Integration of GIS and logistic regression to develop a habitat suitability model for predicting seagrass distribution

By

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Abstract

Habitat suitability modeling can reveal connections between seagrass and environmental variables namely water quality. This can help predict seagrass distribution affected by changing water quality variables. Seagrass, and its density distribution in Tampa Bay, Florida, is an important environmental and economic resource providing a habitat for fish and invertebrates, a food source for larger vertebrates, and a significant source of primary productivity. The overall goal of this study was to loosely couple GIS and logistic regression methods to analyze relationships between seagrass distribution and water quality variables. Specific objectives were i) to determine key water quality variables that influenced seagrass occurrence and ii) to analyze prediction error including difference between continuous and patchy seagrass beds. Water quality variables, such as light attenuation, salinity, and temporal variability of salinity (TVS), were used with GIS and the logistic regression model in order to predict their relationship with seagrass occurrence. Preliminary results showed that light attenuation was a significant predictor of distribution, with salinity and TVS to a lesser extent. The predictive model was validated using known seagrass polygon data. Results indicated that the model predicted continuous coverage more accurately (55%) than patchy coverage (33%). The difference of nearly 20% could be attributed to complex shape dynamics in patch delineation. Future error analysis should include incorporation of fractal dimensions of seagrass polygons for both continuous and patchy beds. Results of this study can give managers and planners information on the relationship between seagrass occurrence and the water quality variables investigated in this study.

Introduction

Coastal environments cover approximately 6.4% of the earth's surface, but are responsible for 43% of the estimated value of the world's ecological services (Dawes et al. 2004). Seagrass is an important resource in these marine environments but has been declining globally since the earliest quantitative records in 1879 ((Waycott et al. 2009), (Unsworth and Unsworth 2010)). The cumulative effect of the reported losses of seagrass and other marine species signals a serious deterioration of coastal environments around the world (Waycott et al. 2009). The degradation of seagrass meadows has been most frequently documented in estuaries experiencing frequent and concentrated nutrient loads with reduced tidal flushing ((Hauxwell and Valiela 2004), (Orth et al. 2006), (Ralph et al. 2006), (Short et al. 2006)). Reduction in water clarity has been implicated as the cause for decreased distribution (Orth et al. 2006). As a benthic organism, seagrass is an important indicator of estuarine health because of its sensitivity to changes in local water quality conditions ((Kirk 1983), (Johansson and Greening 2000)). Seagrass has been nicknamed a "coastal canary" because of its role as a sentinel of increasing anthropogenic influences (Orth et al. 2006).

The seagrass habitat is important to the ecological functioning of coastal lagoons and estuaries because of its value as a refuge for juvenile fish and invertebrates, a sediment stabilizer, and a food source for larger organisms ((Gallegos and Kenworthy 1996), (Kellogg 2002), (FWS 2014), (Greening et al. 2014)), such as waterfowl, manatees and green sea turtles (Bergquist 1997). Seagrass beds in the gulf coast of Florida are also important for the economic health of the region ((Dawes et al. 2004), (Larkum et al. 2007)). Some of the species that find refuge in seagrass beds are economically important to fisheries (FWS 2014) for commercial and sport fishing purposes (Kellogg 2002). The impact of commercial and recreational saltwater fishing in Florida has been reported to be $6 billion annually (FWC 2014). Seagrass meadows and submerged algal beds were estimated to provide ecosystem services worth approximately $3,116 per hectare per year in 1994 (Costanza et al. 1997). Global declines in seagrass coverage may threaten not only local ecosystems, but larger scale systems as well. Methods of predicting change in seagrass coverage in response to fluctuating environmental conditions are needed for the effective management of seagrass populations.

Study Area

Tampa Bay (Figure 1) is located on the gulf coast of west-central Florida and is one of the largest open water estuaries in the southeastern United States (Xian and Crane 2005). The gulf coast region has low energy characteristics as a result of a gently sloping shelf, a divergence of wave trains, an upwind direction of the coast, and wave dampening effects from old submerged beaches and seagrass meadows (Murali 1982). Covering 1,036 square kilometers, Tampa Bay is a Y-shaped embayment with a mean depth of 4 meters (Greening et al. 2011). The natural bathymetry of the bay has been altered to create 13 meter-deep shipping canals between the city of Tampa and the Gulf of Mexico (Greening et al. 2014).

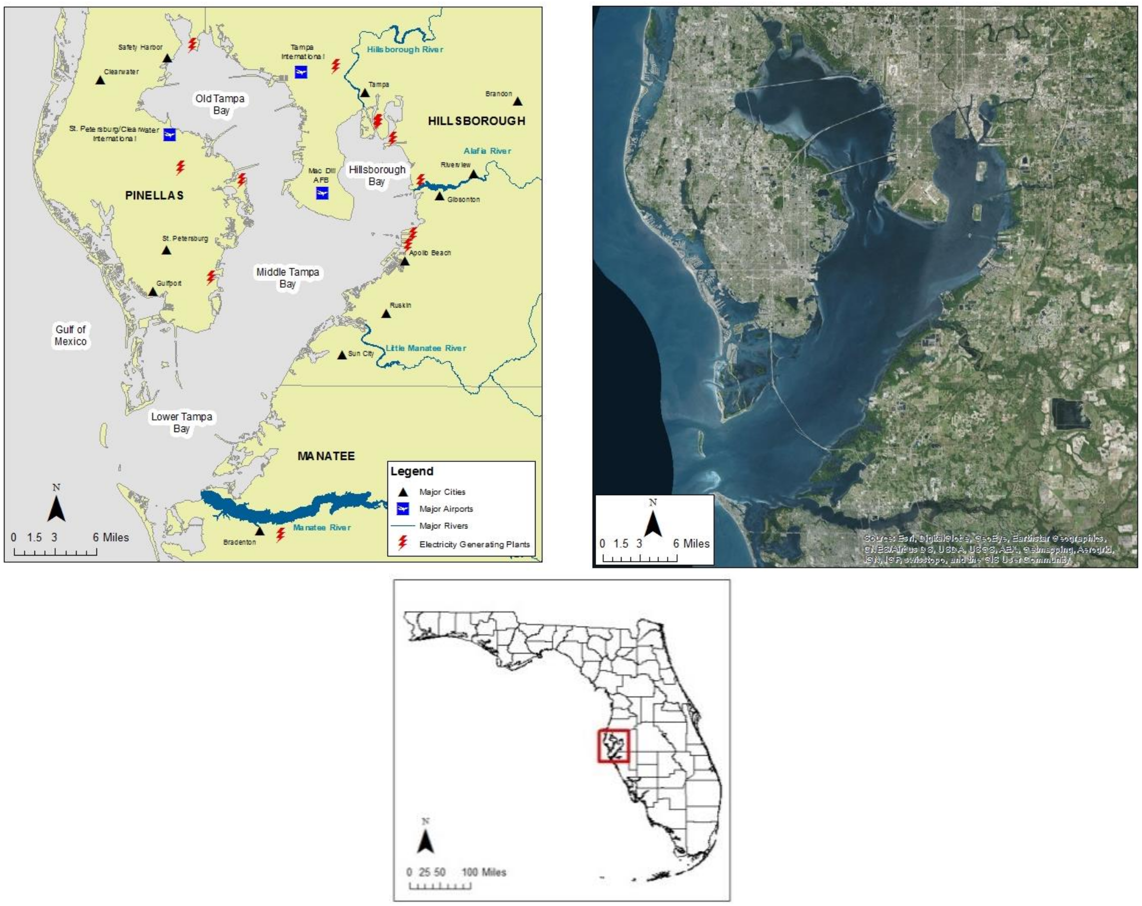

Figure 1: Tampa Bay, Florida study area. Data obtained from Florida Geographic Data Library. Aerial imagery obtained from ESRI.

Tampa Bay has been categorized into 7 smaller water bodies: Old Tampa Bay, Hillsborough Bay, Middle Tampa Bay, Lower Tampa Bay, Boca Ciega Bay, Terra Ceia Bay, and the Manatee River (Lewis and Whitman 1985). Streams and four major rivers, the Hillsborough River, the Alafia River, the Little Manatee River and the Manatee River, empty into Tampa Bay ((Xian and Crane 2005), (Crane and Xian 2006)) and contribute approximately 70% of freshwater inflow (Goodwin 1987). Tampa Bay and its contributing watershed support highly diverse ecosystems because of the bay's large size, its gradient of freshwater and saltwater, and its location in a transition zone between the warm temperature "Louisiana" and tropical "West Indian" biogeographic provinces ((Lewis and Estevez 1988), (TBEP 2006)).

The Tampa Bay watershed covers approximately 5,700 square kilometers (Greening et al. 2011) encompassing parts of Pinellas, Hillsborough, Manatee, Pasco, Polk, and Sarasota Counties and containing the major metropolitan areas of Tampa, St. Petersburg, Clearwater, and Bradenton ((Bergquist 1997), (Crane and Xian 2006)). With more than 2 million people living in the watershed (Xian and Crane 2005), most of the surrounding land is urbanized on the northern and western sides with rapidly increasing suburban expansion on the eastern side (Greening et al. 2014). Freshwater discharge into the bay comes from the rivers and streams, industry outflow, and treated municipal water. In the upper portions of the bay, these inflows total approximately 60 million gallons per day (Wang et al. 1999). Water quality was poor in these areas during the 1970's and 1980's with especially high chorophyll-a concentrations in Hillsborough Bay between 1974 and 1982 (Wang et al. 1999). In 1979 a federal grant allowed the city of Tampa to install an advanced wastewater treatment plant (Wang et al. 1999) reducing the flow of nutrient loads by approximately 90% (Lewis and Estevez 1988). Water quality in the bay has been monitored for more than two decades by the Hillsborough Environmental Protection Commission (Boler 1995).

Human population density in coastal areas, made popular by warm climates and abundant recreational activities, has increased by more than double that in inland areas ((Nicholls and Small 2002), (McGranahan et al. 2007)). Urban development in the Tampa Bay watershed has been ongoing since the 1880's ((Crane and Xian 2006), (Xian et al. 2007)). The populations of Pinellas and Hillsborough Counties grew by 148% and 158% respectively between 1960 and 2001 (Xian et al. 2007). This increase in human population was accompanied by a proportional increase in the use of bay water (Estevez and Upchurch 1985) as well as dredge-and-fill projects modifying the coastline (Morrison and Greening 2011). Three major bridge structures were installed in Old Tampa Bay that altered the previous hydrodynamics of the marine landscape (Greening et al. 2014).

Suburban development changes the inland landscape of the watershed by increasing impervious surface cover (Zarbock et al. 1994). In the Tampa Bay watershed, imperviousness increased significantly between 1995 and 2005 (Reistetter and Russell 2011). Imperviousness greater than 10% characterized 27% of the total watershed in 2002 (Crane and Xian 2006) and is predicted to continue increasing (Xian and Crane 2005). Impervious surfaces alter the natural hydrologic condition of an area by increasing the volume and rate of surface runoff (Moscrip and Montgomery 1997). This runoff can lead to the degradation of water quality when it transports non-point source pollutants into the bay ((Gove et al. 2001), (USEPA 2001)). Pollutants dissolved or suspended in water, such as nutrients, heavy metals, oil, and grease, can accumulate and wash away more easily across impervious ground cover since they are not absorbed into the soil (Xian et al. 2007). An inverse relationship between urbanization and seagrass has been observed as a result of increased impervious land cover (Crane and Xian 2006) (Figure 2).

Figure 2: Flow chart detailing how an increase in human population can negatively influence seagrass distribution.

Anthropogenic Influence on Coastal Ecosystems

Coastal populations contribute high nutrient loads to coastal waters through sewage and other secondary impacts of development ((Vitousek et al. 1997), (NRC 2000)). Rivers and streams contribute a large portion of nutrient addition to larger water bodies ((Zarbock et al. 1994), (Janicki and Wade 1996)) transporting nutrient-enriched water from watershed inputs ((Caraco 1995), (Vitousek et al. 1997)). Increased nutrient addition can lead to increased growth of phytoplankton, epiphytes, and macroalgae ((Orth and Moore 1983), (Borum 1985), (Twilley et al. 1985), (Dennison et al. 1993), (Harlin 1993), (Dunton 1994), (Lapointe et al. 1994), (Short et al. 1995), (Wear et al. 1999), (Hauxwell and Valiela 2004), (Ralph et al. 2006)). Phytoplankton cells and colonies scatter and absorb light while macroalgae can produce thick canopies between 0.75 and 2 meters in height (Hauxwell et al. 2001), all making a significant contribution to the total scattering behavior of incident light in the water-column (Kirk 1983).

As phytoplankton, macroalgae, and epiphytes respond to nutrient enrichment, oxygen production and respiration become increasingly uncoupled leading to hypoxic and anoxic conditions (Sand-Jensen and Borum 1991). Prolonged anoxia, occurring more frequently in warm periods ((Sfriso et al. 1992), (D'Avanzo and Kremer 1994)), increases the energy demand for trans-locating oxygen into the roots of benthic plants (Pregnall et al. 1984). Decreased oxygen availability in the roots leads to reduced photosynthesis, smaller leaves, and a reduced number of leaves per shoot ((Holmer and Bondgaard 2001), (Smith et al. 1988)). Eutrophic conditions help faster-growing organisms, like phytoplankton and macroalgae, reproduce more quickly than slower-growing organisms, like seagrass (Diersing 2011). Nutrients are rapidly taken up by these faster-growing organisms ((Suttle and Harrison 1988), (Suttle et al. 1990)), making the direct measure of water-column nutrients difficult and generally ineffective ((Tomasko et al. 1996), (Morris and Virnstein 2004)).

Nutrients available in the water-column for seagrass have been analyzed by observing leaf tissue content ((Gerloff and Krombholtz 1966), (Atkinson and Smith 1984), (Fourqurean et al. 1992a), (Fourqurean et al. 1992b)). Spatial and temporal variations in carbon, nitrogen, and phosphorus ratios have been used to track the nutritional status of seagrass and nutrient supply sources (Touchette and Burkholder 2000). But quantitative prediction of changes in seagrass due to nutrient enrichment proves difficult because of the ecological complexities involved in seagrass response to environmental conditions ((Duarte 1995), (Hauxwell and Valiela 2004)).

Cultural eutrophication from the over-enrichment of nitrogen and phosphorus has degraded coastal waters and has been linked to seagrass disappearance worldwide ((Cambridge and McComb 1984), (Short and Wyllie-Echeverria 1996), (Bricker et al. 1999), (Green and Short 2003)). Seagrass coverage is strongly impacted by eutrophication among macrophytes (Burkholder et al. 2007) and has declined with increased nutrient loading due to a "cascade" of negative effects (Duarte 1995). The response of seagrass to nutrient enrichment has been categorized into four reactions: i) growth and physiology respond positively when nutrients are low, ii) physiology responds favorably with no growth when nutrients are low, iii) no response occurs when nutrients are in excess (Udy and Dennison 1997), and iv) physiology responds negatively with inhibited growth when nutrients are in excess ((Burkholder et al. 1992), (van Katwijk et al. 1997), (Touchette and Burkholder 2001)). The most common mechanism for seagrass decline under nutrient enrichment is the stimulation of high-biomass algal overgrowth (Shepherd et al. 1989). Phytoplankton, macroalgae, and epiphytes are considered superior competitors for light relative to seagrass ((Duarte 1995), (Valiela et al. 1997)).

Between 1985 and 1994, light attenuation and inorganic nitrogen were the primary factors limiting phytoplankton growth, measured in cholorphyll-a concentrations, in Tampa Bay ((Janicki and Wade 1996), (Wang et al. 1999)). Between 1974 and 1995, chlorophyll-a was highest in Hillsborough Bay, intermediate in Old Tampa Bay and lowest in Lower Tampa Bay (Morrison et al. 1997). The bay is also consistently enriched in phosphorus (Fanning and Bell 1985) from the large-scale phosphorus mining that occurs in the Tampa area, providing 20% of the world's phosphate production (Tiffany and Wilkinson 1989). Nutrient loading is greatest in Hillsborough Bay because of point-source loads from this industry as well as Tampa's wastewater treatment plant (Greening et al. 2014). The overall decline of chlorophyll-a and corresponding increase in water clarity is believed to have resulted from reduced nutrient loadings from the updated wastewater facility ((Johansson 1991), (Boler 1995)) while non-point source pollution in Hillsborough County is still a concern and appears to be related to land-use characteristics within the watershed (Xian et al. 2007).

Environmental Variables Affecting Seagrass Distribution

Light characteristics of the water medium represent the primary determinants of submerged aquatic vegetation distribution in offshore areas of the Gulf of Mexico (Livingston et al. 1997). Seagrass requires light for photosynthesis in the photosynthetically active range (PAR) for growth and survival ((Dennison et al. 1993), (Morrison and Greening 2011)). Seagrass meadow extent is constrained by depth limit imposed by light attenuation through the water-column ((Borum 1983), (Bulthuis 1983), (Iverson and Bittaker 1986), (Dennison 1987), (Orth and Moore 1988), (Hillman et al. 1989)) since irradiance decreases with increasing water depth ((Kirk 1983), (Dennison et al. 1993)). The vertical attenuation coefficient for downward irradiance is a measure of the rate of change of downward irradiance with depth, and is valuable because it provides information about how much light is available for photosynthesis at bottom (Kirk 1983). Temperate and tropical seagrass species growing in areas with similar light attenuation characteristics have been shown to have similar light requirements (Duarte 1991). In the Indian River Lagoon on the east coast of Florida, both Halodule wrightii (shoal grass) and Syringodium filiforme (manatee grass) grew to the same maximum depth (Kenworthy and Fonseca 1996). In Lemon Bay on the west coast of Florida, water clarity alone was used to accurately predict seagrass depth limits (Tomasko et al. 2001).

The limiting of seagrass distribution due to light attenuation can result from a variety of sources ((Kenworthy and Haunert 1991), (Morris and Tomasko 1993)). Light attenuation is a function of not only the water itself, but also dissolved and particulate components reflecting, absorbing, and scattering incident radiation (Dennison et al. 1993). The absorption of light by photosynthetic pigments in these sediments and organisms contributes to diminishing light attenuation with increasing depth (Kirk 1983). Seagrass cannot survive in depths past the light compensation point (Gallegos and Kenworthy 1996) where oxygen production is unequal to carbon dioxide consumption (Kirk 1983). Low light levels result in a negative carbon balance due to decreased photosynthetic rates and increased respiration rates (Burkholder et al. 2007). Epiphytic cover on seagrass blades can also block light from reaching their photosynthetic pigments and is expected to increase with increasing nutrients or increasing distance northward in Tampa Bay (Dixon and Leverone 1995).

The concentration of particulate matter, thus the intensity of light scattering, in coastal environments is strongly influenced by the physical environment and local land-use (Kirk 1983). Suspended solids and chlorophyll-a were the dominant causes of diminished light attenuation in a study in the Indian River Lagoon (Christian and Sheng 2002). Suspended sediments depress gas exchange by increasing the thickness of the diffusion boundary layer across the leaf surface (Ralph et al. 2006). Erosion or coastal disturbance causes particulate matter to be resuspended (Kirk 1983) and can accelerate and maintain further seagrass loss ((Bulthuis 1983), (Larkum and West 1983), (Clarke and Kirkman 1989), (Walker et al. 2006)). Seagrass cover typically promotes stable sediment, but when seagrass cover has decreased between 25% and 50%, re-suspension has been shown to increase (Moore et al. 1996). Therefore, a significant decrease in seagrass coverage can initiate a positive feedback loop resulting in an even greater loss.

Seagrass requires much higher light attenuation than phytoplankton and macroalgae for long-term survival and growth ((Duarte 1991), (Dennison et al. 1993), (Gallegos and Kenworthy 1996), (Kenworthy and Fonseca 1996), (Dawes et al. 2004)). Phytoplankton and macroalgae only require between 1% and 5% of incident light for survival (Kenworthy and Fonseca 1996), while submerged aquatic vegetation requires a minimum between 4% and 29% (Dennison et al. 1993). Seagrass colonizes littoral zones with suitable sediments down to an average available irradiance of 10.8% (Duarte 1991). The edges of seagrass beds extend to depths at which between 10% and 30% of surface irradiance is available ((Duarte 1991), (Kenworthy and Haunert 1991), (Dennison et al. 1993)), although light attenuation levels at the deep edges of Thalassia testudinum (turtle grass) beds in Tampa Bay were significantly higher than 11% (Dawes et al. 2004). The annual average light attenuation required for stable edges of T. testudinum beds in Tampa Bay has been found to be 20.5% (Dixon 2000) with a maximum limit of 22.5% (Dixon and Leverone 1995). In Charlotte Harbor, to the south of Tampa Bay, seagrass requires an average of 15% to 30% (Dixon 2000) while 23% to 37% is required in the Indian River Lagoon (Dawes et al. 2004). The refractive index of light also varies with environmental conditions such as salinity, temperature, and wavelength of light (Kirk 1983) causing light attenuation to vary across mediums.

Salinity has been shown to affect local distribution of seagrass on the gulf coast of Florida ((Phillips 1960a), (Phillips 1960b)) as a result of freshwater inflows, hurricanes, wastewater disposal, desalination plant discharge, and modified watersheds (Tomasko and Hall 1999). Photosynthetic responses in seagrass beds have been used to evaluate the response to changes in salinity and temperature ((Dawes et al. 1987), (Dawes et al. 1989)). River output is essential to maintaining salinity levels for seagrass survival, and atypically higher rates of salinity variability can result in seagrass die-off (Doering et al. 2002). Diversion and withdrawal of freshwater from the Caloosahatchee River on the west coast of Florida has resulted in negative ecological consequences including the loss of submerged aquatic vegetation (Doering et al. 2002). The effect of salinity alone on seagrass distribution is difficult to separate from other contributing factors (Dawes et al. 2004). All seagrass species can tolerate short-term fluctuations in salinity (Koch et al. 2007), with an optimum range between 24 and 35 parts per trillion (FWS 2014). In deeper portions of Tampa Bay, bottom salinity remains relatively stable (Dixon and Leverone 1995) while in shallower portions the water-column is more homogeneous as mixing is more likely to occur (Wang et al. 1999).

Temperature can affect how much carbon dioxide is consumed by seagrass (Kirk 1983). Shading of the seagrass species, Zostera marina (eelgrass), for three weeks reduced non-structural carbohydrate concentrations in leaves, rhizomes, and roots by up to 51% (Burke et al. 1996). Temperatures elevated 3 to 4 degrees Celsius above the local mean exacerbated the inhibitory effects of water-column nitrate enrichment on root growth in the same species ((Touchette et al. 2003), (Touchette and Burkholder 2007)).

Table 1: Summary of studies relating water quality variables to seagrass growth and distribution.

Light attenuation	**Region**	**Location**
Dixon and Leverone 1995	Southwest Florida	Tampa Bay
Miller and McPherson 1995	Southwest Florida	Tampa Bay
Janicki and Wade 1996	Southwest Florida	Tampa Bay
Wang et al. 1999		
Dixon 2000	Southwest Florida	Tampa Bay
Dixon 2002	Southwest Florida	Tampa Bay
Tomasko et al. 2005	Southwest Florida	Tampa Bay
		Sarasota Bay
		Lemon Bay
		Upper Charlotte Harbor
Tomasko et al. 2001	Southwest Florida	Lemon Bay
Corbett and Hale 2006	Southwest Florida	Charlotte Harbor
Tomasko and Hall 1999	Southwest Florida	Charlotte Harbor
SFWMD	Southwest Florida	Caloosahatchee Estuary
Fourqurean and Zieman 1991	Southwest Florida	Florida Bay
Livingston et al. 1997	Northwest Florida	Apalachee Bay
Dawes et al. 2004	Florida gulf coast	Various
Iverson and Bittaker 1986	Florida gulf coast	Big Bed
		Florida Bay
Dunton 1994	Eastern Texas coast	Various
Gallegos and Kenworthy 1996	Atlantic coast Florida	Indian River Lagoon
Kenworthy and Fonseca 1996	Atlantic coast Florida	Indian River Lagoon
Christian and Sheng 2002	Atlantic coast Florida	Indian River Lagoon
Touchette and Burkholder 2007	Atlantic coast	Mesocosm (North Carolina)
Orth and Moore 1988	Atlantic coast	Chesapeake Bay
Batiuk et al. 1992	Atlantic coast	Chesapeake Bay
Dennison et al. 1993	Atlantic coast	Chesapeake Bay
Batiuk et al. 2000	Atlantic coast	Chesapeake Bay
Dennison 1987	Atlantic coast	Woods Hole, MA
Duarte 1991	Literature review	
Kenworthy and Haunert 1991	Literature review	
Morris and Tomasko 1993	Literature review	
Larkum et al. 2007	Literature review	

Salinity		
Wang et al. 1999	Southwest Florida	Tampa Bay
Phillips 1960a	Southwest Florida	Tampa Bay
		Boca Ciega Bay
		Tarpon Springs
Tomasko and Hall 1999	Southwest Florida	Charlotte Harbor
Doering et al. 2002	Southwest Florida	Caloosahatchee Estuary
SFWMD 2007	Southwest Florida	Caloosahatchee Estuary
Koch et al. 2007	Southwest Florida	Florida Bay
Phillips 1960b	Florida gulf coast	Crystal River
Dunton 1996	Eastern Texas coast	Various
Lirman and Cropper 2003	South Florida	Biscayne Bay
Touchette and Burkholder 2007	Atlantic coast	Mesocosm (North Carolina)

Temporal variability of salinity		
Phillips 1960a	Southwest Florida	Tampa Bay
		Boca Ciega Bay
		Tarpon Springs
Tomasko and Hall 1999	Southwest Florida	Charlotte Harbor
Doering et al. 2002	Southwest Florida	Caloosahatchee Estuary
SFWMD 2007	Southwest Florida	Caloosahatchee Estuary
Koch et al. 2007	Southwest Florida	Florida Bay
Phillips 1960b	Florida gulf coast	Crystal River
Lirman and Cropper 2003	South Florida	Biscayne Bay

Temperature		
Phillips 1960a	Southwest Florida	Tampa Bay
		Boca Ciega Bay
		Tarpon Springs
Tomasko and Hall 1999	Southwest Florida	Charlotte Harbor
SFWMD 2007	Southwest Florida	Caloosahatchee Estuary
Phillips 1960b	Florida gulf coast	Crystal River
Touchette et al. 2003	Atlantic coast	Mesocosm (North Carolina)
Touchette and Burkholder 2007	Atlantic coast	Mesocosm (North Carolina)
Burke et al. 1996	Atlantic coast	Chincoteague Bay

Tampa Bay Seagrass

There are three predominant seagrass species in Florida: T. testudinum, H. wrightii, and S. filiforme (Bergquist 1997). Changes in Tampa Bay seagrass distribution have reflected changes in water quality (Morrison et al. 2011). Field research has shown that water clarity and light attenuation are key habitat requirements for all seagrass species ((Duarte 1991), (Fourqurean and Zieman 1991), (Batiuk et al. 1992), (Dennison et al. 1993), (Gallegos and Kenworthy 1996), (Batiuk et al. 2000), (Larkum et al. 2007)).

In Tampa Bay, H. wrightii is the most abundant species of submerged aquatic vegetation at approximately 39.7% and is found in all Tampa Bay regions. H. wrightii is a euryhaline species tolerating a wide range of salinity levels (Dunton 1996) allowing it to be more widely distributed and dominate ecosystems when salinity is constant (Lirman and Cropper 2003). T. testudinum is the next most abundant species at approximately 16.8% and is found in lower regions of Tampa Bay. T. testudinum can tolerate intermediate levels of salinity change, but has been shown to experience a decrease in photosynthetic rate when it experiences a decrease in salinity (FWS 2014). S. filiforme is the third most abundant at approximately 12.5% (FWC 2011). S. filiforme is stenohaline tolerating a narrow range of salinity levels (FWS 2014). All three species prefer temperatures between 20 and 30 degrees Celsius (FWS 2014). Reductions in seagrass coverage occurred in Tampa Bay during a period of rapid coastal development and bay eutrophication that occurred between the 1950's and the 1980's ((Johansson 1991), (Johansson 2002), (Xian and Crane 2003), (Crane and Xian 2006)). Dredge-and-fill projects during the period of heightened coastal development resulted in the direct loss of seagrass beds (Janicki et al. 1995). Indirect seagrass losses were largely caused by reductions in light availability from algal build-up following excessive inputs of nutrients (Johansson and Greening 2000).

Seagrass coverage in Tampa Bay has been increasing in recent years, expanding by 4.8% between 2006 and 2008 (FWC 2011). Management strategies have been directed at improving water quality. Wastewater treatment plants were required to reduce nitrogen concentrations in outflow water by 90%, storm-water treatment regulations were put in place, and the fertilizer industry instituted practices to reduce nitrogen and phosphorus spills (NOAA 2012). While nitrogen loads were reduced by 60% between 1985 and 2003 (NOAA 2012), seagrass coverage is not increasing at the same rate throughout the bay (Greening et al. 2011). Non-point source pollution still contributes to water quality degradation (NOAA 2012). Chlorophyll-a and light attenuation levels are monitored by thirty seagrass management areas (Morrison et al. 2011). Transplantation efforts and the development of new beds have struggled due to a lack of nursery beds (Kellogg 2002).

Mapping and Restoration

The systematic mapping of seagrass patches in Tampa Bay was first initiated in 1982 (Greening et al. 2014). Landscape ecology of seagrass beds has focused on the spatial configuration of patches as the elementary component of seagrass landscape and dimension (Farina 1998). Patches display relative homogeneity with respect to some property at a particular scale and exhibit relatively abrupt boundaries adjacent to areas of different quality (Stine and Hunsaker 2001). The explicit examination of patch dynamics and spatial arrangement of seagrass beds has only recently been used (Bell et al. 2006).

Since 1988, seagrass polygons have been mapped approximately every two years by the Southwest Florida Water Management District (SWFWMD) (Greening et al. 2011) (Figure 3). Aerial photographs are used to classify seagrass beds as patchy or continuous, with patchy beds usually surrounding continuous beds (Robbins 1997). Both continuous and patchy coverage types provide critical and valued habitat functions (Dawes et al. 2004), although studies have shown that certain hard clam and scallop species thrive more successfully in continuous seagrass coverage ((Irlandi 1994), (Irlandi et al. 1995)). Issues arise in the interpretation of aerial photographs when turbidity leads to difficulty in identifying the presence of seagrass; estimations are usually considered to be underestimates (Bergquist 1997). The minimum observed cell size also presents an issue since fluctuations in patch dynamic can occur within the extent of a single recognized cell (Robbins 1997).

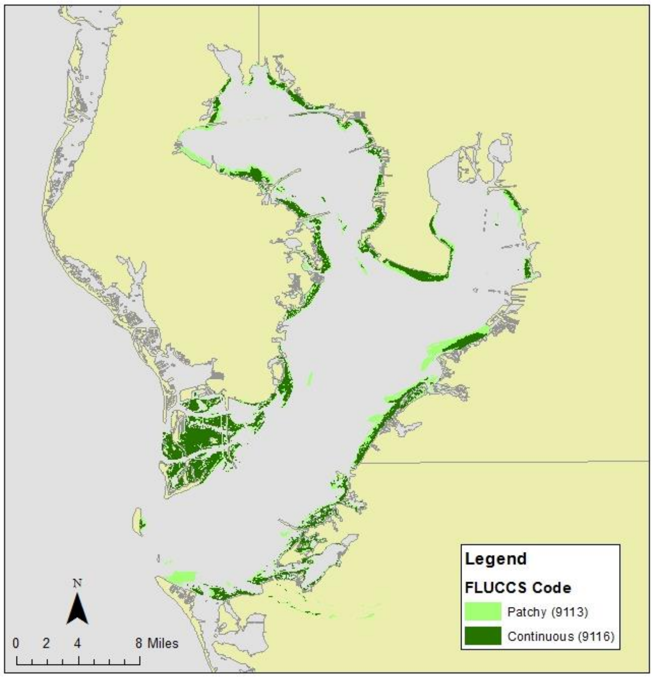

Figure 3: Seagrass distribution across Tampa Bay in 2012. Data obtained from SWFWMD.

Resource management strategies have focused on long-term maintenance of present seagrass coverage through the implementation of water clarity targets (Corbett and Hale 2006). The seagrass light attenuation requirement of 20.5% in Tampa Bay, developed by Dixon (2000), was used to establish initial water quality targets for seagrass restoration ((Greening 2002), (TBEP 2006)). Increases in seagrass coverage are expected if bay segment-specific water clarity targets are continuously met (Janicki and Wade 1996). The ultimate goal of the Tampa Bay Estuary Program is to increase coverage to 15,380 hectares baywide (Greening et al. 2011).

Habitat Modeling of Seagrass

Since the relationship between seagrass depth limit and light attenuation is common to a broad range of species, light attenuation has the potential for high predictive power of seagrass occurrence across species (Duarte 1991). Predictive mapping can provide a means to observe approximated spatial patterns of seagrass, predict the response of a habitat to disturbance, and select potential restoration sites (Kelly et al.

2001). Water quality data and seagrass presence data has been used in Florida Bay to construct "habitat requirements" by overlaying water quality conditions with existing seagrass extent to observe relationships among variables (Fourqurean et al. 2003). Logistic regression is commonly used for habitat suitability modeling because of its ability to predict the likelihood of binary dependent variables like seagrass presence and absence (Álvarez-Arbesú and Felicísimo 2002).

The predictive power of statistical models is constrained by the nature of available data (Hobbie 2000). Optical modeling and spatial interpolation of water quality point data are often used in conjunction with in situ observations to observe water quality characteristics across large areas ((Kirk 1983), (Gallegos and Kenworthy 1996), (Holtmann et al. 1996)). Predictive model success has been based on whether or not actual seagrass was observed in areas identified as having a high probability for seagrass occurrence, greater than 0.5 (Zarri et al. 2008). Geo-statistical analysis provides reportable error by taking spatial autocorrelation of data into account (Bell et al. 2006).

Kelly et al. (2001) investigated the spatial pattern of seagrass populations in coastal North Carolina in relation to bathymetry and developed a relative exposure index measuring seagrass exposure to wind-induced wave action. Shallow water and protection from waves were found to be significant factors influencing seagrass population success. Logistic multiple regression was used to create a meaningful habitat suitability model. Boolean logic was applied to determine possible restoration sites where seagrass populations could thrive. This model was also meant to be useful in predicting environmental response of the seagrass habitat to disturbance.

In southern Florida Santos and Lirman (2012) observed seagrass response to varying salinity levels. The sheet-flow pattern of water typical in the Everglades has been restructured into canals for human water management increasing centralized freshwater inputs into western Biscayne Bay. Light attenuation, depth, temperature, salinity at bottom, and seagrass presence by species were recorded across approximately nine hundred sites. This data was interpolated using ArcGIS, and factor maps were overlaid. Using BioMapper software each cell was scored for habitat suitability based on map layers. From this analysis it was observed that different seagrass species react differently to changes in salinity. H. wrightii increased by 71% when salinity was reduced while T. testudinum only increased by 18%. The model was used to predict the species' reaction to further increases in freshwater inflows.

Z. marina and Zostera noltei (dwarf eelgrass) are two temperate species of seagrass found in western Europe. Van der Heide et al. (2009) identified twenty-six environmental variables affecting growth of these seagrass species: surface water quality variables, sediment pore water variables, sediment variables, and hydrodynamic variables. A multiple stepwise logistic regression method was used with forward selection to identify which variables had the largest impact on seagrass success. Light attenuation, sediment pore-water reduction oxidation, and tidal state accounted for predicting 77% of the Z. marina and 86% of the Z. noltei populations. These conclusions showed that seagrass success is likely dependent on multiple factors. Since many factors often correlate with each other, observation of only a few variables may be necessary for accurately predicting habitat suitability.

Z. marina can also be found off the coast of Massachusetts in Plum Island Sound. Novak and Short (2012) created a habitat suitability model for the sound's seagrass population using bathymetry, sediment type, water quality and clarity, location of tidal flats, and wave exposure. Each variable was assigned a weight determined by its level of influence on seagrass populations. These variables were combined into a multiplicative rating system in order to spatially observe where seagrass habitats may be successful, and a map ranking areas of Plum Island Sound for seagrass habitat suitability was generated. The habitat suitability model was then used to identify possible locations for transplant communities.

Across several Maryland coastal bays, Carruthers and Wazniak (2004) investigated a multitude of water quality variables including chlorophyll-a, dissolved inorganic nitrogen, dissolved inorganic phosphorus, total suspended solids, and Secchi depth. Criterion for a successful seagrass habitat was identified for each variable by observing seagrass presence and the related variable levels. The model did not predict as well as expected, and the authors attributed this to a lack of data on total nutrients. They argued that dissolved nutrient levels were within an expected range while total nutrients were not, and total nutrients should be included in the model since they appear to be limiting seagrass growth.

Swett et al. (2013) attempted to classify regions of light attenuation in Estero Bay, Florida. PAR readings were recorded at locations throughout the bay. ArcGIS software was used to interpolate the observed in situ values spatially. Light attenuation was calculated across the bay using known bathymetry data combined with the interpolated data. Since seagrass success is often attributed to light attenuation, categorized light regions were created for the bay for further seagrass analysis.

In coastal Mississippi and Mobile Bay, Alabama Cho et al. (2011) investigated variables affecting seagrass populations in an attempt to classify categories of submerged aquatic vegetation. Variables such as substrate type, shoreline type, shoreline curve, number of inlets and outlets, stream order, distance to sound, shore aspect, stream width, shore-face profile, depth, and shoreline vegetation were recorded. After species analysis, three categories of submerged aquatic vegetation were classified based on similar combinations of variable requirements.

Meyer (2013) used seagrass populations in St. Joseph Sound and Clearwater Harbor, Florida to develop a habitat suitability model for the local population. Recorded variables included pH, salinity, total suspended solids, turbidity, light attenuation, depth, chlorophyll-a, and bottom type. Chlorophyll-a and turbidity were found to be highly correlated with light attenuation so they were removed from the model. Bathymetry and light attenuation were used to calculate the beam attenuation coefficient of transmittance, and it was found to be the key water quality variable affecting habitat prediction when compared to seagrass occurrence. A habitat suitability model was created using the attenuation coefficient and performed well predicting seagrass occurrence. Patchy areas of seagrass coverage provided the majority of the error in predictions.

In order to make the results of a habitat suitability model more accessible for resource managers, Mazzotti et al. (2007) incorporated seagrass stressor variables into an interactive GIS. A habitat suitability index was developed for T. testudinum and H. wrightii in the Caloosahatchee Estuary of Florida using light, salinity, temperature, and nutrient level variables. The effects of management and policy decisions were able to be modeled in the GIS to observe results. In addition to making the data accessible, a protocol and framework for analyzing the seagrass habitat was established for integrating the model into resource management activities.

Methodologies

A habitat suitability model for seagrass in Tampa Bay was developed using statistical analysis loosely coupled with data organized and manipulated in a GIS. The model was designed to predict seagrass distribution based on water quality variable influences on distribution. Data compilation, model development, model validation, and error analysis were the steps used to develop and evaluate the model.

Data Compilation

Data used in model development was retrieved from multiple agencies and manipulated in a GIS. In order to prepare all data for integration into the model, the following operations were performed: vector to raster conversion, point-to-raster conversion, reclassification, and map algebra (Table 2).

Table 2: Data sources and manipulation for model development preparation.

Layer	**Source**	**Manipulation**
Seagrass coverage	SWFWMD	Vector to raster conversion Reclassification
Water quality (temperature and salinity)	Hillsborough EPC	Point to raster conversion (interpolation) Map algebra (average)
Water quality (vertical attenuation coefficient)	MODIS Aqua	Map algebra (PLW equation)
Bathymetry	NOAA	Map algebra (PLW equation)

Data Layers

In order to maximize the predictive power of the developed model, seagrass data was collected from 2006, 2008, 2010, and 2012 aerial surveys. Upon review of literature relating seagrass success to certain limiting water quality variables, light attenuation, salinity, temporal variability of salinity, and temperature were chosen as potential model input variables. Since seagrass species are benthic, variables were observed at bottom. Variable data was averaged across a six-month period (reflecting the six-month growing season) preceding seagrass data collection in order to most accurately observe the water quality conditions affecting the observed seagrass presence for each year (Table 3). All data was integrated in ESRI's ArcMap 10.1 software and extensions and projected using the GCS_North_American_1983 projection and D_North_American_1983 datum. Seagrass and water quality variable data were masked to the extent of the study area. Statistical analysis was performed using SPSS software.

Table 3: Water quality averaging periods based on seagrass imagery flight dates.

Annual Seagrass Dataset (SWFWMD)	**Imagery Flight Dates**	**6 month water quality averaging period** (Hillsborough EPC and MODIS Aqua)
2006	February 2006	August 2005 – January 2006
2008	January – February 2008	July 2007 – December 2007
2010	January 2010	July 2009 – December 2009
2012	December 2011 – April 2012	June 2011 – November 2011

Data Sources

Seagrass

All seagrass coverage data (2006, 2008, 2010, and 2012) was obtained from SWFWMD. Coverage data was photo-interpreted from natural color aerial imagery at a scale of 1:24,000. Surveys are flown once every two years typically in early winter. Coverage data was classified using a modified form of the Florida Department of Transportation (FDOT) Florida Land Use, Cover and Forms Classification System (FLUCCS). Classes found in Tampa Bay include: open water (5400), tidal flats (6510), patchy seagrass coverage (9113), continuous seagrass coverage (9116), and green algae cover (9121) (Figure 4). This modified land use was digitized into polygons with a minimum mapping unit of 0.5 acres. Ground-truthing was performed by SWFWMD during the photo-interpretation phase to ensure classification accuracy. Data had a minimum classification accuracy of 90%.

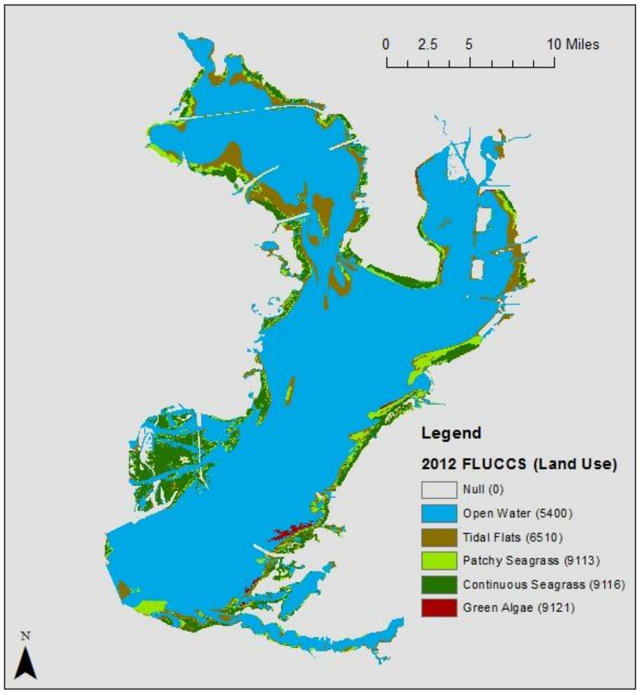

Figure 4: Benthic habitat classification in 2012 in Tampa Bay. Data obtained from SWFWMD.

Water Quality

All temperature and salinity data (2006, 2008, 2010, and 2012) was obtained from the U.S. Environmental Protection Agency's online STORET (STOre and RETrieval) water quality data retrieval system made available through the Environmental Protection Commission of Hillsborough County (Hillsborough County EPC). Water quality was sampled monthly from distinct station locations in Tampa Bay (Figure 5) and presented in spreadsheet form. Temperature and salinity data was taken in situ. Temperature was measured at bottom in degrees Celsius, and salinity was measured at bottom in parts per trillion.

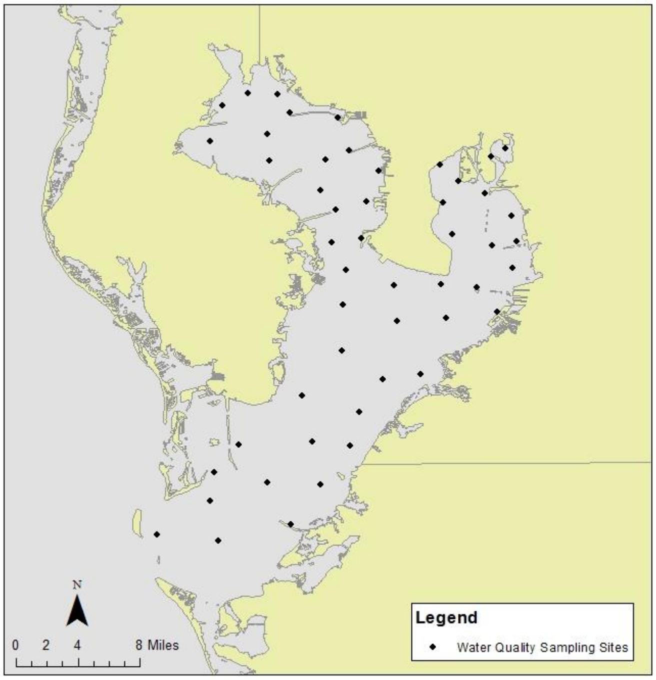

Figure 5: Hillsborough County EPC STORET water quality sampling sites in Tampa Bay.

Vertical attenuation coefficient data (2006, 2008, 2010, and 2012) was obtained from the National Aeronautics and Space Administration (NASA) made available through the Giovanni online interface. The Giovanni interface provides access to remotely sensed data acquired using a moderate resolution imaging spectroradiometer (MODIS) on NASA's Aqua satellite. The vertical attenuation coefficient was recorded at 490 nanometers at a horizontal scale of 4 kilometers every eight days.

Bathymetry

Bathymetry data was obtained from the National Ocean Service (NOS) and the U.S. Geological Survey (USGS). Bathymetric and topographic data sets were merged into a hybrid elevation model for Tampa Bay and made available by the National Oceanic and Atmospheric Administration (NOAA). Hydrologic data was derived from eighteen sounding surveys conducted between 1945 and 1958 and reported in meters below mean low water. Depth data was taken at a 30 meter horizontal resolution and 0.1 meter vertical resolution.

Data Manipulation

Seagrass

Seagrass coverage available as vector polygons was classified according to the following FLUCCS codes: patchy coverage (9113) and continuous coverage (9116). Coverage data was converted to a raster grid. A grid cell size of 9 meters (one fifth of the original minimum mapping unit) was chosen based on the Whittaker-Shannon Sampling Theorem. Raster grid cells were then reclassified to identify the presence or absence of seagrass. Patchy and continuous seagrass coverage was classified as 1. All other benthic habitats were classified as 0.

Water Quality

Monthly in situ temperature and salinity data were interpolated using the Inverse Distance Weighting (IDW) algorithm, available with the ESRI ArcMap 10.1 Spatial Analyst extension. The interpolated surface was created based on point data by assigning a proportionate weight to nearby cells based on the value at the source cell, producing a raster grid. Interpolation was performed with a cell size of 0.0001, power of 2, and 12 points. A raster grid was created for each variable for each month in the corresponding water quality-averaging period. The six raster grids for each year were averaged (Figure 6) to create final temperature and salinity raster grids for each year of seagrass data (2006, 2008, 2010, and 2012). Temporal variability of salinity (TVS) was determined using the standard deviation of in situ salinity measurements at each sampling station across the sixth month water quality averaging period. The standard deviation values at each sampling station were also interpolated using the IDW algorithm to create a TVS raster grid for each corresponding year.

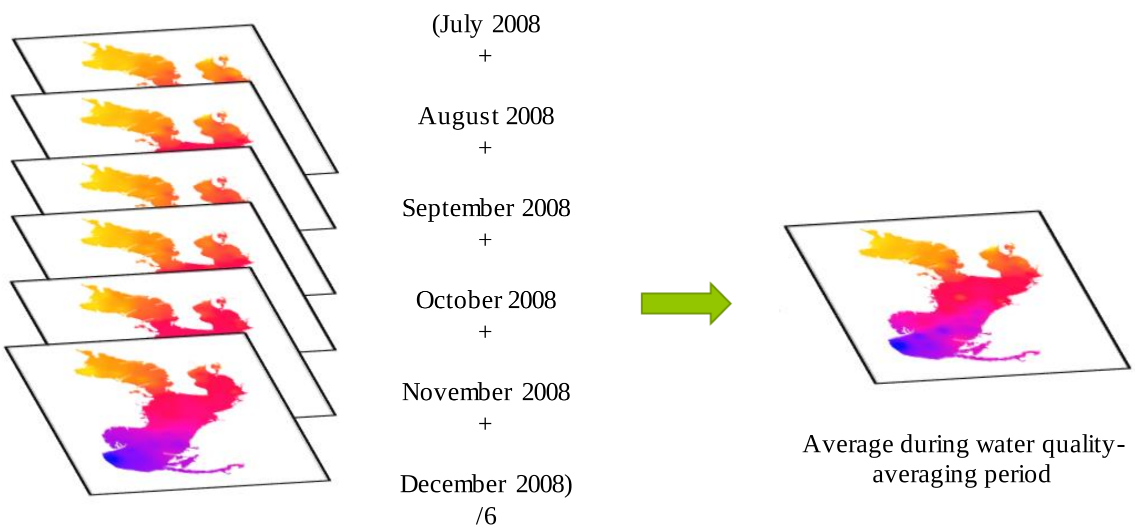

Figure 6: Water quality averaging example using Raster Calculator in ArcMap.

Remotely sensed vertical attenuation coefficient data was used with bathymetry data to calculate the percent light through water at depth (PLW) (Figure 7) (Figure 8). Vertical attenuation coefficient data was time-averaged by the Giovanni interface in relation to the water quality averaging periods. The following equation was used to calculate the percent light through water at given depth (z) in the Raster Calculator in ArcMap 10.1:

PLW = 100*e^(-k*z)

where, k is the vertical attenuation coefficient, and

z is the bathymetry depth value.

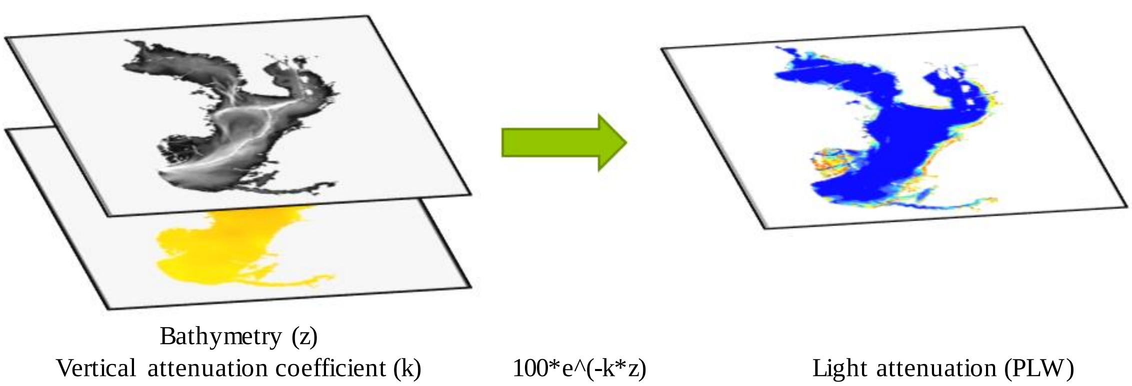

Figure 7: Light attenuation raster created with the Beer-Lambert equation in the Raster Calculator using bathymetry and vertical attenuation coefficient rasters.

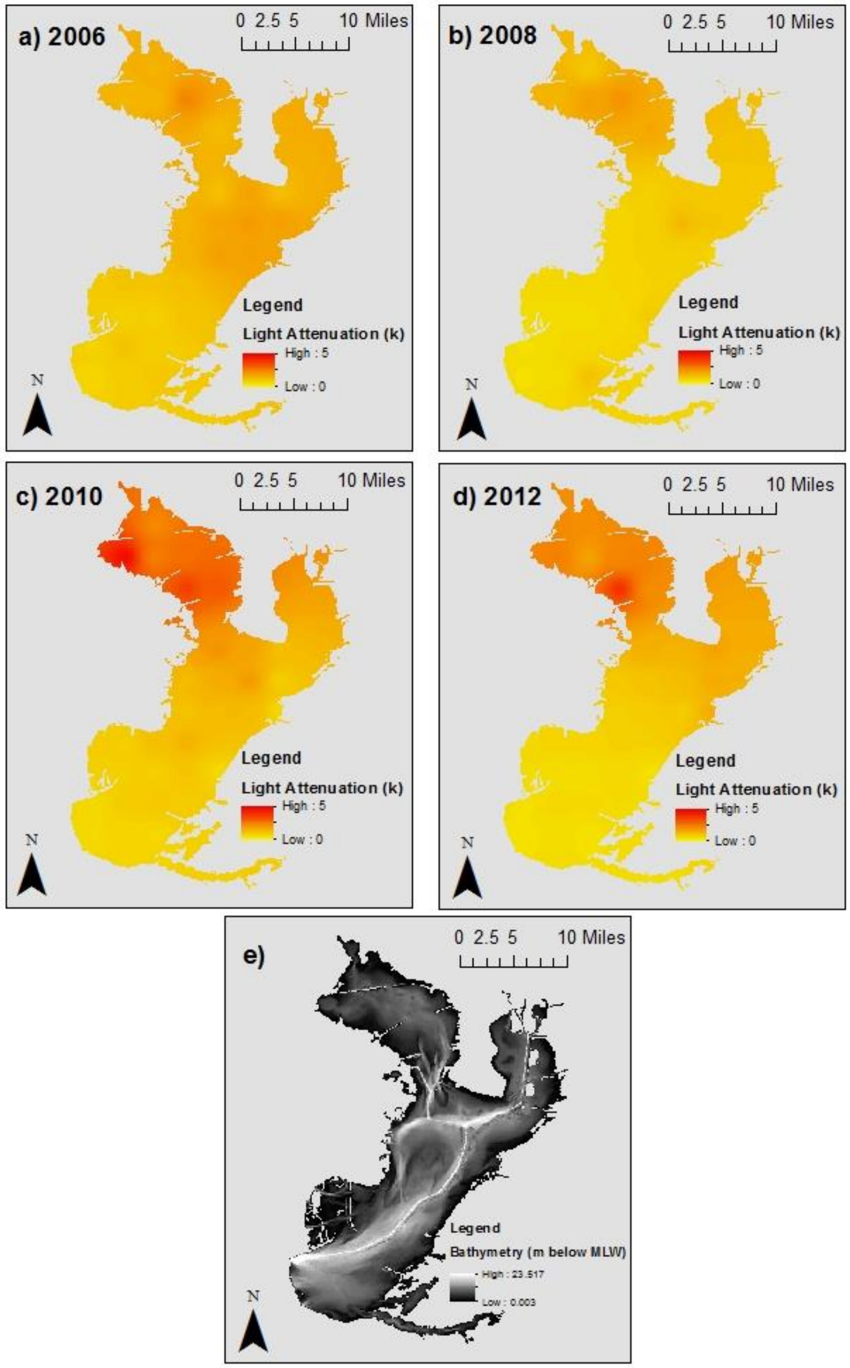

Figure 8(a-e): Remotely sensed vertical attenuation coefficient (k) for the six months preceding seagrass data collection in 2006 (a), 2008 (b), 2010 (c) and 2012 (d). Vertical attenuation coefficient combined with bathymetry data (e) to determine light attenuation (PLW). Vertical attenuation coefficient data obtained from NASA. Bathymetry data obtained from NOAA.

Model Development

All final data layers for each year were overlaid in ArcMap with seagrass as the dependent variable and the water quality variables (light attenuation, salinity, temporal variability of salinity, and temperature) as the independent variables (Figure 9). The GIS “Extract Multi Values to Points” tool was used to create a dataset of points on a 60-meter grid containing information from all overlaid map layers. This was repeated for each year and merged into one data set resulting in a sample size of 975,673 cells.

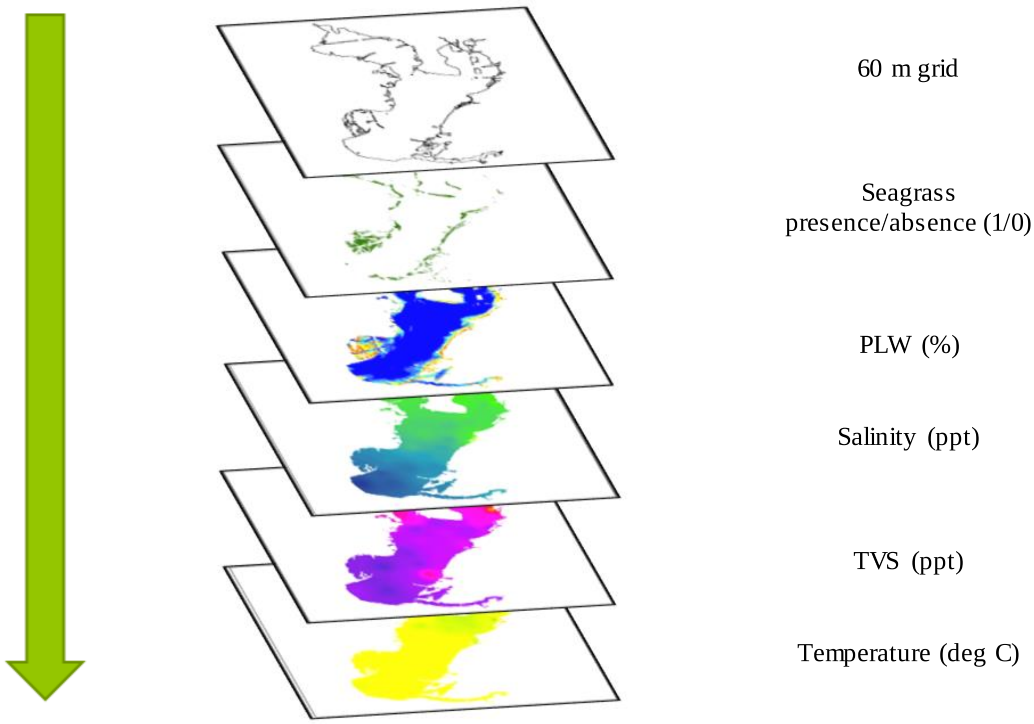

Figure 9: Intersection of dependent and independent variables.

Using the random selection tool in the ESRI ArcMap 10.1 Hawth's Analysis Tools extension, 50% of the points (487,837) were randomly selected and exported to a statistical program (SPSS). With seagrass presence reclassified as present (1) or absent (0), step-wise logistic regression was performed on the binary dependent variable in order to create a regression model. The developed model was intended to predict the probability of seagrass presence based on a combination of the water quality variables. Habitat suitability models often rely on logistic regression because of its ability to predict the occurrence of a binary variable like presence or absence (Álvarez-Arbesú and Felicísimo 2002). The logit transformation of the logistic regression model is the following:

$P = 1 / [1 + \exp^{(-Y)}]$ where,

P is the probability of seagrass presence from 0 to 1

Y is the predictor linear model $\beta_0 + \beta_{PLW} + \beta_{Sal} + \beta_{TVS} + \beta_{Temp}$

β_0 is the constant coefficient, and

$\beta_{PLW, x\ldots}$ are the model variable coefficients.

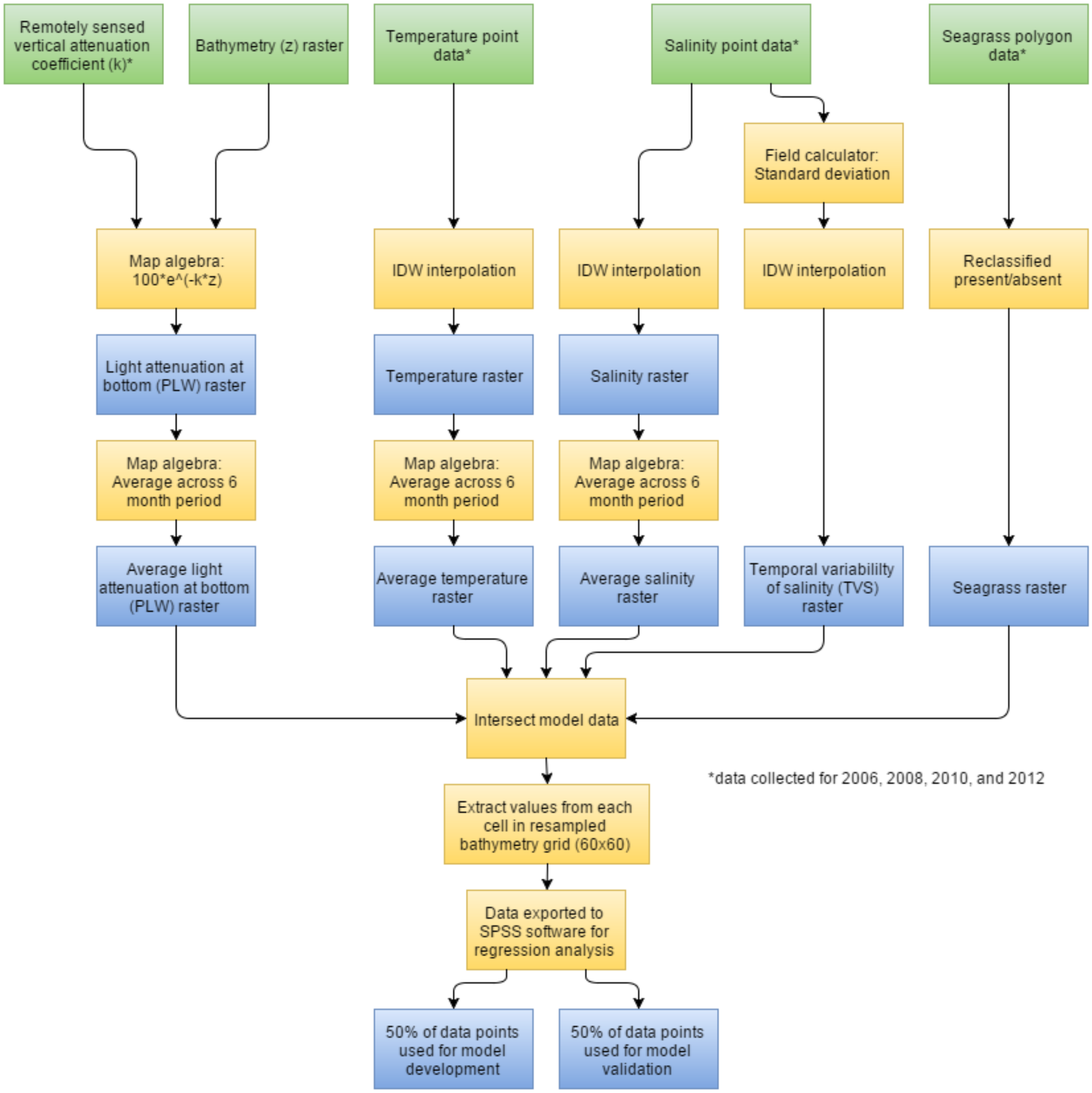

Figure 10: Flowchart of data compilation and manipulation.

Model Validation

The regression model was validated by comparing the predicted probability of seagrass presence with observed seagrass presence data. Using the 50% of data points reserved for model validation (487,836), the logistic regression model was applied to the data using the Field Calculator in ArcMap 10.1 in order to determine a probability of seagrass presence for each data point. The probability values were then reclassified as unlikely (0) with a probability less than 0.5 or likely (1) with a probability greater than or equal to 0.5 (Zarri et al. 2008). The success of the model was evaluated based on how accurately it matched a likely probability with the presence of seagrass or an unlikely probability with the absence of seagrass.

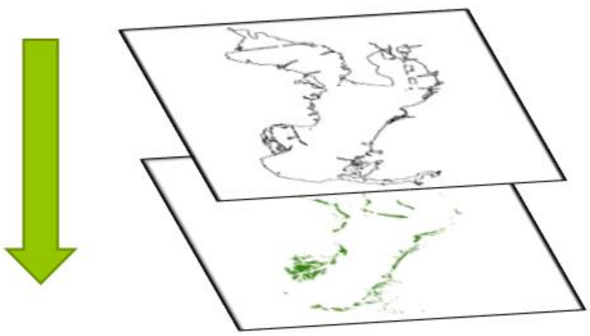

Figure 11: Comparison of predicted seagrass distribution with observed seagrass distribution.

Error Analysis

Prediction error was evaluated by subtracting seagrass occurrence (0 or 1) from the reclassified probability (0 or 1) (Table 4). If the probability was unlikely (0), but seagrass was present (1), seagrass was under-predicted (-1). If the probability was unlikely (0), and seagrass was not present (0), or the probability was likely (1), and seagrass was present (1), seagrass was predicted accurately (0). If the probability was likely (1), and seagrass was not observed (0), seagrass was over-predicted (1). Bias in the model was investigated by displaying under-prediction and over-prediction values as a function of each water quality variable to observe error distribution.

Table 4: Error calculations using predicted seagrass distribution compared to observed seagrass distribution.

Under-prediction		
Probability	Low	0
Seagrass	Present	-1
		-1
Accurate		
Probability	Low	0
Seagrass	Absent	-0
		0
Probability	High	1
Seagrass	Present	-1
		0
Over-prediction		
Probability	High	1
Seagrass	Absent	-0
		1

Results

Data Compilation

Seagrass

Seagrass coverage increased from approximately 8,648 hectares to 11,225 hectares between 2006 and 2012 (Figure 12). Continuous coverage increased slightly each year. Patchy coverage decreased between 2006 and 2008 but increased significantly between 2008 and 2012. Patchy coverage made up 35.4% of all seagrass coverage in 2012.

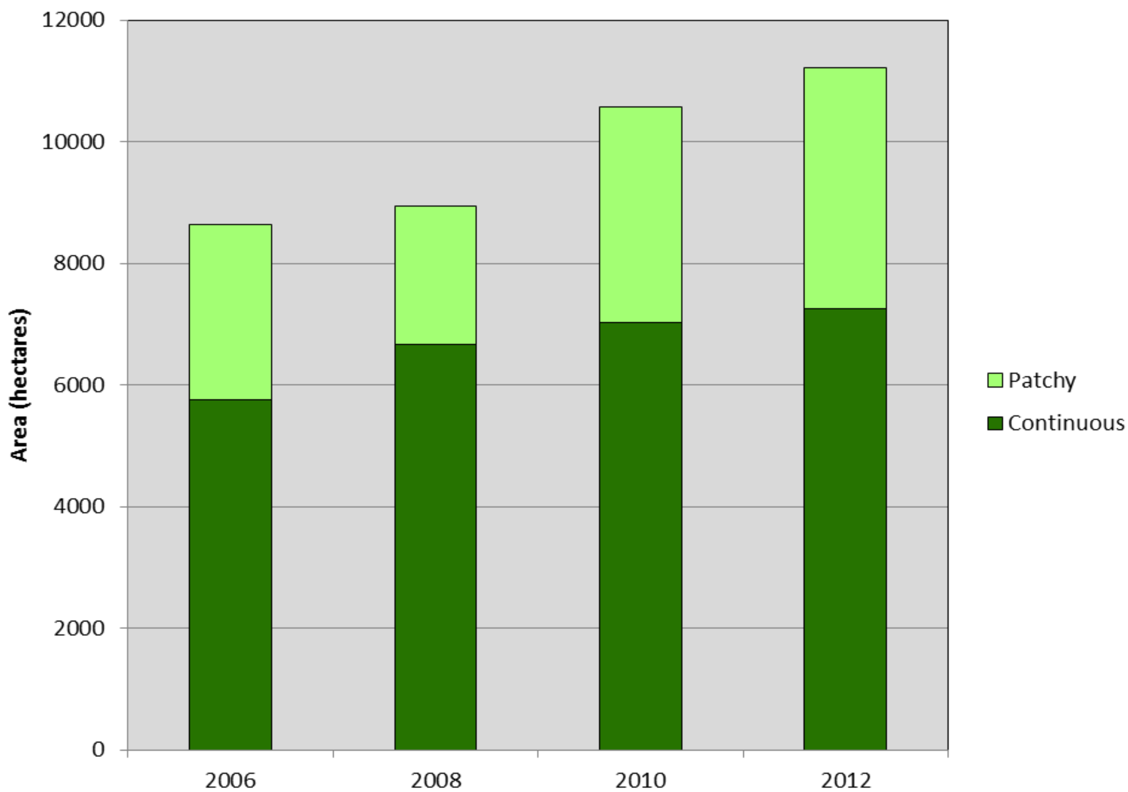

Figure 12: Total seagrass coverage by year classified as patchy (9113) or continuous (9116).

PLW

Total seagrass coverage generally increased with increasing light attenuation with a consistent decrease above 90% light attenuation across all years (Figure 13). Most seagrass was observed in areas receiving between 70% and 90% of available light. As light attenuation increased, continuous seagrass coverage increased while patchy seagrass coverage tended to decrease. As a result, continuous coverage seems to dominate areas of high light attenuation while patchy coverage is found more abundantly in areas of lower light attenuation.

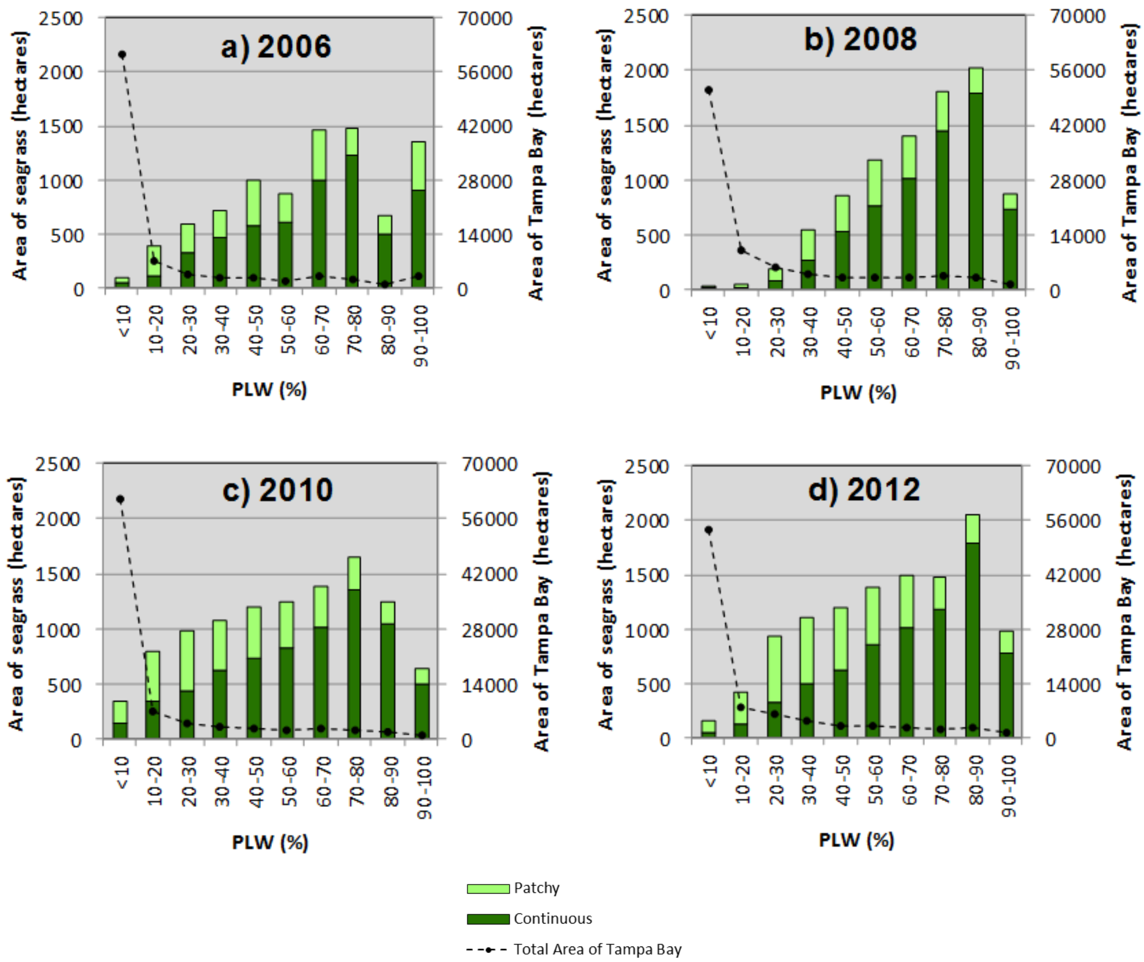

Figure 13(a-d): Seagrass coverage classified as patchy (9113) and continuous (9116) and total coverage of Tampa Bay at fixed percent light attenuation intervals in 2006 (a), 2008 (b), 2010 (c) and 2012 (d).

The increasing trend of total coverage was expected due to the light requirements of seagrass. The drop-off in total coverage observed above 90% light attenuation was not expected, but may be due to increased disturbance in extremely shallow areas. This disturbance may be a result of exposure to wave action at the surface, human-related activities, or the dominance of macroalgae.

Spatial distribution of light attenuation appeared to be related to bathymetry. Light attenuation was highest along the coasts with zones of significant area in the southwestern and southeastern portions of the bay (Figure 14). Old Tampa Bay and Hillsborough Bay experienced a slight decrease in light attenuation between 2006 and 2012 while Lower Tampa Bay experienced an increase.

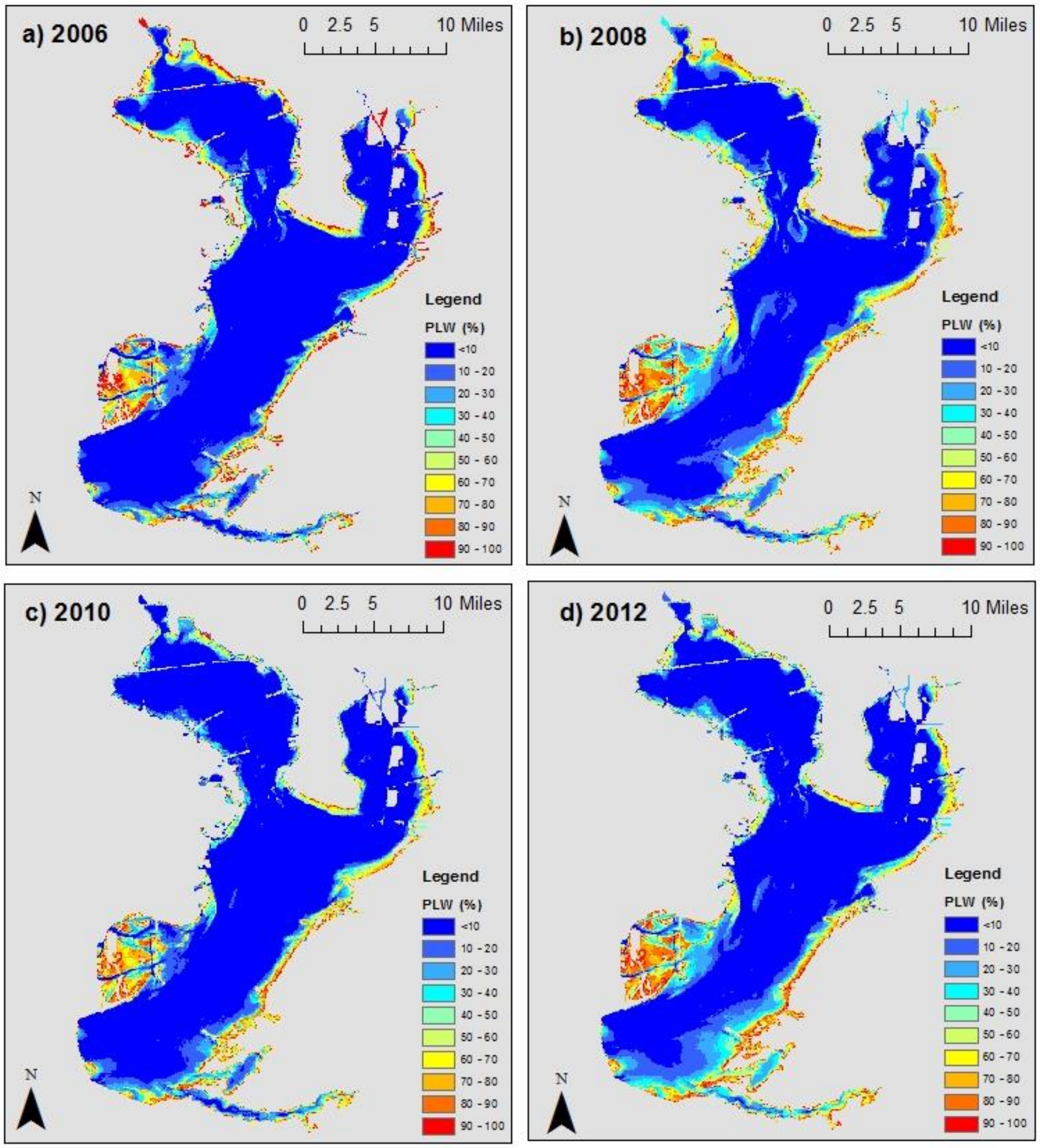

Figure 14(a-d): Average light attenuation for the six months preceding seagrass data collection in 2006 (a), 2008 (b), 2010 (c) and 2012 (d).

Salinity

Seagrass coverage was greatest in two different salinity zones (Figure 15). The salinity level of these zones varied across all years, although the greatest amount of continuous coverage seemed to occur most frequently around 32 parts per trillion. These two zones appear to be related to the salinity characteristics of the bay itself since the total coverage of Tampa Bay in relation to salinity also consistently showed two zones. This suggests that Tampa Bay has two major areas of differing salinity. Seagrass was found in both, although seemed to prefer the higher salinity level zone.

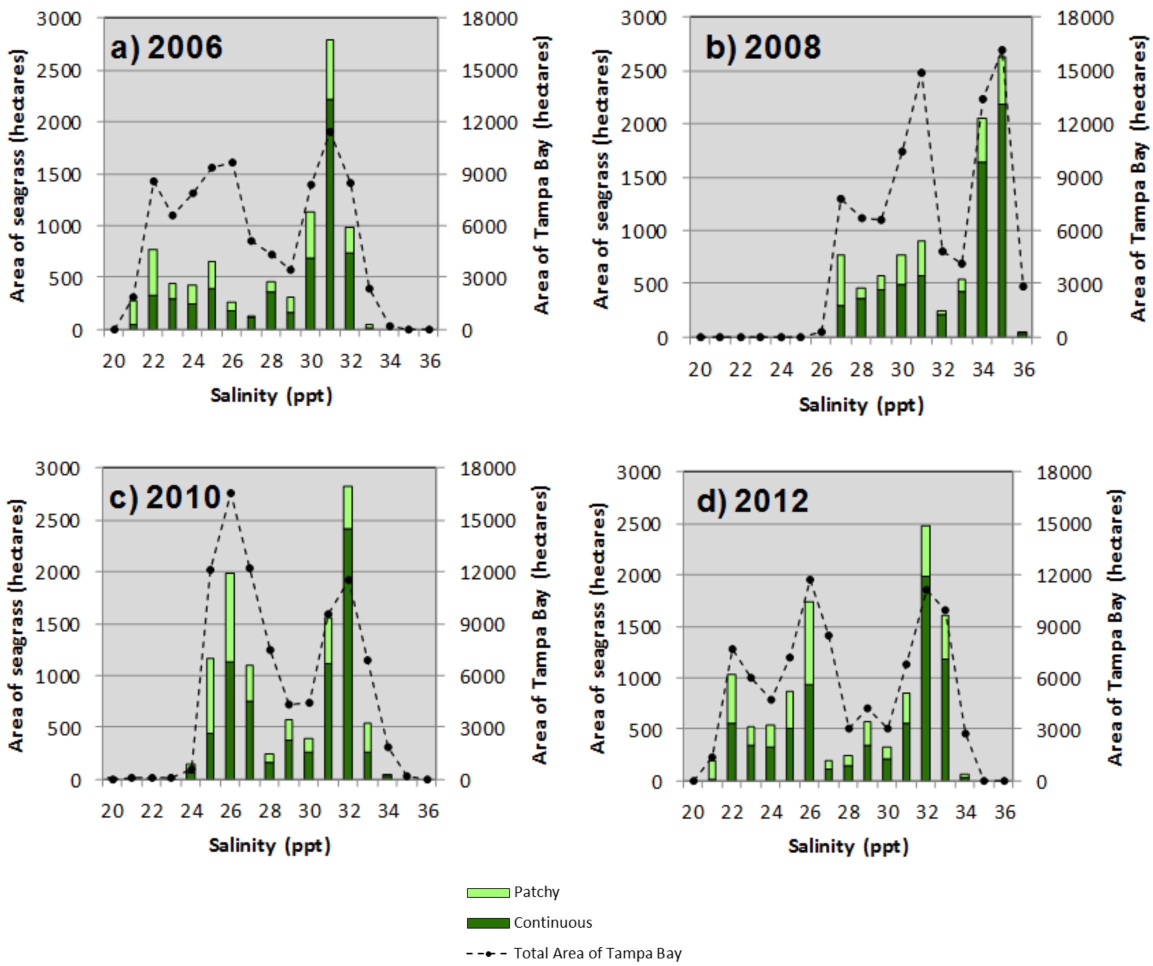

Figure 15(a-d): Seagrass coverage classified as patchy (9113) and continuous (9116) and total coverage of Tampa Bay at fixed salinity intervals in 2006 (a), 2008 (b), 2010 (c) and 2012 (d).

Spatial distribution of salinity appeared to be heavily influenced by the Gulf of Mexico (Figure 16). Areas in Lower Tampa Bay near the Gulf of Mexico consistently experienced the highest levels of salinity in all years. Areas in Old Tampa Bay, farthest from the Gulf of Mexico, experienced the lowest levels of salinity in all years. Slight decreases in salinity along the eastern coast of Hillsborough Bay were also observed between 2006 and 2012.

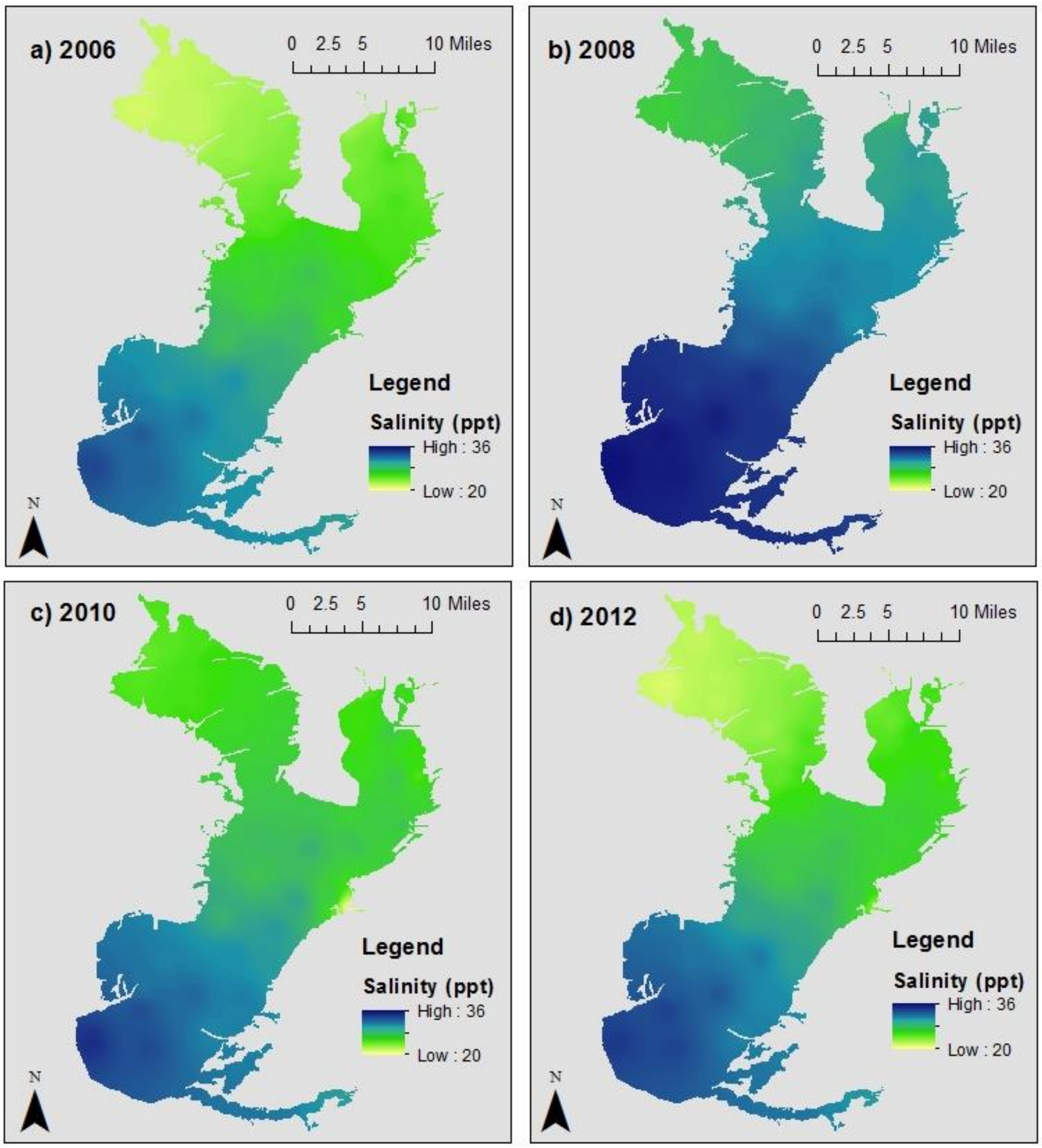

Figure 16(a-d): Average salinity for the six months preceding seagrass data collection in 2006 (a), 2008 (b), 2010 (c) and 2012 (d).

TVS

Seagrass coverage was greatest in areas of lower temporal variability (Figure 17), although the amount of variability experienced by Tampa Bay was inconsistent between years. In 2006, 2008 and 2010, the greatest amount of coverage existed most frequently in areas experiencing fluctuations of about 0.7 parts per trillion. In 2012, the greatest amount of coverage was near 1.8 parts per trillion. Despite differing lower limits of variability, all years showed a decreasing trend in seagrass coverage as variability increased.

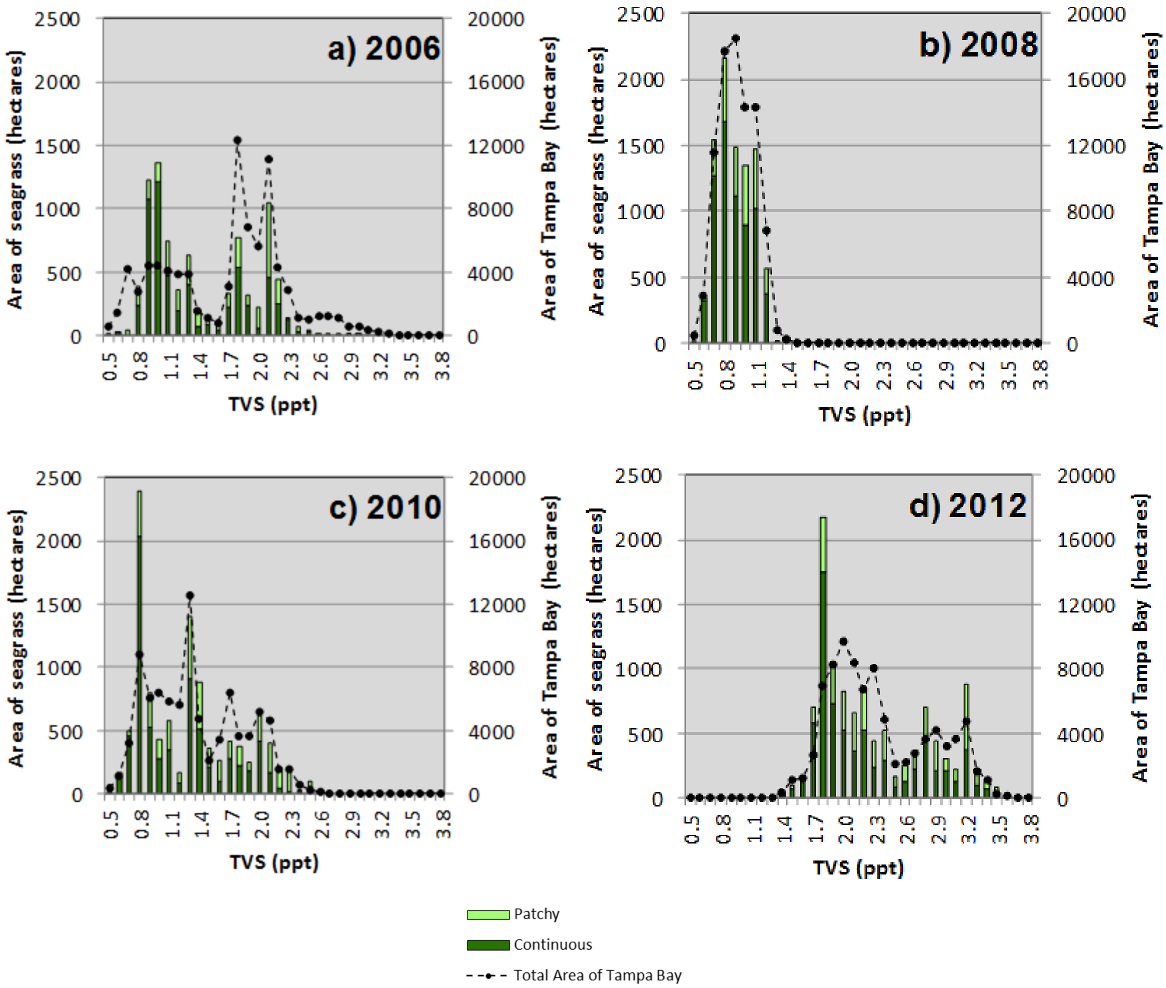

Figure 17(a-d): Seagrass coverage classified as patchy (9113) and continuous (9116) and total coverage of Tampa Bay at fixed TVS intervals in 2006 (a), 2008 (b), 2010 (c) and 2012 (d).

Spatial distribution of temporal variability showed the highest variability in Old Tampa Bay and Hillsborough Bay (Figure 18) across years with high variability. The lowest variability was consistently at the mouth of the bay near the Gulf of Mexico across all years. 2012 showed the greatest variability bay-wide with the greatest amount in Old Tampa Bay.

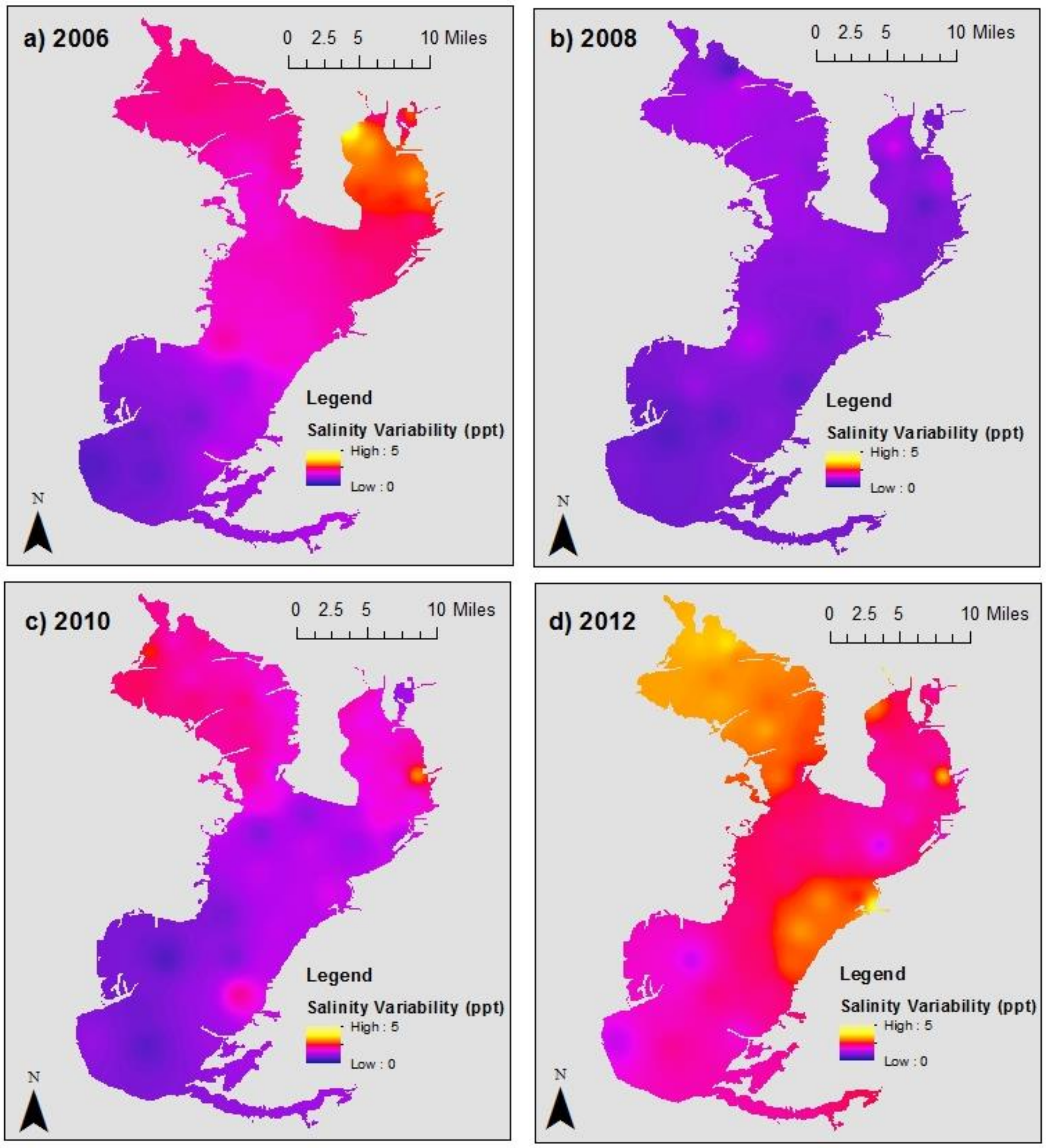

Figure 18(a-d): Average TVS for the six months preceding seagrass data collection in 2006 (a), 2008 (b), 2010 (c) and 2012 (d).

Temperature

Seagrass coverage appeared to adapt to changing bay water temperatures. The temperature of the bay increased between 2006 and 2012, and the greatest seagrass coverage occurred in areas of the most common average temperature (Figure 19) in each year. Despite the increasing temperature of the bay as a whole between years, each individual year showed moderate homogeneity in temperature across the bay with the exception of heightened temperatures in Hillsborough Bay in 2012 (Figure 20). Across all years observed bay waters never fluctuated outside of the ideal temperature range for Tampa Bay seagrass species (20 and 30 degrees Celsius) (FWS 2014).

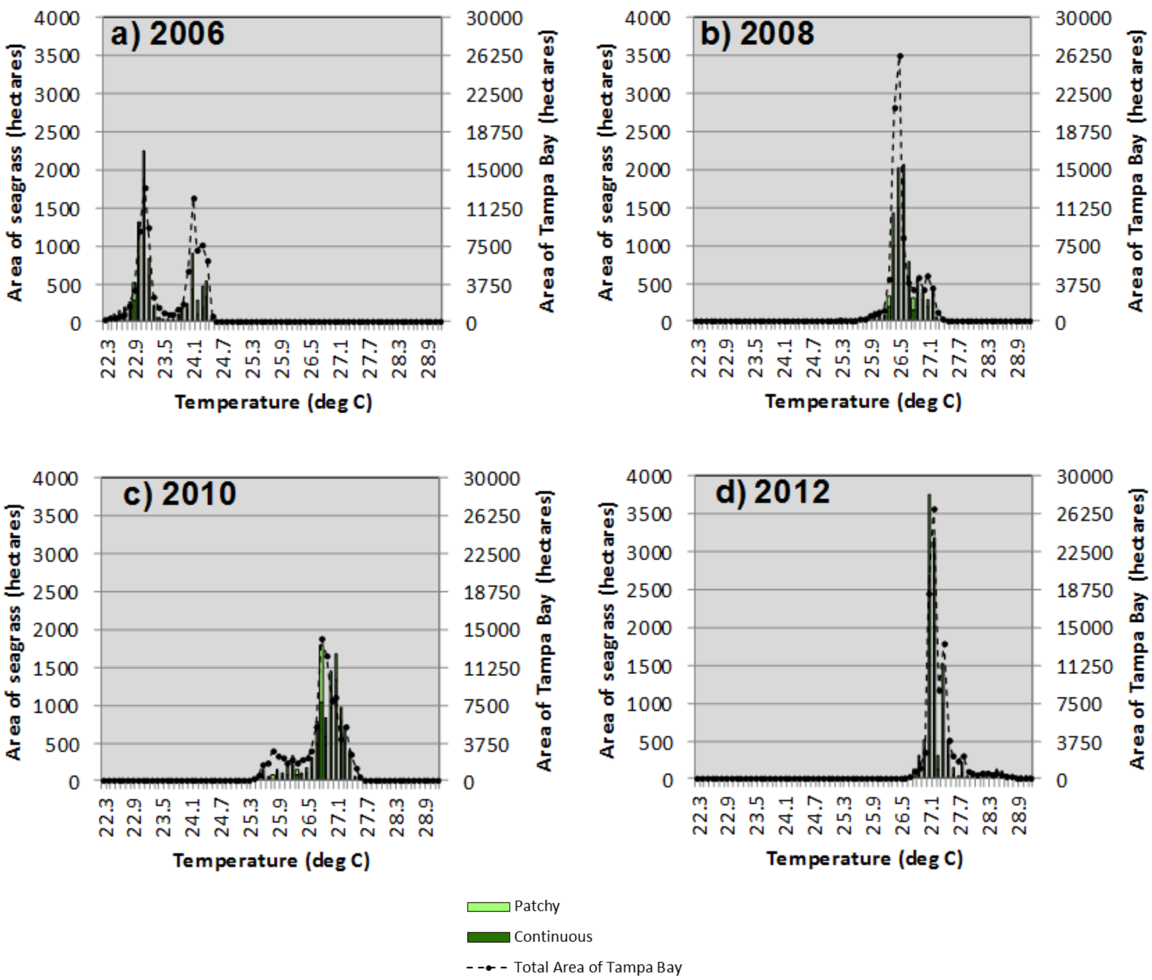

Figure 19(a-d): Seagrass coverage classified as patchy (9113) and continuous (9116) and total coverage of Tampa Bay at fixed temperature intervals in 2006 (a), 2008 (b), 2010 (c) and 2012 (d).

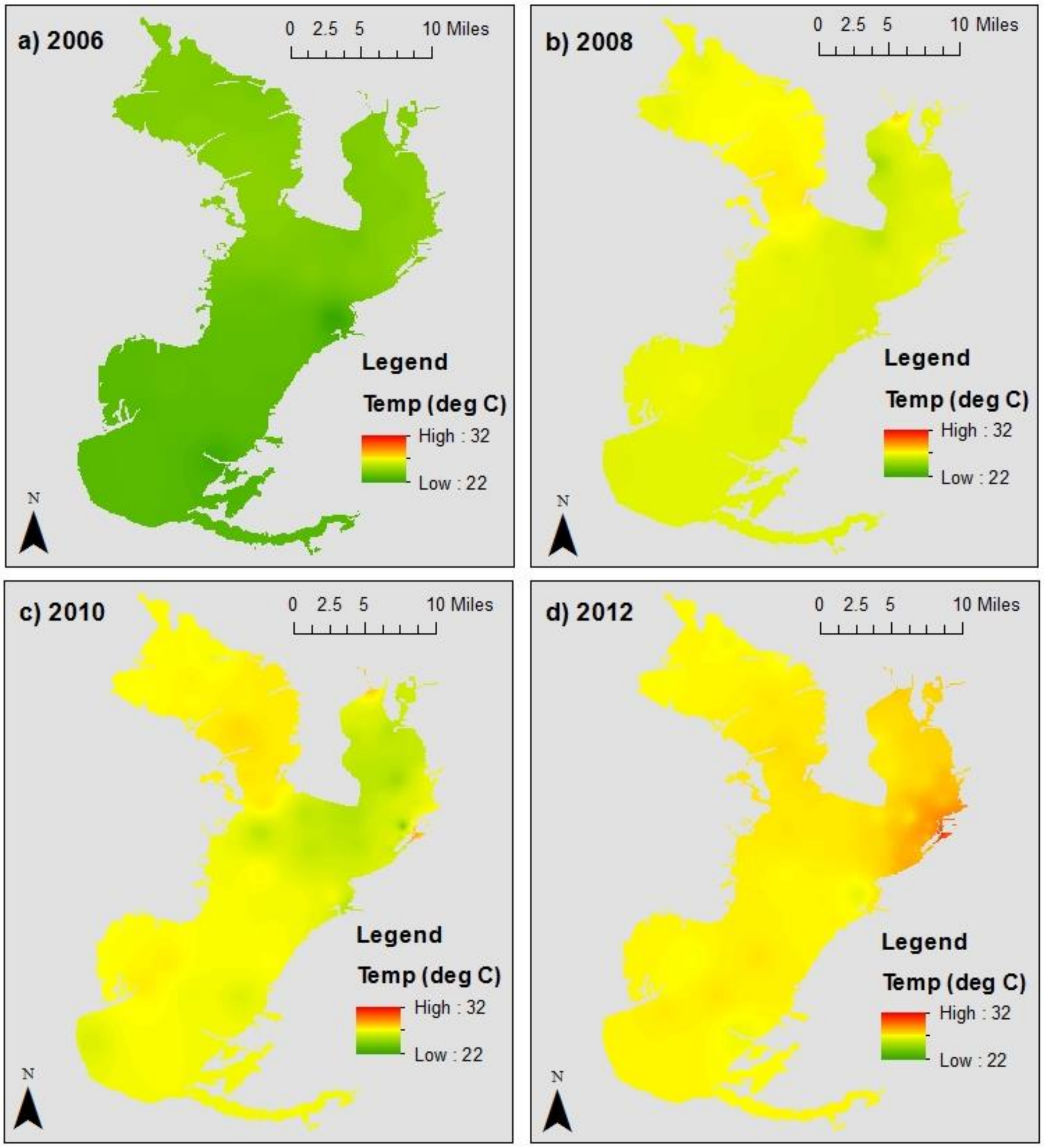

Figure 20(a-d): Average temperature for the six months preceding seagrass data collection in 2006 (a), 2008 (b), 2010 (c) and 2012 (d).

Model Development

Since temperature was found to remain within tolerated levels, it was removed from consideration in the model. Multiple logistic regression with a forward stepwise maximum likelihood method was used on data from the GIS to determine how significantly light attenuation, salinity and temporal variability of salinity affected seagrass occurrence (Table 5). The logit transformation of the logistic regression model was the following:

$P = 1 / [1 + \exp^{(-Y)}]$ where,

$Y = -3.210 + 0.061_{PLW} - 0.026_{Sal} - 0.036_{TVS}$

Y = linear model equation, listing model terms in the same order they were entered in the forward stepwise model

Based on the logistic regression results, seagrass was most accurately predicted by light attenuation with a positive relationship; increasing light attenuation resulted in an increasing amount of seagrass coverage. Salinity and temporal variability of salinity had a negative relationship to seagrass occurrence; increasing salinity levels and temporal variability of salinity resulted in a decreasing amount of seagrass coverage.

Table 5: Logistic regression output used for model development.

		B	S.E.	Wald	df	Sig.	Exp(B)	95% C.I.for Exp(B)		Nagelkerke R Square
								Lower	Upper	
Step 1[a]	PLW	0.06	0	93698.287	1	0	1.062	1.062	1.062	
	Constant	-4.003	0.011	144369.4	1	0	0.018			0.493
Step 2[b]	PLW	0.061	0	92647.951	1	0	1.062	1.062	1.063	
	Sal	-0.021	0.002	176.338	1	0	0.979	0.976	0.982	
	Constant	-3.401	0.046	5403.679	1	0	0.033			0.494
Step 3[c]	PLW	0.061	0	92692.087	1	0	1.062	1.062	1.063	
	Sal	-0.026	0.002	135.314	1	0	0.975	0.97	0.979	
	TVS	-0.036	0.012	9.128	1	0	0.964	0.942	0.987	
	Constant	-3.21	0.079	1670.154	1	0	0.04			0.494

a. Variable(s) entered on step 1: PLW.

b. Variable(s) entered on step 2: Sal.

c. Variable(s) entered on step 3: TVS.

Model Validation

A large amount of seagrass was present in areas with a low predicted probability (Figure 21) and declined with increasing probability. At probabilities above 0.5, the presence of seagrass increased with increasing probability, but decreased at probabilities above 0.8. Significantly higher percentages of continuous coverage compared to patchy coverage were observed at probabilities above 0.5. Patchy coverage and continuous coverage were more evenly distributed at probabilities below 0.3.

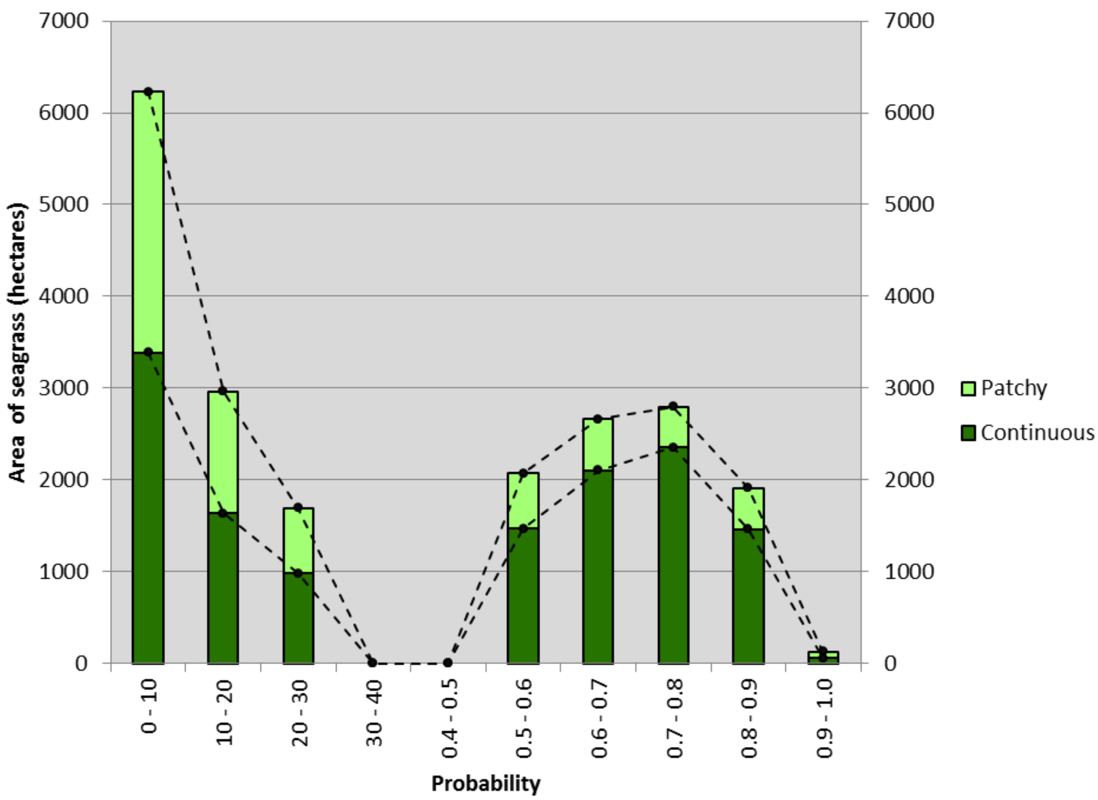

Figure 21: Probability of seagrass coverage classified as patchy (9113) and continuous (9116) predicted by the logistic regression function compared to data reserved for model validation.

As probability increased, light availability increased as well. Observed seagrass coverage matched this increasing trend between probabilities of 0.5 and 0.8. Salinity levels matched the trend of observed seagrass coverage except at probabilities below 0.3. Temporal variability of salinity showed an inverse relationship to observed seagrass coverage except at probabilities below 0.3 (Figure 22).

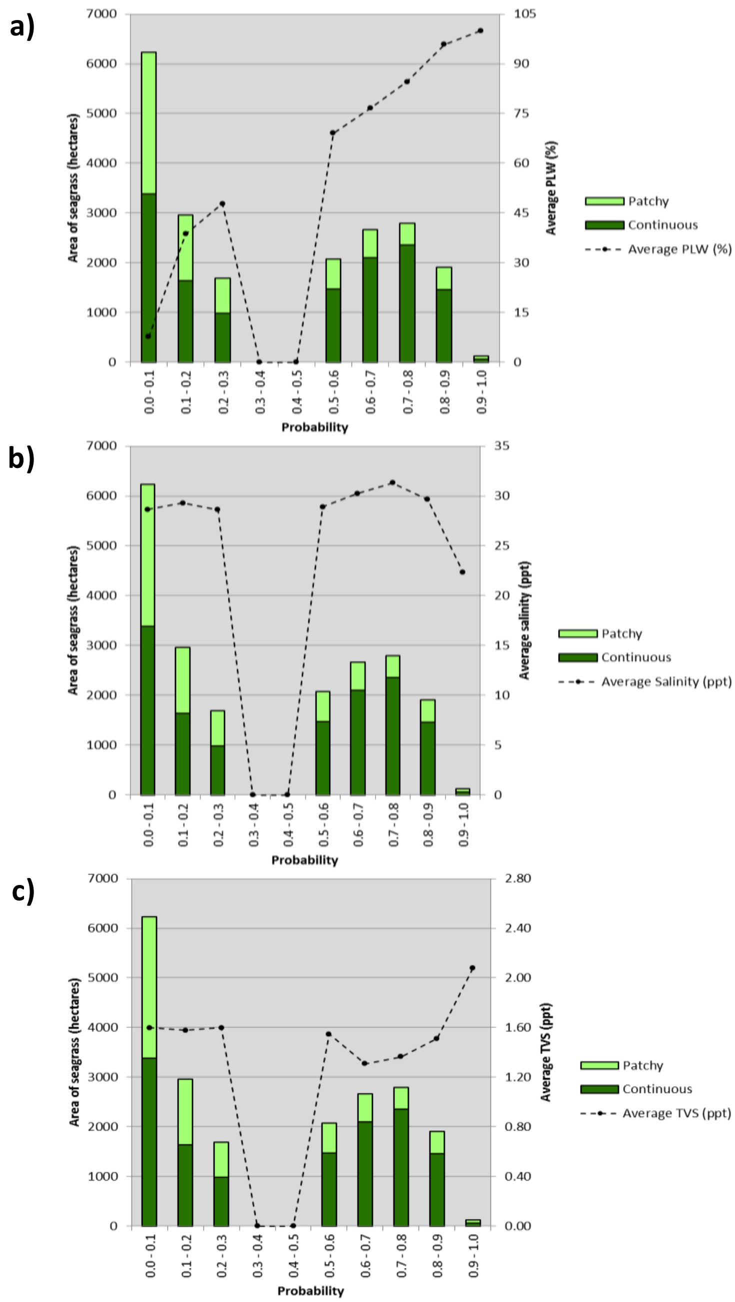

Figure 22(a-c): Probability of seagrass coverage classified as patchy (9113) and continuous (9116) predicted by the logistic regression function compared to data reserved for model validation and light attenuation (a), salinity (b), and TVS (c).

Error Analysis

Model prediction was classified as an under-prediction (-1), accurate (0), or an over-prediction (1) in order to perform a residual analysis. This was determined by comparing model prediction probability with observed seagrass occurrence. The habitat suitability model accurately predicted approximately 48.2%, under-predicted 50.5%, and over-predicted 1.3% of seagrass coverage in Tampa Bay (Figure 23). There was a significant difference in prediction accuracy of patchy versus continuous seagrass coverage. The model accurately predicted approximately 55.8% of continuous coverage and only 33.5% of patchy coverage.

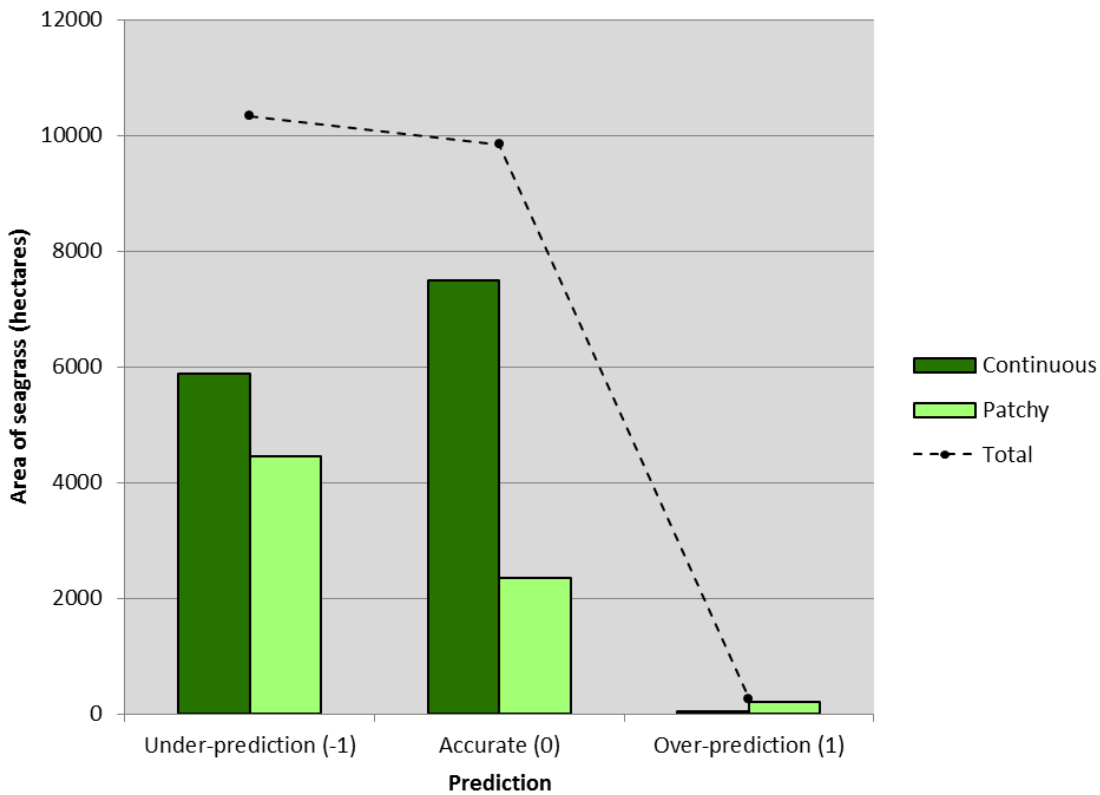

Figure 23: Seagrass coverage classified as patchy (9113) or continuous (9116) cover in each category of error.

Prediction error was observed as a function of each water quality variable in order to analyze bias in the model. Significant bias was observed in light attenuation values since under-prediction was more common in levels below 60%, and over-prediction was more common in levels above 70%. Little bias was observed in salinity values since prediction accuracy was more evenly distributed over the range of observed salinity. Little bias was also observed in temporal variability of salinity since prediction accuracy was also evenly distributed (Figure 24).

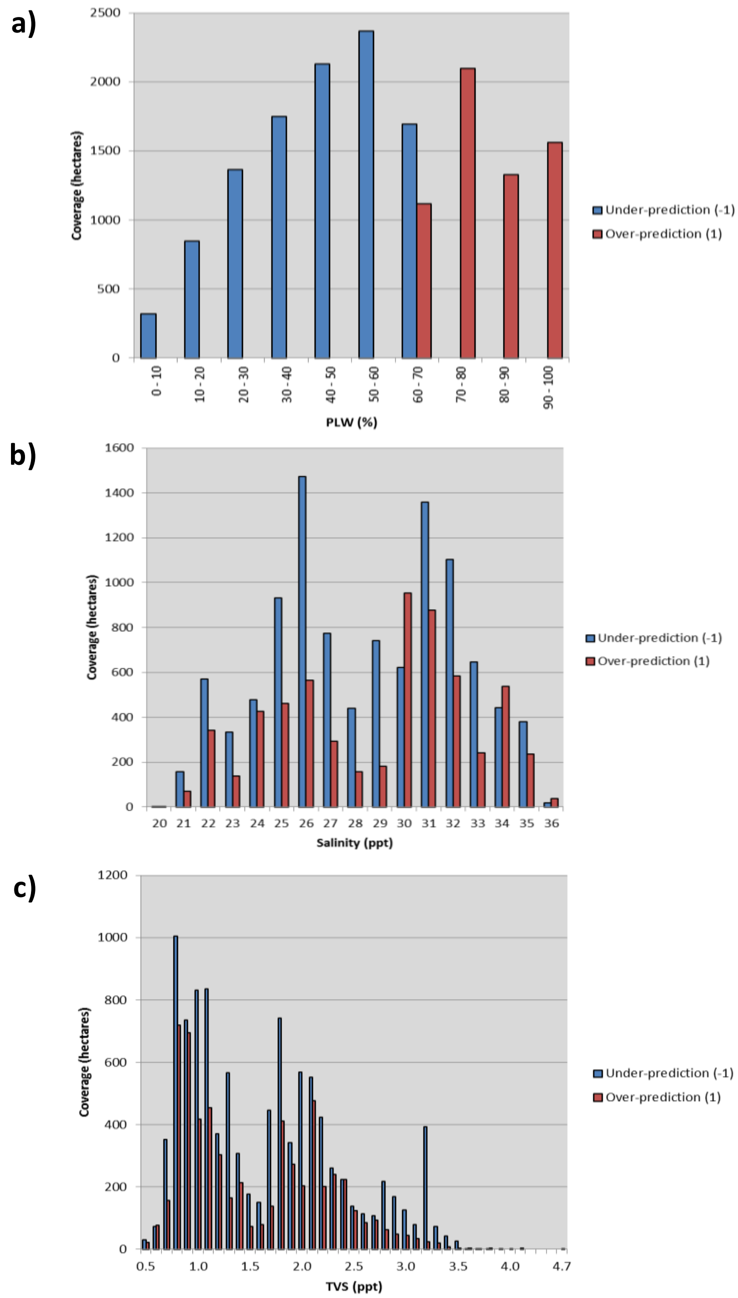

Figure 24(a-c): Error distribution as a function of light attenuation (a), salinity (b), and TVS (c) for predicted cover.

In 2006 and 2008, most over-prediction was observed in near-shore areas of Hillsborough Bay, and most under-prediction was observed in offshore areas around the outer perimeter of continuous seagrass beds. In 2010, over-prediction declined in Hillsborough Bay while under-prediction increased in Old Tampa Bay. In 2012, prediction accuracy remained similar to 2010 except for an increase in over-prediction in eastern Lower Tampa Bay (Figure 25).

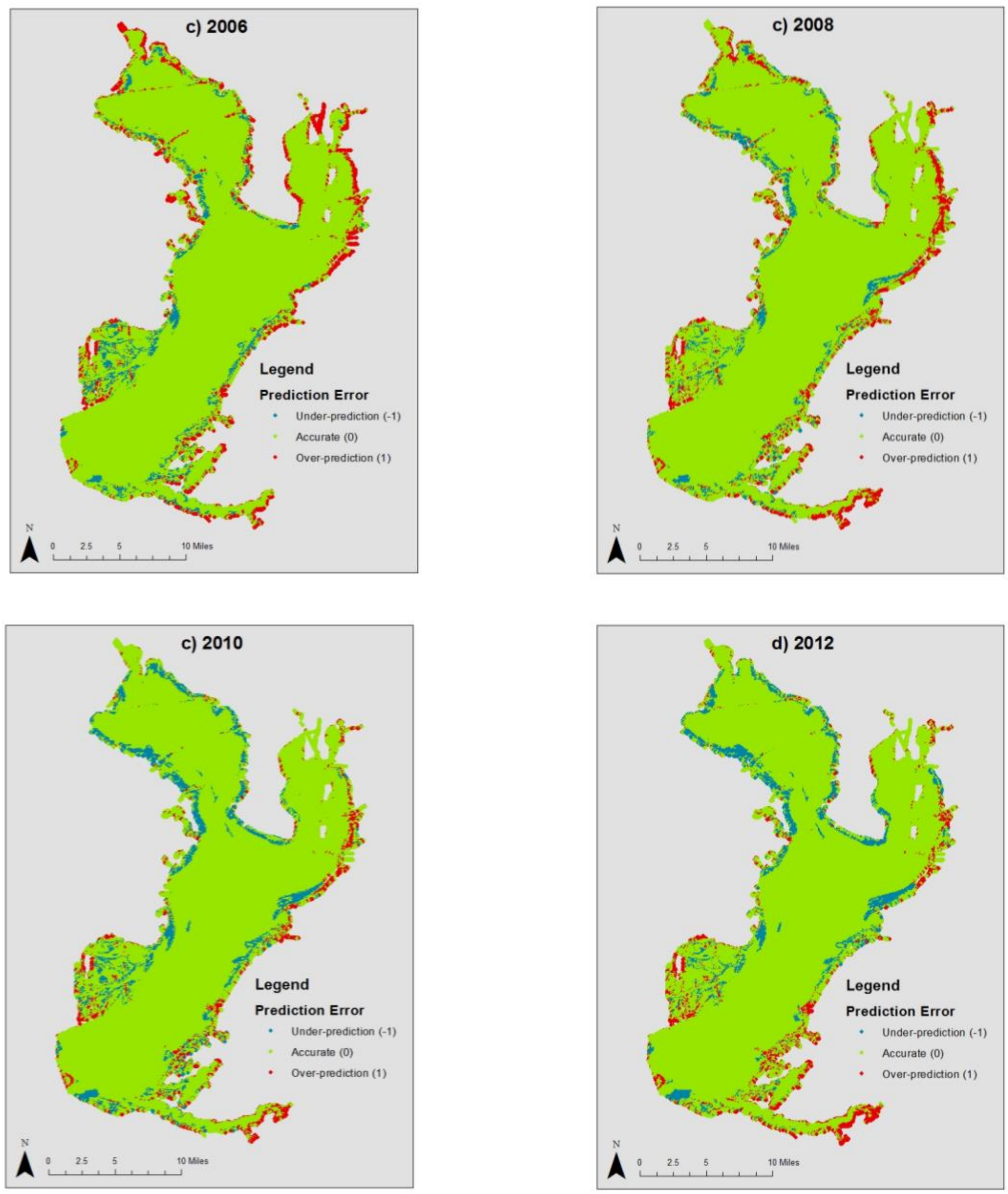

Figure 25(a-d): Prediction error classified as an under-prediction (-1) if the calculated probability was less than 0.5, but seagrass was present. Prediction error classified as accurate (0) if the calculated probability was less than 0.5, and seagrass was absent, or probability was greater than 0.5, and seagrass was present. Prediction error classified as over-prediction (1) if the calculated probability was greater than 0.5, and seagrass was absent.

Discussion

The predictive model performed well at predicting continuous seagrass coverage. Some patchy coverage was accurately predicted, but most was observed in areas of under-prediction. A significant amount of continuous coverage was also under-predicted. Areas of under-prediction contained the most hectares of seagrass coverage in total among the prediction error classes (under-prediction, accurate, over-prediction). Very small amounts of seagrass were observed in areas of over-prediction with the largest percentage from patchy coverage.

Seagrass coverage was most accurately predicted in areas of medium to high light availability in Middle and Lower Tampa Bay. These areas experience less point-source nutrient loading than Hillsborough Bay and Old Tampa Bay with fewer major metropolitan areas near their shores. These areas are also closer to the Gulf of Mexico experiencing higher salinity levels and lower temporal variability of salinity. Continuous coverage was also most accurately predicted overall. These seagrass beds show homogeneity across each observed cell and are more likely to survive a disturbance than seagrass beds exhibiting patchy coverage (Robbins 1997).

The predictive model performed poorly in areas of increased freshwater inputs. Error was observed in Hillsborough Bay and Old Tampa Bay that receive inputs from the Hillsborough and Alafia Rivers as well as run-off from urban centers, such as Tampa, Clearwater and St. Petersburg. Increasing error was also observed between 2010 and 2012 in Lower Tampa Bay near the mouth of the Manatee River. Patchy seagrass coverage was also poorly predicted by the model. Most patchy coverage was under-predicted. Even though the total amount of over-prediction was relatively small, patchy coverage made up the largest percentage.

Error in model development likely affected the predictive power of the model. Model development was based on relating observed seagrass occurrence with water quality parameters of the surrounding water-column. Error may have occurred when seagrass was observed, but may have been surviving under water quality conditions that were not ideal. This would have resulted in identifying ideal water quality conditions for seagrass that may have not been accurate. The reclassification of patchy and continuous seagrass coverage into the binary format of present (1) and absent (0) may have contributed to this error since it is possible that patchy coverage may exist under conditions that are not ideal for survival.

Seagrass species' preference of different salinity levels may also have caused error in model development. The three most predominant species of seagrass in Tampa Bay prefer different salinity conditions ((Dunton 1996), (FWS 2014)). The predictive model assumes that all seagrass has the same desired salinity condition. Since salinity and temporal variability of salinity are influenced spatially by the Gulf of Mexico, error may have occurred when attempting to relate seagrass presence with certain salinity levels. Different species of seagrass appear to thrive in different salinity conditions.

Prediction error was likely influenced by similar issues. Patchy seagrass coverage reclassified with continuous coverage may have influenced the model to prefer non-ideal water quality conditions resulting in under-prediction. The reclassification of patchy coverage may also have caused under-prediction throughout offshore areas of the bay where light attenuation is decreasing. Water quality conditions considered ideal by the model may not have identified with increasing patchy coverage existing under non-ideal conditions. As patchy coverage increased between 2010 and 2012 as did under-prediction of the model.

Prediction error also likely occurred as result of increased exposure near in-shore areas. The model related high light attenuation with the presence of seagrass coverage. It is assumed that areas with high light attenuation are also shallow areas that may experience increased wave action and anthropogenic influence. The bias in model prediction as a function of light attenuation showed that seagrass was mostly over-predicted in areas of high light attenuation and under-predicted in areas of lower light attenuation. Due to high light attenuation the model predicted seagrass presence in these exposed areas, but seagrass was not observed because this exposure may have limited its growth.

Two issues arose in model implementation due to seagrass mapping and classification methods. The first was absence of seagrass species data. All seagrass species found in Tampa Bay have similar light attenuation requirements (Duarte 1991) but different salinity preference ((Dunton 1996), (FWS 2014)). Seagrass species have been found to tolerate moderate fluctuations in salinity (Koch et al. 2007), but increased freshwater inflows have also been found to affect seagrass coverage in other gulf coast estuaries (Doering et al. 2002). Since Tampa Bay has multiple freshwater inflows from rivers, wastewater treatment plants, and surface run-off, salinity appears to be an important variable to consider when attempting prediction of seagrass response to changing freshwater inflows. If the species of seagrass in an area are known, more accurate probability of response could be predicted based on knowledge of the salinity tolerance levels of each species.

The second issue in model implementation was the classification system of continuous and patchy coverage. Despite the discontinuity of patchy seagrass, it has been shown to provide valuable ecosystem function (Dawes et al. 2004). But for modeling purposes, patchy coverage increased error in model development and was mostly predicted inaccurately. Each recognized cell of patchy coverage in seagrass mapping efforts can exhibit fluctuations in patch dynamic (Robbins 1997). Therefore, it is difficult to determine what specific water-column conditions are positively or negatively affecting seagrass success in these cells. It can be assumed that water quality conditions are not consistently ideal in areas of patchy seagrass, although these conditions may be inconsistent across a single cell. Cells containing patchy seagrass coverage are calculated into total counts assuming that the entire cell is covered in seagrass, although this may not be the case. This leads to significant under-prediction in habitat suitability model development and prediction.

Conclusion

This study developed a habitat suitability model for Tampa Bay seagrass that most accurately predicted continuous seagrass coverage. An explicit logistic regression function was developed based on interpolation of in situ and remotely sensed water quality data. The regression function used light attenuation, salinity, and temporal variability of salinity to predict the probability of seagrass occurrence. In order to improve the model an exposure index should be included to account for decreased seagrass coverage in shallow exposed areas. In situ transect data should also be included in order to more accurately observe water quality variables and related seagrass coverage dynamics not made obvious through interpolation and reclassification. Error analysis should incorporate fractal dimensions of patchy and continuous seagrass polygons to investigate the influence of false positive and false negative results. This study can help managers and planners understand how seagrass occurrence may be related to the water quality variables observed in this study.

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