the spectral signatures of benthic habitats greater than 9 m in depth, a great deal of time was wasted in trying to achieve accuracies greater than 85% for these areas. To avoid this scenario, all areas greater than 9 m in depth should be masked out prior to classifying benthic habitat with Landsat. The analyst should also be aware that because Landsat alone cannot be used to distinguish seagrass from algae, these habitats should be aggregated into one class, not two separate classes. Seagrass and algae are differentiated in the postclassification change map. Areas that changed from coral to vegetated are classified as a shift from coral to algae.

Landsat as a Planning Tool

The spectral bands available on the TM and ETM sensors provide an interpretation tool for distinguishing developed from pristine areas on land, and for differentiating coral, vegetation, and sand in shallow benthic habitats. In coastal territories, states, or nations with no historical data on terrestrial and benthic habitat change, Landsat maps may serve as baseline data for future analyses. Studies based on Landsat may be used to differentiate watersheds past the point of recovery from watersheds that may benefit from a greater level of protection. They may also assist in identifying watersheds where additional research, such as in situ analysis, field surveys, or interpretation of high resolution aerial or satellite imagery, should be conducted.

To understand the impact of land use change at the watershed level, a digital elevation data set could be merged with the Landsat data using a GIS. Tools with the GIS can be used to segment the elevation file into watersheds that could be characterized by determining the land cover type present in each watershed (Merry, C., pers. comm., 2006). Other information useful for determining the impact of land use on benthic habitats, such as the location of paved and unpaved roads, location of permanent and temporary streams, location of wetlands, and population density may be incorporated as layers of data within the GIS.

The soil erosion potential for each slope in a watershed could be determined with the Revised Universal Soil Loss Equation (RUSLE), which

incorporates information on a watershed's rainfall, slope steepness, land cover, and management (Del Mar Lopez, et al., 1998) to determine the soil erosion potential for each slope within a watershed.

A Landsat analysis that indicates a greater shift from coral to algae at the mouths of streams or guts than in other benthic areas may indicate that land-based runoff is a primary factor in coral ecosystem decline. To establish a link between land-based runoff and benthic ecosystem change, in-situ samples may be obtained at periodic locations in guts, as well as at the benthic areas where runoff is deposited. In the USVI, the Department of Environmental Protection, which monitors water quality by measuring Total Maximum Daily Loads (TMDLs) of pollutants, would be able to take the lead in such assessments (Sayeli, pers. comm., 2006).

Landsat data may be combined with real-time field data to differentiate algae from seagrass over large areas. Digital spectral values from a Landsat image can be compared to the information derived from a field spectrometer near the image date (Gullstrom, et al., 2006). A supervised classification that uses the spectral information from the field spectrometer to train or classify Landsat pixels can be used to map large areas of coral and seagrass. Because the map would be based on field data acquired near the scene date, it would be a more accurate representation of benthic vegetation than a postclassification change map.

Another method to improve discrimination of benthic categories in a Landsat scene is to utilize higher resolution imagery, such as CASI (Compact Airborne Spectrographic Imager). CASI may be used to accurately discriminate

and map different species of coral and vegetation (Mumby and Edwards, 2002) for areas in a Landsat scene targeted for more in-depth analysis. Aerial photography may also be used to achieve greater discrimination of Landsat benthic data.

Landsat for Policy Application

Landsat data results may be used to provide probable cause as to the factors that contribute to benthic ecosystem decline. Probable cause may be used in formulating policies that regulate the extent or nature of construction activities more stringently. In the USVI, harsher fines for violators and stricter standards for granting construction permits may serve to reduce the amount of soil that erodes to benthic areas from construction sites (Rohring, pers. comm., 2006).

By showing a correlation between land use change and benthic ecosystem decline, Landsat analyses may serve as the basis for a host of studies that definitively show a connection between land use and benthic health at the watershed level. This evidence may be used to formulate land use policies that strictly regulate land use throughout the watershed and not only near the coastal boundary. In the USVI, while land use near the coast is strictly regulated and monitored through the Coastal Zone Management Program (CZMP), fines for violating land use regulations on other portions of the islands are minimal. Studies that show a stronger link between land use change and ecosystem health may aid in passage of the USVI Development Law, which would place land use regulation under the strict jurisdiction of the CZMP.

Landsat data offer the opportunity to approach issues that impact the coastal environment, from a regional or national perspective. Based on this perspective, the combined impact of fisheries, transportation, industry, energy, housing, tourism and other sectors that impact benthic ecosystem health (Winsemius, 1995) can begin to be ascertained. A GIS may be used to incorporate information about the impacts that each sector has on coastal ecosystem health. When used in conjunction with Landsat maps of land cover and change, the GIS layers may aid in formulating a coastal policy that takes into account the many factors on land and in the water that impact coastal areas.

Because Landsat imagery may be used to map entire regions, the data may be used to ascertain the state of benthic and terrestrial habitats for several countries. A comparison of the Landsat maps of the terrestrial and benthic habitats in each country may be used to ascertain the relative effectiveness of each nation's coastal policy.

Because the public plays an important role in interpreting science and in communicating its preference to policy makers (CGER, 1995), Landsat data may be used to provide a picture of the impact of land use change over time. Using the rates of change derived from change maps, thematic maps may be created that show the potential size and distribution of land cover categories, 10 or 20 years into the future. By presenting a possible future scenario in a clearly discernible visual format, the Landsat data may be used to inform policy makers and the public of the potential impact of continuing the present course of action.

CHAPTER 7

CONCLUSIONS AND RECOMMENDATIONS

Land cover is changing rapidly in the Caribbean both on land and in the benthic environment. Coastal managers need a way to measure and monitor the change in both terrestrial and benthic ecosystems (Merry, pers comm.., 2006). Landsat offers historical data that extends from the 1970s to the present (Lillesand and Keifer, 2000). By using Landsat to measure the degree and type of change in terrestrial and benthic habitats, this study has demonstrated the potential use of Landsat as a planning and monitoring tool for coastal managers.

The data results for terrestrial areas were as expected for the study period, showing as shift from vegetated to rural or urban land cover. Over the 1986 to 2000 period, there was an increase in rural (9.99%) and urban (7.81%) land cover for St. Croix, for St. John (6.48% for rural and 1.3% for urban) and for St. Thomas (7.49%). That urban land cover did not increase for St. Thomas

during this period may be attributed to the extensive development that occurred on the island prior to the study period, leaving little undeveloped land near the island's urban centers.

The data results for the benthic habitats were not as expected. It was anticipated that a dramatic decrease in benthic vegetation would be observed, due to a decrease in seagrass density. Any dramatic shift in seagrass density would serve as an overall indicator of ecosystem health. However, the results showed a significant increase in benthic vegetation (18.91%) for St. Croix for the 1986 to 2000 period. Because it was determined that the increase in vegetation was due to a shift from live coral to dead, algae-covered coral, the data results were consistent. A decrease in seagrass density was observed for both St. John (- 20.57%) and St. Thomas (-3.51%) for the same time period.

The largest change category in St. Croix's benthic habitat for the 1986 to 2000 period was "coral to algae" (22.98%), indicating that 22.98% of live coral in 1986 had changed to dead algae-covered coral. The second largest category of change, "seagrass to coral" (11.08%), indicated a change from seagrass to sand or a decrease from high to low seagrass density. While benthic areas for St. Thomas and St. John also pointed to a shift from live coral to dead, algae-covered coral, the largest class in the 1986 to 2000 change map was "seagrass to coral" (11.24% for St. John and 11.75% for St. Thomas).

The observed shift from coral to algae is supported by numerous studies that have reported an increase over the past decade in algal cover in coral ecosystems of the Caribbean (Gardner et al., 2003; Rogers and Beets, 2001;

Dustan et al., 2001). In the USVI, the apparent shift from live to dead, algae-covered coral appears to have been caused by both anthropogenic and natural factors.

The anthropogenic factors may include the increase in sediments and nutrients to USVI benthic habitats. The sediments would smother corals or diminish the light required for photosynthesis by coral zooxanthellae. Increased levels of nitrogen or phosphorous input into the characteristically oligotrophic waters of the Caribbean would impact corals negatively and promote the growth of algae. Natural factors that contribute to an increase in algae include destruction of coral reefs by hurricanes and a decrease in the number of algae-grazers, such as the black spine urchin. The observed decrease in seagrass density is supported by studies that point to blow-outs by hurricanes, as well as damage caused by boat anchors as the primary reasons for a decrease in USVI seagrasses.

The inductive nature of this research requires determining whether the methods and results inform a larger body of knowledge. It also requires determining whether the methods and results address a specific gap in this body of knowledge. In the remote sensing field, a literature review at the inception of this research indicated there were no published studies that utilized Landsat for a change detection analysis of both terrestrial and benthic habitats. This study has shown the usefulness of Landsat in providing an overall snapshot of change for both the benthic and terrestrial ecosystems of an entire territory, state, region, or country (Figure 5.9). The research results support the claim that remote sensing

may be used to provide quantitative estimates of coastal and estuarine and habitat conditions (Muttitaton and Tirpathi, 2004; Klemas, 2001).

Due to benthic ecosystem interactions, an increase in nutrients or sediment levels or a decrease in light availability impact both corals and seagrasses negatively. Severe disturbances in nutrient, light, or sediment levels may limit or prevent seagrass growth and transform corals into dead hard bottom, usually dominated by algae. The study results support these findings, by suggesting an increase in land-based runoff, which contains both sediments and nutrients. For St. Croix, these components contributed to 22.98% of live coral cover changing to dead, hard bottom dominated by algae over the 1986 to 2000 period. Elevated levels of nutrients contributed to the growth of algae, which have the ability to out-compete corals for light and oxygen. Increased sediment runoff decreased the amount of light available to coral zooxanthellae and hindered the physical function of coral organisms.

Data from 30 sites around St. John indicate that bays with developed watersheds have higher turbidity and lower light transmission than bays associated with less disturbed watersheds (Rogers and Beets, 2001). Nemeth and Nowlis (2001) found that coral cover in a St. Thomas bay declined 14% from July, 1997 to March, 1999 following exposure to sediments from construction sites.

Although Section 319 of the Clean Water Act and Section 6217 of the Coastal Zone Act Reauthorization Amendment mandate the control and mitigation of nonpoint source pollution in coastal areas, there is a lack of a clearly

defined methodology that monitors the success of these efforts at the watershed level. In addition, the science to conclusively link sedimentation from higher parts of the watershed to those in the coastal areas is still being developed. Findings from Landsat data that help to show a correlation between land use and benthic ecosystem decline at the watershed level may aid policy makers and managers to bridge the science-policy gap that has characterized nonpoint source pollution management.

Recommendations

Future studies can build upon the current methodology by conducting further studies of watersheds targeted through the use of Landsat. Studies that utilize a hydrological model to determine the flow of water and sediments within a watershed or that monitor the content of land-based runoff at the base of streams or guts can be used to determine the impact of land use activities, such as housing or road construction on benthic ecosystems. To determine amount of soil loss from a watershed, it is recommended that he Revised Universal Soil Loss Equation (RUSLE) be used, as it differentiates regions within a watershed based on sensitivity to erosion. TMDL and other data from EPA gauging stations may be used to determine the amount of sediment loads in past years.

Arerial photographs and images from CASI may be used to provide greater detail to benthic maps created from Landsat data. To determine whether a decrease in the terrestrial vegetation of a watershed contributed to an increase in benthic algal cover at the base of the watershed requires the assessment of ancillary data for the study period, including changes in population size, sewage

effluent and construction. To allow for a valid comparison of watersheds, it would be necessary to account for natural variations caused by slope, soil type, hydrology, rainfall level, wind speed, and exposure to storms, as well as variations due to population size and economy.

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