2014


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Ecosystem Service Potential Capacity Scenarios: Effects from Sea Level Rise and Management Practices, Chesapeake Bay, Virginia

A Dissertation
Presented to
The Faculty of the School of Marine Science
College of William and Mary in Virginia

In Partial Fulfillment
of the Requirements for the Degree of
Doctor of Philosophy

by
Cielomar Rodriguez-Calderón
2014
APPROVAL SHEET

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Doctor of Philosophy

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# TABLE OF CONTENTS

AKNOWLEDGEMENTS ...........................................................................................................vii
LIST OF TABLES ..................................................................................................................viii
LIST OF FIGURES ...............................................................................................................xi
ABSTRACT ......................................................................................................................xvii
GENERAL INTRODUCTION ............................................................................................xviii

CHAPTER 1: Ecosystem Services Model: Tidal Shoreline’s Capacity to Provide Habitat and Water Quality Services in Mathews County and City of Hampton, VA..................................................................................1

ABSTRACT.........................................................................................................................2
INTRODUCTION ..................................................................................................................3
ECOSYSTEM SERVICES .....................................................................................................4
   COASTAL ECOSYSTEM SERVICES ..............................................................................7
   HABITAT AND WATER QUALITY SERVICES ..........................................................9
ECOSYSTEM SERVICES MODELING .............................................................................11
SHORELINE COMPONENTS ............................................................................................14
STUDY SITES ....................................................................................................................23
   CHESAPEAKE BAY ........................................................................................................23
   MATHEWS COUNTY AND CITY OF HAMPTON, VIRGINIA ...................................24
      MATHEWS COUNTY ..........................................................................................25
      CITY OF HAMPTON ...............................................................................................26
METHODS ..........................................................................................................................27
   SHORELINE CLASSIFICATION AND SAMPLING SIZE ........................................28
   SAMPLE SIZE PER SHORELINE CLASS ..................................................................30
   DETERMINING THE ASSESSMENT BUFFER SIZE ..................................................31
   ASSESSMENT ZONES .................................................................................................33
   DIGITIZING AND CLASSIFICATION OF COMPONENTS ........................................35
   MODELING OF ECOSYSTEM SERVICES ..................................................................37
   SPATIAL VARIATIONS ...............................................................................................42
RESULTS AND DISCUSSION ............................................................................................43
HABITAT SERVICES MODEL .............................................................................................43
   Habitat Services Model: Mathews County ...............................................................45
      Temporal and spatial Changes in Habitat Capacity ..............................................45
      Habitat Service Capacity: Mathews 1968 ............................................................46
      HSM Variations in Trends for Shoreline Components: Mathews 1968 ..............47
      Habitat Service Capacity: 2007 ............................................................................48
      HSM Variations in Trends for Shoreline Components: Mathews 2007 .............49
      Mathews’ HSM: General Findings .......................................................................50
WATER QUALITY SERVICES MODEL .............................................................................44
   Water Quality Services Model: Mathews County .....................................................51
Temporal and Spatial Changes in Water Quality Service Capacity .............................................................. 51
Water Quality Service Capacity: Mathews 1968 ................................................................. 52
WQM Variations in Trend for Shoreline Components: Mathews 1968 .............................................................. 53
Water Quality Service Capacity: Mathews 2007 ................................................................. 54
WQM Variations in Trends for Shoreline Components: Mathews 2007 .............................................................. 55
Mathews' WQM: General Findings .............................................................................................. 55
Capacity by Shoreline Type: Mathews .................................................................................. 56
Capacity for Habitat Services by Shoreline Type: Mathews .................................................. 57
Capacity for Water Quality Services by Shoreline Type: Mathews .................................................. 57
Mathews: Overall Changes in Shoreline Components and Capacity .............................................................. 58
Habitat Services Model: City of Hampton .................................................................................. 60
Temporal and Spatial Changes in Habitat Capacity ........................................................................ 60
Habitat Service Capacity: Hampton 1963 .................................................................................. 61
HSM Variations in Trends for Shoreline Components: Hampton 1963 .............................................................. 62
Habitat Service Capacity: Hampton 2009 .................................................................................. 63
HSM Variations in Trends for Shoreline Components: Hampton 2009 .............................................................. 65
Hampton’s HSM: General Findings .............................................................................................. 67
Water Quality Model: City of Hampton .................................................................................. 68
Temporal and Spatial Changes in Water Quality Service Capacity .............................................................. 68
Water Quality Service: Hampton 1963 .................................................................................. 68
WQM Variations in Trends for Shoreline Components: Hampton 1963 .............................................................. 69
Water Quality Service Capacity: Hampton 2009 ........................................................................ 70
WQM Variations in Trends for Shoreline Components: Hampton 2009 .............................................................. 71
Hampton’s WQM: General Findings .............................................................................................. 71
Capacity by Shoreline Type: City of Hampton .................................................................................. 72
Capacity for Habitat and Water Quality Services by Shoreline Type: Hampton .......................... 73
Hampton: Overall Changes in Shoreline Components and Capacity .............................................................. 74
Habitat and Water Quality Services Models Performance ........................................................................ 76
Variations in Model Scores .............................................................................................. 76
Model’s Limitations .................................................................................................................. 78
CONCLUSIONS .............................................................................................................................. 79
TABLES AND FIGURES .............................................................................................................. 82
APPENDIX ........................................................................................................................................ 129
LITERATURE CITED .................................................................................................................. 137
CHAPTER 2: Modeling Shoreline Change: Influence of Physical and Vegetation Components over Shoreline Change and Effects of Marshes on Land Inundation in Mathews County and City of Hampton, Virginia, Chesapeake Bay

ABSTRACT
INTRODUCTION
STUDY SITES
CHESAPEAKE BAY
MATHEWS COUNTY AND CITY OF HAMPTON, VIRGINIA
MATHEWS COUNTY
CITY OF HAMPTON
METHODS
SHORELINE CHANGE
SHORELINE INVENTORY
APPROACH 1
APPROACH 2
Land Slope
Observed vs. Expected Inundated Lands
APPROACH 3
MODEL CALIBRATION
MODEL VERIFICATION
RESULTS AND DISCUSSION
APPROACH 1
Mathews County
Unmanaged Shoreline Units: Mathews
Managed Shoreline Units: Mathews
City of Hampton
Unmanaged Shoreline Units: Hampton
Managed Shoreline Units: Hampton
General Findings: Approach 1
APPROACH 2
Mathews County
Shoreline Change: Mathews
Land Slope: Mathews
City of Hampton
Shoreline Change: Hampton
Land Slope: Hampton
Influence of Shoreline Features on Inundation
Marshes
Beaches
Managed Shorelines
Influence of Physical and Vegetation Components in Marshes
Shoreline Change
Mathews
Hampton
Shoreline Change Variations in Marshes Based on Land Use
CHAPTER 3: Influence of Sea Level Rise and Management Practices on Capacity to Provide Habitat and Water Quality Services in Tidal Shorelines by 2050 ........................................223

ABSTRACT .....................................................................................................................224
INTRODUCTION ..........................................................................................................225
SEA LEVEL RISE ..............................................................................................226
MANAGEMENT PRACTICES: LIVING SHORELINES ..............................................228
STUDY SITES ................................................................................................................231
CHESAPEAKE BAY .................................................................................................231
MATHEWS COUNTY AND CITY OF HAMPTON, VIRGINIA .........................232
MATHEWS COUNTY ..............................................................................................232
CITY OF HAMPTON ...............................................................................................233
METHODS ......................................................................................................................234
ECOSYSTEM CAPACITY: DRIVERS OF CHANGE ..............................................235
    Future Land Inundation due to Sea Level Rise: Scenario 1 .........................235
    Management Practices: Scenario 2 ...............................................................237
RESULTS AND DISCUSSION .....................................................................................238
    POTENTIAL CAPACITY TO PROVIDE ECOSYSTEM SERVICES BY
    2050: INFLUENCE OF LAND INUNDATION AND MANAGEMENT
    PRACTICES ...........................................................................................................238
        Mathews County: Scenario 1 .........................................................................238
        City of Hampton: Scenario 1 .........................................................................239
        Scenario 2: Best Management Practices ....................................................240
CONCLUSIONS .............................................................................................................241
TABLES AND FIGURES .............................................................................................243
LITERATURE CITED ..................................................................................................258
VITA ................................................................................................................................262
ACKNOWLEDGMENTS

I would like to thank the many people who made this project possible. Thanks to my major advisor, Dr. Carl H. Hershner, for all his unconditional guidance, encouragement, support, and patience. Carl, thank you for the opportunity you gave me to accomplish this personal goal. Thanks to my committee members, Dr. Donna M. Bilkovic, Dr. Mark J. Brush, Dr. John T. Wells and Dr. Christopher R. Pyke for all their advice and interest in my project. Special thanks to my dissertation readers, Julie Herman and Marcia Berman for reviewing my prospectus and dissertation and for providing me with ideas and suggestions to make this project a success.

Many thanks to Dawn Fleming, Tamia Rudnick, Karinna Nuñez, Dan Schatt, Dave Weiss, Sharon Killeen, Christine Tombleson, Molly Mitchell and Pamela Mason for their unconditional help and assistance. You help me extensively by providing me with moral support, GIS and statistical support, guidance to prevent many computer crises and by supplying me with information related to management practices and ecosystem services in Mathews and Hampton. In addition, I gratefully acknowledge financial support from the Hall-Bonner Program and for all the great opportunities that I had as a fellow.

Most importantly, thanks to my family for all their love, help, support, prayers, for believing in me and for being there in every step of my journey. Thanks for being the strength, the energy and the compass that keeps me moving forward. All what I have achieved I owe it to all of you. Lastly, I would like to thank Christopher Jump, his love and words of strength kept me sane through all these months. Thank you for giving yourself everyday to make me smile and for reminding me what life is all about!
LIST OF TABLES

Table 1.1 List of some coastal ecosystem services, their processes, functions, and controlling components. Modified from de Groot et al. (2002) and Barbier et al. (2011) ................................................................. 83

Table 1.2 Six different shoreline classes based on fetch (short= ≤300m; long= >300) and bank height classifications. These classes were used for Mathews' and Hampton's shorelines ........................................................................................................ 85

Table 1.3 Shoreline types generated for Mathews County. The column to the right indicates the abbreviation for each class that will be used in the rest of this document. The “M” in the abbreviation is to specify the location of the bank (Mathews), “B” is for Bank., the numbers represent the bank height in meters, and the last letter represents fetch conditions (Long, Short). ........................................................................ 86

Table 1.4 Shoreline types generated for City of Hampton. The column to the right indicates the abbreviation for each class that will be used in the rest of this document. The “H” in the abbreviation is to specify the location of the bank (Hampton), “B” is for Bank., the numbers represent the bank height in meters, and the last letter represents fetch conditions (Long, Short). ........................................................................ 86

Table 1.5 Total sample points generated to classify shorelines and used to select the sites to be assessed in Mathews and Hampton ................................................................. 87

Table 1.6 Percent error calculations for Mathews based on a total sample size of 30 samples. All errors are below a 10% error ....................................................................................... 87

Table 1.7 Percent error calculations for Mathews based on a total sample size of 30 samples. All errors are below a 10% error ....................................................................................... 87

Table 1.8 Digitization error included for the most common components observed in the intertidal, riparian and upland zones. The values represent the standard errors in square meters ........................................................................................................ 93

Table 1.9 Shoreline components per assessment zone. The categorical values and model scores are specified for each component assessed for the HSM and/or WQM. The total maximum and minimum model score that each component can receive are indicated in the last two columns. The two bottom rows at the right indicate the possible maximum and minimum total model scores that shoreline units can receive by model ........................................................................................................ 95

Table 1.10 HSM capacity classifications. Capacity classes were generated applying Jenks Natural Breaks method in GIS. The maximum and minimum scores represent the range of values for each capacity class. The same classes were applied in Mathews and Hampton for historic and current times ......................................................... 96

Table 1.11 Capacity classifications for the Water Quality Model generated by applying Jenks Natural Breaks method in GIS. The maximum and minimum scores represent the range of values for each capacity class. The same classes were applied in Mathews and Hampton for historic and current times ......................................................... 96
Table 1.12  HSM averaged model scores per component in 1968. Individual averaged model scores were calculated for all components observed under each capacity class. Averaged scores represent the general components’ conditions observed under each capacity class (high, moderate, low). To determine the lowest and highest types of component conditions see Table 7.

Table 1.13  Changes in area and/or amount in shoreline components in the HSM and WQM from 1968 to 2007. Changes are displayed by assessment zones and in percent change.

Table 1.14  Changes in area and/or amount in shoreline components in the HSM and WQM for a. Hampton from 1963 to 2009 and for b. Mathews from 1968 to 2007. Changes are displayed by assessment zones and in percent change.

Table 2.1  Shoreline uncertainties for Mathews and Hampton by shoreline type.

Table 2.2  Shoreline components assessed for Mathews and Hampton and specifically applied in Approach 1. Components included under the Database I were assessed by the CCRM. Database II was generated by the current study. The categorical values and model values are specified for each component.

Table 2.3  Description of the type of influence shoreline components have over shoreline change. (Continuation below).

Table 2.4  Six different shoreline classes based on fetch (short = ≤300m; long = >300) and bank height classifications. These classes were used for Mathews’ and Hampton’s shorelines and for the Approach 1.

Table 2.5  Physical and vegetation components assessed for Approach 2 and Approach 3. Components included in the Database I were assessed for the CCRM’s shoreline inventory. Database III was generated by the current study. The categorical values and model values are specified for each component.

Table 2.6  Total number of shoreline units assessed per shoreline feature for Mathews and Hampton and for the Approach 2.

Table 2.7  Total number of shoreline units with eroding marshes that were assessed for Mathews and Hampton and for the Approach 3.

Table 2.8  Global model (GM) for unmanaged shoreline units in Mathews. The table includes the components identified as predictors, the model coefficients and the model intercept.

Table 2.9  Global model (GM) for unmanaged shoreline units in Hampton. The table includes the components identified as predictors, the model coefficients and the model intercept.

Table 2.10  Global model (GM) for managed shoreline units in Hampton. The table includes the components identified as predictors, the model coefficients and the model intercept.

Table 2.11  General shoreline change and slope statistics for Mathews’ and Hampton’s shoreline features.

Table 2.12  Observed and expected horizontal displacement of the shoreline per slope intervals. The table shows lower observed values than the expected values for land slopes under 5°. The opposite was observed for land slopes over 5°.

Table 2.13  Global model (GM) for shoreline units with marsh presence in Mathews. The table includes the components identified as predictors, the model coefficients and the model intercept.

Table 2.14  Global model (GM) for shoreline units with marsh presence in Hampton. The table includes the components identified as predictors, the model coefficients and the model intercept.
Table 2.15 Differences in shoreline change (EPR), fetch and slope conditions per land use type in Mathews and Hampton. ......................................................................................................................... 212

Table 2.16 Global model (GM) for shoreline units with eroding marshes in Mathews. The table includes the components identified as predictors, the model coefficients and the model intercept. ......................................................................................................................... 214

Table 2.17 Global model (GM) for shoreline units with eroding marshes in Hampton. The table includes the components identified as predictors, the model coefficients and the model intercept. ......................................................................................................................... 214

Table 3.1 HSM capacity classifications. Capacity classes were generated applying Jenks Natural Breaks method in GIS. The maximum and minimum scores represent the range of values for each capacity class. ........................................................................................................... 245

Table 3.2 Capacity classifications for the Water Quality Model generated by applying Jenks Natural Breaks method in GIS. The maximum and minimum scores represent the range of values for each capacity class. ........................................................................................................... 245

Table 3.3 Shoreline components per assessment zone. The categorical values and model scores are specified for each component assessed for the HSM and/or WQM. The total maximum and minimum model score that each component can receive are indicated in the last two columns. The two bottom rows at the right indicate the possible maximum and minimum total model scores that shoreline units can receive by model. ................. 247

Table 3.4 Mathews. Changes in area (m²) and/or amount in shoreline components in the HSM and WQM for 2050 based on forecasted land inundation from Sc1 and Sc2. Changes are displayed by assessment zones and in percent change. ....................................................... 251

Table 3.5 Hampton. Changes in area (m²) and/or amount in shoreline components in the HSM and WQM for 2050 based on forecasted land inundation from Sc1 and Sc2. Changes are displayed by assessment zones and in percent change. ....................................................... 255

Table 3.6 Total shoreline length where marshes, beaches and riparian buffer (i.e. forested lands and scrub-shrubs) are present in a. Mathews and b. Hampton. ............................................. 257
LIST OF FIGURES

Figure 1.1 a. Map of the Chesapeake Bay, USA indicating the location of b. Mathews (M) and c. Hampton (H).......................................................................................... 84

Figure 1.2 a. Mathews imagery from 2007 showing a point site (red dot) and the spatial extent of the 60m assessment buffer (yellow circle). b. Mathews imagery from 1968 showing the same location from Figure 1a. The point site is in red and the spatial coverage of the 60m assessment buffer is circled in yellow. This site is located at a low bank (0-1.5m or 0-5ft.) with long fetch ......................................... 88

Figure 1.3 Mathews imagery from 2007 showing the different assessment zones within the 60m buffer. b. Mathews imagery from 1968 showing the different assessment zones within the 60m buffer ................................................................. 89

Figure 1.4 Components present in the intertidal zone in 2007 ............................................. 90

Figure 1.5 Components present in the riparian zone in 2007 .................................................. 90

Figure 1.6 Components present in the upland zone in 2007 .................................................. 91

Figure 1.7 Diagram exemplifying how the Clip Tool works. This ArcGIS tool requires a clip feature to define the area to be extracted from the input. The output will include the area that overlaps the clip feature ................................................................. 92

Figure 1.8 Diagram exemplifying the Ecosystem Services Models and the type of models ran in Model Builder from GIS................................................................. 94

Figure 1.9 Location of shoreline units assessed in Mathews County for the HSM. a. Location of sites per capacity class in 1968. b. Location of sites per capacity class in 2007. These figures indicate changes through time in capacity for habitat services. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity. Map source: 2007 VBMP Imagery .................. 97

Figure 1.10 Prediction surfaces indicating capacity scores for habitat services in a. 1968 and b. 2007 for Mathews County. (A). Piankatank River, (B). Mobjack Bay (C). Gwynn’s Island. Dark orange and red colors indicate high capacity shorelines, light blue and yellow tones represent areas with moderate capacity and dark blue represents low capacity. Interpolation values are only showed for a 500m wide buffer. This buffer size was only used for presentation purposes and to make values adjacent to the shoreline discernible ................................................................. 98

Figure 1.11 a. Number of sites per capacity class in 1968 and 2007 for the HSM in Mathews County. b. Total area in square meters for three different land use types (natural, agriculture, developed) in the riparian and upland zones. Areas for land use in 1968 are specified by capacity class................................................................. 99

Figure 1.12 a. Area fractions for vegetation composition in the riparian zone during 1968. The graph indicates variations in vegetation types within each capacity class. Even though more vegetation types were identified in the high capacity class, based on the database generated by the HSM, all sites presented a low vegetation composition. This indicates that only one or two different types of vegetation were observed at the sites. In the moderate class, 13 sites were classified with high vegetation composition indicating the sites presented 3 or more types of vegetation. This class also showed the largest area size for most
vegetation types. The low capacity only presented 1 site with high composition. b. Riparian land use for 1968 indicating higher anthropogenic activities under the moderate and low capacity classes. The moderate and low capacity classes also presented the largest area size for secondary vegetation (i.e. scrub-shrubs and grass).

Figure 1.13 Total area in square meters for three different land use types (natural, agriculture, developed) in the riparian and upland zones. Areas for land use in 2007 are specified by capacity class.

Figure 1.14 a. Changes in beach area per capacity class. A larger area size was identified under the high capacity class in 1968 and under the moderate class in 2007. b. Area fraction for vegetation composition in the riparian zone in 2007. c. Riparian land use for 2007 indicating higher anthropogenic activities for sites under the moderate capacity class. The moderate and low capacity classes also presented the largest area size for secondary vegetation: scrub-shrubs and grass.

Figure 1.15 Location of shoreline units assessed in Mathews County for the WQM. a. Location of sites per capacity class in 1968. b. Location of sites per capacity class in 2007. These figures indicate changes in capacity through time. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity. Map source: 2007 VBMP Imagery.

Figure 1.16 Prediction surfaces indicating capacity scores for water quality services in 1968 and 2007 for Mathews County. (A). Mobjack Bay, (B). Gwynn’s Island. Dark orange and red colors indicate high capacity shorelines, light blue and yellow tones represent areas with moderate capacity and dark blue represents low capacity. Interpolation values are only showed for a 500m wide buffer. This buffer size was only used for presentation purposes and to make values adjacent to the shoreline discernible.

Figure 1.17 a. Number of sites per capacity class in 1968 and 2007 for the WQM in Mathews County. b. Total area in square meters for three different land use types (natural, agriculture, developed) in the riparian and upland zones. Areas for land use in 1968 are specified by capacity class.

Figure 1.18 a. Total area for tidal and inland marshes in square meters in 1968. Inland marshes total area includes marshes in the riparian and upland zones. b. Area fractions for vegetation composition in the riparian zone during 1968. Sites with moderate capacity showed the largest area fraction for most of the vegetation types (n=3) in addition to the largest number of sites with high vegetation composition. However, more vegetation types were identified in the high capacity class (n=4), but all sites presented low vegetation composition. c. Riparian land use in 1968 indicating higher anthropogenic activities under the moderate and low capacity classes. The moderate and low capacity classes also presented the largest area size for secondary vegetation (i.e. scrub-shrubs and grass). d. Total area for forested lands in the riparian and upland zones in 1968.

Figure 1.19 a. Total area in square meters for three different land use types (natural, agriculture, developed) in the riparian and upland zones for 1968 and 2007. Areas for land use are specified by capacity class. b. Total area for vegetation
cover in the riparian and upland zones in 1968 and 2007. The total vegetation cover was 710,516 m$^2$ in 1968 and 623,946 m$^2$ in 2007 indicating a total vegetation loss of 86,570 m$^2$.

**Figure 1.20** Conditions for forested lands, riparian land use and riparian vegetation composition per capacity class in 2007. a. Riparian and upland forested lands showed a larger area under the moderate capacity class in 2007. b. Total riparian land use in 2007 per capacity class. Larger area for natural and developed lands was observed in the moderate capacity class. c. Riparian vegetation composition in 2007 indicated the presence of all vegetation types and the largest area fractions for secondary vegetation under the moderate capacity class.

**Figure 1.21** Averaged model scores for the a. HSM and b. WQM in 1968 and 2007 in Mathews County. Shoreline type in the x-axis (bank height (m) = 0 -> 9.1; fetch = Long (L), Short (S)). c. Changes through time in averaged model scores for the upland zone in the HSM and WQM. d. Riparian and upland land use averaged model scores for the HSM and WQM by shoreline type in 1968 and 2007.

**Figure 1.22** Location of shoreline units assessed in the City of Hampton for the HSM. a. Location of sites per capacity class in 1963. b. Location of sites per capacity class in 2009. These figures indicate changes through time in capacity for habitat services. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity. Map source: 2009 VBMP Imagery.

**Figure 1.23** Prediction surfaces indicating capacity scores for habitat services in a.1963 and b. 2009 for the City of Hampton. (A) Grunland Creek, (B) Harris River, (C) Stony Point, (D) Tabbs Point, (E) Marsh Point, (F) Salt Ponds. Dark orange and red colors indicate high capacity shorelines, light blue and yellow tones represent areas with moderate capacity and dark blue represents low capacity. Interpolation values are only showed for a 500m wide buffer. This buffer size was only used for presentation purposes and to make values adjacent to the shoreline discernible.

**Figure 1.24** a. Number of sites per capacity class in 1963 and 2009 for the City of Hampton. b. Land use patterns in percent for Mathews County in 1968 and for c. the City of Hampton in 1963.

**Figure 1.25** Conditions for a. SAV area and b. beach area in 1963 per capacity class.

**Figure 1.26** a. Area fractions for vegetation composition in the riparian zone. The graph indicates variations in vegetation types within each capacity class. A larger area fraction was identified under the moderate capacity class. b. Riparian land use conditions in 1963 indicating higher anthropogenic activities under the moderate and low capacity classes. The moderate and low capacity classes also presented the largest area size for secondary vegetation (i.e. scrub-shrubs and grass).

**Figure 1.27** Change in area for land use types from 1963 to 2009 in Hampton. Natural lands lost 139,628 m$^2$, agricultural lands were completely lost by 2009 and a total of 104,252 m$^2$ were converted to developed lands.

**Figure 1.28** Conditions for SAV and beach area per capacity class in 2009. a. Area in meter square for SAV in the subaqueous zone. A larger area size was identified in the low capacity class in 2009. b. Beach conditions in 2009 showed a larger
area size under the moderate capacity. However, most shoreline units with beach presence were identified under the low capacity class. An increase in sites with beach presence under the low capacity class coincided with a loss of 3,948 m² in beach area by 2009.

**Figure 1.29** Area size for a. mudflats, b. forested lands, c. riparian vegetation composition and d. riparian land use in 2009 was larger under the low capacity class. Conditions in vegetation composition were similar as observed in Mathews and seemed to be influenced by anthropogenic activities as well.

**Figure 1.30** a. Decrease in area for tidal and inland marshes between 1963 and 2009. Inland marshes showed the largest change with a 51% area loss. b. Changes in area per marsh type and per capacity class between 1963 and 2009. Most of the marsh components were identified in high capacity sites.

**Figure 1.31** Location of shoreline units assessed in the City of Hampton for the WQM. a. Location of sites per capacity class in 1963. b. Location of sites per capacity class in 2009. These figures indicate changes in capacity through time. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity. Map source: 2009 VBMP Imagery.

**Figure 1.32** Prediction surfaces indicating capacity scores for water quality services in a. 1963 and b. 2009 in the City of Hampton. (A). Grunland Creek, (B). Harris River, (C). Tabbs Point, (D). Marsh Point. Dark orange and red colors indicate high capacity shorelines, light blue and yellow tones represent areas with moderate capacity and dark blue represents low capacity.

**Figure 1.33** a. Number of sites per capacity class in 1963 and 2009 for the WQM in the City of Hampton. b. Percent area for land use types in Mathews County and the City of Hampton during historic times.

**Figure 1.34** Area for marsh components in meter square per capacity class in 1963.

**Figure 1.35** a. Area in meter square for SAV in the subaqueous zone. A larger area size was identified in the moderate capacity class. b. Vegetation composition showed a larger diversity and larger area size for secondary vegetation under the moderate capacity. c. Riparian land use conditions seemed to influence vegetation composition. Even though the low capacity class showed the largest area size for developed lands, many shoreline units (n=14) showed no vegetation reducing the total amount of vegetation, especially secondary vegetation.

**Figure 1.36** Conditions for riparian vegetation composition, forested lands, riparian land use and marsh components in 2009. a. Area fraction for vegetation composition indicated a larger area size for almost all vegetation types under the low capacity class. b. Larger total area for forested lands was observed under the low capacity class as well. c. These conditions coincided with a larger area size for natural and developed lands under the low capacity. d. However, most marsh components were observed in high capacity sites.

**Figure 1.37** Averaged model scores for the a. HSM and the b. WQM in 1963 and 2009 by shoreline type. c. Changes in averaged model scores for the intertidal zone in the HSM and WQM between 1963 and 2009. d. Changes in averaged model scores for the upland zone in the HSM and WQM for 1963 and 2009.

**Figure 2.1** a. Map of the Chesapeake Bay, USA indicating the location of. b. Mathews (M) and c. Hampton (H).
Figure 2.2 This diagram summarizes the three different approaches applied in this study. Each approach assessed shoreline change at different spatial scales and different shoreline types. A series of predictors or shoreline components were used per approach to statistically determine their influence over shoreline change.

Figures 2.3 a. Linear regression for the sea level rise trend at the Gloucester Point/Yorktown and at the b. Sewells Point stations, VA.

Figure 2.4 Conceptual model indicating predictors of shoreline change based on Approach 1. Plus and minus symbols next to the predictors indicate the type of correlation with shoreline change. Solid lines indicate the component showed a high model coefficient, a dashed line is a moderate model coefficient, and a dotted line is a low model coefficient.

Figure 2.5 Box plot describing rates of shoreline change for beaches, marshes and defended (i.e. managed) shorelines in Mathews and Hampton. The values included in the graph are the averaged shoreline change per shoreline feature.

Figure 2.6 Box plot describing land slope for beaches, marshes and defended (i.e. managed) shorelines in Mathews and Hampton. The values included in the graph are the averaged slope in degrees per shoreline feature.

Figures 2.7 Residuals and observed values for marshes in a. Mathews and b. Hampton. c. Mathews’ and Hamptons’ residual values combined.

Figures 2.8 Residuals and observed values for beaches in a. Mathews and b. Hampton. c. Mathews’ and Hamptons’ residual values combined.

Figures 2.9 Residuals and observed values for defended shorelines in a. Mathews and b. Hampton. c. Mathews’ and Hamptons’ residual values combined.

Figure 2.10 Conceptual model indicating predictors of shoreline change based on Approach 2. Plus and minus symbols next to the predictors indicate the type of correlation with shoreline change. Solid lines indicate the component showed a high model coefficient, a dashed line is a moderate model coefficient, and a dotted line is a low model coefficient.

Figure 2.11 Conceptual model indicating predictors of shoreline change based on Approach 3. Plus and minus symbols next to the predictors indicate the type of correlation with shoreline change. Solid lines indicate the component showed a high model coefficient, a dashed line is a moderate model coefficient, and a dotted line is a low model coefficient.

Figures 2.12 Slope vs. shoreline change for marshes in Mathews County. a. Rates of shoreline change by land slope for all marshes from Approach 2. b. Rates of shoreline change by land slope for eroding marshes from Approach 3.

Figures 2.13 Model verification for a. Mathews’ and b. Hampton’s eroding marsh models from Approach 3. This is an example of the lack of strength observed in the models generated in this study.

Figures 3.1 a. Map of the Chesapeake Bay, USA indicating the location of b. Mathews (M) and c. Hampton (H). Map source: 2007 VBMP Imagery.

Figures 3.2 Location of shoreline units assessed in a. Mathews County and b. City of Hampton.

Figures 3.3 Capacity of ecosystem services to provide habitat services by 2050 in Mathews County. a. Capacity based on Sc1 for sea level rise b. Capacity based on Sc2 for sea level rise. Green circles indicate high capacity, yellow is moderate capacity and red
circles are sites with low capacity.  c. Graph indicating changes in the number of shoreline units per capacity from 1968 to 2050. The percent change is based on the difference in sites between 2007 and the scenarios.  Map source: 2007 VBMP Imagery.  

Figures 3.4 Capacity of ecosystem services to provide water quality services by 2050 in Mathews County.  a. Capacity based on Sc1 for sea level rise b. Capacity based on Sc2 for sea level rise. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity.  c. Graph indicating changes in the number of shoreline units per capacity from 1968 to 2050. The percent change is based on the difference in sites between 2007 and the scenarios.  Map source: 2007 VBMP Imagery.  

Figures 3.5 Prediction surfaces indicating capacity scores for habitat services in Mathews by 2050 under a. Sc1 and b.Sc2.  c. Prediction surface indicating water quality services under Sc1 and d. Sc2.  

Figures 3.6 Capacity of ecosystem services to provide habitat services by 2050 in Hampton.  a. Capacity based on Sc1 for sea level rise b. Capacity based on Sc2 for sea level rise. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity.  c. Graph indicating changes in the number of shoreline units per capacity from 1963 to 2050. The percent change is based on the difference in sites between 2009 and the scenarios.  Map source: 2009 VBMP Imagery.  

Figures 3.7 Capacity of ecosystem services to provide water quality services by 2050 in Hampton.  a. Capacity based on Sc1 for sea level rise b. Capacity based on Sc2 for sea level rise. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity.  c. Graph indicating changes in the number of shoreline units per capacity from 1963 to 2050. The percent change is based on the difference in sites between 2009 and the scenarios.  Map source: 2009 VBMP Imagery.  

Figures 3.8 Prediction surfaces indicating capacity scores for habitat services in Hampton by 2050 under a. Sc1 and b.Sc2.  c. Prediction surface indicating water quality services under Sc1 and d. Sc2.  

Figures 3.9 Location and type of management practices suitable for each shoreline unit in a. Mathews and b. Hampton.  Legend:  I (Inundated), E/MM (Enhance/Maintain Marsh), WM (Widen Marsh), PMS (Plant Marsh with Sill), WM/EB (Widen Marsh/Enhance Buffer), ER/MB (Enhance Riparian/Marsh Buffer), ER/MB/BN (Enhance Riparian/Marsh Buffer or Beach Nourishment), E/MB (Enhance/Maintain Buffer), E/MB (Enhance/Maintain Beach), MB/OBBN (Maintain Beach or Offshore Breakwaters with Beach Nourishment).
ABSTRACT

Ecosystem services in tidal shoreline systems in the Chesapeake Bay experienced an increase in environmental pressure during the last decades mainly due to population growth, land development, and increasing sea levels. These changes jeopardized the potential capacity of shoreline ecosystems to provide habitat and water quality services which are vital for coastal resources, the economy and the coastal population’s welfare.

This dissertation’s main goal was to develop a local scale methodology capable of determining potential capacity of tidal shorelines to provide habitat and water quality services by 2050 based on the effects of sea level rise and management practices in Mathews County and the City of Hampton, VA. In this study, the potential capacity of tidal shorelines to provide water quality and habitat services was determined by the conditions of shoreline components. A primary emphasis was placed on the conditions of vegetation cover and vegetation composition present within the system. Chapter 1 generated a practical methodology consisting of two categorical models used to determine the potential capacity of tidal shorelines to provide habitat and water quality services during historic and current times. The methods applied allowed a spatially explicit identification of a decline in capacity through time. For Chapter 2, an empirical analysis including three different approaches was developed to identify the most important physical and natural predictors of shoreline change and to determine the response of different shoreline types (i.e. marshes, beaches and managed shorelines) to shoreline change and land inundation. The multiple models generated for each approach showed high variability by shoreline features and by locality in predictors and in the strength of their effects. Marshes showed the lowest erosion rate and were identified as the most efficient shoreline feature at attenuating land inundation. Chapter 3 includes scenarios for the potential capacity to provide habitat and water quality services by 2050 based on two accelerated scenarios for sea level rise and alternative management practices. Based on the scenarios, potential capacity will be highly compromised by 2050 due to land inundation. However, living shoreline methods could provide a potential solution to help mitigate the effects from sea level rise and maintain ecosystems.
GENERAL INTRODUCTION

Tidal shorelines are among the most important and productive resources that support the widest range and most significant areas for ecosystem services (Millennium Ecosystem Assessment, 2005). Services are generated across a variety of ecosystems such as marshes, wetlands, beaches, dunes, and seagrass beds. However, tidal shoreline ecosystems are currently one of the most threatened natural systems globally (Greenberg et al., 2006). In 2000, the Chesapeake Executive Council established habitat and water quality services as priority objectives to restore the health of the Chesapeake Bay. Ultimately, the quality of these services and the capacity of tidal shorelines to provide them are reflected in the economy, health, and security of coastal populations.

Tidal shorelines have unique interactions between terrestrial areas and the marine environments (Duxbury & Dickinson, 2007). It is based on these critical interactions that tidal shorelines are experiencing the most distressing changes and challenging management issues. Increasing coastal population and development have led to an extensive conversion of land use and land cover compromising many services. These changes alter the natural distribution and conditions of shoreline components or shoreline structure that ultimately defines the capacity to provide services. Currently, the stakeholder and scientific community are in need of a practical tool capable of collecting data regarding the capacity of tidal shorelines to provide habitat and water quality services. This type of information is essential to understand the changes experienced in shoreline systems and to proactively plan for the changes to come.

Today, coastal ecosystems are more physically vulnerable to rising sea levels. Studies have shown that shoreline change varies with shoreline settings (Anderson et al., 2009). This increases the difficulty of identifying the type and magnitude of the risks that will be experienced at a particular shoreline ecosystem. However, most of the impacts from sea level rise are expected to take place on low lying areas where natural vegetation buffers have been removed. This shoreline condition makes tidal shorelines more susceptible to flooding, accelerated erosion, and seawater intrusion into freshwater environments (Church et al., 2001). These impacts are expected to be exacerbated by future rates of sea level rise (IPCC, 2007).
Management of coastal zones is challenged by the interrelationships between human activities and natural systems. During the last decades Virginia's shorelines were heavily armored to help reduce impacts from storm surges and flood events (CCRM, 2012). As a consequence, acres of tidal marshes and other riparian vegetation were lost reducing the capacity to provide habitat and water quality services in many areas. With an increasing number of manmade structures, shorelines might experience an even greater decrease in vegetation cover. However, recent efforts to reduce the adverse impact from armored shorelines incorporate the use of natural shoreline habitats as buffers for erosion protection (Erdle et al., 2006). This management practice could provide a solution to reduce the risks from future climate change while ecosystem services are being preserved.

The main goal of this dissertation was to develop a local scale methodology capable of determining potential capacity of tidal shorelines to provide habitat and water quality services by 2050 based on the effects of sea level rise and management practices. In this study, the potential capacity of tidal shorelines to provide water quality and habitat services was determined by the conditions of shoreline components. A primary emphasis was placed on the conditions of vegetation cover and vegetation composition present within the system. Chapter 1 generated a practical methodology consisting of two categorical models used to determine the potential capacity of tidal shorelines to provide habitat and water quality services during historic and current times. For Chapter 2, three different approaches were developed to identify the most important physical and natural shoreline change predictors and to determine the response of different shoreline types (i.e. marshes, beaches and managed shorelines) to sea level rise. In Chapter 3, scenarios indicating potential capacity to provide habitat and water quality services by 2050 were generated based on two accelerated scenarios for sea level rise and management practices.

The study sites selected for this study were Mathews County and the City of Hampton in Virginia, U.S. These two sites have similar coastal physical conditions, but different socioeconomic characteristics. There is reason to anticipate that sea level rise and management practices will influence future effects on ecosystem services quite
differently. In addition, the study sites have a variety of shoreline conditions commonly found in Chesapeake Bay, allowing extrapolation of findings to a wide range of settings.

To achieve the study’s main goal several objectives were pursued. They were to:

- Determine historic and current potential capacity to provide habitat and water quality services in tidal shorelines to identify whether there was a trend in the changes experienced in ecosystems and the possible drivers of change.
- Identify the main shoreline change predictors and variations in the response to sea level rise from different shoreline types to better determine future changes in ecosystems due to land submergence.
- Estimate potential capacity to provide habitat and water quality services by 2050 based on impacts from sea level rise inundation and management practices.

The results and methods developed in this dissertation should be useful for the improvement of integrated coastal management plans in Virginia and the Chesapeake Bay. The analytical construct and methods generated in this study should provide practical tools for scientists and coastal managers interested in assessing potential capacity of ecosystems at any point in time. The methods are exportable and with some basic data should be applicable in other estuarine and coastal systems around the world.
Chapter 1

Ecosystem Services Model: Tidal Shoreline’s Potential Capacity to Provide Habitat and Water Quality Services in Mathews County and City of Hampton, VA
ABSTRACT

Ecosystem services in tidal shoreline systems in the Chesapeake Bay experienced an increase in environmental pressure during the last decades mainly due to population growth, land development, and increasing sea levels. These changes jeopardized the potential capacity of shoreline ecosystems to provide habitat and water quality services which are vital for coastal resources, the economy and the coastal population's welfare.

Due to the lack of a reliable, scale appropriate and continuous data set for water quality and habitat services along the Bay's shoreline, this study modified two categorical models generated by the Center for Coastal Resources Management with the main objective of estimating potential capacity for ecosystem services. Based on a GIS analysis and field observations, temporal and spatial changes in potential capacity were determined. A series of natural and anthropogenic components in the sub-aqueous, intertidal, riparian and upland zones were included and assessed as part of the modeling process. These components were identified at randomly selected sites in Mathews County and the City of Hampton. These two localities provided different socioeconomic settings, but similar physical coastal environments.

The Habitat Services Model and the Water Quality Services Model showed a similar trend that indicated a decrease in potential capacity through time in both localities. Although Hampton showed a more acute degradation in ecosystems services since historic times, the potential capacity to provide habitat and water quality services in these two localities seemed to be mainly impacted by anthropogenic activities, specifically development. Increasing impervious surfaces registered since the 1960s in both localities was identified as the main cause for the loss of vegetation and other natural components, consequently decreasing capacity.
INTRODUCTION

ECOSYSTEM SERVICES: HABITAT AND WATER QUALITY SERVICES MODELS

Ecosystem services are essential for the sustainability of the Chesapeake Bay. However, studies quantifying and analyzing the processes generating services are limited along the Bay's shoreline. The lack of quantitative information regarding water quality and habitat services in tidal shorelines could be due to the time consuming protocols required in the field and in laboratory analyses in addition to the large number of personnel necessary to assess entire coastal localities. With increasing disturbances due to climate change effects, rising coastal population and land development, the need for reliable methods to assess water quality and habitat services are imperative. This type of information will allow generating appropriate management policies for the preservation of natural resources and the protection and security of the coastal population.

This current study modified two categorical models included in the Ecosystem Services Model (ESM) generated by the Center for Coastal Resources Management (CCRM). The Habitat Services Model (HSM) and the Water Quality Model (WQM) were created to categorically assess current ecosystem functions based on field observations. This study's main modification to the HSM and WQM consisted of generating a practical modeling method capable of determining ecosystems conditions not only during current times, but also historically (i.e. early 1960s). The new methods provided the opportunity to define a trend in tidal ecosystem services. Mathews County and the City of Hampton were selected for this study. These two localities have similar physical settings, but different socioeconomic characteristics. The differences were expected to impact the capacity for habitat and water quality services.

The HSM and WQM classified ecosystems based on their potential capacity to provide a flow of benefits rather than a direct measure of the processes that generate the services. To accomplish this, the study categorically assessed system structure to estimate potential capacity. The assessment involved field observations, GIS analyses and the use of historic and current aerial images. The field observations collected by the
CCRM provided current conditions of physical shoreline characteristics such as fetch, bathymetry and bank stability. Most of these physical conditions were assumed to remain the same for both historic and current times. Analysis of aerial images allowed the identification of natural and anthropogenic components along tidal shorelines during both time periods. The modified models incorporated additional subaqueous, intertidal, riparian and upland vegetation components that were assessed in an area 60m in diameter. This differed from the original models that assessed conditions at individual points along the shoreline. The analysis of components included digitization of each of the shoreline components present within a system. These additional steps in the modeling process allowed a more appropriate assessment of the ecosystem’s structure during historic and current times.

Vegetation composition was the main indicator of potential capacity for habitat and water quality services. The HSM assumed a high variety of vegetation could provide a more diverse habitat space for multiple organisms. For the WQM, vegetation cover was used to indicate the presence or absence of root systems with the potential of providing a filtering function.

This study’s goal was to determine tidal shorelines’ potential capacity to provide habitat and water quality services in Mathews and Hampton during the 1960s and the late 2000s. A trend based on the changes observed in potential capacity was generated and possible drivers of change were identified. In addition, by defining potential capacity conditions during historic and current times it was possible to generate assumptions of possible future conditions.

ECOSYSTEM SERVICES

Ecosystems are the product of living and non-living processes on Earth (Millennium Ecosystems Assessment Panel, 2005). These living systems maintain and replenish the composition of the air and soil, the cycling of elements through air and water bodies, and many other ecological properties (de Groot et al., 2002; Costanza et al.,
The services produced by these ecosystems have always been essential and vital to humanity.

Recognition of coastal ecosystems as a source of numerous and varied services to human populations and as capital assets is being accepted more and more within society (EPA, 2009; Turner and Daily, 2008). During the last decades, coastal areas have been under a constant increase in environmental pressure. Coastal ecosystem changes originate from a range of driving forces involving natural and human processes. Ecosystem components and services are mainly affected by an increasing population, expanding socioeconomic system, and the gradual increase in sea level rise. This makes coastal areas particularly difficult to manage (Turner, 2000; Boesch, 1999; Turner et al., 1996).

Ecologists define the term “ecosystem” as a dynamic complex of plants, animals, and microorganisms communities. This includes the abiotic environment as well, which in conjunction with the living components interacts as a system (EPA, 2009). Ecosystem functions or processes consist of a subset of natural processes (the flow, storage, and transformation of materials and energy within and through ecosystems) (EPA, 2009; de Groot et al., 2002). These processes and functions describe biophysical relationships that generate ecosystem services (EPA, 2009). In addition, these processes and functions are influenced by ecosystem structures or components (physical, chemical, and biological) and as a result, the condition and quality of these structures and components can affect the services provided (EPA, 2009; de Groot et al., 2002).

The concept “ecosystem services” dates back to the mid-1960s and early 1970s. It has been investigated using two approaches (de Groot et al., 2002). The first way describes the function of an ecosystem through the study of its components. It focuses on the natural functioning of the system and the interconnection of its parts. The second way defines the benefits derived by human population, directly or indirectly from the properties and processes of ecosystems (Pinto et al., 2010; Costanza et al., 1997). It is completely anthropocentric.

De Groot et al. (2002) grouped ecosystem functions into four primary categories. Regulation functions are functions in charge of maintaining ecosystems and the biosphere’s health by the regulation of ecological processes and life support systems
through bio-geochemical cycles and biospheric processes (De Groot et al., 2002). Habitat functions supply protection and reproduction habitat for plants and animals. This function helps maintain and preserve the biological and genetic diversity and the evolutionary processes of species. Production is a function that integrates the food chain from the conversion of energy, carbon dioxide, water, and nutrients into edible components by autotrophs, to secondary consumers, and ultimately into ecosystem goods for humans (e.g. consumption, food, raw materials, energy sources, and genetic materials). Lastly, information functions involve the opportunities that ecosystems provide to humans by maintaining health, spiritual enhancement, recreation, and aesthetic experience, among others.

Ecosystem services or goods are the benefits people obtain from ecosystems (Bennett et al., 2009; Millennium Ecosystem Assessment, 2005). A series of services are obtained from this natural capital: provisioning services such as food, water, and fiber; regulating services that regulate floods, drought, land degradation, and disease; supporting services such as soil formation and nutrient cycling; and cultural services, which provide nonmaterial benefits such as places for recreation and spiritual or religious inspiration. (Bennett et al., 2009; Millennium Ecosystem Assessment, 2005). Table 1.1 shows an overview of some principal functions, goods, and services that are provided by ecosystems and their respective structures and processes.

This natural capital can take different forms, most notably in physical forms (e.g. trees, minerals, atmosphere), and manufactured capital (e.g. machines and buildings) (Costanza et al., 1997). They have become an integral part of every aspect of society and today their continued health depends on society’s use and management. However, despite the increase in interest and publications on ecosystem services and goods, a comprehensive framework for integrated assessment of ecosystem services remains vague (Kremen & Ostfeld, 2005; de Groot et al., 2002). Based on Nelson et al. (2009), there are two different paradigms that scientists are currently applying to generate quantitative ecosystem assessments that ultimately influence policy decisions. The first paradigm involves a broad scale (e.g. regions, planet) assessment of the services and an extrapolation of values based on habitat types (e.g. Turner et al., 2007; Troy and Wilson, 2006; Costanza et al., 1997). This paradigm is definitely a simple approach, and it
incorrectly assumes that the assessment values are consistent throughout an entire habitat type. This paradigm generates general values without incorporating the particularity of each habitat and the uniqueness of the external factors surrounding the system. The second paradigm is known as the "ecological production function" (e.g. Ricketts et al., 2004; Kaiser and Roumasset, 2002). This paradigm is based on modeling the production of one service, in a small area, with the goal of determining the dependency of a provision of the service based on local ecological variables. Nelson et al. (2009) considered that studies based on this approach lack both the scope (number of services) and scale (geographic and temporal) to be considered useful for most policy questions. In order for humanity to continue to benefit from these services, it is necessary to assess this natural capital at a temporal and spatial scale that is deemed most useful for stakeholders. It is also necessary to understand these ecosystems, the interconnections between systems and between external factors, and their behavior to be able to protect their existence and integrity.

An efficient and effective management of living natural capital is necessary to be able to sustain and provide vital ecosystems services such as climate stabilization, drinking water supply, flood protection, pollination, and recreation, and the control of diseases and pests (EPA, 2009; Millennium Ecosystems Assessment, 2005; Turner et al., 2003; Balmford et al., 2002; Daily, 1997; Westman, 1977; Holden and Ehrlic, 1974). The contributions of ecosystems to human populations through ecosystem services are, in part, dependent upon the effectiveness of policies that regulate or impact ecosystems. This requires detailed information at scales that can be useful for decision makers on how specific services are generated. It is also important to explore the outcomes of environmental changes and to determine possible future conditions of ecosystem components and their services.

**COASTAL ECOSYSTEM SERVICES**

Coastal systems are a complex interrelationship of habitats that include aquatic and terrestrial elements. They can be defined based on biophysical features and policy-
oriented definitions. If biophysical characteristics are considered, the upland that interacts with water-borne characteristics such as tides, salinity, winds, and with biota at the land-sea interface, must be included (Duxbury & Dickinson, 2007; Davis & Fitzgerald, 2004). In other words, the coastal system is comprised of parts of the land that are affected by their proximity to the sea and areas of the sea that are affected by their proximity to the land (Burke et al., 2001; Hinrichsen, 1998). This holds true in estuaries as well. These interactions can extend several meters to hundreds of meters inland and offshore. In cases where the physical and ecological connections extend far inland, the coastal area encompasses watersheds and rivers that drain into coastal waters (Beatly et al., 2002). The policy-oriented definitions are mainly used for coastal planning and management purposes. These are determined by legislation and local ordinances which are influenced by distance definitions (e.g. limits of landward municipalities that front the ocean, or based on land use) (Kay and Alder, 2005).

Coastal areas are also subdivided by physical properties that include a range of marine environments. These environments represent diverse dynamic habitats that often coexist and the boundaries that distinguish them are not always clear (Burke et al., 2001). Tidal shoreline systems are part of the coastal ecosystem and include three different environments or zones: riparian buffer, intertidal, and subaqueous zones. The definition and extension of these environments vary depending on the context of their application. For practical purposes, the Center for Coastal Resources Management defines the riparian buffer as the terrestrial area within 9m of the high tide line. The intertidal zone is the land and seabed area that is under the influence of tides, exposed to the air at low tide and underwater during high tide (i.e. between MLW and MHW). The subaqueous zone is considered the area from mean low water line out to a depth of 2m.

Physical coastal components are coastal features that depend on the natural balance and interconnection of land and marine processes. These components can be described as physical units such as wetlands, dunes, riparian areas, and the littoral zone (Hinrichsen, 1988). These components can also be defined in terms of the features that make up these physical units (e.g. plants, animals, microbes, sediments, water, etc.). It is the organization of the components (internally and collectively) that affects the generation of services. In other words, these components represent the infrastructure of
coastal ecosystems and indicate how the system functions and the services rendered (International Union for Conservation of Nature and Natural Resources, 2007). The quality of the complex linkages between components must be considered when trying to identify the capacity of coastal ecosystems to maintain and provide services (Boumans et al., 2002). The interconnections between physical features also suggest that the physical components can influence more than one ecosystem service (Barbier et al., 2011). Ecosystem components are often changed by stakeholders in ways that potentially compromise the long-term provision of important services for the well-being of society and ecosystems: water quality and habitat services (Carpenter et al., 2006).

HABITAT AND WATER QUALITY SERVICES

Habitat and water quality services are fundamental in coastal ecosystems. Society benefits from these natural services, directly and indirectly. Anthropogenic and natural disturbances can influence, limit, and reduce the capacity of ecosystems to provide these and other important services (Millennium Ecosystems Assessment, 2005).

Habitats are generally defined as living spaces in which organisms occur (Millennium Ecosystems Assessment, 2005). Because ecosystem services are produced by the living components found in ecosystems, the maintenance of healthy habitats is essential for the provision of all services, directly or indirectly (de Groot et al., 2002). Habitat services are mainly dependent on the availability of living space for plants and animals, both resident and transient. In addition, the connectivity between and among living spaces is important for the integrity of the service. Although habitat amount is one main factor in determining the size and persistence of populations, the spatial arrangement of habitat patches becomes increasingly more important as habitat is lost (Dobson et al., 2006; Flather and Bevers, 2002). Cumming (2002) and Flather and Bevers (2002) identified a rapid decline in connectivity once 30–50% of habitat is lost. This may have a significant impact on population dynamics and interactions between species (Cumming, 2002, Flather and Bevers, 2002).
Water quality sustains ecological processes that support habitat quality for both plant and animal populations. The indispensable service of water quality depends mainly on the filtering function which is performed by several different ecosystem components such as riparian vegetation, marsh vegetation, and marine organisms such as submerged aquatic vegetation, and oysters (de Groot et al., 2002). For example, in some settings wetlands can remove 20 to 60% of metals in water, trap 80 to 90% of sediment particles from runoff, and eliminate 70 to 90% of nitrogen present in surface water run-off (Daily, 1997).

Society also depends on these services to maintain certain economic activities (e.g. irrigation, fishing, recreation) and for the health of the population. A healthy ecosystem is defined as a system with water quality and habitat conditions that can support a rich biodiversity and protect public health. However, for an ecosystem to provide a service it must have both the opportunity and the capacity for the service (Sutter et al., 2009). In other words, water quality and habitat services are closely linked to the surrounding environment and land use. For example, pollutants can enter wetland waters from point sources (e.g. industrial, waste water treatments) or non-point sources (e.g. agricultural lands, urban areas, failed septic tanks) and destabilize the ecosystem. These alterations in the natural conditions of a wetland generate an opportunity for the system to provide a service. In addition, the wetland must have the internal capacity to hold the runoff and remove pollutants before releasing the water. The opportunity to perform a service is determined by factors external to the system, or the need and demand for the service created by human use of the system. The capacity of a system to provide a service comes from the properties or structure of the system (i.e. the type and organization of the components) along with its landscape position. This means that a wetland can have the capacity to filter pollutants, but if there are no pollutants present in the system, it does not have the opportunity to provide the service. A system that does not have the opportunity to provide certain services today does not mean that it will not have the opportunity in the future.

Water quality and habitat services provide an important framework on which to build our understanding of the natural coastal environment. The Millennium Ecosystem Assessment (2005) also indicated that an efficient way to determine the condition and
capacity of an ecosystem to produce these services is by examining the structure of the system as a whole and the relationship among all its components.

**ECOSYSTEM SERVICES MODELING**

The lack of sufficient high resolution spatial and temporal data limits the understanding of coastal ecosystems. Scientists often use abstract and simplified representations of real world systems to simulate their behavior and test assumptions. Understanding is generated through the application of models analyzed and verified by statistical methods (Hettelingh, 1990).

A model in ecology is considered a representation of the real world. In the past, models have been classified by many authors in different ways depending on their target application and desired output (Hettelingh, 1990). Ecosystem modeling is currently focusing on a more holistic approach that simulates the links and causal relationships between natural systems and anthropogenic processes. (e.g. Nobre et al., 2010; Nobre and Ferreira, 2009). The majority of ecosystem models seek to explain complex scientific knowledge through an output that is easy to comprehend and accessible. In addition, most ecological models are constructed for ecosystem forecasting and management purposes (Rykiel et al., 1996).

Ecosystem models are powerful tools that can provide the required scientific basis to estimate the current capacity of ecosystems to provide services, and to simulate or project their future conditions. Modeling can also be useful for: 1) providing insights about ecological interactions within the ecosystem (Dumbauld et al., 2009); 2) estimating the impacts of multiple activities within a coastal area; and 3) evaluating the susceptibility of an ecosystem to a variety of pressures through scenario simulation (Ferreira et al., 2008).

The usefulness of ecosystem models and any other model is limited by available data and different types of uncertainty. Is the complexity of a system properly reflected in the model structure? Are the data used in the model representative of the system? Is the temporal and spatial scale used in the model appropriate for understanding the
system’s behavior? These uncertainties may lead to unexpected and unrealistic results. Elements such as data collection, model formulation, and estimated parameters contribute to model uncertainty.

The understanding of ecosystem services, characterization of capacity, and sustainability are not possible to achieve without an assessment of the system. This must include identifying the impacts of human activities and natural processes, and tracing their condition and importance over time (Pinto et al., 2010). However, the characterization of ecosystem services has only recently emerged as a field study (Naidoo et al., 2008; Metzger et al., 2006; Schroter et al., 2005). Despite the recent interest in this field there is still little quantitative evidence available to date, and the different approaches and methods available have led to mixed conclusions (Bohensky et al., 2006; Chan et al., 2006).

The identification and quantification of ecosystem services is currently considered a valuable tool for the efficient allocation of environmental resources (Millennium Ecosystem Assessment, 2005). Beaumont et al. (2006) and Pinto et al. (2010) identified two main assessment types and approaches to determine ecosystem services conditions and importance: the economist and the ecological approach.

The first approach is the “economist” approach, which incorporates economic valuation or value transfer approach, focuses on the exchange values of ecosystem services (Troy and Wilson, 2006). This approach is not restricted to economic benefits. It also incorporates the analysis of potential costs, as well as welfare functions. The value transfer approach has become a very essential and practical way to inform stakeholders’ decisions when the collection of data are limited or not feasible (Environmental Protection Agency, 2000). This approach is now considered a very important tool that can easily and quickly generate an estimate of the economic values associated with a particular landscape (Iovanna and Griffiths, 2006). This way of valuing ecosystem services is becoming more appealing due to its versatility and efficiency. However, there exists some controversy in the academic community related to the validity of the method (Wilson and Hoehn, 2006). This is due to the general agreement that primary valuation research always represents the “first best” option for gathering
information related to ecosystem services. However, when primary valuation is not feasible, value transfer is considered a practical way to inform decisions.

Among the limitations of the value transfer approach is the need to use values generated in sites with different socio-economic and biophysical contexts than the areas under assessment (Troy and Wilson, 2006). Another limitation of this approach is the development of values by scaling values down from larger geographic regions or global scales. Fotheringham et al. (2000) and Openshaw et al. (1987) indicate that when values of a geographic phenomenon are spatially aggregated, local patterns of heterogeneity tend to be obscured. The Costanza et al. (1997) study estimating the value of the world's ecosystems is a well known example of the application of aggregated values. Economists are aware of the importance of considering the spatial and ecological context of sites when applying the value transfer approach (Bateman et al., 2002). Currently their challenge is to link the economic valuation of ecosystem services to landscapes based on typological characterizations that are functionally meaningful. Geographic Information Systems (GISs) and the public availability of high quality land cover data sets and biogeographic entities are becoming essential in facilitating and allowing the linkage between valuations of ecosystem services and landscapes types.

The second approach is the “ecological” valuation, defined by Roberts (1992) as the approach that achieves multiple use management by blending the needs of people and environmental values to improve the health, diversity, productivity, and sustainability of ecosystems. This approach highlights the need to incorporate assessment and modeling of biophysical components of ecosystems with the social and economic features of their surroundings. The ability to incorporate biophysical, social, and economic aspects for the assessment of ecosystem services is a relatively new phenomenon, but also a difficult task to achieve (Kreuter et al., 2001). This approach relies heavily on the notion of the “surrogate” parameter and the “objective” parameter (van Jaarsveld et al., 1998). A “surrogate” parameter is considered an “indicator” (Sarkar et al., 2005). The “objective” parameter is considered a target parameter or what scientists and stakeholders ultimately plan to conserve. The term surrogacy implies the relationship and interconnection between the concepts “surrogate” and “indicator”. Surrogacy is exemplified by biologists’ use of well-studied taxa as surrogates for poorly studied groups (Egoh et al.,
In the case of ecosystem services, ecosystem components and functions, and other social and economic characteristics are used as surrogates or proxies for assessing, modeling, and mapping the distribution and quality of ecosystem services. Troy and Wilson (2006) indicated that as scientists improve the match between the biophysical and socio-economic context of an ecosystem with the services of interest, the more accurate the estimates will be.

Similar to the value transfer approach, ecological valuation also deals with two very common scale-related problems. The first problem is related to the scale at which certain functions become important. This scale usually varies. The second problem arises when incorporating and comparing data at different scales. This method can generate inaccuracies in terms of the interrelations and feedback loops between the components and services.

To move forward in this new field of interest, assessments of ecosystem services must generate better and more accurate maps indicating the types of services produced and their quality. They need to determine the likelihood of land use conversion and possible scenarios of impact on service provision, and must identify and understand the flow of goods to nearby and distant human communities (Naidoo et al., 2008). This will require a major interdisciplinary effort, but a vital one to improve informed decision making.

**SHORELINE COMPONENTS**

Shoreline ecosystems are composed of a diverse array of natural components that define the potential capacity of shoreline systems to provide services. The most commonly observed and most feasible to identify are physical (e.g. fetch, beach, land use) and biological components (e.g. vegetation type, vegetation cover). The effects of these components are highly variable within small spatial scales. Some components have the capacity to improve services by providing a living space such as vegetation canopy, sandy or muddy environments for nesting, or food. Other components provide a strong and deep enough root system capable of reducing nitrogen and phosphorus
concentrations at subsurface levels (Dosskey et al., 2010). However, some other features can become a threat to the natural conditions of shoreline systems. Fetch conditions, if too long, can weaken the stability of a shoreline and decrease the capacity of an ecosystem to provide services. Anthropogenic activities are known to adversely impact and disturb natural processes and dynamics. In some cases impacts involve complete removal of natural structures that are required to sustain ecosystem services.

In this study ecosystem services will be determined based on the systems' components, not the actual processes that generate these services. A series of components that can present a negative or positive effect in potential capacity were considered. To determine how these structures or components influence the capacity to provide water quality and habitat services and how these components were assessed, it is necessary to first define the main parameters that this model applied. The ecosystem services of interest in this study and the shoreline components that were assessed are described below. In addition, the main influence (i.e. negative/positive) of these components on the ESM is also specified.

- **Potential capacity**: Using the ecological definition, the capacity of a shoreline ecosystem to provide services will be defined by the system structure, properties, components and organization of components in the landscape. This study determined ecosystem services based on the system structure and not the actual process that generates these services. For a system to have a high capacity to provide services, the conditions of the system structure and components have to be adequate for the system to be able to maintain a sustained flow of benefits. Ultimately the system has to be able to support rich biodiversity. The adequate conditions that shoreline components must have present to support rich biodiversity in a system are based on peer-reviewed literature and best state of the science. These conditions are identified below.

- **Habitat services**: Provide spaces where organisms occur and are essential for the provision of all services. Habitat services are defined from an ecological perspective and represent the potential of providing living space for a diverse community of organisms. To determine this potential, vegetation composition
was used as a proxy. A system with a variety of vegetation can provide potential habitat space for diverse organisms.

- **Water quality services:** Sustain ecological processes and support habitat quality for plant and animal populations. Water quality represents the opportunity to provide the filtering function. This function was assessed by determining the system’s vegetation cover conditions. The identification of vegetation in a system indicates the presence of a root system with the potential to provide the filtering function.

- **Fetch:** This coastal component is mainly applied as a simple measure of relative wave energy. It has been observed in previous studies that long fetch conditions trigger most of the shoreline erosion, especially during high energy storm events (Hardaway et al., 1992). Increasing erosion rates in shoreline systems promote instability of the ecosystem structure by removing material from the area and degrading stability of the riparian bank. Bank sediments, along with stored nutrients and other chemical constituents will be ultimately released into tidal waters impacting water quality and habitat services. Fetch is also highly correlated with marsh planting, a strategy promoted for shoreline protection and viable along low energy shorelines (Knutson et al., 1981). Several studies suggest that marsh creation or natural marsh areas did poorly in areas with fetch conditions exceeding 1,600m (Hardaway and Byrne, 1999). Williams (2001) suggests limiting fetch to <300 m when trying to establish salt marsh plantings and insure natural sedimentation. Knutson et al. (1982) also found that naturally occurring salt marshes in areas with fetch >3,000m had only a 44% probability of survival.

**The ESM considers:** fetch to determine both water quality and habitat conditions. Long fetch conditions (>300m) will be defined as a deteriorating factor for shoreline systems by increasing instability, reducing the total area of natural structures such as beaches and marshes, among others, and by reducing the
opportunity for natural sedimentation to occur. However, in sandy environments, long fetch could increase the opportunity for natural sedimentation through beach and bank erosion. Short fetch will yield an opposite effect.

- **Bathymetry:** Shoreline stability is also influenced by depth conditions in the nearshore zone. Shallow depths, like those observed in tidal flats and sand bars, can attenuate wave energy before reaching the shoreline more effectively than deeper waters (Hardaway et al., 1999). Möller (2006) studied wave attenuation at three different macrotidal saltmarshes in the United Kingdom. The experiment showed an increment in wave attenuation with decreasing depth (0.8% reduction in 0.4m depth to 33% in 0.2m depth). A recent manual specifically for regulators and property owners was generated by the Center for Coastal Resources Management (2010) to help make decisions regarding coastal resources. This manual facilitates decision-making in selecting the most efficient approach for management of a shoreline. Results indicate that living shorelines need to be located in areas where the distance of the 2m depth contour to the shoreline is greater than 10m. These conditions will increase the probability of success in planted marshes, increase the intertidal width, and reduce wave energy.

The ESM considers: shallow nearshore are defined as areas where the distance between the shoreline position and the 2m depth contour is greater than 10m. Deep water areas have the 2m contour located within 10m of the shoreline position. Shallow conditions are considered a positive physical component because they reduce disturbances in existing and planted marshes, allowing for these communities to have the capacity to provide water quality and habitat services.

- **Submerged Aquatic Vegetation (SAV):** Seagrass is considered one of the most heterogeneous landscape structures of shallow water estuarine ecosystems in the world (Boström et al., 2006). These aquatic communities provide physical
structure with ecological functions that resemble mangroves, saltmarshes, and coral reefs (Boström et al., 2006). In Chesapeake Bay, SAV is an important habitat for fish and blue crabs, among other species. Seagrass beds provide a nursery for juveniles, shelter from predators, and are a source of food for many organisms (Horinouchi, 2007; Boström et al., 2006; Lipcius et al., 2005; CCRM, 1999). SAV also modifies energy regimes, stabilizes sediments, and plays a major role in nutrient cycling (Deaton et al., 2010; Fonseca and Calahan, 1992). SAV communities provide functions that enhance water quality by removing suspended solids from the water column, improve water clarity, and add dissolved oxygen to the system (Cerco and Moore, 2001; Churchill et al., 1978).

The ESM considers: the presence of SAV communities enhances the capacity of shoreline systems to provide water quality and habitat services in the subaqueous zone.

- **Beach:** Beaches are components composed primarily of permeable, unconsolidated sediment. In sandy environments, beaches are typically gently sloping. In this study, beaches extend from the mean high water line landward, to where a change in material is observed or where the vegetation line begins. This natural feature provides conditions suitable for different flora and fauna such as saltmeadow cordgrass, insects, arthropods, amphipods, invertebrates, turtles, and bird species that forage the intertidal zone (Dexter, 1967; Pearse et al., 1942). In addition, based on Rosen (1980), beaches can have the largest vertical buffer to the impact of storm surge and waves.

The ESM considers: beaches enhance shoreline systems because of the biodiversity they support, and the erosion protection they offer natural structures located landward. Therefore, this component enhances habitat services in shoreline systems.
• **Mudflat**: Low energy areas and high deposition rates of clay, silt, and biological detritus are necessary conditions for the formation of mudflats. Habitats for organisms in this type of component vary with season and tide. The types of organisms observed in these environments range from large organisms such as birds, angiosperms, and invertebrates to microscopic plants (Little, 2003).

The ESM considers: that the presence of this type of structure increases the biodiversity of the shoreline system and as a consequence, habitat conditions are enhanced.

• **Tidal and inland marsh**: Tidal marsh communities in Chesapeake Bay are distributed along gradients of salinity and tidal inundation (Perry & Atkinson, 2008). In areas where inundation and salt water become a combined stress for marshes, only a few species of vascular plants can survive, biota are limited, but it provides a variety of wetland habitats. In tidal fresh water zones, more species of vascular plants can survive (Odum, 1988).

Inland marshes are defined as marshes not subjected to tidal variations. These marshes can be observed where the water table is at or near the surface (Groffman & Taylor, 1996). Tidal and inland marshes are natural structures that provide a number of important functions such as primary production and detritus availability, wildlife and waterfowl support, shoreline erosion buffering, and water quality control (Perry & Atkinson, 2008). Marshes are considered spawning and nursery habitats. Several important fisheries in Chesapeake Bay are dependent on wetlands for one or more life stage: blue crabs, oysters, clams, striped bass, spot, croaker, and menhaden. Several species of turtle, a vast diversity of birds, among other organisms, also benefit from marsh communities (Erwin, 1996).

Knutson et al. (1982) concluded that over 50% of wave energy was dissipated within the first 2.5m of marshes. This reduces erosion of the adjacent upland, thereby reducing sediment and nutrient inputs in tidal waters from the introduction of bank-derived sediments. As suggested by Rosen (1980) who
identified marsh margins as the least erodible shorelines in Chesapeake Bay, the natural cohesive properties of the fine-grained sediment that comprise marshes make them more resilient to wave erosion than unconsolidated beach material.

The ESM considers: the presence of tidal and inland marshes in shoreline systems increases the capacity of shoreline systems to provide water quality and habitat services. Therefore, the ESM considers tidal and inland marshes valuable natural capital.

- **Phragmites australis**: Phragmites is an invasive species in Virginia and generally is considered undesirable. It tends to invade in areas where disturbances have occurred along the shoreline (e.g., construction sites). Rooth et al. (2003) concluded that the *P. australis* community was associated with higher depositional patterns and faster increase in substrate elevation over relatively short periods compared to other marsh communities. Several studies found different responses from fauna present in Phragmites communities (Weis & Weis, 2003; Wainright et al., 2000) versus native communities. Marsh flora seems to be replaced with different species, which consequently generates a shift in habitat type and changes in other biotic assemblages (Chambers et al., 1999). Although Phragmites is considered an invasive species that over-takes native marsh communities, it helps remove sediments from the water, increases the rate of deposition, and contributes to food webs. However, Phragmites communities are also known to reduce biodiversity (Weis & Weis, 2003).

The ESM considers: these communities provide habitat services for a different assemblage of organisms. Based on this, the presence of Phragmites in shoreline systems are considered to positively influence habitat services.

- **Defended shorelines**: Engineered or hard structures in shoreline systems can negatively impact the physics, geology, biology, and chemistry of an area. Studies have found that structures alter hydrodynamics; wave regime; and
sediment size, transport, and deposition (Dugan et al., 2011; Martin et al., 2005; Griggs, 2005). These alterations can impact vegetation communities in subaqueous and intertidal zones by changing nutrient cycling and sediment deposition. Hard structures also function as barriers for marsh communities preventing landward migration. Consequently, water filtration and habitat connectivity is reduced (Bilkovic and Roggero, 2008; Bilkovic et al., 2006).

The ESM considers: the presence of engineered or any type of hard structure as a negative factor in the provision of water quality and habitat services.

- **Bank height**: Bank height is defined as the approximate height of the riparian bank. Based on a tool generated by the CCRM (2010), Decision Tree for Undefended Shorelines and Those with Failed Structures, all high banks could have greater adverse impacts on water quality and habitat services than unstable lower banks. A failing high bank will erode large volumes of sediments and remove large amounts of vegetation if any are present. A failing bank will wash sediments into estuarine waters along with nutrients stored in the bank, and water quality services will be degraded. In low banks, the loss of sediments typically is less. As a result, most of the vegetation communities and other natural structures providing water filtration, sediment trapping, and nutrient cycling functions may be able to gradually migrate or adapt.

The ESM considers: low bank heights to provide a better water quality service due to the more stable and less erosive conditions that they exhibit in the system.

- **Bank stability**: Defines the potential of a bank to fail due to gravitational forces combined with erosional processes (Dosskey et al., 2010). The instability of a bank is generated by different factors that can act individually or as an integrated unit. Some of the factors that promote instability are bank height, wave action, storm surge, rainfall impact, surface water runoff, groundwater seepage, sediment starvation, bank slope, bank vegetation cover and boat wakes (Hardaway et al.,
All these factors increase the probability of generating an unstable bank and consequent failure. The result is the introduction of large volumes of sediments directly into estuarine waters. Vegetation cover can reduce impacts from erosive factors by anchoring sediment in place. In low lying scrub-shrub and forest-dominated shorelines, vegetation can baffle waves and thereby dissipate energy. However, this effect can vary depending on vegetation conditions. Undercut banks result from a number of scenarios including boat wake activity, and rapid tidal currents in restricted waters. The bank face can be stable or unstable. Instability of the bank face can be exacerbated if vegetation is removed or if elevated water levels increase the zone of impact on the bank face. This is a concern during storm conditions and future sea level rise.

The ESM considers: a stable bank condition to positively influence water quality services in a system. Undercut banks are considered less stable with fewer impacts on services, and an unstable bank is considered the worst condition.

- **Forested lands**: Forested lands provide shade, regulate temperature, and provide habitat for aquatic and terrestrial species (Price and Leigh, 2006). Forested lands also provide both vertical (i.e. height) and horizontal (e.g. trunk thickness, stem lengths, and subsurface root system diameter) dimensions that can benefit many organisms by generating multiple services. Forested uplands along the coast are also known for providing a buffer system that contributes to reduced effects from flooding events. They contribute small and large debris to the soil and nearby waters. This input affects soil chemistry by increasing organic matter in surface soils. In nearshore waters, large debris can provide roughness to the channel bed and bank toe-slopes reducing water velocity and increasing deposition. Debris is also known to provide habitat for various aquatic and terrestrial fauna. Riparian forests can also increase filtration and play a major role in nutrient recycling (Dosskey et al., 2010).
The ESM considers: forested lands one of the best components capable of reducing impacts on both water quality and habitat services in riparian and upland zones.

- **Land use**: Land use is known as one of the most influential components on ecosystem services. Changes in the natural integrity of a system due to land use practices can reduce the biological, chemical, and physical integrity of both terrestrial and aquatic habitats (Van Holt et al., 2006; Henry et al., 1999). Changes in land use can also reduce water infiltration in the soil and increase surface water runoff, nutrients and sediment loadings into streams. In addition these land disturbances can alter the natural cycles of aquatic habitats (Bilkovic et al., 2006; Burcher and Benfield, 2006).

The ESM classifies: the upland into three types of land uses. Developed lands are considered the most negative land use impacting natural services in ecosystems in the riparian and upland zones. For this study, developed lands were classified as impervious surfaces, paved or unpaved roads, shoreline structures such as piers and bulkheads, and cleared lands used as parking lots or bare lands. Agricultural activities are considered the second most negative condition. Natural cover is considered the best condition under the land use component. In this study, natural cover was classified as any natural component present in the land surface. These natural components were: beaches, mudflats, marshes, phragmites, grass, scrub-shrubs, and trees.

**STUDY SITES**

**CHESAPEAKE BAY**

The Chesapeake Bay is the largest estuary in the United States and one of the largest in the entire world (Figure 1.1a). This system receives salt water from the Atlantic
Ocean and fresh water from more than 50 rivers and innumerable smaller tributaries. The strong interactions between land, freshwater, and saltwater make this estuary a very complex, but also very productive system. It annually generates revenues that exceed the six hundred billions due to tourism and commercial fishing, among others. The Bay also provides humans with a highway for commerce, a playground, about 500 million pounds of food and cultural and aesthetic value. However, the recent State of the Bay (CBP, 2012) indicates that the Bay currently exhibits very poor water quality, a reduction in natural habitats, and compromised conditions of many coastal resources and organisms. These circumstances jeopardize the system's capacity to provide many ecosystem services.

The 2000 Chesapeake Executive Council specified that water quality and habitat services are central for the restoration of the Chesapeake Bay's health (Chesapeake Executive Council, 2000). However, the restoration of the Bay's watershed is becoming more difficult from the effects of increasing development associated with a growing population. More than 16.6 million people currently live in the Bay’s watershed and the states of Maryland and Virginia account for 68% of this population.

The unpredictability of climate change also increases the challenge in the restoration of the Bay's former conditions. Based on a 35 year database from 10 tide gauges from Norfolk, VA and Baltimore, MD the relative rates of sea level rise in the Chesapeake Bay range from 2.91 to 5.80mm per year. These rates are higher than the rates observed in many other areas in the U.S. East Coast (Boon et al., 2010). Ramhstorf (2007) predictions indicate that the Chesapeake Bay will be experiencing an increase of 0.7m (700mm) to 1.6m (1,600mm) in sea level by 2100. Based on different CO2 scenarios, more variations are expected in the climatic conditions of the Bay during the 21st century (Pyke et al., 2008).

MATHEWS COUNTY AND CITY OF HAMPTON, VIRGINIA

This study focused on the shorelines along Mathews County and City of Hampton in the state of Virginia (Figure 1b-c). The socioeconomic characteristics differ between
localities with more rural lands observed in Mathews and a highly developed landscape in Hampton. However, these localities share similar physical coastal conditions (i.e. mean tidal range, coastal slope, rate of relative sea level rise, shoreline erosion and accretion rates, mean wave height, geomorphology) (Boruff et al, 2005). More importantly, the coastal area of both localities lies below the 6m elevation contour (Titus and Wang, 2008). This implies future greater risks of inundation for developed coastal areas and the loss of shoreline features.

MATHEWS COUNTY

Mathews County is located on the Middle Peninsula of southeastern Virginia. The county is bordered by Mobjack Bay to the south, Chesapeake Bay to the east, North River to the west, and the Piankatank River to the north (Figure 1.1b). Mathews County is adjacent to Gloucester County to the northwest. According to the U.S. Census Bureau, the county has a total area of 652.677 km² of which 222.739 km² is land, 429.938 km² is water, with 559.04 kilometers of shoreline.

Mathews is considered a rural area with a slow growth rate compared with other localities in the vicinity. The majority of the land use is classified as either rural or low density residential (Berman et al., 2000). Most of the residential development currently occurring in Mathews is on the waterfront. However, in accordance with the Chesapeake Bay Preservation Act the County established 30m (100 ft.) buffers landward of all streams, adjoining wetlands, and related sensitive areas (Resource Protection Areas (RPAs)). In addition to the RPAs, the County also incorporated the Resource Management Areas (RMAs), which is an extension of the inland limit of the RPA buffer.

The shorelines in Mathews County show a high variability in physical properties. The fetch characterizing the shores range from fetch-limited creeks to open Bay long fetch conditions. Most of the tidal shorelines in Mathews County are found in narrow, small creeks and rivers with low wave energy (Hardaway et al., 2010).

The types of shorelines vary along the County’s coast. The North River is characterized by having very low uplands and marsh coasts. The eastern part of the coast
has very high energy barrier beaches and marshes. High uplands are commonly observed along the Piankatank River. For 2010, about 80 kilometers of Mathews' 559.04 km of shoreline were already hardened (Hardaway et al., 2010). From these 80 km, 27.36 km were built in the last ten years and this amount is expected to increase greatly in the years to come.

The intertidal zone is mainly characterized by the presence of marshes, wetlands, maritime forests, high and low energy shorelines, beaches, and dunes. In the subaqueous zone, submerged aquatic vegetation and oyster reefs are still present in some areas. These coastal components are currently providing habitat for different aquatic and terrestrial species, reducing wave energy and erosion, and stabilizing shoreline sediments.

Historically, shoreline change rates varied from 0 m/yr to over ±2.44 m/yr along the Bay coast (Byrne and Anderson, 1978). A recent study from Hardaway et al. (2005) calculated a shoreline change rate from 1937 to 2002 for Mathews County that varies from 0.88m/yr to -3.17m/yr. Accelerating sea levels are converting some of the Mobjack Bay-facing marshes to marginal marshes (Strange et al., 2008). Some marshes and unnourished beaches will be completely lost in the Piankatank River due to greater than 3.0m bank elevations in this area. Beaches facing the Chesapeake Bay are currently showing signs of high erosion rates. Marshes and beaches with sufficient sediments to accrete and keep pace with a 7 to 16mm/yr increase of sea level are likely to continue migrating inland, but most marshes are likely to be lost with a predicted 7mm per year of sea level increase.

CITY OF HAMPTON

The City of Hampton is an independent city and one of the seven major cities that compose the Hampton Roads metropolitan area. It is located on the southeastern end of the Peninsula. The City shares physical boundaries with Newport News and York County to the west and it is contiguous to the Chesapeake Bay waters to the east and the James River to the south (Figure 1.1c). Based on the U.S. Census Bureau, the City of Hampton has a total area of 352.76 km², of which 134.16 km² is land, and 218.60 km² is
water (CCRM, 2011; Hardaway et al., 2005b). This includes 12.07 kilometers of tidal shoreline along the James River, 12.87 kilometers along the Chesapeake Bay, and 8.05 kilometers along the Back River.

The City of Hampton also established 30m (100 ft.) buffers landward of all streams, wetlands, and related sensitive areas and RMAs with an additional 30m extension of the inland limit of the RPA buffer. This is in accordance with the Chesapeake Bay Preservation Act.

Shorelines are characterized by a wind climate defined by a long fetch exposure mainly to the northeast and east across the Chesapeake Bay (Hardaway et al., 2005b). Most of the shorelines along Hampton River are bulkheaded. The bayfront shorelines and lowland areas prone to tidal flooding are occupied by extensive marshes.

The coasts in the City of Hampton have experienced strong impacts in the past due to coastal flooding during hurricanes and nor’easters (Boon et al., 2010). In addition, the combination of effects from sea level rise and land subsidence in this city will expose many shorelines and coastal communities to greater risks from sea level rise in the future. Observations already confirm the inundation of marsh areas, converting these to tidal flats and then open water.

Historically (1937-2002), the shoreline rate of change for Hampton shorelines varied between 0.21 m/yr to -1.25 m/yr (Byrne and Anderson, 1978). Hardaway et al. (2005b), calculated similar rates between 1937-2002 of 0 to -1.25 m/yr. Based on the expected future increase in sea level, planners indicate that the developed portion of the City is almost certain to be protected by defended shorelines while other areas east of the city are already experiencing shoreline erosion (Strange et al., 2008).

METHODS

Historic and current aerial photographs, digitization and assessment of shoreline components, and a Geographic Information System (GIS) were used to determine historic and current capacity of shoreline systems to provide services. Ecosystems’ capacity to provide habitat and water quality services was determined by assessing eighteen different
shoreline components within a buffer 60m in diameter. The components' conditions were assessed from the 2m depth contour in the subaqueous zone to the adjacent upland zone. A total of 150 sites were assessed in Mathews County and 120 in City of Hampton. Capacity to provide services was determined for two different years at the same sites in both localities (i.e. Mathews: 1968 and 2007; Hampton: 1963 and 2009). Capacity was calculated numerically then classified as High, Moderate, or Low. This assessment allowed for identifying temporal and spatial variations in capacity as well as possible drivers of change.

SHORELINE CLASSIFICATION AND SAMPLING SIZE

Shoreline systems are dynamic environments that can be characterized by different processes, physical conditions, and dynamics. Depending on the type of shoreline system, conditions in shoreline components can vary and consequently different capacities to provide services can be observed. In these natural environments, studies have identified two main components that define shoreline types: fetch and bank height (CCRM, 2010; Hardaway and Byrne, 1999). To select the sites to be assessed appropriately for this analysis and to determine the adequate sample size per county, all shoreline systems were classified based on fetch and bank height conditions.

To classify shorelines based on different fetch and bank heights, an existing shoreline inventory database was modified for Mathews and Hampton (CCRM, 20011; CCRM, 2009b). This inventory was performed based on a set of protocols developed by the Comprehensive Coastal Inventory Program (CCI), a part of the Center for Coastal Resources Management (CCRM). These protocols were created to describe shoreline conditions along Virginia’s tidal shorelines. The shoreline inventory assessed and characterized coastal components in the shorezone, which extends from a portion of the riparian zone seaward to the shoreline (CCRM, 2011). The assessment was based on observations made from a moving shallow draft vessel, navigating at slow speed and parallel to the shoreline. In the field, the data was logged using a handheld Trimble GeoExplorer III, GeoExplorer XT, or GeoExplorer XH GPS unit. These units collected
georeferenced data, which was then processed in the lab to generate highly accurate records of shoreline features and conditions.

ArcGIS, a GIS software, was used to process and integrate the data. The shoreline inventory data are line features and contain categorical data for a series of shoreline components (CCRM, 2011). In addition, bathymetric data from NOAA was converted from raster format to vector, specifically as a polygon feature. SAV data collected and published by VIMS’ as polygon features was downloaded from the SAV Mapping Program (VIMS, 1995). Ultimately, bathymetric data and SAV data were combined with the shoreline inventory using the Identity Tool in ArcGIS. For the bathymetric data, the 2m depth contour was extracted using a 10m buffer from shoreline position. The 2m depth contour selection is based on a series of factors included under the Shoreline Components section. A 100m buffer from the shoreline position was used to extract SAV communities for 1971, 2007, and 2009. From the union of all these components, shoreline units or reaches of shoreline were generated. Shoreline units are defined as shoreline segments where the shoreline components do not change.

The shoreline units were then converted from line features to point features in ArcGIS. The population of points was classified based on six different combinations of fetch (long= >300m or short = ≤300m) and bank height (0-1.5m, 1.5-9m, >9m) (Table 1.2). The selection of fetch and bank height intervals are explained under the Shoreline Components section. Five shoreline classes were generated for Mathews County and four for Hampton (Tables 1.3 and 1.4). Shoreline classes with high bank heights (i.e. 1.5-9.1m and >9.1m.) were not well represented in the population of points generated for Mathews and Hampton. No presence of bank height class >9.1m were observed in Hampton and only bank heights >9.1m with long fetch were observed in Mathews. Due to the unequal representation of some shoreline classes, the points were regenerated at equal intervals of 10m using the Construct Points Tool under the Editor menu. This generated thousands of shoreline units or points that became the sample population (Table 1.5). Individual files were generated for each shoreline class containing just the point sites within the same classification and their respective shoreline components. After classifying the shoreline units based on fetch and bank height, the sample size per shoreline class was determined.
SAMPLE SIZE PER SHORELINE CLASS

A total of 200 random points were selected from each class in both localities using the "Subset Feature" tool. These randomly selected points were only used for this test and later discarded from the total population of point samples. For each file containing a shoreline class a new field in the attribute table called "Total Score" was created twice, one to calculate water quality services and a second time to calculate habitat services. The total scores were calculated for each point by adding up categorical values previously assigned to each shoreline component based on their conditions present at each specific point. This test exemplifies how the actual ecosystem model works, but in a more simplistic way.

To determine the number of samples per class required for an adequate analysis of the data with an error <10%, the total scores were exported to a statistical package and the following equation was applied:

\[ N = \frac{\text{Population Variance}}{(\text{Allowable error})^2 (\text{Population Mean})^2} \quad \text{Equation 1} \]

where \( N \) equals total number of samples for a specific error percentage (Murphy & Willis, 1996). This test determined that a total of 30 shoreline units (i.e. points) per class was an adequate sample size for the type of data and analysis used in this project. In addition, 30 sites per shoreline class describe most of the variability and natural complexity of shoreline units. The maximum and minimum errors calculated per county are shown in Tables 1.6 and 1.7.

A total of 150 point samples, 30 for every shoreline class, were randomly selected from Mathew’s sample population and 120 points from Hampton’s. These two localities are not characterized by having extensive areas with high bank conditions. This generated some clustering of point samples in high bank classes specifically in Hampton under the 1.5-9.1m Short Fetch class and in Mathews under the 1.5-9.1m Short Fetch and >9.1m Long Fetch class. The points selected for Mathews and Hampton were used for both historic and current time analyses.
After calculating the appropriate sample size and identifying the sites to be assessed, the buffer zone necessary to define the area to be assessed at each point was determined.

DETERMINING THE ASSESSMENT BUFFER SIZE

Literature reviewing ecosystem assessment buffer sizes for ecological studies demonstrates that there is no ideal buffer size for all applications in all areas (Wenger, 1999). Accordingly, some judgment and setting of priorities is necessary to obtain a buffer width for a specific set of functions. The effectiveness of a buffer size depends on the objective of interest. In many ecological studies factors such as the function of interest, environmental risk, and sustainability are considered before selecting a buffer size. In addition, the size can also differ depending on the parameters of interest, and conditions present in the study site, among others.

Studies have found that a buffer size of 60m can provide enough area to assess stream bank stabilization and aquatic food webs, water temperature moderation, nutrient removal, sediment control, and flood control (Wenger, 1999). Wildlife habitat assessment may require larger buffer sizes depending on the nature of the organism of interest. However, it is not an objective of the current study to determine migration, feeding patterns or other types of processes that can define habitat services. The current study will assess primarily, but not exclusively, vegetation components as proxies for habitat and water quality services. To accomplish this, a buffer 60m in diameter will be used to assess a variety of shoreline components to ultimately determine both water quality and habitat services.

For this study several factors were considered to determine the buffer size or area where the shoreline components were assessed. The buffer comprised an extent big enough to:

- include presence of each type of component;
- include four different types of vegetation (trees, scrub-shrubs, grass, and marshes);
• represent shoreline system variability; and
• include adequate image resolution

In addition, the buffer allowed to 1) define an absolute location, 2) provide a spatial scale sufficient for the occurrence, integrated assessment, and classification of shoreline components based on their conditions in the system, and ultimately 3) generate a protocol and facilitate future on site assessments of capacity of a shoreline unit to provide water quality and habitat services.

Two additional factors were considered for establishing an adequate buffer size. The first factor is based on the practicality necessary for the collection of shoreline components. As part of CCRM’s shoreline inventory protocol, components were only collected for the riparian and intertidal zones. The protocol states that a riparian zone corresponds to an area with a 9m width that extends from the shoreline position inland. This width was defined by the Chesapeake Bay Program (2002) and based on the state of science at that time (Kirk Havens, personal communication, February 15, 2012). The surface area encompassed by this 9m width corresponds to where approximately 90% of runoff is produced, specifically nitrogen (Palone and Todd, 1997). This definition has been incorporated into VA, MD, DE, and NC’s shoreline inventory protocols. In the field, the riparian zone is not a physically defined boundary. It is mainly used to define the dominant use of the land parcel most proximal to the shoreline and not to define the function of the system.

The second factor is based on the Chesapeake Bay Preservation Act, which requires local governments in the Commonwealth of VA to create a vegetated buffer no less than 30.5m (100ft) wide as part of the Resource Protection Area (RPA). An RPA consists of tidal shores, tidal wetlands, non-tidal wetlands or water bodies with perennial flow. The main purpose of the RPA is to improve water quality function by removing or decreasing negative impacts from groundwater and surface water entering Chesapeake Bay and its tributaries (Baird and Wetmore, 2006).

Based on the Chesapeake Bay Preservation Act, the buffer size pertinent for this analysis required it to be 30.5m in width or wider. To determine the most appropriate size based on the type of data and the type of analysis applied in this study, a test to determine the length of the most common type of shoreline unit was applied. This test
consisted of 1) determining the mode of the length of shoreline units, 2) determining the mode of the length of individual components, and 3) calculating how much variability of the most common lengths is present in the entire population of shoreline units. Determining the buffer size based on the most common length of shoreline units and components will allow the final model to represent most of the different types of shoreline systems present in Mathews and Hampton localities.

The output from this test showed that the most common length of shoreline units in Mathews was 17m. This length in shoreline unit represents 77% of the locality. A length of 11m was identified as the most common length of shoreline units in Hampton, representing 62% of the locality. In Mathews County the smallest mode among the components was 19m in length and the largest mode was 56m in length. In Hampton the range in modes was between 1m to 56m in length.

A buffer size of 60m in radius was selected as appropriate for this study. This 60m radius represents a shoreline unit length of a 120m. The buffer was determined to be a practical, effective size for the type of analysis, and useful for management practices. It was generated the same way at each point site that was assessed to allow comparisons between sites. It also incorporated the most common length in shoreline units in Mathews and Hampton and the most common lengths in shoreline components (Figures 1.2a-b). In addition, it doubles the area required by the Chesapeake Bay Preservation Act allowing the opportunity to assess not just the riparian zone, but also to assess possible interactions between the RPA zone and adjacent upland zone.

ASSESSMENT ZONES

The buffer generated to assess shoreline components, with a 60m radius in size was subdivided into different assessment zones (Figures 1.3a-b). These five zones were identified as: Subaqueous, Shoreline Position, Intertidal, Riparian, and Adjacent Upland (Figures 1.3-1.6). The boundaries defined for each zone are not solely based on variations in natural characteristics of the landscape. These zones are mainly defined by practical definitions and local management stipulations (i.e. RPA). In addition, the
characterization and boundaries of these zones were mainly based on the nature of the model and the data available. Each of the zones was delineated and digitized for historic and current conditions using aerial photographs in ArcGIS. Historic aerial photographs for Mathews (1968) and Hampton (1963) were scanned and ortho-rectified by the Shoreline Studies Program (Hardaway et al., 2005a; Hardaway et al., 2005b). Current geo-referenced digital aerial images were collected by the Virginia Base Mapping Program. In each of these zones, different shoreline components were identified, assessed, and classified based on the conditions that each component exhibited. Below are the characterizations and boundaries of each of the assessment zones.

• **Subaqueous Zone (SZ):** The SZ extends from the shoreline position seaward. Within this zone two physical structures were assessed: fetch and bathymetry. Fetch and bathymetry conditions in historic times were assumed to exhibit the same conditions as in present time. In addition, within the SZ the conditions for the SAV community were determined.

• **Shoreline Position (SP):** The shoreline position was defined differently depending on the type of shoreline system. For a marsh shoreline, the shoreline position was delineated at the edge of the marsh. For a defended shoreline, the shoreline was positioned at the seaward or outer side of the hard structure. For a beach shoreline, the shoreline position was generated by identifying the dark edge defining the boundary between wet and dry sand material.

Based on the 60m radius, the total shoreline length assessed is ~120m long. A segment of shoreline defining the shoreline position within a 60m buffer is also identified as a *shoreline unit*. SP for historic years (1963 and 1968) in Mathews and Hampton were digitized by the Shoreline Studies Program and adjusted to this study’s preferences.

• **Intertidal Zone (IZ):** The IZ boundary extends from the SP to the inland edge of a beach, mudflat, and/or tidal marsh in the site. If no beach, mudflat or tidal marsh was present, the IZ was defined as the SP.

• **Riparian Zone (RZ):** The RZ extends from the inland edge of the IZ to 9m landward based on the Chesapeake Bay Program (2002) definition (Havens, K.,
personal communication, February 15, 2012). The RZ is the zone of most interest in this study since it is where most of the ecosystem services are generated (Klapproth and Johnson, 2009). Within this zone, the following shoreline components were assessed: bank height, bank stability, forested lands, inland marshes, vegetation cover, vegetation composition, and riparian land use.

- **Adjacent Upland Zone (UZ):** The UZ was defined to determine possible effects in the riparian zone induced by changes in the adjacent lands. The UZ extends from the inland edge of the RZ (9m distance) to the outer boundary of the 60m buffer. In this zone forested lands, inland marshes, and upland land use were assessed.

DIGITIZING AND CLASSIFICATION OF COMPONENTS

All components observed within the 60m buffer were digitized and assessed. The components were digitized in ArcGIS v.10.1 as polygon features in order to obtain area calculations. The digitization was done using historic and current aerial photos. The digitization of components, especially using historic images was mainly based on texture differences, changes in gray and black shades, and identification of defined shapes (e.g. buildings). Due to differences in resolution between historic and current aerial photographs, a comparable resolution scale between both types of photographs was identified. For 2007 images, all components were digitized based on a resolution of 1:600. For historic images all components were digitized based on a 1:1,000 resolution. All components were digitized using NAD_1983_UTM_Zone_18N projected coordinate system. The same system was used for the aerial photographs.

The inventory generated by CCRM was used and modified for Mathews and Hampton. This database contained categorical classifications for a series of shoreline components assessed in the field. The components that were included in this inventory and used in this model were: Phragmites australis, defended shorelines, bank stability, and bank height. By digitizing all the components present within the 60m buffer, it was
possible to calculate an area for each component at each site using GIS. However, not all components could be digitized from historic imagery: fetch, bathymetry, Phragmites, bank height, and bank stability. These components were assumed to be the same during historic and current times. Additional components were added to the original inventory by digitizing them when possible using aerial photographs: fetch, bathymetry, marshes, beaches, mudflats, vegetation cover, vegetation composition, riparian inland marsh, riparian land use, upland inland marsh, upland forested lands, and upland land use. These components were included to better assess the conditions for water quality and habitat services at each site.

Forested lands were classified differently from the other components. The definition of forested lands varies depending on the study type and depending on the type of landscape assessment. For this reason, it was necessary to develop a protocol to identify tree polygons in GIS that could be classified as forested lands. Forested lands were determined based on the original definition of the CCRM Ecosystem Services Components. The definition applied by CCRM identifies forested lands as areas covered with trees with a width > 9m and greater than 5.5m (18 ft.) high. This 9m width is based on the total maximum area that CCRM assessed in coastal areas to generate the shoreline inventory. This is the same width of the riparian zone applied in this study.

Most assessed components were digitized as polygons in GIS. However, ArcGIS does not calculate the widths and lengths for polygons. The only two calculations that GIS provides are area and perimeter. For this reason, it was necessary to know how much area is in a polygon 9 m in diameter. To determine this it was necessary to first classify tree polygons in two different ways: 1) Individual or group of trees: an area less than 9m diameter; and 2) Forested Lands: an area equal or bigger than a 9m diameter.

Historic marshes, beaches, and inland water bodies were verified using the Tidal Marsh Inventory report for Mathews (1974) and Hampton (1975) (Silberhorn, 1974; Barnard, 1975). For marshes that were difficult to identify in historic images, but whose presence was verified using the report, a polygon ≤5m wide was created. Defended shorelines in historic times were verified using the Shoreline Situation Report for Mathews (1975) and Hampton (1975) (Hobbs et al., 1975a; Hobbs et al., 1975b). Current
marsh communities were verified using the Tidal Marsh Inventory for Mathews (2007) and Hampton (2009) (CCRM, 20011; CCRM, 2009b).

After digitizing all components for all sites in Mathews and Hampton, the “Clip” tool from ArcGIS was used to “cut” the area of each shoreline component that belonged to a particular assessment zone (Figure 1.7).

A digitization error test was performed to determine an approximate error value included in the model due to the digitizing procedure. The digitization was done by two different people to incorporate variations in perception. A group of sites was selected for each locality to represent different land uses (i.e. natural vs. developed sites). The sites were defined by the 60m buffer generated for this study. All the components present in the buffer were digitized. Sites were digitized three different times to generate replicates. Replicates were done using both historic and current aerial photos. The standard errors of the areas were calculated for the most common components found in the intertidal, riparian and upland zones (Table 1.8).

MODELING OF ECOSYSTEM SERVICES

Each component was assigned a categorical value that represented the component condition at each site and at each assessment zone (Table 1.7). These categorical values, as specified in CCRM (2010) also depended on the effect of the component condition on water quality and/or habitat services. The highest categorical values represent the best components conditions for a particular service and the low values represent the less adequate conditions. Not all components were considered for both models, but some were applied in both.

The categorical values were used as model values and ultimately applied in model equations to generate a model score. The model score generated for each component represents the component’s condition and influence on water quality and/or habitat services. For some components no defined equations were generated and the categorical values were used as model scores. However, for format purposes these model scores will be considered as equations (e.g. fetch). The highest model values were assigned to the
component conditions that have a positive influence or the best positive effect on water quality and/or habitat services. Low model values represent conditions with negative or poor influence on water quality and habitat services. Below are the model equations assigned to each component. Each component contains in parentheses the initials of the Water Quality Services Model (WQM) and/or the Habitat Services Model (HSM) to indicate which model(s) the component was applied. An example of how these equations work uses the SAV component, which was applied in both the WQM and HSM. Using GIS the SAV area was calculated for each site and this area was divided by the total area of the subaqueous zone. After determining the proportion that SAV represents in the subaqueous zone, the proportion was multiplied by the SAV’s model score (Table 1.9). If SAV was present the proportion was multiplied by a model value of 3; if it was absent the model score was 0. In the case of the fetch, no equation was applied and the model values were used as model scores.
**Subaqueous Zone Model Score:** (Equation 3 + Equation 4 + Equation 5)  

SAV (WQ & H): SAV area / Total area of subaqueous zone * 3  

Fetch (WQ & H): Short = 1.0; Long = 0.5  

Bathymetry (WQ & H): Shallow = 1.0; Deep = 0.5  

**Intertidal Zone Model Score:** (Equation 7 + Equation 8 + Equation 9 + Equation 10 + Equation 11)  

Beach (H): Beach length / Shoreline length * 3  

Mudflat (H): Mudflat length / Shoreline length * 3  

Tidal marsh (WQ & H): Marsh length / Shoreline length * 3  

Phragmites (H): Present = 3; Absent = 0  

Defended shorelines (WQ & H): Present = 1 ; Absent = 3
**Riparian Zone:** (Equation 13 + Equation 14 + ......+Equation 18 + Equation 22)

**Bank height (WQ):**  
0-1.5m = 3; 1.5-9.1m = 2; >9.1m = 1

**Bank stability (WQ):** Stable = 3; Undercut = 2; Unstable = 1

**Forested lands (WQ & H):** Total forested land area in riparian zone /Total area in riparian zone * 3

**Inland marshes (WQ & H):** Marsh area in riparian zone / Total area in riparian zone * 3

**Vegetation cover (WQ & H):** Total vegetation in riparian zone / Total area in riparian zone * 100; then, this percentage was classified as:  
Total (>75%) = 3; Partial (25-75%) = 2; Bare (<25%) = 1

**Vegetation composition (i.e. grass, scrub-shrubs, trees, inland marsh) (WQ & H):**  
High composition (presence of 3 or more types of vegetation) = 3 ; Low composition (presence of 1 or 2 types of vegetation) = 2; None = 0

**Riparian land use (WQ & H):**

Natural area in riparian zone / Total area in riparian zone * 3

Agriculture area in riparian zone / Total area in riparian zone * 2

Developed area in riparian zone / Total area in riparian zone * 1;

Then, (Equation 19 + Equation 20 + Equation 21)

Model rule: If land use contains paved or industrial land use, then model score is 0
**Upland Zone:** (Equation 24 + Equation 25 + ....+ Equation 29)

**Forested lands (WQ & H):** Total forested land area in upland zone / Total area in upland zone * 3

**Inland marshes (WQ & H):** Marsh area in upland zone / Total area in upland zone * 3

**Upland land use (WQ & H):**
- Natural area in upland zone / Total area in upland zone * 3
- Agriculture area in upland zone / Total area in upland zone * 2
- Developed area in upland zone / Total area in upland zone * 1;

Then, (Equation 26 + Equation 27 + Equation 28)

Model rule: If land use contains paved or industrial land use, then model score is 0

**Final Model Score:** (Equation 2 + Equation 6 + Equation 12 + Equation 23)
These model equations were applied and run in ArcGIS using the Model Builder tool (Figure 1.8). As described earlier, model scores were computed for individual components. In addition, model scores were calculated for each individual assessment zone. These zone scores were computed by adding up the model scores of all the components within the zone (i.e. Equation 2, Equation 6, Equation 12, and Equation 23). The final model scores for Water Quality and Habitat Models were computed by adding all the zones model scores (i.e. Equation 30). Final scores defined the capacity to provide services at a specific site. These final scores were generated individually for each of the shoreline classes and for each of the shoreline units in Mathews County and the City of Hampton.

Final scores were classified as High Capacity, Moderate Capacity, or Low Capacity. Because of the nature of the data (i.e. categorical) applied in these models and the limited amount of analysis that can be performed, the final scores were classified using Jenks Natural Breaks in GIS. This method creates an optimal number of classes in the data by minimizing the variance within a class and maximizing variance between classes (Smith et al., 2013; Jenks & Caspall, 1971). The maximum and minimum model scores that each capacity class can have are shown in Tables 1.10 and 1.11. These model scores are nondimensional scores that can only be viewed as a state of capacity and not as a measure of capacity. These scores can also be applied to determine differences and averages of states from multiple sites.

**SPATIAL VARIATIONS**

The spatial analysis for the HSM and WQM consisted of the application of an interpolation method. The interpolation was based on the ordinary kringing method from the Geostatistical Analyst package in GIS. The output included a prediction surface indicating the distribution of the total model scores for each ecosystem model and for each year that was assessed. This continuous surface predicts the final model scores for areas that were not included in this study using scores from shoreline units that were assessed. However, kringing methods incorporate mathematical and statistical methods
that include probability. Due to this reason, the values calculated for the predictive surfaces generated for this study are not perfectly predictable. Ultimately, these continuous surfaces were only used to provide an approximate overall view of the distribution of capacity along the localities tidal shoreline systems and to identify spatial variations in capacity during historic and current times.

RESULTS AND DISCUSSION

HABITAT SERVICES MODEL

The HSM defined habitat services from an ecological perspective. Habitat services represented the potential of an ecosystem to provide living space for a diverse community of organisms (Fiedler et al., 2008). To determine the potential of a shoreline unit to provide living space, the modified version of the HSM used vegetation composition as a proxy. This shoreline component specifies the presence of multiple vegetation types (i.e. trees, forested lands, scrub-shrubs, grass, tidal and inland marshes) and the area fraction that the vegetation occupies within a shoreline unit. A varied vegetation composition (i.e. presence of 3 or more types of vegetation) and larger area fraction will indicate a high capacity to provide organisms with a diverse living space area. This will ultimately be translated into high biodiversity present within the shoreline unit.

In addition to vegetation composition many other natural and anthropogenic components were included and assessed to generate this model. The inclusion of a variety of shoreline components (Table 1.9) was necessary to generate integrated assessments for shoreline units. This allowed determining total habitat model scores based on the interconnection of components present in the unit and not based solely on individual components. The inclusion of natural and anthropogenic components in this model provided a better interpretation of the landscape and the opportunity to identify patterns and relationships that ultimately described historic and current capacity conditions to provide services.
Table 1.9 indicates a possible maximum total model score of 44.0 and a minimum of 5.0 for the HSM. This table shows the range of final model score values that the assessed shoreline units were able to receive. These final model scores were generated at each individual shoreline unit by aggregating model scores from each shoreline component present. All scores generated in this model are non-dimensional. They are, however, generated and classified using the same protocol, making it possible to compare between assessed years and between localities.

In addition, final total model scores were assessed at different spatial scales. Temporal variations in capacity were determined at the locality level, for each assessment zone (Table 1.9) and for different shoreline types (Tables 1.3 and 1.4). Spatial variations based on differences in capacity were assessed as well. Final total model scores were classified based on different capacities to provide services: high, moderate, and low (Table 1.10). Temporal and spatial variations in capacity were assessed for both Mathews and City of Hampton.

WATER QUALITY SERVICES MODEL

The WQM is based on the premise that water quality functions are limited to the capacity of a shoreline unit to filter nutrients and other pollutants. The filtering function in this modified version of the WQM was determined using riparian vegetation cover as a proxy. As previous studies established, dense vegetation cover in the riparian zone is an indication of a strong root system capable of filtering high concentrations of nutrients and contaminants in the land surface and underground and to help reduce sedimentation in adjacent estuarine waters.

Many other natural and anthropogenic components were included and assessed to generate this model. The inclusion of a variety of shoreline components (Table 1.9) was necessary to generate an integrated assessment of shoreline units. This allowed determining total water quality model scores based on the interconnection of shoreline components present in the unit and not based solely on individual components. The inclusion of components representing natural and anthropogenic activities provided a
better interpretation of the landscape and the opportunity to identify patterns and relationships that ultimately helped explain changes in capacity.

Table 1.9 indicates a maximum final model score of 41.0 and a minimum of 7.0 for the WQM. These final scores were generated by aggregating model scores from each of the shoreline components present in the unit. All scores generated in this model are non-dimensional. They are, however, generated and classified using the same protocol, making it possible to compare between assessed years and between localities.

In addition, final total model scores were assessed at different spatial scales. Temporal variations in capacity were determined at the locality level, for each assessment zone (Table 1.9) and for different shoreline types (Tables 1.3 and 1.4). Spatial variations based on differences in capacity were assessed as well. Final total model scores were categorically classified as high, moderate, and low (Table 1.11). Temporal and spatial variations in capacity were assessed for both Mathews and City of Hampton.

**Habitat Services Model: Mathews County**

*Temporal and Spatial Changes in Habitat Capacity*

Based on 150 shoreline units assessed in Mathews County (Figure 1.9a), the HSM showed temporal and spatial variations in capacity to provide habitat services from 1968 to 2007 (Figures 1.9a-b). An increase in developed lands and a decrease in areal size for most natural components, especially vegetation components, were identified during a 39 year period. However, high vegetation composition was observed in developed shoreline units. The loss of most vegetation components was followed by a decline in capacity to provide habitat services. Capacity to provide habitat services seemed to decrease through time along the eastern shorelines in Mathews while an increase in capacity was observed at the west coast (Figures 1.9-1.10).
Habitat Service Capacity: Mathews 1968

The HSM indicated a total of 87 sites (58%) from the total amount of sites that were classified with high capacity to provide habitat services. The moderate capacity had 59 sites or 39% of the total number of sites and 4 sites (3%) were classified as low capacity sites (Figure 1.11a).

High capacity sites were mainly observed along the northeast and northwest shorelines of the county, specifically along the Piankatank River (A) shorelines and Mobjack Bay (B) (Figure 1.10a). An additional area characterized with high capacity was also observed at the southeast.

Mathews County showed mostly natural conditions and low disturbances from anthropogenic activities during 1968. This can be confirmed by the overall land use conditions in 1968 which were predominantly comprised of natural lands (Figure 1.11b). For the HSM, high capacity areas were identified as forested lands and high banks (north east), extensive marshes (northwest and south east) and as tidal shorelines with SAV, beach, and low bank components (south east).

Agriculture and developed lands showed a smaller total area. This land use distribution was observed for both capacity classes high and moderate. However, agricultural and developed lands showed a larger area in the low capacity class. As observed in the continuous surface in Figure 1.10a, low capacity areas were identified at Gwynn Island’s northwest side. This area was already heavily developed during historic times and presented shoreline component conditions lower than average. Additional sites with low capacity along the county’s shorelines were mainly due to the presence of agricultural lands, developed lands, or due to the absence of marshes in the system. These land use conditions helped explain the low capacity observed at the 4 sites in Figure 1.9a and located in sheltered environments. The sheltered location of three of these low capacity sites was expected to help increase capacity compared to other sites exposed to the Bay conditions. However, these low capacity sites were characterized by developed or agricultural lands contrary to most long fetch sites where extensive beaches and marshes were mainly observed.
A closer look into these three capacity groups (Table 1.12) confirms that conditions for nearly all shoreline components within the high capacity class exhibited the highest averaged model scores. The low capacity class presented the opposite for the most part. This pattern was expected due to the nature of the model. Model scores in Table 1.12 represent the averaged conditions of a component under different capacity classes. However, these averages can be influenced by the total number of sites within each class. Based on this, instead of relying on averaged model scores to identify patterns and variations in capacity, results presented below are mainly based on changes in shoreline components areal size.

**HSM Variations in Trends for Shoreline Components: Mathews 1968**

Even though high capacity sites were expected to present the best conditions for shoreline components, high vegetation composition was commonly observed in sites with moderate capacity. From a total of 14 sites with high vegetation composition identified during 1963, 13 sites were under the moderate capacity and only 1 site under the low capacity class. Interestingly, no site with high composition was observed in the high capacity class, but based on area fraction data this class was the only one to have presence of all types of vegetation. The high capacity class was characterized by a large area fraction of trees, forested lands (i.e. trees = individual trees with area <64m² / forested trees = trees area ≥64m²) and inland marshes (Figure 1.12a). This study considered these vegetation types as some of the most important components for habitat services and as the main original land covers in Mathews during historic times.

The unexpected presence of high vegetation composition in moderate capacity sites seemed influenced by differences in land use between capacity classes. Data for riparian land use (Figure 1.12b) confirmed that in 1968 the areal extent for developed lands was larger under the moderate and low capacity classes. Land development usually, if not always, removes the natural land cover of an area transforming it into impervious surfaces and allows for smaller and less dense vegetation to grow (Peterson et al., 2012; McKinney, 2002). Larger areas of developed lands in moderate capacity sites
may explain the larger area fractions for scrub-shrubs and grass. This type of vegetation was considered by Clagget et al. (2013) as secondary vegetation and the dominant ground cover in developed areas. This study defined scrub-shrubs and grass as pervious surfaces that function similarly to impervious surfaces due to the compaction that occurred during the development process.

As part of the HSM’s output it was expected to observe sites with high vegetation composition classified as high capacity sites. However, the results showed this shoreline component was influenced by development. Ultimately, the interaction between these two components defined the capacity of sites with high composition as lower than high. The use of vegetation composition as the main proxy for habitat services in this study will need to be reassessed in future applications of the model. However, this component was treated the same way as the rest of the components indicating that the classification used for vegetation composition was similar to other vegetation components.

**Habitat Service Capacity: 2007**

In 2007 a decrease in capacity for habitat services was observed. The total number of sites under the high capacity class reduced in number while the low capacity class experienced an increase (Figure 1.11a). High capacity sites went down by 28% where 19 sites no longer provided high capacity for habitat services. Only 52 sites from the original 87 in 1968 retained their classification as high capacity sites, 29 sites reduced capacity to moderate and 6 sites to low capacity.

Moderate capacity sites went up by one additional site. Thirty one sites classified as moderate in 1968 kept the same classification by 2007, 16 sites were reclassified as high capacity and 12 sites reduced their classification to low capacity.

A large increase in sites was identified for the low capacity class increasing from 4 sites in 1968 to 22 sites by 2007. The 4 low capacity sites found in 1968 were classified the same by 2007. Low capacity shoreline units increased in number by 15% indicating weaker conditions in shoreline components.
This decline in sites was mainly observed around the northeast area (Figures 1.9b and 1.10b). However, the opposite was observed along the western shorelines where an increase in sites with high capacity was observed. The pattern observed in Mathews County indicates a spatial shift in sites with high capacity from the east to the west by 2007. In addition, this change in capacity distribution also represents a shift in high capacity sites from high bank shorelines to mainly low lying areas. This pattern was mainly due to a higher increase in developed lands at the northeast and eastern side of the county. Another factor that could have influenced this change in distribution is the decrease through time in agricultural lands at the west coast. In 2007 many of these lands previously used for agricultural activities and now abandoned presented a transition in land cover to secondary vegetation types.

Most sites with low capacity were observed at the eastern shorelines of the county. These sites were mainly observed along the northeast, east, and at Gwynn’s Island shorelines that were already heavily developed during 1968.

The increase in the number of sites coincided with an increase in area for developed lands and a decline in vegetation components (Figure 1.13). The expansion through time in anthropogenic activities could explain the deterioration observed in habitat services from 1968 to 2007 in Mathews County.

Most site reclassifications from moderate capacity to high capacity were driven mainly by an increase in vegetation cover. An increase in total vegetation area in some sites was due to abandoned agricultural lands overgrown by trees, grass, and/or scrub-shrubs. In some other sites forested lands showed a growth in area. Site reclassifications from high capacity to moderate or low were in most cases driven by a loss of natural cover due to development. Sites with no change in capacity through time were mainly comprised of large natural areas such as extensive marshes, forested lands or were already highly developed.

In 2007 beach and vegetation composition presented better conditions under the moderate capacity class. Even though more high capacity sites (n = 29) showed beach presence than in the moderate class (n = 19), the latter exhibited a larger areal extent for the component (Figure 1.14a). Most beach shoreline units (n=16) in the moderate capacity class were characterized by wide sandy beaches and high capacity during 1968. By 2007 all 19 sites experienced an increase in development and a loss in total vegetation area reducing the capacity from high to moderate. This pattern could be an indication of the increase in population in lands closer to the shore as it was well documented in the Chesapeake Bay region for the last decades (Gill et al., 2009). In addition, the increase experienced in development by 2007, specifically for the moderate capacity class, coincided with the largest number of defended shorelines since 1968 (Appendix I).

As identified in historic times, a larger representation of sites under the moderate capacity class showed high vegetation composition in the riparian zone in 2007 (Figure 1.14b). The same interaction between developed lands and vegetation composition was observed (Figure 1.14c). Development triggered the transformation of the original land cover into secondary vegetation (i.e. scrub-shrubs and grass). Ultimately, the secondary vegetation does not provide the same vertical and horizontal spatial scales that trees and marshes provide for habitat services (Dosskey et al., 2010).

Mathews' HSM: General Findings

Habitat services experienced a negative impact from 1968 to 2007 in Mathews County. The number of high capacity sites declined 28% while low capacity sites increased by 15%. However, high capacity sites were the most prevalent type of site after a 39 year period. Based on spatial analyses, the decrease in capacity seemed to be triggered by the loss of vegetation components generated by an increase in developed lands.
In 1968 the high capacity class showed the largest count of sites followed by the moderate capacity class. Based on the results, natural conditions dominated in Mathews during historic times and explained the overall high capacity for habitat services observed in the county. Almost all shoreline components presented the best conditions under the high capacity class. Only vegetation composition showed a different trend than expected by having the largest number of sites with high composition under the moderate capacity class. This particular finding seemed to be mainly defined by a large presence of developed lands in sites with moderate capacity that generated a larger presence of secondary vegetation and consequently increasing the vegetation composition of the area.

By 2007 vegetation composition and beach components showed more beneficial conditions for habitat services under the moderate capacity class. These two components were mainly influenced by anthropogenic activities. The interaction between these natural and anthropogenic components diminished the capacity of sites with beach presence and high vegetation conditions to moderate. Based on these findings, it will be necessary to reevaluate the classification for vegetation composition in future applications of the model and the methods used to assess it.

**Water Quality Services Model: Mathews County**

*Temporal and Spatial Changes in Water Quality Service Capacity*

The WQM’s output indicated that most assessed shoreline units were characterized by moderate capacity conditions during historic and current times (Figures 1.15a-b). Based on the results, the number of sites with low capacity increased through time. As observed in the HSM, capacity to provide water quality services was reduced by developed lands. However, the WQM was highly influenced by the presence of marshes by including most shoreline units with these vegetated communities under the high capacity class. The spatial distribution in capacity showed a shift through time from east to west as identified in the HSM (Figures 1.16a-b).
The WQM identified 56 sites as high capacity, 85 as moderate capacity, and 9 sites were classified as low capacity (Figure 1.17a). Fifty seven percent of the sites were characterized by moderate capacity indicating water quality services were more degraded than habitat services during 1968. Only 37% of the total sites during 1968 presented adequate conditions for a constant provision of water quality services and 6% were already highly deteriorated.

The spatial distribution of capacity for the HSM and the WQM was similar during 1968. Sites with high capacity to provide water quality services were mainly observed along the northwest shorelines of the county, specifically along the shorelines of the Mobjack Bay (Figure 1.16a). Most of the highest capacity scores were observed at the southeast. These areas vary in fetch and bank conditions, but presented mainly undisturbed tidal and extensive marshes.

As observed in the continuous surfaces for the HSM, low capacity areas were mainly identified at Gwynn Island’s northwest side. As previously indicated this area was heavily developed during historic times and presented shoreline component conditions lower than average. All sites observed with low capacity in the WQM during 1968 were mainly characterized by the absence of tidal and inland marshes. In addition, all sites presented low conditions for land use in the riparian zone and even lower for the upland zone.

This difference in site distribution between capacity classes suggests that vegetation cover in most assessed shoreline units was insufficient to generate the necessary processes and functions that ultimately define water quality services. However, based on land use data for 1968 natural cover was the largest land use type observed in each capacity class (Figure 1.17b). Most of the vegetated lands were concentrated in moderate capacity sites due to the large number of sites identified with this capacity class.

Even though the natural cover was the main land use type for all capacity classes during historic times, only high capacity sites showed presence of inland marshes and almost all sites (n=51 from 56 total sites) showed presence of tidal marshes (Figure
The presence and condition of tidal and inland marshes seemed to define capacity for water quality services. Table 1.9 indicates that one of the main differences of the WQM from the HSM was the assessment of tidal marshes in the intertidal zone as the only natural component. Based on this, sites with marsh presence in the WQM presented higher water quality total model scores especially if no anthropogenic activities were present.

However, most sites ($n = 53$) under the moderate capacity class showed a riparian zone completely vegetated. A lower amount showed a vegetated upland zone ($n = 40$) and only 30 sites showed both riparian and upland zones covered in vegetation. These vegetation patterns suggest that even though many of the sites classified as moderate showed no presence or a small presence of marsh components, these were providing adequate conditions for water quality services probably to a similar extent as high capacity sites. Currently, there is no comparative study of the influence of different vegetation types on water quality (Dosskey et al., 2009). Many studies have identified the different benefits between herbaceous vegetation (e.g. stabilize soils more rapidly and effectively) and woody plants (e.g. stabilize high, steep banks, stronger and deeper roots that increase shear strength deeper in the soil) (Dosskey et al., 2009). However, based on Simon and Collison (2002) the best conditions for bank stability and water quality processes were observed in areas with a mix of woody and herbaceous vegetation.

WQM Variations in Trends for Shoreline Components: Mathews 1968

Based on shoreline components assessments, conditions for vegetation composition, land use, and forested lands were higher under the moderate capacity class for both riparian and upland zones (Figures 1.18b-d). WQM’s output showed a large number of sites from the total number of sites with high vegetation composition ($n = 12$ out of 14) under the moderate capacity class. This generated a large area for the vegetation types that were assessed (Figure 1.18b). The same pattern was observed for the HSM. High vegetation composition was identified where most developed lands were located (Figure 1.18c). The presence of high development promoted an increase in areal
size for secondary vegetation that ultimately increased vegetation composition (Peterson et al., 2012; Mckinney, 2002).

Areal size for natural land use and forested lands was larger under the moderate capacity class as well (Figures 1.18c-d). This pattern in the moderate capacity class can be explained by the low presence of tidal marshes, relative to the high capacity class, and by the absence of inland marshes. Based on observations of aerial images, tidal and inland marshes were mainly observed where forested lands spatial extent was small or absent. This could be explained by the fact that most roots from woody plants do not grow far below the water table because this soil is poorly drained. In addition, most sites with extensive marshes at low bank heights were classified as high capacity as well. This indicates that most natural lands for high capacity sites were comprised of marshes while natural lands for the moderate class were characterized by woody vegetation including secondary vegetation types.

**Water Quality Service Capacity: Mathews 2007**

Tidal shorelines experienced a reduction in total model scores for the capacity to provide water quality services from 1968 to 2007. The number of sites under the high capacity class was reduced from 56 sites in 1968 to 40 by 2007(Figure 1.17a). This indicates a reduction of 29% of high capacity sites in 39 years. In addition, this decrease translates into 16 fewer sites with high capacity for water quality services in the county. From the 56 sites identified with high capacity in 1968, only 29 were classified as high capacity in 2007. Of the remainder 17 sites were reclassified as moderate capacity sites, and 10 sites reduced their capacity to low.

Site count for the moderate capacity class also experienced a decrease of 10 sites since 1968. From the 85 sites with moderate capacity in historic times, 10 sites increased to high capacity, and 19 sites reduced to low capacity. Even though this capacity class experienced a decrease in sites through time, it was the most common capacity observed in sites during 2007.
Low capacity sites increased through time from 9 sites to 35 sites by 2007. Twenty six additional sites showed low capacity to provide services. Six of the original 9 sites identified in 1968 retained their low capacity classification. Two other sites increased capacity to moderate and 1 site showed high capacity improvements.

The spatial patterns for the WQM in 2007 coincided with the trends from the HSM in 2007 (Figures 1.9b, 1.10b, 1.15b and 1.16b). High capacity for water quality services showed a shift from the east coast in 1968 to the west coast by 2007. This could be attributable to an increase in developed lands along shorelines at the eastern side of the county. However, some areas at the southeast showed high capacity by 2007 and were mainly dominated by extensive marsh areas protected by the Chesapeake Bay Preservation Act. As observed in the WQM in 1968 and in the HSM, low capacity areas were at the east side specifically at Gwynn’s Island and at the northeast of Mathews.

The distribution in sites per capacity class and the spatial trends observed for the WQM in 2007 coincided with an increase in developed lands and a loss in vegetation cover through time (Figures 1.19a-b).

**WQM Variations in Trends for Shoreline Components: Mathews 2007**

As identified in 1968, forested lands, natural land use and vegetation composition showed a different pattern from expected (Figures 1.20a-c). Conditions for these components were higher in moderate capacity sites. As previously suggested, the components’ patterns were mainly driven by influences from developed lands and the presence or absence of marsh components.

**Mathews’ WQM: General Findings**

Mathew’s sites were characterized by moderate capacity for water quality services in 1968 and 2007. Vegetation cover conditions declined through time generating a decrease in sites for the high and moderate capacity classes of 29% and 12% respectively.
On the contrary, a 74% increase was experienced in low capacity sites. During historic times most high capacity sites were concentrated along shorelines located at the southeast. By 2007, a decline in capacity was observed at the east and the number of high capacity sites increased at the west side probably due to an increase in area of developed lands.

Moderate capacity for water quality services was widespread among shoreline units during 1968. Differences in sites distribution between capacity classes seemed influenced by the presence or absence of marsh components. Most marsh components were observed in high capacity sites. In addition, vegetation composition, forested lands, and land use components showed a larger presence under the moderate capacity class. This pattern was the same for 1968 and 2007. Vegetation composition was influenced by developed lands as observed in the HSM. Ultimately, the decrease in capacity to provide water quality services seemed driven by the loss of thousands of square meters of native vegetated lands and development expansion in Mathews County during a 39 year period.

Capacity by Shoreline Type: Mathews

Capacity to provide habitat and water quality services in Mathews County was reduced through time at almost all shoreline types (Figures 1.21a-b). Shoreline units with high banks and long fetch showed the best capacity for habitat services while low bank heights were observed with the best capacity for water quality services in 1968 and 2007.

The upland zone was the most impacted during the 39 year period (Figure 1.21c). All bank types showed a decrease in capacity due to a decrease in natural components conditions and increase in development. These changes in land use ultimately reduced the capacity to provide services by 2007.
Capacity for Habitat Services by Shoreline Type: Mathews

The best capacity for habitat services during 1968 and 2007 was observed in shoreline units with high bank heights and long fetch conditions (MB>9.1L) (Figure 1.21a). Even though this model considered long fetch conditions to reduce the capacity for ecosystem services by increasing the physical forces shorelines are exposed to, long fetch sites presented mostly natural land use with dense forested lands and a small presence or complete absence of developed lands. Sites with moderate bank heights and long fetch (MB1.5-9.1L) showed the opposite conditions with a larger number of polygons with trees, a possible indication of forest fragmentation. In addition, the low capacity observed in this shoreline type was mainly due to the large influence from anthropogenic activities observed in the riparian and upland zones and its increase through time (Figure 1.21d).

Capacity for Water Quality Services by Shoreline Type: Mathews

Based on the WQM, banks with low height (i.e. MB0-1.5L and MB0-1.5S) presented the best capacity to provide water quality services (Figure 1.21b). Low banks provide the elevation conditions necessary for the development and horizontal migration of marsh components (Cahoon et al., 2009). As indicated in the WQM outputs, this model was influenced by the presence of tidal and inland marshes. Shoreline units with presence of marsh components showed higher model scores than sites with no presence of these vegetated communities. However, high banks presented the lowest capacity due to an increased level of instability from potential bank failure and because the elevations are not suitable for marsh formation.

The highest averaged model scores during 1968 and 2007 were observed in low banks with short fetch (MB0-1.5S) (Figure 1.21b). MB0-1.5S was characterized by a high presence of marsh components and relatively low anthropogenic influence. Studies have identified these physical conditions as more adequate and preferable for some shoreline components, specifically tidal marshes (Williams, 2001; Hardaway et al.,
1992). The lowest capacity for the WQM in 1968 was observed at shoreline units with high banks and long fetch (MB>9.1L). This type of shoreline showed low presence of tidal marshes and no presence of inland marshes. In 2007 shorelines with bank height 1.5-9.1m and long fetch showed the lowest capacity. This bank type experienced the largest influence from developed lands, especially in the riparian zone, reducing its capacity (Figure 1.21d).

**Mathews: Overall Changes in Shoreline Components and Capacity**

Table 1.13 summarizes the changes in area or amount identified for each of the shoreline components that were assessed in the HSM and WQM. The patterns observed in this table provided possible explanations to the decrease in capacity through time in Mathews County. Even though these changes are based on the assessment of 150 sites, it is assumed that similar trends were experienced along the rest of the tidal shorelines in the county.

Based on the assessment generated by this study a clear decline was identified in the total areal size for most natural shoreline components (i.e. tidal marsh, riparian vegetation composition, riparian and upland trees, inland marsh, forested lands, land cover and natural land use). The decline in vegetation components coincided with an increase in anthropogenic components through time represented by an increase in the number of defended shorelines and in the areal size for riparian and upland developed lands. Of all the anthropogenic components that were assessed, only agricultural lands were drastically reduced by 2007. This was expected due to changes in the economic structure observed in Mathews County since the 1960s.

Compared to the riparian zone, the upland zone showed a larger area for vegetation composition and cover as well as for developed lands in 1968 and 2007. The upland assessment zone also presented the largest loss for vegetation components possibly due to a larger total area of developed lands. However, the largest increase in developed lands was observed in the riparian zone. Even though the upland zone
comprised the largest assessment area among the rest of the zones (i.e. subaqueous, intertidal and riparian zones), the percent difference in developed lands between the riparian (71.67%) and the upland zone (57.56%) may indicate a shift through time in anthropogenic disturbances from the upland zone to the riparian zone.

SAV, beach, mudflats, scrub-shrubs and grass were the only natural components to show an increase in area by 2007. The increase in SAV communities could be explained by the nature of the historic data used in this study. Data from 1971 was used to determine the total area of these vegetated communities within the subaqueous zone. During this year a drastic decline in SAV was registered (Orth & Moore, 1983 under SAV). After the decline, multiple efforts to bring back these communities slowly helped in the restoration and increase in SAV area in different areas of Chesapeake Bay. By 2007 the presence of SAV was observed for almost all the different shoreline types that were assessed. Sites mainly located upriver or in small tributaries currently show a small presence of SAV. Moore et al. (1999) indicated these shoreline types are now too enriched with sediments and nutrients thereby reducing the opportunity for SAV to grow. Even though SAV communities have not fully recovered, their current presence in some sites that were assessed could indicate better capacity for habitat and water quality services.

Beach and mudflats also experienced an increase in area through time, and there are several possible explanations. In some sites a genuine increase in sand or fine clay material could be responsible for the increase identified in these two components. However, as was indicated previously, most shoreline units with beach presence experienced high development and shoreline armoring during the last decades. It is possible that beach nourishment projects in some beach systems generated a pattern of accretion through time. Another possible explanation for the pattern observed in beaches and mudflats is based on an error during the digitization process. It is clear that part of the increase in area for these two components is due to omission error specifically during 1968 due to low image resolution. Variations in shoreline components due to this type of error are discussed below in more detail.

As identified previously, the presence of high vegetation composition increased where high development was taking place. Table 1.13 also shows secondary vegetation
(i.e. scrub-shrubs and grass) increasing with increasing area of developed lands. However, an overall decrease in vegetation composition was observed mainly due to a major loss in the original vegetation cover of the county comprised of inland marshes, trees and forested lands. This transformation of the land’s surface and in vegetation could also indicate that since 1968 many areas along Mathew’s shorelines experienced a possible displacement of organisms and a conversion in the biological structure of the system from a very complex habitat structure to a more simple, constantly changing, and more fragmented one (Mckinney, 2002). In addition, many other services could be affected by degradation of the water quality.

Natural components assessed in this study were considered essential for a high capacity provision of habitat and water quality services. The decline in conditions observed in these components can represent a decrease in the availability and diversity of living space as well as an inefficient filtering system of contaminants in waters along tidal shorelines. Studies along the Chesapeake Bay shorelines have identified signs of these changes in habitat and water quality services (Cooper, 1995). This degradation in tidal shorelines could cause an impact in organism diversity, the loss of essential resources for natural and economical purposes, and a possible further deterioration of ecosystems by an increasing number of invasive species in the county. Future conditions along Mathew’s tidal shorelines are expected to show a decline in the original vegetation cover if development keeps expanding especially in the riparian zone and if secondary vegetation keeps growing. Ultimately, habitat and water quality services along most tidal shorelines in Mathews could be adversely compromised if the pattern defined since 1968 continues.

Habitat Services Model: City of Hampton

Temporal and Spatial Changes in Habitat Capacity

A total of 120 sites were assessed in the City of Hampton to determine the capacity of tidal shorelines to provide habitat services during 1963 and 2009 (Figures
1.22a-b). Based on the HSM’s output, Hampton was characterized by heavy development during historic times. Anthropogenic activities increased drastically over time reducing the natural vegetation cover by 2009. Consequently, capacity for habitat services was greatly impacted through time and to a greater extent than observed in Mathews County. High capacity for habitat services was concentrated in shorelines along the northern side of the county and mostly low capacity areas were observed at the southern side where most of the developed lands are located (Figures 1.23a-b). This spatial distribution was similar for both time periods.

**Habitat Service Capacity: Hampton 1963**

In 1963 a total of 47 (39%) sites from the total number of assessed sites were classified as high capacity sites. Forty three sites (36%) were classified with moderate capacity and 30 (25%) sites showed low capacity (Figure 1.24a). Sites with high capacity for habitat services in Hampton were mainly located along the southwest branch of the Back River (Figure 1.23a). Areas such as Grundland Creek, Harris River, Stony Pt., Tabbs Pt., and Marsh Point presented the highest capacities. These sites were characterized with forested lands, high vegetation cover, tidal and inland marshes, and fewer defended shorelines. An additional high capacity conglomerate was observed at the Salt Ponds. This is a semi-enclosed area with high presence of extensive marshes. Low capacity shorelines during 1963 were mainly concentrated at the south, where most of anthropogenic activities and loss of vegetation was observed during historic times.

Hampton’s sites distribution per capacity class clearly presented a much lower capacity for habitat services than Mathews at about the same time period. Due to the lack of historic published data in ecosystem services for Hampton and Mathews County it is not possible to corroborate these findings. However, based on Figures 1.24b-c the land use percent patterns between the localities’ high and moderate capacities were very similar. The only exception was a lower percentage for natural land use and a higher percentage for developed lands under the moderate capacity for Hampton. This indicates that a larger proportion of lands under the moderate capacity were developed in Hampton.
than what was observed for Mathews. This larger presence of developed lands and lower presence for natural lands could explain the overall lower capacity conditions observed in the City. The low capacity class showed similar percentage for developed lands, but a larger proportion of agricultural lands for Mathews County as it was expected. The difference in natural lands between localities could be due to the large dissimilarity in the number of sites classified with low capacity during historic times (i.e. Hampton = 30; Mathews= 4).

**HSM Variations in Trends for Shoreline Components: Hampton 1963**

A large area size for SAV and most sites with presence of these communities were identified under the moderate capacity class (Figure 1.25a). These submerged vegetated communities were mainly observed at the northeast area of the City during the 1970s (Orth et al., 1979). Sites located in this area were characterized by long fetch, high bank conditions, and presence of anthropogenic activities (i.e. mostly marinas). These conditions explain the classification of the sites with moderate capacity.

Shoreline units with beach presence showed a larger area size under the low capacity class (Figure 1.25b). Most of these sites were exposed to long fetch conditions and located at the north, east, and southern shorelines along Hampton. These sites also showed some of the widest beaches during historic times. In addition, the low capacity class showed the largest number of defended shorelines (Appendix II). This could indicate that the wide beaches observed in the locality were due to nourishment projects. The Shoreline Situation Report for the City of Hampton from 1975 indicates the presence of artificially stabilized shorelines in these areas, however there is no direct indication that these sites were nourished. The combination of long fetch conditions and the presence of anthropogenic activities could have influenced the classification of these sites as low capacity areas.

Another possible explanation for the classification of most beaches as low capacity sites is the size of the assessment buffer used in this study. To assess shoreline
components, a buffer 60m in diameter was used. Due to the large width characterizing some beaches only the sandy and unvegetated land surface was included in the assessment buffer. In this case, other shoreline components were excluded at the site generating a low total model score.

As observed in Mathews County, a large number of moderate capacity sites presented high vegetation composition in the riparian zone ($n = 9$ of a total of 15). The rest of the sites were identified under low capacity sites. Figure 1.26a shows presence of four different types of vegetation under the moderate capacity class. This class also showed the largest area fraction for scrub-shrubs and grass. This secondary vegetation type coincided with highly developed sites (Figure 1.26b). As Mckinney (2002) and Clagget et al. (2013) indicated and as observed in Mathews County, developed lands seemed to promote the increase of secondary vegetation and less permeable surfaces. The low capacity class also presented four vegetation types and showed the largest area size for developed lands. However a large number of sites ($n=13$) were identified with no presence of vegetation due to highly developed conditions. This lack of vegetation in a large number of shoreline units reduced the presence of high vegetation composition.

Contrary to Hampton’s vegetation composition, Mathews County’s inland marshes were not observed under the moderate or low capacity classes; they were only present under the high capacity class. This suggests that human disturbances in Hampton were found at every shoreline type including sites with marsh presence. However, most of the original vegetation cover in Hampton during historic times was observed under the high capacity class. No secondary vegetation type was found under this class indicating that high capacity sites were mostly pristine shoreline units.

**Habitat Service Capacity: Hampton 2009**

By 2009, major changes in the land surface were observed impacting already compromised habitat services (Figure 1.27a). From 120 sites, only 33 sites were classified as high capacity (Figure 1.24a). This corresponds to a 30% reduction in high capacity or 14 less sites providing the necessary shoreline component conditions for
habitat services. From the original 47 sites that were observed in 1963, a total of 25 sites remained as high capacity, 6 sites reduced their capacity to moderate, and 16 were lowered to low capacity. Even though the high capacity class experienced a decline in the number of sites, this capacity class consisted mostly of the best shoreline components conditions and the highest total averaged scores per sites.

The moderate capacity class presented a decline in sites as well. In 1963 a total of 43 sites were classified as moderate capacity sites and by 2009 only 22 sites were identified indicating a 49% decrease. Almost half the amount of sites with moderate capacity was lost after 46 years. From the original 43 sites identified in 1963, only 15 sites retained the moderate capacity classification, 8 sites increased capacity to high, and 20 sites reduced capacity to low.

A total of 65 sites were identified with low capacity by 2009 doubling the total count of sites observed in 1963. Thirty five additional sites were identified with low shoreline components conditions suggesting a general state of degradation of the city's habitat services. Almost all sites with low capacity in 1963 were low conditions in 2009. Only one site showed improved conditions in shoreline components to moderate capacity. With most sites remaining as low capacity and only 1 site reclassified to a higher capacity, this probably indicates the loss of resilience.

Most sites with the best capacities were still located around the same locations identified in 1963 however, Grundland Creek area showed a decrease in capacity through time (Figures 1.22b and 1.23b). These clusters of high capacity areas were mainly high bank shorelines. No sites with high capacity were observed at the southeast and southern sides of the locality.

Low capacity areas were evenly distributed around the City by 2009 (Figure 1.23b). This coincided with the impacts registered at all shoreline types discussed in previous sections. These impacts were mainly linked to a widespread development in Hampton. This spread can also be observed in Figures 1.22-1.23, where development seemed to keep spreading through time from south to north, from highly developed shorelines to more natural lands.

The decrease in capacity for habitat services coincided with a decrease in natural land use and an increase in developed lands (Figure 1.27). The increase in developed
lands not only converted natural land surface into impervious surfaces, but it also reduced vegetation cover, and it is highly probable it caused vegetation fragmentation (Mckinney, 2002). Ultimately these processes can: 1) alter the natural dynamics of local and migratory organisms; 2) decrease presence of native plants; 3) promote the growth of exotic or secondary vegetation; and/or 4) completely eliminate vegetation from shoreline units. In addition, these possible alterations to land surface and vegetation components could change the biological structure of shoreline units and ultimately affect the natural processes and functions that generate services (Peterson et al., 2012).

Even though Mathews County presented a decrease in capacity through time, the observed changes were not as acute and drastic as observed in Hampton. This is exemplified by the land use patterns for both localities. Developed lands in Hampton more than doubled by 2009 and comprised a much more extensive area than observed in Mathews for a similar time period.

**HSM Variations in Trends for Shoreline Components: Hampton 2009**

As identified in 1963 SAV and beach components showed a different pattern than expected. These components were identified with a larger spatial extent under the moderate capacity class. SAV and beach showed the same physical and anthropogenic conditions observed during 1963 (i.e. high energy, defended shorelines and presence of marinas) (Figures 1.28a-b). The only difference in patterns between 1963 and 2009 was a shift in classification from low to moderate capacity of sites with the widest beaches along the eastern shoreline. This change suggests that most of the healthiest beaches were influenced by nourishment projects and were less developed than beaches less exposed to the Bay. However, most shoreline units with beach presence during 2009 were identified as low capacity sites. The classification of most beaches as low capacity sites coincided with a decline in beach area from 1963 to 2009 (Figure 1.28c).

Three additional components showed a different pattern than expected by 2009 (Figures 1.29a-d). Mudflats, riparian and upland forested lands, vegetation composition
and natural land use showed a larger area size under the low capacity class. The fact that most of the area for these key components was identified under the low capacity class suggests that most of these natural components were highly disturbed by anthropogenic activities during 2009.

Mudflats are components mainly found in low energy areas with high rates of sediment deposition. The increase in area size through time observed for this component, specifically under the low capacity class, may be an indication of an increase in sedimentation due to increasing anthropogenic activities in these areas (Figure 1.29a). It is known that sedimentation is higher where developed lands are present due to higher erosion rates observed in these shoreline types (Swaney et al., 1996). Although, other factors such as changes in riverine sediment discharge due to climate change and the omission error can explain the increase in area as well.

Forested lands, vegetation composition and natural land use showed a larger area size under the low capacity class in 2009 (Figures 1.29b-d). This pattern indicates that by 2009 human activities, specifically development, were spread around Hampton even in areas characterized by the original vegetation cover. This may suggests that the demand for lands to expand development increased since 1963. This need for additional space for impervious surfaces then impacted most natural lands reducing the area size for forested lands while increasing secondary vegetation.

Even though tidal and inland marshes experienced a reduction in area by 2009 (Figures 1.30a-b), the fact that these components were mostly present under the high capacity could indicate that less development was occurring in these areas. This could be due to two different reasons. Development in marsh areas is a more complex process due to the geology characterizing these lands (e.g. wet conditions, unsuitable soils for building). Another possibility is that development in these sensitive areas is currently more actively controlled by the implementation of the Chesapeake Bay Preservation Act that regulates development in natural areas essential for the improvement of water quality conditions in the Chesapeake Bay (Baird and Wetmore, 2006).
Hampton's HSM: General Findings

Capacity to provide habitat services in the City of Hampton was highly impacted from 1963 to 2009. In 1963 only 39% of the sites showed low potential to provide a constant flow of services. By 2009, 73% of the sites showed low capacity to provide habitat services. The weakening of habitat services seemed widely spread and bigger in magnitude since historic times compared to Mathews County. This could imply that habitat services in Hampton were compromised for a longer period of time reducing their resilience to improve shoreline components conditions. Based on the results and visual assessment of the sites, some areas showed the absence of habitat services especially where development was the main and only component of the land surface.

During 1963, Hampton’s tidal shorelines showed high to moderate capacity to provide habitat services. However, the difference in number of sites between capacity classes was very small. A more altered land surface than observed in Mathews and a larger area size of developed lands characterized Hampton during historic times. In addition, Mathews County presented a larger natural land use area suggesting a higher capacity for habitat services during the 1960s.

As expected, most of the best component conditions in Hampton were identified under the high capacity class. This class showed mostly sites with the original land cover. Only SAV, beach and vegetation composition showed better habitat conditions under a different capacity class. Conditions for SAV and beach were mainly determined by the shorelines' physical properties and by anthropogenic influences. As identified in Mathews, vegetation composition showed better conditions under the moderate capacity class and was influenced by developed lands.

By 2009, a decrease in almost all vegetation components, except in secondary vegetation, was widespread around the city indicating an impacted capacity for habitat services in most tidal shorelines. Due to the high population density that characterizes Hampton, the loss of vegetation could also jeopardize the security of its citizens. The removal of the natural buffer in tidal shorelines will increase the impacts from waves, storm surges, and sea level rise increasing the risk of land inundation.
Water Quality Model: City of Hampton

Temporal and Spatial Changes in Water Quality Service Capacity

The capacity to provide water quality services in the City of Hampton was assessed at 120 randomly selected sites. The results indicated a drastic reduction in capacity for water quality services since 1963 (Figures 1.31a-b). A possible deterioration in the filtering function in shoreline units was registered in a 46 year period due to an exorbitant loss in vegetation components generated by an increase in development. As observed in the HSM, high capacity areas were mainly observed at the north side of the county coinciding with the location of most of Hampton’s marsh communities (Figures 1.32a-b). In addition, Hampton’s capacity for water quality services showed lower total model scores than identified for Mathews County.

Water Quality Service Capacity: Hampton 1963

In 1963 a total of 46 sites (38%) were classified as high capacity sites (Figure 1.33a). Thirty-eight sites were classified as moderate capacity and 36 sites presented the lowest capacity to provide water quality services. This indicates that 32% of the sites showed lower than adequate conditions for the provision of water quality services during historic times. As observed in the HSM for Hampton unlike for Mathews, the majority of the sites were classified under the moderate and low capacity classes suggesting an overall low condition in shoreline components and mainly compromised water quality services.

The spatial distribution of capacity for water quality services showed similar patterns as in the HSM (Figures 1.23a and 1.32a). High capacity was observed mainly around areas such as Grundland Creek, Harris River, Tabbs Pt., and Marsh Point (Figure 1.32a). Interestingly, high capacity for habitat services was more commonly observed along the southern side of the locality than for water quality services. As indicated in the previous section this may be due to the influence of marsh components in the WQM.
The best conditions for tidal and inland marshes were mainly observed at the northern region of the City explaining the large distribution of high capacity for water quality services along this area. Most of the low capacity areas were observed at the southern side of the City where most of the anthropogenic activities and development were concentrated during 1963.

The WQM and HSM outputs for Hampton indicated a larger number of sites with low capacity than observed in Mathews. Based on historic land use data for the localities, the amount of developed lands was 11% higher in Hampton even with a 36% more land area assessed in Mathews (Figure 1.33b). This level of disturbance due to anthropogenic activities could explain the low number of sites within the high capacity class and the clear difference in capacity conditions between Mathews and the City of Hampton.

Interestingly, the largest total number of sites under the low capacity class in Hampton was identified in the WQM. Due to the similar components assessed for the HSM and WQM, it appeared marshes were the main component driving the differences between the HSM and the WQM outputs. Tidal marshes in the WQM were the only natural component assessed within the intertidal zone. Figure 1.34 shows most tidal and inland marshes under the high capacity class inferring that shoreline systems with this type of natural feature were less influenced by anthropogenic activities. This same model behavior was observed in Mathews.

**WQM Variations in Trends for Shoreline Components: Hampton 1963**

Shoreline components in the WQM showed the same patterns identified in the HSM for SAV and riparian vegetation composition. These two components showed the largest area size under the moderate capacity class (Figures 1.35a-b). As described under the HSM, moderate capacity for sites with SAV presence were driven by the long fetch, high bank heights and anthropogenic activities in the area. The patterns observed for vegetation composition were defined by high presence of developed lands that ultimately generated a higher presence of secondary vegetation (Figure 1.35c).
Water Quality Service Capacity: Hampton 2009

Hampton’s WQM in 2009 showed the most drastic decrease in capacity observed for both localities. The number of sites with high capacity was reduced from 46 in 1963 to 30 sites by 2009 (Figure 1.33a). This represents a 35% decrease in sites with high capacity or 16 less sites with sustained water quality services provision in the City. Only 24 sites classified as high capacity during 1963 retained their classification by 2009. Four sites decreased in capacity to moderate and 18 sites were reclassified as low capacity sites.

Moderate capacity sites showed a decline in number as well. Only 21 sites were classified as moderate by 2009 from an original total of 38 sites in 1963. This 45% reduction was due to the reclassification of most sites (53%) as low capacity sites (Figure 1.33a).

In contrast to the high and moderate capacity classes, the low capacity class experienced an increase in number with 33 additional sites by 2009. A change from 36 sites in 1963 to 69 sites by 2009 indicates a rise of 48% in a 46 year period. Thirty-one sites retained the low capacity classification and 5 sites increased conditions to moderate capacity. This increase in sites with low capacity also indicates that 58% of the total sites for Hampton were characterized by providing low capacity for water quality services during 2009.

As observed in the WQM for 1963 and in the HSM, high capacity areas were located at the northern side of Hampton where most of the vegetated areas were present (Figure 1.32b). From the sites identified with high capacity, mostly Grundland Creek showed a decrease in capacity through time. This area is part of a natural reserve. However the area exposed to the Bay showed most of the decrease in water quality services probably due to shoreline erosion.

For the WQM in 2009 low capacity areas were also evenly distributed around the City coinciding with the widespread development identified in the HSM. Figures 1.31 and 1.32 show the increase in low capacity areas moving from the southern side to the northern area through time and probably impacting vegetation conditions in this area.
An assessment of shoreline components indicated that the largest areal extent for riparian vegetation composition, riparian and upland forested lands, and natural land use was under the low capacity class (Figures 1.36a-c). This pattern was also observed in the HSM. As indicated previously, this pattern could be mainly explained by a well distributed presence of anthropogenic activities around the city. A need of space for urban expansion could be the main reason for recent development in different types of shoreline systems and the alteration of lands that consisted of the original vegetation cover of the city. This can be confirmed by Figure 1.36c indicating a bigger total area for developed lands than for natural cover. This suggests that most sites in Hampton were mainly characterized by impervious surfaces and not vegetated lands. Different to what was observed under the low capacity class, the high capacity class presented no sites with high vegetation composition. All sites under the high capacity class were identified with presence of tidal marshes and most sites showed presence of inland marshes as well (Figure 1.36d).

Hampton’s WQM: General Findings

In summary, the WQM for the City of Hampton presented the largest decline in capacity determined in this study. The number of sites with high capacity class decreased 35% and the number of sites for the low capacity class increased by 48% from 1963 to 2009. By 2009, 58% of 120 total sites were characterized by low capacity to provide water quality services. Most sites experienced a reduction in area for natural components, specifically vegetation components. However, most sites experienced a considerable increase in anthropogenic activities, specifically of developed lands.

The WQM and HSM presented a similar output for 1963. Both models showed very low conditions in shoreline components. However, more sites with low capacity were identified for the WQM during historic times. The distribution of sites by capacity
class was similar between models indicating that anthropogenic activities were highly spread around Hampton’s land surface altering most of the natural components of the area and reducing water quality services.

Based on total area calculations, the best components conditions were identified under the high capacity class. However, SAV and vegetation composition showed the best conditions specifically under the moderate capacity class. The trend identified for these two components was mainly driven by the effect of anthropogenic activities, specifically defended shorelines and developed lands. Vegetation composition was influenced by developed lands in both models and in both localities.

Even though Mathews and Hampton presented a similar trend for both models, the magnitude of the changes in the land surface observed in Hampton was much larger. The total area of developed lands in Hampton by 2009 was almost four times bigger than the developed area observed in 1963 and almost three times bigger than what was defined for Mathews during 2007. Effects from developed lands could be the main cause of the negative trend observed in shoreline components in the HSM and WQM. The vast changes experienced in the land surface and the spatial arrangement of shoreline components could have degraded the overall health and ultimately the capacity of shoreline systems to provide services. The capacity for water quality and habitat services could be dangerously compromised and these conditions could be exacerbated by future threats such as sea level rise and future alterations to the landscape.

Capacity by Shoreline Type: City of Hampton

Averaged model scores for the different shoreline types in Hampton showed similar trends for the HSM and WQM. Both models identified all shoreline types experiencing a decrease in capacity (Figures 1.37a-b). Shorelines with low bank heights and short fetch were identified with the best capacity for both habitat and water quality services. Shoreline units with long fetch and bank heights between 0-9.1m showed the lowest capacity.
In Hampton the intertidal and upland zones presented a decline in capacity through time at each bank type (Figures 1.37c-d). The intertidal zone experienced a decrease through time in the conditions for the main natural components (i.e. beach and tidal marshes) while defended shorelines showed an increase (Appendix III). The upland zone experienced similar conditions with mostly a reduction in vegetation components and an increase in developed land use.

**Capacity for Habitat and Water Quality Services by Shoreline Type: Hampton**

The HSM and WQM outputs identified shorelines with low bank height and short fetch (HBO-1.5S) as the bank type with the highest averaged model score in 1963 and 2009 (Figures 1.37a-b). This indicates that most of the sites with the best shoreline conditions and capacities during both years were located at this shoreline type. The assessment of this bank type presented the best conditions for most of the vegetation components and less developed land surface. However, drastic declines in shoreline components conditions were observed through time mainly induced by an increase in anthropogenic activities. The lowest model scores for both models in 1963 were observed at shorelines with low bank height and long fetch (HBO-1.5L). This shoreline type was identified with the lowest conditions for vegetation cover during historic times because most of the extensive beaches were observed in these shorelines. In 2009 shorelines with high bank heights and long fetch (HB1.5-9.1L) showed the lowest capacity. A major increase in defended shorelines and developed lands was observed under this shoreline type reducing vegetation cover and capacity. Interestingly, B1.5-9.1L shoreline type was identified with the lowest averaged model scores in Mathews County during 2007 as well.

Mathews’ and Hampton’s variations in capacity were influenced by different factors. Differences in capacity for shoreline types in Mathews during historic and current times were influenced mainly by conditions in natural shoreline components (i.e. bank height, tidal and inland marshes, forested lands) (Appendix IV). In Hampton,
trends were mostly defined by conditions in natural components during 1963, but mainly influenced by differences in anthropogenic activities (i.e. defended shorelines, land use) during 2009 (Appendix III and IV).

**Hampton: Overall Changes in Shoreline Components and Capacity**

As observed in Mathews County, the modified HSM and WQM allowed identifying variations in ecosystem's capacity to provide services in Hampton from historic to current times. However, Hampton experienced a larger decrease in capacity from 1963 to 2009 compared to Mathews. In Table 1.14a a summary of changes observed in shoreline components is shown. Clearly most natural components experienced a loss in area through time (i.e. tidal marsh, riparian and upland vegetation composition, riparian and upland inland marshes, riparian and upland forested lands, riparian and upland vegetation cover, and riparian and upland natural land use). This decrease in natural components coincided with an increase in all the anthropogenic components that were assessed. This same pattern was observed for Mathews’ components.

The increase through time in the number of sites with low capacity could be related to the drastic increase in developed lands observed in Hampton by 2009. This type of land use doubled since historic times. Conversely, the total area for vegetation components experienced a large decrease since 1963. Based on these changes, it is expected that large alterations in the land surface were experienced in Hampton since 1963. These changes may have prevented the healthy maintenance of shoreline components and consequently reduced possible improvements in capacity.

However, some natural components showed an improvement during the time period that was assessed. SAV, mudflats, riparian and upland trees, riparian grass, and riparian and upland scrub-shrubs experienced an increase in area through time.

Based on previous studies SAV communities showed improvements in many areas of the Chesapeake Bay due to recent efforts to protect and propagate these
important resources along Virginia’s shorelines (Moore et al., 2001). The observed increase in mudflat area could be mainly explained by the occurrence of omission error due to low resolution of historic aerial images.

Secondary vegetation in the riparian zone was expected to increase through time due to the areal expansion observed in developed lands by 2009. However, the increase in areal size for trees in the riparian and upland zones was not anticipated. As indicated previously the vegetation type classified as trees is defined as a single tree or a group of trees with an area $\leq 64 \text{ m}^2$. On the contrary, forested lands are lands with an area of $> 64 \text{ m}^2$ with trees as the main cover. An increase in the number of polygons with trees may indicate that forested lands experienced fragmentation since 1963. Table 1.14a shows a decline in forested lands in the riparian and upland zones. However this loss could be due to the clearing of the land or due to fragmentation. This type of alteration in shoreline units is translated into a more inefficient process for contaminant removal and for sediment trapping by the disruption of the root system in the area. In addition, this fragmentation could also represent a threat for the capacity to provide habitat services. As observed in Table 1.14a the overall areal size for vegetation composition showed a decrease through time. This is mainly due to the loss of the original vegetation cover of the city and ultimately due to a decrease in the total vegetation cover.

In comparison to Mathews, Hampton presented close to half the area size for natural components and consequently a smaller count of sites with the most adequate conditions to provide high capacity during historic times (Tables 1.14a-b). In addition, Hampton presented more alterations in the land surface by anthropogenic activities in the riparian zone compared to the upland zone. The opposite was observed in Mathews County. Based on the nature of the models, the total area covered by the riparian zone is much smaller than the upland zone. However, during 1963 these two zones presented a similar area size for developed lands in Hampton (Riparian= 32,512 m$^2$; Upland= 36,735 m$^2$), but a huge difference in area for the natural land use (Riparian= 118,433 m$^2$; Upland= 306,763 m$^2$). This indicates that the riparian zone was much more disturbed by anthropogenic activities compared to the upland zone during 1963. Because most of the habitat services are generated and found within the riparian zone (Klapproth and Johnson,
2001), this large anthropogenic impact observed in Hampton’s riparian zone could have compromised capacity in a faster and more drastic manner.

In 2009, both riparian and upland zone land surfaces seemed greatly changed. The migration of anthropogenic activities from the riparian to the upland zone was observed under the WQM and HSM. The drastic increase in development in the upland zone could be due to the lack of space available in the riparian zone, loss of land due to shoreline erosion, storm inundation or sea level rise, and/or the implementation of management policies regulating development in the riparian zone. However, if no changes are applied to the current development pattern in this locality and if no proactive management is implemented in face of future sea level rise scenarios, most tidal shoreline ecosystem services could be lost during the next century.

**Habitat and Water Quality Services Models Performance**

The modeling process showed temporal and spatial variations in tidal shorelines capacity to provide habitat and water quality services in Mathews and Hampton. The application of these categorical models provided a method capable of identifying a trend in capacity and variations in shoreline components from historic times to current times. Based on the models outputs, capacity for habitat and water quality services deteriorated through time in both localities. Possible drivers of change were determined by identifying a connection between low capacity conditions, an increase in developed lands and a decrease in vegetation components. Based on these results, the HSM and WQM met this study’s goals and showed their capability of providing a practical method to continue assessing ecosystems services along the Chesapeake Bay’s shoreline or any other coastal locality.

**Variations in Model Scores**

Based on the models’ results some components experienced an increase in conditions or area through time, others showed a decrease, and a few remained
unchanged. One possible explanation for these variations is based on the total number of sites under each capacity class as mentioned previously. For example, higher number of sites under the high capacity class in 1968 showed a different areal extent compared to the high capacity class in 2007. This is mainly explained by the reduction in the number of sites under the high capacity class by 2007.

Trends indicating an increase through time for SAV component could be due to the historic database used for this study. Historic SAV data applied in this study was collected in 1971. This is the oldest database available and exportable to GIS. By 1971, SAV communities in the Chesapeake Bay experienced a dramatic decline (Orth & Moore, 1983). This explains why SAV communities were almost nonexistent during historic times. Since 1987 multiple efforts have been made to help restore and protect SAV communities in different areas including Mobjack Bay in Mathews County (Moore et al., 2001; Orth & Moore, 1983). These efforts of restoration could explain the larger area for SAV observed during current years.

Some components showed no change or stayed the same for both years that were assessed. The only component expected to not change among sites was bathymetry, classified as shallow in every site. This classification was based on actual bathymetric data from NOAA. Other parameters such as fetch and Phragmites remained the same for both years due to lack of historic data for these two components. It is probable that conditions for some of these components have changed at some assessed sites, especially Phragmites' conditions due to the resilience and rapid propagation that characterizes this vegetation. This study assumes it is less probable that bathymetric and fetch conditions have changed drastically since the 1960s.

Omission error and poor image resolution are other possible explanations for variations in shoreline components conditions. The omission error was present during the digitization of shoreline components based on historic aerial photographs (Goodchild, 1994). This error negatively altered the components' conditions in 1968 mainly by underestimating the total area these components occupied. This generated a possible false improvement in conditions by 2007. Due to the importance of determining the possible error included during the digitization process, a digitization error was calculated for this study. Table 1.8 indicates the error results.

77
Based on the digitization process performed for this study, higher probability of omission error could be found in components such as mudflats along low bank heights and tidal marshes adjacent to high bank heights. Mudflats were difficult to differentiate from water surface due to the low resolution of the images. Tidal marshes were also difficult to identify in areas with high bank heights specifically in sites with thick forested lands. This error was reduced for 2007 due to the higher image resolution and the existence of current tidal marsh inventories that allowed the identification of these components.

Model's Limitations

Capacity to provide habitat services was based on the premise that high vegetation composition can provide a complex vegetation structure that can support, regulate, and provide services that ultimately increases species richness (Millennium Ecosystem Assessment, 2005). Based on this principle, the current study considered shoreline units with presence of only one type of vegetation as a system with low vegetation composition. This assumes that sites with extensive tidal and inland marshes provided low capacity for habitat services. Most sites observed under the high capacity class were characterized by extensive marshes or forested lands explaining the lack of sites with high vegetation composition.

Tidal and inland marshes support a vast variety of aquatic and terrestrial organisms, but habitat services are limited to organisms that can adapt and survive to conditions present in marsh communities. This also indicates that an extensive marsh area will be classified with a lower score for vegetation composition than areas with secondary vegetation types that are less beneficial for habitat services (Peterson et al., 2012; Mckinney, 2002). However, this limitation in the assessment of vegetation composition was corrected by including two additional components that only assessed the benefits of tidal and inland marshes (i.e. tidal marsh, riparian inland marsh, and upland inland marsh). The additional components captured differences in distribution of marsh communities and their influence on habitat services.
CONCLUSIONS

Capacity for habitat and water quality services showed strong temporal and spatial variations in Mathews County and the City of Hampton. The Habitat Services Model (HSM) and Water Quality Model (WQM) output indicated a decline in capacity from the 1960s to the late 2000s for both localities. Sites with high capacity for habitat services in Mathews and Hampton were reduced in number from historic to current times by 28% and 35%, respectively. However, the number of sites with low capacity increased by 15% in Mathews and 48% in Hampton. A similar pattern was observed for the WQM with a decrease of 29% in high capacity sites for Mathews and 35% decrease for Hampton. The highest increase in the number of low capacity sites through time was identified under the WQM with an increase of 74% for Mathews and 48% for Hampton. Even though a higher percentage for low capacity sites was identified in Mathews, 50% of the total assessed sites were classified as moderate capacity sites during 2007. Opposite conditions were observed in Hampton where 58% of the total sites were classified as low capacity in 2009. The overall assessment generated by the HSM and WQM identified Hampton as the locality with the lowest capacity for ecosystem services since historic times. This could indicate that Hampton has experienced changes longer than Mathews, or Hampton’s changes are of a bigger magnitude than Mathews in tidal shorelines and the adjacent land surface.

The decline in capacity for habitat and water quality services seemed defined by the large loss of vegetation components in both localities since historic times. The decrease in vegetated lands coincided with an increase in developed lands through time. However, vegetation composition was the only vegetation component to show an increase in area through time. The improvement observed in this shoreline component used as the main proxy for habitat services was explained by the increase in secondary vegetation (i.e. scrub-shrubs and grass) triggered by the presence of development on the land surface. Due to the interaction between this vegetation component and development, this study recommends the reassessment of this component in future applications of the models.
Land use patterns also were reflected in the spatial distribution of capacity. Mathews County showed a spatial shift in high capacity sites from shorelines in the eastern region of the county to the western side from 1968 to 2007. The east coast underwent an increase in development and defended shorelines thereby reducing vegetated lands and impacting the natural dynamics of many shoreline components. Conversely, shorelines in the west coast experienced the abandonment of agricultural lands increasing the presence of mainly grass lands and scrub-shrubs that provided additional land space for habitat and water quality services.

Hampton showed higher capacity for ecosystem services along the north coast. Since historic times, most of the locality’s population and development occurred along the southern shorelines. Changes in capacity through time indicated an increase in low capacity sites migrating from the highly developed southern region to the northern region where most marsh lands are found. It is expected that development expansion will keep migrating north disrupting the natural dynamics of shoreline components in this region and will jeopardized the already compromised services in this area.

Development showed a larger spatial area in Hampton since historic times explaining the widespread low capacity conditions in this locality since 1963. In addition, most of the development was concentrated within the riparian zone where most of the ecosystem services are generated. In Mathews development took place mainly in the upland zone characteristic of a rural locality. The patterns observed in Hampton’s land use could have compromised capacity in a faster and more drastic way than in Mathews.

Ecosystems in Mathews and Hampton showed different conditions in capacity temporally and spatially. However, a similar trend in both models was observed by indicating a decrease in capacity through time in both localities. Although Hampton showed a more acute degradation in ecosystems services since historic times, the capacity to provide habitat and water quality services in these two localities seemed to be mainly impacted by anthropogenic activities, specifically development. Increasing impervious surfaces registered since the 1960s in both study localities was identified as the main cause for the loss of vegetation and other natural components, consequently decreasing capacity. Currently, many tidal shorelines along the Chesapeake Bay are facing low
capacities to provide ecosystems services. With an expanding coastal development and future sea level rise, coastal managers and decision makers will be facing new and more challenges. Future shoreline conditions could deteriorate habitat and water quality services even more by diminishing the ecosystems' resilience and ultimately by completely degrading the natural processes and functions that generate services. This study aims to help understand the changes observed in tidal shorelines to help public and private decision makers cope with uncertainty while trying to develop policy that can effectively manage the problems coastal ecosystems are currently facing, and to prepare for future changes.
TABLES AND FIGURES
<table>
<thead>
<tr>
<th>Ecosystem services</th>
<th>Ecosystem processes and functions</th>
<th>Controlling components</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coastal protection</td>
<td>Attenuates and/or dissipates waves, floods, winds</td>
<td>wave height and length, wind climate, beach slope, presence of vegetation, seagrass beds, water depth, land use, sea level rise</td>
</tr>
<tr>
<td>Erosion control</td>
<td>Provides sediments for shoreline stabilization and soil retention in vegetation structure</td>
<td>sea level rise, subsidence, tides, coastal geomorphology, wave climate, sediment supply, presence of vegetation, land use</td>
</tr>
<tr>
<td>Water purification</td>
<td>Provides nutrients and contaminants uptake</td>
<td>presence of vegetation, nutrient load, hydrodynamic conditions, light availability, land use, sea level rise</td>
</tr>
<tr>
<td>Habitat</td>
<td>Provides suitable living space for wild plants and animals, reproduction space</td>
<td>presence of vegetation, habitat quality, food resources, land use, sea level rise</td>
</tr>
<tr>
<td>Tourism, recreation,</td>
<td>Provides unique terrestrial and marine areas suitable for marine and terrestrial organisms diversity, and natural processes</td>
<td>biological productivity, natural and human disturbances, habitat quality, presence of vegetation, land use, sea level rise</td>
</tr>
<tr>
<td>education</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 1.1 List of some coastal ecosystem services, their processes, functions, and controlling components. Modified from de Groot et al. (2002) and Barbier et al. (2011).
Figure 1.1 a. Map of the Chesapeake Bay, USA indicating the location of b. Mathews (M) and c. Hampton (H).
<table>
<thead>
<tr>
<th>Shoreline Types</th>
<th>Fetch = Long</th>
<th>Fetch = Long</th>
<th>Fetch = Long</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Bank height = 0-1.5m</td>
<td>Bank height = 1.5-9.1m</td>
<td>Bank height = &gt;9.1m</td>
</tr>
<tr>
<td>Fetch = Short</td>
<td>Fetch = Short</td>
<td>Fetch = Short</td>
<td></td>
</tr>
<tr>
<td>Bank height = 0-1.5m</td>
<td>Bank height = 1.5-9.1m</td>
<td>Bank height = &gt;9.1m</td>
<td></td>
</tr>
</tbody>
</table>

Table 1.2 Six different shoreline classes based on fetch (short= ≤300m; long= >300) and bank height classifications. These classes were used for Mathews’ and Hampton’s shorelines.
### Mathews Shoreline Types

<table>
<thead>
<tr>
<th>Class #</th>
<th>Shoreline Type</th>
<th>Class Abbreviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Class 1</td>
<td>Bank height 0-1.5m &amp; Fetch &gt;300m (Long)</td>
<td>MB0-1.5L</td>
</tr>
<tr>
<td>Class 2</td>
<td>Bank height 0-1.5m &amp; Fetch ≤300m (Short)</td>
<td>MB0-1.5S</td>
</tr>
<tr>
<td>Class 3</td>
<td>Bank height 1.5-9.1m &amp; Fetch &gt;300m (Long)</td>
<td>MB1.5-9.1L</td>
</tr>
<tr>
<td>Class 4</td>
<td>Bank height 1.5-9.1m &amp; Fetch ≤300m (Short)</td>
<td>MB1.5-9.1S</td>
</tr>
<tr>
<td>Class 5</td>
<td>Bank height &gt;9.1m &amp; Fetch &gt;300m (Long)</td>
<td>MB&gt;9.1L</td>
</tr>
</tbody>
</table>

Table 1.3 Shoreline types generated for Mathews County. The column to the right indicates the abbreviation for each class that will be used in the rest of this document. The “M” in the abbreviation is to specify the location of the bank (Mathews), “B” is for Bank., the numbers represent the bank height in meters, and the last letter represents fetch conditions (Long, Short).

### Hampton Shoreline Types

<table>
<thead>
<tr>
<th>Class #</th>
<th>Shoreline Type</th>
<th>Class Abbreviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Class 1</td>
<td>Bank height 0-1.5m &amp; Fetch &gt;300m (Long)</td>
<td>HB0-1.5L</td>
</tr>
<tr>
<td>Class 2</td>
<td>Bank height 0-1.5m &amp; Fetch ≤300m (Short)</td>
<td>HB0-1.5S</td>
</tr>
<tr>
<td>Class 3</td>
<td>Bank height 1.5-9.1m &amp; Fetch &gt;300m (Long)</td>
<td>HB1.5-9.1L</td>
</tr>
<tr>
<td>Class 4</td>
<td>Bank height 1.5-9.1m &amp; Fetch ≤300m (Short)</td>
<td>HB1.5-9.1S</td>
</tr>
</tbody>
</table>

Table 1.4 Shoreline types generated for City of Hampton. The column to the right indicates the abbreviation for each class that will be used in the rest of this document. The “H” in the abbreviation is to specify the location of the bank (Hampton), “B” is for Bank., the numbers represent the bank height in meters, and the last letter represents fetch conditions (Long, Short).
<table>
<thead>
<tr>
<th>County</th>
<th>Shoreline Type</th>
<th>Sample points</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mathews</td>
<td>MB0-1.5L</td>
<td>25,881</td>
</tr>
<tr>
<td></td>
<td>MB0-1.5S</td>
<td>23,334</td>
</tr>
<tr>
<td></td>
<td>MB1.5-9.1L</td>
<td>1,553</td>
</tr>
<tr>
<td></td>
<td>MB1.5-9.1S</td>
<td>578</td>
</tr>
<tr>
<td></td>
<td>MB&gt;9.1L</td>
<td>32</td>
</tr>
<tr>
<td>Hampton</td>
<td>HB0-1.5L</td>
<td>10,700</td>
</tr>
<tr>
<td></td>
<td>HB0-1.5S</td>
<td>11,756</td>
</tr>
<tr>
<td></td>
<td>HB1.5-9.1L</td>
<td>326</td>
</tr>
<tr>
<td></td>
<td>HB1.5-9.1S</td>
<td>212</td>
</tr>
</tbody>
</table>

Table 1.5 Total sample points generated to classify shorelines and used to select the sites to be assessed in Mathews and Hampton.

<table>
<thead>
<tr>
<th>Shoreline Class</th>
<th>Water Quality Model % Error</th>
<th>Habitat Model % Error</th>
</tr>
</thead>
<tbody>
<tr>
<td>MB05L</td>
<td>6.7</td>
<td>3.00</td>
</tr>
<tr>
<td>MB05S</td>
<td>5.5</td>
<td>3.30</td>
</tr>
<tr>
<td>MB530L</td>
<td>4.0</td>
<td>2.75</td>
</tr>
<tr>
<td>MB530S</td>
<td>5.1</td>
<td>3.20</td>
</tr>
<tr>
<td>MB30L</td>
<td>2.6</td>
<td>4.50</td>
</tr>
</tbody>
</table>

Table 1.6 Percent error calculations for Mathews based on a total sample size of 30 samples. All errors are below a 10% error.

<table>
<thead>
<tr>
<th>Shoreline Class</th>
<th>Water Quality Model % Error</th>
<th>Habitat Model % Error</th>
</tr>
</thead>
<tbody>
<tr>
<td>HB05L</td>
<td>9.3</td>
<td>6.10</td>
</tr>
<tr>
<td>HB05S</td>
<td>8.4</td>
<td>5.60</td>
</tr>
<tr>
<td>HB530L</td>
<td>6.8</td>
<td>4.00</td>
</tr>
<tr>
<td>HB530S</td>
<td>8.7</td>
<td>5.50</td>
</tr>
</tbody>
</table>

Table 1.7 Percent error calculations for Mathews based on a total sample size of 30 samples. All errors are below a 10% error.
Figures 1.2a-b  a. Mathews imagery from 2007 showing a point site (red dot) and the spatial extent of the 60m assessment buffer (yellow circle). b. Mathews imagery from 1968 showing the same location from Figure 1a. The point site is in red and the spatial coverage of the 60m assessment buffer is circled in yellow. This site is located at a low bank (0-1.5m or 0-5ft.) with long fetch.
Figures 1.3a-b  

a. Mathews imagery from 2007 showing the different assessment zones within the 60m buffer.  
b. Mathews imagery from 1968 showing the different assessment zones within the 60m buffer.
Figure 1.4 Components present in the intertidal zone in 2007.

Figure 1.5 Components present in the riparian zone in 2007.
Figure 1.6 Components present in the upland zone in 2007.
Figure 1.7 Diagram exemplifying how the Clip Tool works. This ArcGIS tool requires a clip feature to define the area to be extracted from the input. The output will include the area that overlaps the clip feature.
<table>
<thead>
<tr>
<th></th>
<th>Standard Error</th>
<th>Standard Error</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Natural (m²)</td>
<td>Developed (m²)</td>
</tr>
<tr>
<td><strong>1968</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Trees</td>
<td>21.45</td>
<td>18.16</td>
</tr>
<tr>
<td>Shrubs</td>
<td>2.20</td>
<td>7.83</td>
</tr>
<tr>
<td>Grass</td>
<td></td>
<td>43.33</td>
</tr>
<tr>
<td>Marsh</td>
<td>25.10</td>
<td>7.34</td>
</tr>
<tr>
<td>Anthro</td>
<td></td>
<td>20.20</td>
</tr>
<tr>
<td>Agric.</td>
<td>4.78</td>
<td>9.30</td>
</tr>
<tr>
<td><strong>2007</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Trees</td>
<td>17.80</td>
<td>21.28</td>
</tr>
<tr>
<td>Shrubs</td>
<td></td>
<td>0.53</td>
</tr>
<tr>
<td>Grass</td>
<td></td>
<td>26.87</td>
</tr>
<tr>
<td>Marsh</td>
<td>13.98</td>
<td>75.03</td>
</tr>
<tr>
<td>Anthro</td>
<td></td>
<td>44.33</td>
</tr>
<tr>
<td>Agric.</td>
<td>13.13</td>
<td></td>
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</tbody>
</table>

Table 1.8 Digitization error included for the most common components observed in the intertidal, riparian and upland zones. The values represent the standard errors in square meters.
Figure 1.8 Diagram exemplifying the Ecosystem Services Models and the type of models ran in Model Builder from GIS.
<table>
<thead>
<tr>
<th>Component</th>
<th>Habitat Model</th>
<th>Water Quality Model</th>
<th>Categorical Values/Model Scores</th>
<th>Maximum Score</th>
<th>Minimum Score</th>
</tr>
</thead>
<tbody>
<tr>
<td>SAV</td>
<td>✔</td>
<td>✔</td>
<td>Present / 3.00 Absent / 0.00</td>
<td>3.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Beach</td>
<td>✔</td>
<td>✔</td>
<td>Present / 3.00 Absent / 0.00</td>
<td>3.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Mudflat</td>
<td>✔</td>
<td>✔</td>
<td>Present / 3.00 Absent / 0.00</td>
<td>3.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Total marsh</td>
<td>✔</td>
<td>✔</td>
<td>Present / 3.00 Absent / 0.00</td>
<td>3.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Pharmrutus</td>
<td>✔</td>
<td>✔</td>
<td>Present / 3.00 Absent / 0.00</td>
<td>3.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Deflated shoreline</td>
<td>✔</td>
<td>✔</td>
<td>Present / 1.00 Absent / 3.00</td>
<td>3.00</td>
<td>3.00</td>
</tr>
<tr>
<td>Bank height</td>
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<td>✔</td>
<td>0-5ft / 3.00 5-30ft / 2.00 &gt;30ft / 1.00</td>
<td>3.00</td>
<td>1.00</td>
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<tr>
<td>Bank stability</td>
<td>✔</td>
<td>✔</td>
<td>Stable / 3.00 Undercut / 2.00 Unstable / 1.00</td>
<td>3.00</td>
<td>1.00</td>
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<tr>
<td>Riparian forested lands</td>
<td>✔</td>
<td>✔</td>
<td>Present / 3.00 Absent / 0.00</td>
<td>3.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Riparian island march</td>
<td>✔</td>
<td>✔</td>
<td>Present / 3.00 Absent / 0.00</td>
<td>3.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Vegetation cover</td>
<td>✔</td>
<td>✔</td>
<td>Total (&gt;75%) / 3.00 Partial (25-75%) / 2.00 Bare (&lt;25%) / 1.00</td>
<td>3.00</td>
<td>1.00</td>
</tr>
<tr>
<td>Vegetation composition</td>
<td>✔</td>
<td>✔</td>
<td>High (1 or more types vegetation) / 3.00 Low (1 or 2 types vegetation) / 2.00 None / 0.00</td>
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<td>0.00</td>
</tr>
<tr>
<td>Riparian land use</td>
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<td>✔</td>
<td>Natural / 3.00 Agriculture / 2.00 Developed / 1.00</td>
<td>3.00</td>
<td>1.00</td>
</tr>
<tr>
<td>Upland forested lands</td>
<td>✔</td>
<td>✔</td>
<td>Present / 3.00 Absent / 0.00</td>
<td>3.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Upland island march</td>
<td>✔</td>
<td>✔</td>
<td>Present / 3.00 Absent / 0.00</td>
<td>3.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Upland land use</td>
<td>✔</td>
<td>✔</td>
<td>Natural / 3.00 Agriculture / 2.00 Developed / 1.00</td>
<td>3.00</td>
<td>1.00</td>
</tr>
<tr>
<td>Total Score</td>
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<td></td>
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<td>Habitat Model Possible Total Score</td>
<td></td>
<td></td>
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<td>44.00</td>
<td>5.00</td>
</tr>
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</table>

Table 1.9 Shoreline components per assessment zone. The categorical values and model scores are specified for each component assessed for the HSM and/or WQM. The total maximum and minimum model score that each component can receive are indicated in the last two columns. The two bottom rows at the right indicate the possible maximum and minimum total model scores that shoreline units can receive by model.
### Habitat Model Capacity Classification

<table>
<thead>
<tr>
<th>Capacity</th>
<th>Maximum Score</th>
<th>Minimum Score</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>44.00</td>
<td>22.08</td>
</tr>
<tr>
<td>Moderate</td>
<td>22.07</td>
<td>15.54</td>
</tr>
<tr>
<td>Low</td>
<td>15.53</td>
<td>5.00</td>
</tr>
</tbody>
</table>

Table 1.10 HSM capacity classifications. Capacity classes were generated applying Jenks Natural Breaks method in GIS. The maximum and minimum scores represent the range of values for each capacity class. The same classes were applied in Mathews and Hampton for historic and current times.

### Water Quality Model Capacity Classification

<table>
<thead>
<tr>
<th>Capacity</th>
<th>Maximum Score</th>
<th>Minimum Score</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>41.00</td>
<td>26.86</td>
</tr>
<tr>
<td>Moderate</td>
<td>26.85</td>
<td>20.98</td>
</tr>
<tr>
<td>Low</td>
<td>20.97</td>
<td>7.00</td>
</tr>
</tbody>
</table>

Table 1.11 Capacity classifications for the Water Quality Model generated by applying Jenks Natural Breaks method in GIS. The maximum and minimum scores represent the range of values for each capacity class. The same classes were applied in Mathews and Hampton for historic and current times.
Figures 1.9a-b Location of shoreline units assessed in Mathews County for the HSM. a. Location of sites per capacity class in 1968. b. Location of sites per capacity class in 2007. These figures indicate changes through time in capacity for habitat services. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity. Map source: 2007 VBMP Imagery.
Figures 1.10a-b Prediction surfaces indicating capacity scores for habitat services in a. 1968 and b. 2007 for Mathews County. (A). Piankatank River, (B). Mobjack Bay (C). Gwynn's Island. Red colors indicate low capacity shorelines, pale orange and yellow represents moderate capacity and green represents high capacity. Interpolation values are only showed for a 500m wide buffer. This buffer size was only used for presentation purposes and to make values adjacent to the shoreline discernible.
Figures 1.11a-b  a. Number of sites per capacity class in 1968 and 2007 for the HSM in Mathews County.  b. Total area in square meters for three different land use types (natural, agriculture, developed) in the riparian and upland zones. Areas for land use in 1968 are specified by capacity class.
### Habitat Services Model Averaged Scores Per Component (1968)

<table>
<thead>
<tr>
<th>Habitat Shoreline Components</th>
<th>Total Population of Sites</th>
<th>Averaged Model Scores</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>High</td>
<td>Moderate</td>
</tr>
<tr>
<td>SAV</td>
<td>0.05</td>
<td>0.00</td>
</tr>
<tr>
<td>Bathymetry</td>
<td>1.00</td>
<td>1.00</td>
</tr>
<tr>
<td>Fetch</td>
<td>1.00</td>
<td>1.00</td>
</tr>
<tr>
<td>Phragmites</td>
<td>0.03</td>
<td>0.00</td>
</tr>
<tr>
<td>Defended shorelines</td>
<td>2.82</td>
<td>2.76</td>
</tr>
<tr>
<td>Beach</td>
<td>1.29</td>
<td>0.60</td>
</tr>
<tr>
<td>Tidal marsh</td>
<td>1.62</td>
<td>1.44</td>
</tr>
<tr>
<td>Mudflats</td>
<td>0.02</td>
<td>0.01</td>
</tr>
<tr>
<td>Vegetation composition</td>
<td>2.00</td>
<td>2.22</td>
</tr>
<tr>
<td>Vegetation cover</td>
<td>2.98</td>
<td>2.86</td>
</tr>
<tr>
<td>Riparian forested lands</td>
<td>2.31</td>
<td>1.67</td>
</tr>
<tr>
<td>Riparian non-tidal marsh</td>
<td>0.59</td>
<td>0.00</td>
</tr>
<tr>
<td>Riparian land use</td>
<td>3.00</td>
<td>2.91</td>
</tr>
<tr>
<td>Upland forested lands</td>
<td>2.24</td>
<td>1.02</td>
</tr>
<tr>
<td>Upland non-tidal marsh</td>
<td>0.58</td>
<td>0.00</td>
</tr>
<tr>
<td>Upland land use</td>
<td>2.97</td>
<td>2.45</td>
</tr>
<tr>
<td><strong>Sum of Averaged Scores 1968</strong></td>
<td>24.19</td>
<td>19.61</td>
</tr>
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</table>

Table 1.12  HSM averaged model scores per component in 1968. Individual averaged model scores were calculated for all components observed under each capacity class. Averaged scores represent the general components' conditions observed under each capacity class (high, moderate, low). To determine the lowest and highest types of component conditions see Table 7.
Figures 1.12a-b. a. Area fractions for vegetation composition in the riparian zone during 1968. The graph indicates variations in vegetation types within each capacity class. Even though more vegetation types were identified in the high capacity class, based on the database generated by the HSM, all sites presented a low vegetation composition. This indicates that only one or two different types of vegetation were observed at the sites. In the moderate class, 13 sites were classified with high vegetation composition indicating the sites presented 3 or more types of vegetation. This class also showed the largest area size for most vegetation types. The low capacity only presented 1 site with high composition. b. Riparian land use for 1968 indicating higher anthropogenic activities under the moderate and low capacity classes. The moderate and low capacity classes also presented the largest area size for secondary vegetation (i.e. scrub-shrubs and grass).
Figure 1.13 Total area in square meters for three different land use types (natural, agriculture, developed) in the riparian and upland zones. Areas for land use in 2007 are specified by capacity class.
Figures 1.14a-c  a. Changes in beach area per capacity class. A larger area size was identified under the high capacity class in 1968 and under the moderate class in 2007.  
c. Riparian land use for 2007 indicating higher anthropogenic activities for sites under the moderate capacity class. The moderate and low capacity classes also presented the largest area size for secondary vegetation: scrub-shrubs and grass.
Figures 1.15a-b Location of shoreline units assessed in Mathews County for the WQM. a. Location of sites per capacity class in 1968. b. Location of sites per capacity class in 2007. These figures indicate changes in capacity through time. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity. Map source: 2007 VBMP Imagery.
Figures 1.16a-b Prediction surfaces indicating capacity scores for water quality services in a. 1968 and b. 2007 for Mathews County. (A) Mobjack Bay, (B) Gwynn’s Island. Red colors indicate low capacity shorelines, pale orange and yellow tones represent areas with moderate capacity and green represents high capacity. Interpolation values are only showed for a 500m wide buffer. This buffer size was only used for presentation purposes and to make values adjacent to the shoreline discernible.
Figures 1.17a-b  a. Number of sites per capacity class in 1968 and 2007 for the WQM in Mathews County.  b. Total area in square meters for three different land use types (natural, agriculture, developed) in the riparian and upland zones. Areas for land use in 1968 are specified by capacity class.
Figures 1.18a-d  

a. Total area for tidal and inland marshes in square meters in 1968. Inland marshes total area includes marshes in the riparian and upland zones.  

b. Area fractions for vegetation composition in the riparian zone during 1968. Sites with moderate capacity showed the largest area fraction for most of the vegetation types (n=3) in addition to the largest number of sites with high vegetation composition. However, more vegetation types were identified in the high capacity class (n=4), but all sites presented low vegetation composition.  

c. Riparian land use in 1968 indicating higher anthropogenic activities under the moderate and low capacity classes. The moderate and low capacity classes also presented the largest area size for secondary vegetation (i.e. scrub-shrubs and grass).  

d. Total area for forested lands in the riparian and upland zones in 1968.
Figures 1.19a-b  a. Total area in square meters for three different land use types (natural, agriculture, developed) in the riparian and upland zones for 1968 and 2007. Areas for land use are specified by capacity class.  b. Total area for vegetation cover in the riparian and upland zones in 1968 and 2007. The total vegetation cover was 710,516 m² in 1968 and 623,946 m² in 2007 indicating a total vegetation loss of 86,570 m².
Figures 1.20a–c Conditions for forested lands, riparian land use and riparian vegetation composition per capacity class in 2007. a. Riparian and upland forested lands showed a larger area under the moderate capacity class in 2007. b. Total riparian land use in 2007 per capacity class. Larger area for natural and developed lands was observed in the moderate capacity class. c. Riparian vegetation composition in 2007 indicated the presence of all vegetation types and the largest area fractions for secondary vegetation under the moderate capacity class.
Figures 1.21a-d Averaged model scores for the a. HSM and b. WQM in 1968 and 2007 in Mathews County. Shoreline type in the x-axis (bank height (m) = 0 - >9.1; fetch = Long (L), Short (S)). c. Changes through time in averaged model scores for the upland zone in the HSM and WQM. d. Riparian and upland land use averaged model scores for the HSM and WQM by shoreline type in 1968 and 2007.
### Mathews County

<table>
<thead>
<tr>
<th>Shoreline Component</th>
<th>1968</th>
<th>2007</th>
<th>Change in Area (m²) or Amount (#)</th>
<th>% Change</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Subaqueous Zone</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>SAV (m²)</td>
<td>6,878.42</td>
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<tr>
<td><strong>Intertidal Zone</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Defended shoreline (# of structures)</td>
<td>18</td>
<td>65</td>
<td>47</td>
<td>72.31</td>
</tr>
<tr>
<td>Beach (m²)</td>
<td>23,657.65</td>
<td>26,309.68</td>
<td>2,652.03</td>
<td>10.08</td>
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<tr>
<td>Tidal Marsh (m²)</td>
<td>154,738.57</td>
<td>116,248.17</td>
<td>-38,490.40</td>
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<tr>
<td>Mudflats (m²)</td>
<td>314.71</td>
<td>447.56</td>
<td>132.85</td>
<td>29.68</td>
</tr>
<tr>
<td><strong>Riparian Zone</strong></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Vegetation composition (m²)</td>
<td>182,016.89</td>
<td>170,463.53</td>
<td>-11,553.36</td>
<td>-6.78</td>
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<tr>
<td>Trees (m²)</td>
<td>12,495.57</td>
<td>1,334.04</td>
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<td>Scrub-shrubs (m²)</td>
<td>3,772.75</td>
<td>8,938.58</td>
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<td>Grass (m²)</td>
<td>20,909.55</td>
<td>25,325.64</td>
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<tr>
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<td>24,680.99</td>
<td>20,939.79</td>
<td>-3,741.20</td>
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<td>120,158.03</td>
<td>113,925.48</td>
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<td>Riparian vegetation cover (m²)</td>
<td>182,016.89</td>
<td>170,463.53</td>
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<td>-6.78</td>
</tr>
<tr>
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<td></td>
<td></td>
<td></td>
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<tr>
<td>Natural (m²)</td>
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<td>-11,553.36</td>
<td>-6.78</td>
</tr>
<tr>
<td>Agriculture (m²)</td>
<td>54,867.56</td>
<td>13,364.59</td>
<td>-41,502.97</td>
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</tr>
<tr>
<td>Developed (m²)</td>
<td>4,339.87</td>
<td>15,319.94</td>
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<tr>
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<td>Trees (m²)</td>
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<td>3,788.98</td>
<td>10,677.98</td>
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<td>Grass (m²)</td>
<td>88,817.10</td>
<td>122,914.30</td>
<td>34,097.20</td>
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<td>Inland marsh (m²)</td>
<td>72,540.63</td>
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<td>-78.26</td>
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<td>Upland forested lands (m²)</td>
<td>308,874.08</td>
<td>276,640.86</td>
<td>-32,233.22</td>
<td>-11.65</td>
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<tr>
<td>Upland vegetation cover (m²)</td>
<td>528,499.81</td>
<td>453,481.91</td>
<td>-75,017.90</td>
<td>-16.54</td>
</tr>
<tr>
<td>Upland land use</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Natural (m²)</td>
<td>528,499.81</td>
<td>453,481.91</td>
<td>-75,017.90</td>
<td>-16.54</td>
</tr>
<tr>
<td>Agriculture (m²)</td>
<td>54,867.56</td>
<td>13,364.59</td>
<td>-41,502.97</td>
<td>-75.64</td>
</tr>
<tr>
<td>Developed (m²)</td>
<td>19,488.65</td>
<td>45,922.61</td>
<td>26,433.95</td>
<td>57.56</td>
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</tbody>
</table>

**Table 1.13** Changes in area and/or amount in shoreline components in the HSM and WQM from 1968 to 2007. Changes are displayed by assessment zones and in percent change.
Figures 1.22a-b  Location of shoreline units assessed in the City of Hampton for the HSM.  

a. Location of sites per capacity class in 1963.  

b. Location of sites per capacity class in 2009.  

These figures indicate changes through time in capacity for habitat services.  
Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity.  
Map source: 2009 VBMP Imagery.
Figures 1.23a-b Prediction surfaces indicating capacity scores for habitat services in a. 1963 and b. 2009 for the City of Hampton.

(A) Grunland Creek, (B) Harris River, (C) Stony Point, (D) Tabbs Point, (E) Marsh Point, (F) Salt Ponds. Red colors represent low capacity shorelines, pale orange and yellow tones represent areas with moderate capacity and green represents high capacity. Interpolation values are only showed for a 500m wide buffer. This buffer size was only used for presentation purposes and to make values adjacent to the shoreline discernible.
Figures 1.24a-c  a. Number of sites per capacity class in 1963 and 2009 for the City of Hampton. b. Land use patterns in percent for Mathews County in 1968 and for c. the City of Hampton in 1963.
Figures 1.25a-b  Conditions for a. SAV area and b. beach area in 1963 per capacity class.
Figures 1.26a-b  a. Area fractions for vegetation composition in the riparian zone. The graph indicates variations in vegetation types within each capacity class. A larger area fraction was identified under the moderate capacity class.  

b. Riparian land use conditions in 1963 indicating higher anthropogenic activities under the moderate and low capacity classes. The moderate and low capacity classes also presented the largest area size for secondary vegetation (i.e. scrub-shrubs and grass).
Figure 1.27 Change in area for land use types from 1963 to 2009 in Hampton. Natural lands lost 139,628 m², agricultural lands were completely lost by 2009 and a total of 104,252 m² were converted to developed lands.
Figures 1.28a-c  a-b. Conditions for SAV and beach area per capacity class in 2009.  

- **a.** Area in meter square for SAV in the subaqueous zone. A larger area size was identified in the low capacity class in 2009.

- **b.** Beach conditions in 2009 showed a larger area size under the moderate capacity. However, most shoreline units with beach presence were identified under the low capacity class. An increase in sites with beach presence under the low capacity class coincided with a loss of 3,948 m² in beach area by 2009.
Figures 1.29a-d  Area size for a. mudflats, b. forested lands, c. riparian vegetation composition and d. riparian land use in 2009 was larger under the low capacity class. Conditions in vegetation composition were similar as observed in Mathews and seemed to be influenced by anthropogenic activities as well.
Figures 1.30a-b  

a. Decrease in area for tidal and inland marshes between 1963 and 2009. Inland marshes showed the largest change with a 51% area loss.

b. Changes in area per marsh type and per capacity class between 1963 and 2009. Most of the marsh components were identified in high capacity sites.
Figures 1.31a-b  Location of shoreline units assessed in the City of Hampton for the WQM.  
a. Location of sites per capacity class in 1963.  
b. Location of sites per capacity class in 2009.  
These figures indicate changes in capacity through time.  
Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity.  
Map source: 2009 VBMP Imagery.
Figures 1.32a-b Prediction surfaces indicating capacity scores for water quality services in a. 1963 and b. 2009 in the City of Hampton. (A). Grunland Creek, (B). Harris River, (C). Tabbs Point, (D). Marsh Point. Red colors indicate low capacity shorelines, pale orange and yellow tones represent areas with moderate capacity and green represents high capacity.
Figures 1.33a-b  
a. Number of sites per capacity class in 1963 and 2009 for the WQM in the City of Hampton. 
b. Percent area for land use types in Mathews County and the City of Hampton during historic times.
Figure 1.34 Area for marsh components in meter square per capacity class in 1963.
Figures 1.35a-c  

a. Area in meter square for SAV in the subaqueous zone. A larger area size was identified in the moderate capacity class. 

b. Vegetation composition showed a larger diversity and larger area size for secondary vegetation under the moderate capacity. 

c. Riparian land use conditions seemed to influence vegetation composition. Even though the low capacity class showed the largest area size for developed lands, many shoreline units (n=14) showed no vegetation reducing the total amount of vegetation, especially secondary vegetation.
Figures 1.36a-d  Conditions for riparian vegetation composition, forested lands, riparian land use and marsh components in 2009.  
a. Area fraction for vegetation composition indicated a larger area size for almost all vegetation types under the low capacity class.  
b. Larger total area for forested lands was observed under the low capacity class as well.  
c. These conditions coincided with a larger area size for natural and developed lands under the low capacity.  
d. However, most marsh components were observed in high capacity sites.
**City of Hampton**

<table>
<thead>
<tr>
<th>Shoreline Component</th>
<th>1963</th>
<th>2009</th>
<th>Change in Area (m²)</th>
<th>% Change</th>
</tr>
</thead>
<tbody>
<tr>
<td>SAV (m²)</td>
<td>4,040.91</td>
<td>81,090.15</td>
<td>76,050.24</td>
<td>94.32</td>
</tr>
<tr>
<td><strong>Subaqueous Zone</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Defended shoreline (no. structures)</td>
<td>35</td>
<td>64</td>
<td>31</td>
<td>46.97</td>
</tr>
<tr>
<td>Beach (m²)</td>
<td>36,747.92</td>
<td>32,800.12</td>
<td>-3,947.80</td>
<td>-12.04</td>
</tr>
<tr>
<td>Tidal Marsh (m²)</td>
<td>395,965.74</td>
<td>222,308.26</td>
<td>-173,657.48</td>
<td>-44.13</td>
</tr>
<tr>
<td>Mudflats (m²)</td>
<td>0.00</td>
<td>196.19</td>
<td>196.19</td>
<td>100.00</td>
</tr>
</tbody>
</table>

| Vegetation composition (m²) | 118,432.96 | 110,086.64 | -8,346.32 | -7.18 |
| Trees (m²)                | 1,011.03 | 7,281.85 | 6,270.82 | 85.98 |
| Scrub-shrubs (m²)         | 3,913.81 | 13,614.07 | 9,700.26 | 71.23 |
| Grass (m²)                | 27,953.91 | 32,625.67 | 4,671.76 | 13.73 |
| Inland marsh (m²)         | 68,004.54 | 39,762.41 | -28,242.13 | -41.28 |
| Riparian forested lands (m²) | 17,549.67 | 16,803.25 | -746.42 | -4.43 |
| Riparian vegetation cover (m²) | 118,432.98 | 110,086.69 | -8,346.29 | -7.18 |
| Riparian land use         | 118,432.98 | 110,086.69 | -8,346.29 | -7.18 |
| Natural (m²)              | 118,432.98 | 110,086.69 | -8,346.29 | -7.18 |
| Agriculture (m²)          | 2,321.42 | 726.09 | -1,595.33 | -68.80 |
| Developed (m²)            | 32,512.61 | 53,105.31 | 20,592.71 | 63.28 |

| Upland Zone | Vegetation composition (m²) | 375,538.54 | 244,256.98 | -131,281.56 | -53.72 |
| Trees (m²)  | 1,281.00 | 12,249.45 | 10,968.45 | 87.45 |
| Scrub-shrubs (m²) | 5,159.35 | 18,164.86 | 13,005.51 | 71.60 |
| Grass (m²) | 86,853.91 | 72,224.90 | -14,629.01 | -20.25 |
| Inland marsh (m²) | 230,171.40 | 106,231.79 | -123,939.61 | -53.85 |
| Upland forested lands (m²) | 52,072.88 | 35,364.35 | -16,708.53 | -48.25 |
| Upland vegetation cover (m²) | 375,538.54 | 244,256.98 | -131,281.56 | -53.72 |
| Upland land use | Natural (m²) | 375,538.54 | 244,256.98 | -131,281.56 | -53.72 |
| Agriculture (m²) | 13,924.95 | 0.00 | -13,924.95 | -100.00 |
| Developed (m²) | 36,735.63 | 120,394.80 | 83,659.17 | 69.49 |

**Mathews County**

<table>
<thead>
<tr>
<th>Shoreline Component</th>
<th>1968</th>
<th>2007</th>
<th>Change in Area (m²)</th>
<th>% Change</th>
</tr>
</thead>
<tbody>
<tr>
<td>SAV (m²)</td>
<td>6,878.42</td>
<td>45,894.70</td>
<td>39,016.28</td>
<td>85.01</td>
</tr>
<tr>
<td><strong>Subaqueous Zone</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Defended shoreline (no. structures)</td>
<td>106</td>
<td>47</td>
<td>59</td>
<td>55.74</td>
</tr>
<tr>
<td>Beach (m²)</td>
<td>23,657.65</td>
<td>26,309.68</td>
<td>2,652.03</td>
<td>-10.08</td>
</tr>
<tr>
<td>Tidal Marsh (m²)</td>
<td>134,738.57</td>
<td>116,248.17</td>
<td>-18,490.40</td>
<td>-14.00</td>
</tr>
<tr>
<td>Mudflats (m²)</td>
<td>314.71</td>
<td>447.56</td>
<td>132.85</td>
<td>41.29</td>
</tr>
</tbody>
</table>

| Vegetation composition (m²) | 182,016.89 | 170,463.53 | -11,553.36 | -6.35 |
| Trees (m²)                | 12,495.57 | 1,334.04 | -11,161.53 | -89.32 |
| Scrub-shrubs (m²)         | 3,772.75 | 8,938.58 | 5,165.83 | 137.79 |
| Grass (m²)                | 20,909.55 | 25,325.64 | 4,416.09 | 23.91 |
| Inland marsh (m²)         | 24,680.99 | 20,939.79 | -3,741.20 | -14.61 |
| Riparian forested lands (m²) | 170,463.53 | 113,925.48 | -56,538.05 | -32.62 |
| Riparian vegetation cover (m²) | 182,016.89 | 170,463.53 | -11,553.36 | -6.35 |
| Riparian land use         | 182,016.89 | 170,463.53 | -11,553.36 | -6.35 |
| Natural (m²)              | 182,016.89 | 170,463.53 | -11,553.36 | -6.35 |
| Agriculture (m²)          | 54,867.56 | 13,364.59 | -41,502.97 | -78.56 |
| Developed (m²)            | 19,488.65 | 45,922.61 | 26,433.96 | 135.56 |

**Upland Zone**

| Vegetation composition (m²) | 528,499.79 | 453,481.88 | -75,017.90 | -14.90 |
| Trees (m²) | 528,499.81 | 453,481.91 | -75,017.90 | -14.90 |
| Scrub-shrubs (m²) | 2,321.42 | 0.00 | -2,321.42 | -100.00 |
| Grass (m²) | 13,924.95 | 0.00 | -13,924.95 | -100.00 |
| Inland marsh (m²) | 36,735.63 | 120,394.80 | 83,659.17 | 69.49 |

**Tables 1.14a-b** Changes in area and/or amount in shoreline components in the HSM and WQM for a. Hampton from 1963 to 2009 and for b. Mathews from 1968 to 2007. Changes are displayed by assessment zones and in percent change.
Appendix I

Habitat Model: Total defended shorelines in 1968 and 2007 per capacity class. Higher number of hardened shorelines was observed in the moderate class by 2007.
Habitat: Defended Shorelines (1968 and 2007)

<table>
<thead>
<tr>
<th>Capacity Class</th>
<th>1968</th>
<th>2007</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>8</td>
<td>5</td>
</tr>
<tr>
<td>Moderate</td>
<td>7</td>
<td>39</td>
</tr>
<tr>
<td>Low</td>
<td>3</td>
<td>21</td>
</tr>
</tbody>
</table>
Appendix II

Habitat Model: Number of defended shorelines per capacity class in Hampton during 1963.
Habitat: Defended Shorelines (1963)

<table>
<thead>
<tr>
<th>Capacity Class</th>
<th># Defended Shorelines</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>2</td>
</tr>
<tr>
<td>Moderate</td>
<td>12</td>
</tr>
<tr>
<td>Low</td>
<td>21</td>
</tr>
</tbody>
</table>
Appendix III

HSM and WQM: Changes in averaged model scores for a. beach, b. tidal marshes, and c. defended shorelines by shoreline type in Hampton from 1963 to 2009.
Appendix IV

HSM and WQM: Changes in averaged model scores for a. inland marshes, b. forested lands and c. land use by shoreline type in Hampton from 1963 to 2009.
HSM & WQM: Inland Marshes Averaged Model Score (1963 and 2009)

HSM & WQM: Forested Lands Averaged Model Score (1963 and 2009)

HSM & WQM: Land Use Averaged Model Score (1963 and 2009)
LITERATURE CITED


Center for Coastal Resources Management. 2009a. Vulnerability of shallow tidal water habitats in Virginia to climate change. College of William & Mary, Virginia Institute of Marine Science, Gloucester Point, VA.


Möller, I. 2006. Quantifying saltmarsh vegetation and its effect on wave height dissipation: Results from a UK East coast saltmarsh. *Estuarine, Coastal and Shelf Science*, v. 69, pp. 337-351.


Chapter 2

Modeling Shoreline Change: Influence of Physical and Vegetation Components over Shoreline Change and Effects of Marshes on Land Inundation in Mathews County and City of Hampton, Virginia, Chesapeake Bay
ABSTRACT

Shoreline change is a dynamic and complex process that varies at multiple spatial and temporal scales. The horizontal displacement of the shoreline position is currently used as an indicator of the threats that shoreline ecosystems are facing. With increasing population growth and increasing effects from climate change, tidal shorelines in the Chesapeake Bay will experience higher pressures from land and sea jeopardizing the safety of coastal populations, their economy and the health of shoreline ecosystems. This makes shoreline change prediction essential and of crucial importance for future coastal management plans. The current study generated an empirical analysis based on three different approaches to determine the influence that physical and vegetation components have over shoreline change in Mathews and Hampton, Chesapeake Bay. Because of the effects that anthropogenic influences have on shoreline systems, the focus of this study was primarily on natural shorelines, specifically on shorelines with marsh presence. The first approach determined shoreline change predictors for managed and unmanaged shorelines. The second approach stratified shorelines as marshes, beaches and managed shorelines. For this approach, the strongest predictors per shoreline type were identified and the importance of marshes in minimizing shoreline retreat was determined. The third approach focused on identifying the shoreline components controlling shoreline retreat in marshes. Ultimately the goal was to generate a model capable of predicting shoreline change.

The empirical models generated for each approach indicated fetch and land slope conditions were the most important physical components controlling shoreline change. Vegetation components were not as strong predictors as hypothesized. However, the approaches applied in this study showed high variability in the predictors for shoreline change and the strength of their influence. The first approach found unmanaged shorelines in Mathews and Hampton mainly influenced by natural components including vegetation. The opposite was observed for managed shorelines where physical components controlled shoreline change. The second approach indicated marshes located at shorelines with a slope \( \leq 5^\circ \) are more efficient at mitigating shoreline change and land inundation than beaches and defended shorelines. In addition, shoreline changes
observed in Mathews' marshes were mainly related to fetch and slope conditions. However, changes in Hampton's marshes were not predicted by these physical conditions. This suggests that sea level rise could be influencing Hampton's shoreline dynamics to a higher degree than in Mathews. Based on this analysis, shoreline change predictors were identified for each model generated for each approach. However, the models were not strong enough to be verified with an independent database. This indicates the complexity of shoreline dynamics and the difficulty in forecasting shoreline change at small spatial and temporal scales.
INTRODUCTION

Shoreline change is a prime indicator of threats to estuarine ecosystems. Current rates of shoreline retreat could lead to the loss of important ecosystems such as tidal marshes and beaches, in addition to the loss of public and private infrastructure. With expected increases in sea level between 0.7 to 1.6 m by 2100 in the Chesapeake Bay, most low-lying areas in the coastal zone are expected to be lost due to land inundation (Gesch et al., 2009; Rahmstorf, 2007). Based on elevation conditions alone, the forecasted sea level will trigger the loss of most current shoreline habitats in the Bay.

Even though sea levels are rising globally, shoreline systems are not responding uniformly (Le Cozannet et al., 2014). Many shoreline components could ultimately affect the response to future sea levels. To generate an adequate and effective coastal management plan for future rates of sea level rise, a better understanding of temporal and spatial trends in shoreline change and identification and characterization of potential components influencing shoreline dynamics is necessary.

Changes in shoreline position are highly variable even with a stable sea level and can change daily and seasonally due to changes in tides or the passage of a storm. Changes in shoreline position are affected by numerous factors such as waves, wind action, sediment supply, morphological feedback, vegetation, and human activities (Cooper and Pilkey, 2004). The poor resolution of data available for most of these factors in many coastal regions limits research and increases uncertainty of the conclusions, reducing the capacity to help inform coastal managers.

Beaches, marshes and managed shorelines are mainly defined by different physical, geological and biological conditions that can consequently generate different responses to cope with erosion (Gesch et al., 2009; FitzGerald et al., 2008; Leatherman, 2001). Due to the ecologic and economic importance of these shoreline features in the Chesapeake Bay, an assessment of their vulnerability to shoreline retreat and flooding is essential to provide guidance for both scientists and coastal managers.

Marshes are well known as primary producers, water filters, and spawning and nursery habitats. These vegetated communities also provide a buffering mechanism that reduces shoreline erosion (Kirwan et al., 2010). However, recent studies have identified
losses of marsh communities (Glick et al., 2008; Reed, 2008; Ward et al., 1998). These studies identified a failure of marsh vertical accretion to keep pace with rising water levels. Based on Glick et al. (2008), the average accretion for Virginia’s marshes is approximately 4.02mm/yr. Vertical accretion in marshes varies depending on the local site characteristics, slope, and sediment availability (Cahoon et al., 2006). In addition, accretion rates can vary at different temporal and spatial scales (Cahoon et al., 2009; Ward et al., 1998; Kearney et al., 1994). It is expected that a tidal marsh with vertical accretion at a rate equal to or higher than sea level rise will be more resilient, reducing the rate of shoreline change.

Beaches and defended shorelines are considered very dynamic shoreline features (Dugan et al., 2011; Schlacher et al., 2007). Settings suitable for beach formation include a constant supply of unconsolidated sediments and moderate to long fetch conditions. Beaches generally provide a wide buffering area that protects the upland zone from erosive forces (Hardaway and Byrne, 1999; Rosen, 1980). Today, beaches are generally migrating inland or being maintained by placement of hard or soft engineering solutions. In many cases, the placement of hard structures along the shore helps reduce sediment inputs from upland areas, but increases erosion rates seaward and downdrift of the structure (Shellenbarger et al., 2007). Dugan et al. (2011) concluded that defended shorelines exhibit a wide range of efficacy to control shoreline change.

Many efforts have been made to help predict shoreline change in the Chesapeake Bay region and the rest of the East Coast. The scientific information available illustrates the complexity and variability of the linkages between sea level rise and shoreline response. Le Cozannet et al. (2014) summarizes two main research approaches used to determine the relationship between sea level rise and shoreline change. The first approach is a model-based approach and the second one is a data-based approach. Modeling is generally beneficial when not enough data is available for the area of interest, but the dynamics of the system are well known. Currently, no model is able to predict all the processes taking place at a yearly or decadal scale in shoreline systems (Hanson et al., 2003). The second approach is based on coastal observations and correlations among observed parameters. This approach uses advanced statistical methods to determine the strength of relationships between multiple factors and shoreline
change (Gutierrez et al., 2001). The use of this approach is very limited due to the lack of data availability for many coastal regions. In addition, the results obtained tend to be highly variable. Some studies applying this approach were able to identify a relation between sea level rise and shoreline change. Other studies found other factors such as storms and anthropogenic activities generating changes of higher importance in shoreline systems. Le Cozannet et al. (2014) justify this disparity noting that shoreline systems are complex due to the variety of local settings causing shorelines to respond differently to the same rate of sea level rise. For example, Webb and Kench (2010) observed shoreline systems with no signs of retreat even with rising water levels.

The hypothesis in this study was that shoreline change is mainly driven by variations in fetch, land elevation and shoreline vegetation. Ultimately, the attempt in this study was to generate an empirical model based on three different approaches to forecast shoreline change. To generate this model, the main objectives were to:

- Identify the most important predictors of shoreline change in each approach
- Determine and describe the type of influence physical and vegetation components have over shoreline change
- Determine the type of influence that marshes, beaches and defended shorelines have over shoreline change and land inundation

STUDY SITES

CHESAPEAKE BAY

The Chesapeake Bay is located in the Mid-Atlantic coast of the United States and is one of the largest estuaries in the world (Figure 2.1a). The strong interactions found in this estuary between land surface, fresh, and saltwater provide the conditions for a variety of ecotones, high biodiversity, and high productivity. However, a recent Chesapeake Bay Program Assessment (2012) indicates poor water quality, a reduction in natural habitats,
and compromised conditions of many coastal resources and organisms in the Bay. These circumstances jeopardize the health and quality of the ecosystem services generated in the estuary.

The unpredictability of climate change also increases the challenge in the restoration of the Bay’s former conditions. Based on a 35 year database from 10 tide gauges from Norfolk, VA and Baltimore, MD the relative rates of sea level rise in the Chesapeake Bay range from 2.91 to 5.80 mm per year. These rates are higher than the rates observed in many other areas in the U.S. East Coast (Boon et al., 2010). Rahmstorf (2007) predictions indicate that the Chesapeake Bay will be experiencing an increase of 0.7 m (700 mm) to 1.6 m (1,600 mm) in sea level by 2100. Based on different CO₂ scenarios, more variations are expected in the climatic conditions of the Bay during the 21st century (Pyke et al., 2008).

MATHEWS COUNTY AND CITY OF HAMPTON, VIRGINIA

This study focused on the shorelines along Mathews County and City of Hampton in the state of Virginia (Figures 2.1b-c). The socioeconomic characteristics differ between localities with more rural lands observed in Mathews and a highly developed landscape in Hampton. However, these localities share similar physical coastal conditions (i.e. mean tidal range, coastal slope, rate of relative sea level rise, shoreline erosion and accretion rates, mean wave height, geomorphology) (Boruff et al., 2005). More importantly, the coastal area of both localities lies below the 6 m elevation contour (Titus and Wang, 2008). This implies future greater risks of inundation for developed coastal areas and the loss of shoreline features.

MATHEWS COUNTY

Mathews County is located on the middle peninsula of the state of Virginia. The county is bordered by Mobjack Bay to the south, Chesapeake Bay to the east, North
River to the west, the Piankatank River to the north, and Gloucester County at the north-west (Figure 2.1b). According to the U.S. Census Bureau, the county has a total area of 652.68 km$^2$ of which 222.74 km$^2$ is land, 429.94 km$^2$ is water, with 559.04 kilometers of shoreline.

Mathew’s shoreline types vary along the County’s coast. Wave climate conditions range from fetch-limited creeks to open Bay high fetch. Most of the tidal shorelines in Mathews County are found in narrow, small creeks and rivers with low wave energy (Hardaway et al., 2010).

The intertidal zone is mainly characterized by the presence of marshes, wetlands, maritime forests, high and low energy shorelines, beaches, and dunes. These coastal components are currently providing habitat for different aquatic and terrestrial species, reducing wave energy and erosion, and stabilizing shoreline sediments. The North River is characterized by having very low uplands and marsh coasts. The eastern part of the coast has very high energy barrier beaches and marshes. High uplands are commonly observed along the Piankatank River. For 2010, about 80 kilometers of Mathews’ 559.04 kilometers of shoreline were already defended (Hardaway et al., 2010). From these 80 km, 27 kilometers were built in the last ten years and this amount is expected to increase greatly in the years to come.

Historically, shoreline change rates varied from 0 m/yr to over ±2.44 m/yr for both erosion and accretion along the Bay’s coast (Byrne and Anderson, 1978). A recent study from Hardaway et al. (2005a) calculated shoreline rates of change from 1937 to 2002 that varies from 0.88 m/yr to -3.17 m/yr. Strange et al. (2008) concluded that an increase of 2mm in water levels will transform marshes in the Mobjack Bay area to marginal marshes. With future increasing water levels it is expected that some marshes and unnourished beaches will be completely lost in the Piankatank River due to bank elevations greater than 3m. Beaches facing the Chesapeake Bay are currently showing signs of high erosion rates. Marshes and beaches with sufficient sediments to accrete and keep pace with a 7 to 16mm/yr increase of sea level are likely to continue migrating inland, but most marshes are likely to be lost with a predicted 7mm per year of sea level increase.
The City of Hampton is one of the seven major cities in Hampton Roads metropolitan area. It is located on the southeastern end of the Virginia Peninsula. The City shares physical boundaries with Newport News and York County to the northeast and it is contiguous to the Back River to the north, Chesapeake Bay waters to the west and the James River to the south (Figure 2.1c). Based on the U.S. Census Bureau, the City has a total area of 352.76 km², of which 134.16 km² is land, and 218.60 km² is water, and 234.964 kilometers of shoreline (CCRM, 2011; Hardaway et al., 2005b). A total of 12.07 km of tidal shoreline extend along the James River, 12.87 km along the Chesapeake Bay, and 8.05 km along the Back River.

Shorelines are characterized by a wave climate defined by a large fetch exposure mainly to the northeast and east across the Chesapeake Bay (Hardaway et al., 2005b). Most of the shorelines along Hampton River are bulkheaded. The bayfront shorelines and lowland areas prone to tidal flooding are occupied by extensive marshes and surrounded by heavy development in the upland zone.

Hampton’s shorelines have experienced strong impacts in the past due to coastal flooding during hurricanes and nor’easters (Boon et al., 2010). In addition, the combination of effects from sea level rise and land subsidence in this city will expose many shorelines and coastal communities to greater risks from sea level rise in the future. Observations already confirmed the inundation of marsh areas, converting these to tidal flats and then open water (Strange et al., 2008).

Historically, shoreline rate of change for Hampton shorelines varied between 0.21 to -1.25 m/yr for both shoreline retreat and accretion (Byrne and Anderson, 1978). Hardaway et al. (2005b), calculated similar rates between 1937-2002 of 0 to -1.25 m/yr. Based on the expected future increase in sea level, planners indicate that the developed portion of the City is almost certain to be protected by defended shorelines while other areas east of the city are already experiencing shoreline erosion (Strange et al., 2008).
METHODS

To determine the type of effects that physical and biological components have over shoreline change, three different approaches were applied in this study. The first approach was based on an analysis of sites selected in a stratified random sampling to represent the various types of shorelines found in the study communities. At these sites the conditions of an extensive set of shoreline components (i.e. physical, biological and land use) were examined to detect relationships with observed rates of shoreline change (Figure 2.2). The second approach refined the parameters examined for relationships with shoreline change. The third approach focused on identifying the main components responsible for most of the shoreline retreat specifically in marshes.

Each approach consisted on an analysis based on different spatial scales as well as different shoreline components. Multiple models were generated for each approach and for each locality using an Akaike Information Criterion (AIC) analysis. The components that were found to explain most of the shoreline changes were selected for a final model. Even though different shoreline types and settings were assessed in this study, shorelines with marsh presence were the main focus. Final models generated by each approach and for each locality were calibrated and verified using an independent database.

SHORELINE CHANGE

To determine the horizontal displacement of shoreline position the Digital Shoreline Analysis System (DSAS) v.4.3 was used to calculate the rate of change for Mathews and Hampton. This public domain software created by the U.S. Geological Survey is an extension to ESRI ArcGIS that enables calculation of shoreline change statistics from multiple historic shorelines (Thieler et al., 2000). DSAS was applied based on Himmelstoss (2009).

Two different shoreline positions were used for Mathews County (1968 and 2007) and Hampton (1963 and 2009). Historic shoreline positions were digitized in GIS by the Shoreline Studies Program (Hardaway et al., 2005a and b). Current shoreline positions
were digitized by the CCRM (2009 and 2011). Historic and current shorelines were
digitized to an approximate MHW. The shoreline position was digitized depending on
the type of shoreline features. For a marsh shoreline, the shoreline position was
delineated at the edge of the marsh. For a beach shoreline, an approximate MHW
position was defined by identifying the dark edge defining the boundary between wet and
dry sand material also known as the land-water interface. For a defended shoreline, the
shoreline was positioned at the seaward or outer side of the hard structure.

The digitization error included in the shoreline change calculations was
determined individually for the three different types of shoreline features that were
assessed in this study. The protocol applied in the test included the selection of one
shoreline site consisting of a 300m shoreline area for each type of shoreline feature that
was of interest in this study (i.e. marshes, beaches and defended shorelines). For the
randomly selected shorelines used in the test, five replicates were digitized for the
historic and current shoreline position. The line features generated in each replicate were
converted to a point feature with a 1 meter interval using Create Points Tool in ArcGIS.
The standard error for the x and y coordinates were determined for the replicates and
compared to the original shoreline used in this study. The error calculated for the
shoreline position during historic and current times is shown in Table 2.1 and based on
Morton et al. (2004) and Romine (2008) methods. Ultimately, a total uncertainty ranging
between ±5.5 m to ±6.1 m was calculated for Mathews’ and Hampton’s rates of shoreline
change.

Inland and offshore baselines were generated parallel to the shoreline’s position
for the entire county. The inland baseline was used to calculate shoreline change for low
energy areas and the offshore baseline to calculate change for high energy shorelines.
Previous applications of DSAS in Mathews and Hampton included rates of shoreline
change calculations only for high energy shorelines (Hardaway et al., 2005a and b). The
exclusion of low energy areas was due to higher error included in the calculations
generated by a more complex coastal morphology in these areas (Cowart et al., 2011).
Because the main objective of this current study is to assess marsh influence in shoreline
retreat and because most of these features are found along low energy shorelines, it was
necessary to generate a new protocol to calculate rates of change for these areas. To
accomplish this, the inland and offshore baselines were applied separately in DSAS to create individual outputs for low and high energy shorelines.

From the baselines, a series of transects with a 25m spacing were used to establish measurement points. Transects were cast perpendicular to the baseline to intersect both historic and current shoreline positions (Himmelstoss, 2009). Most transects were ~25m in length. However, transects were edited before calculating change to assure: 1) both shorelines were intersected at all times; 2) both shorelines were intersected once and 3) the transects intersected both shorelines at a correct angle. Every measuring point generated in DSAS was considered a shoreline unit with a specific rate of change and specific shoreline conditions.

Shoreline change calculations were performed by MATLAB executables within the DSAS installation. DSAS has the capability to show rates of change based on different statistical methods. For this study, the End Point Rate (EPR) was calculated at each measuring point. EPR represents the rate of change (m/yr) and is calculated by dividing the distance of the shoreline movement by the time in between the oldest (i.e. historic) and the youngest shorelines (Himmelstoss, 2009).

SHORELINE INVENTORY

Shoreline inventories generated by the Center for Coastal Resources Management (CCRM 2011, 2009) at the Virginia Institute of Marine Science are some of the largest databases generated in the Chesapeake Bay. A CCRM’s database including a field assessment of a series of shoreline components conditions collected for Mathews in 2007 and for Hampton during 2009, was applied in this study. The inventory corresponds to the same year as the digitized shorelines used in this study. Only the components utilized in this study are shown on Table 2.2 (i.e. Database I). This inventory was generated based on a set of protocols developed by the Comprehensive Coastal Inventory Program (CCI) (CCRM 2011, 2009). These protocols were created to describe shoreline conditions along Virginia’s tidal shorelines. The shoreline inventory assessed and characterized coastal components in the shorezone, which extends from a portion of the
riparian zone seaward to the shoreline. The assessment was based on observations made from a moving shallow draft vessel, navigating at slow speed and parallel to the shoreline. In the field, the data was logged using a handheld Trimble GeoExplorer III, GeoExplorer XT, or GeoExplorer XH GPS unit. These units collected georeferenced data, which was then processed in the lab to generate highly accurate records of shoreline features and conditions as a line feature in ArcGIS. In addition, bathymetric data from NOAA were converted from raster format to vector, specifically as a polygon feature. SAV data collected and published by VIMS as polygon features was downloaded from the SAV Mapping Program (VIMS, 1995). Ultimately, bathymetric data and SAV data were combined with the shoreline inventory using the Identity Tool in ArcGIS. For the bathymetric data, the 2m depth contour was extracted using a 10m buffer from the shoreline position. The 2m depth contour selection is based on a series of factors included under the Shoreline Components section in Chapter 1. From the union of all these components, shoreline units or reaches of shoreline were generated. Shoreline units are defined as shoreline segments where the shoreline components do not change.

Each component was assigned a categorical value that represented the component’s condition at each shoreline unit. These categorical values also indicated the hypothetical effect of the component’s condition on shoreline change. High categorical values represent the best components’ conditions to help reduce shoreline erosion and low categorical values represent the less adequate conditions. The adequate conditions that shoreline components must have present to reduce shoreline retreat in a shoreline system are based on peer-reviewed literature and best professional judgment (Table 2.3). The categorical values were used in the calibration and verification process as model values.

APPROACH 1

To determine the importance of fetch, bank height and vegetation, in addition to land use components on shoreline change, a total of 150 shoreline units were randomly selected for Mathews and 120 units for Hampton. Shoreline units were classified based
on six different shoreline types with specific fetch and bank height conditions (Table 
2.4). The physical conditions that characterize these shoreline classes are considered to 
define most of the dynamics observed in shoreline systems (CCRM, 2010; Hardaway and 
Byrne, 1999). Each shoreline class at each locality was comprised of thirty different 
shoreline units. Some clustering of shoreline units were observed in high bank conditions, 
specifically for the bank heights 1.5-9.1m Short Fetch class in Hampton and for the 1.5-
9.1m Short Fetch and >9.1m Long Fetch in Mathews. This is due to the low 
representation of units with these types of conditions in the two localities. The specific 
methods applied to select the assessed units and their classifications are explained under 
the Shoreline Classification and Sampling Size section and Sample Size per Shoreline 
Class section in Chapter 1.

The shoreline inventory generated by the CCRM was applied in this approach 
(Table 2.2, Database I). The Identity tool from ArcGIS was used to determine the 
specific shoreline components and their conditions at each shoreline unit. An additional 
inventory was generated for this specific approach and for the randomly selected 
shoreline units (Table 2.2, Database II). The 11 additional components allowed better 
assessment of vegetation and land use components in shoreline units. For this new 
inventory, an assessment buffer 60m in diameter was generated (See Determining the 
Assessment Buffer Size in Chapter 1 for more details). This buffer provided a physical 
boundary used to determine the components’ conditions for each shoreline unit (See 
Assessment Zones and Digitizing and Classification of Components for more details). 
All shoreline components included in Table 2.2 and Database II were digitized as 
polygon features using aerial images for Mathews (2007) and Hampton (2009) from the 
Virginia Base Mapping Program. The 11 additional components were digitized based on 
a resolution of 1:600 and using NAD_1983_UTM_Zone_18N as the projected coordinate 
system. The same system was used for the aerial photographs.

For Approach 1, shoreline rate of change was determined only for the randomly 
selected shoreline units in both localities. DSAS transects were generated with a 1m 
spacing, instead of 25m spacing indicated earlier, to increase the number of measurement 
points within the 60m assessment buffer. The 25m spacing previously specified under 
the Shoreline Change section was applied in the other two approaches. All the measuring
points or EPR values within a 60m buffer were averaged and only one EPR averaged value was considered at each shoreline unit.

**APPROACH 2**

This approach was designed based on a management perspective to help determine the type of influence (i.e. increases or reduces shoreline retreat) shoreline features, specifically marshes, beaches and managed shorelines have over shoreline change and land inundation in Mathews and Hampton. The approach was based at a local spatial scale where all shoreline features along Mathews' and Hampton' shorelines were assessed. Only the physical components identified as important in Approach 1 and the vegetation components with a database for the entire Mathews County and Hampton were used (Table 2.5). Land slope was included as an additional component in this approach to increase the accuracy of land elevation data necessary to better asses land inundation. In addition, to determine the type of response shoreline features have over land inundation due to sea level rise, the difference between the observed inundated lands due to shoreline change and the expected inundated lands due to sea level rise was calculated.

For this approach, only shoreline units with beach, marsh, and presence of hard and/or soft engineered structures were selected. Shoreline units were verified to assure that only one of these three features was present at each unit. Due to the artificial shoreline change dynamics found in managed beaches and in defended shorelines, most of the attention in this Approach was centered in shorelines units with marsh presence.

To provide a better sample size to determine variations in shoreline change between shoreline features, rates of change were calculated for the entire shoreline of Mathews and Hampton. A total of 18,444 shoreline units comprised Mathews’ sample size and 7,100 in Hampton. The shoreline change database generated for each shoreline feature and for each locality was inspected and outliers were discarded. Based on the final total number of shoreline units per feature type, shorelines with marsh presence were the most common feature (Table 2.6).
For each shoreline unit, the physical components identified as important in Approach 1 and the vegetation components with data available for the entire shoreline of both localities were assessed using the CCRM inventory (Table 2.5, Database I). The vegetation components included under the Database II were not applied in Approach 2 because data was only available for the selected sites assessed in Approach 1. The Identity Tool in ArcGIS was used to determine the component’s conditions at each shoreline unit.

Land slope at each shoreline unit is a critical aspect when assessing sea level rise and land inundation (Gesch, 2009). This component was calculated in degrees and was included as part of the inventory used in this Approach (Table 2.5, Database III). The methods applied to calculate and apply land slope are described below.

**Land Slope**

Land slope was calculated to determine variations in land elevation between shoreline features. The physical boundary used to determine land slope extended from the digitized current shoreline position to the most inland boundary of tidal marshes. Specifically, this physical boundary was defined based on Virginia's jurisdictional boundary for vegetated marshes.

Virginia’s Tidal Wetland Act defines vegetated marshes as the “...lands lying between and contiguous to mean low water and an elevation above mean low water equal to the factor one and one-half times the mean tide range”. In other words, vegetated marshes are commonly found between the mean low water up to:

\[ 1.5 \times \text{mean tide range} = \text{meters in elevation} \quad \text{Equation 1} \]

This definition describes the general offshore and inland physical boundaries of tidal marshes. However, databases applied in this study are referenced to the mean high water (MHW) placing the offshore marsh boundary at the digitized shoreline position. Even though it was not logistically possible to tidally reference the digitized shorelines
used in this study, it was assumed their position represented the MHW position or the tidal marshes most offshore physical boundary.

To calculate tidal marshes inland boundary, based on Virginia's Tidal Wetland Act, a mean tide range of 0.61m (2ft.) was used for Mathews and Hampton (Hardaway et al., 2005a-b). If the Tidal Wetland Act definition is applied,

\[1.5 \times 0.61\text{m} = 0.915\text{m} (3\text{ft.})\] 

then, this indicates that the landward boundary for tidal marshes extends 0.3048m (1ft.) above the mean tide range in Mathews and Hampton.

Lidar elevation data collected by the USGS and under College of William & Mary domain was used for Mathews (2010) and the City of Hampton (2011) (www.wm.edu/as/cga/VALIDAR/). The original raster was referenced to NAD_1983_HARN_StatePlane_Virginia_South_FIPS_4502_Feet. This data layer was referenced to MHW using a script provided by NOAA. Lidar data provided high quality elevation data, high vertical and high spatial resolution (Gesch, 2009). These layers had a ±0.14 m-0.23 m (±0.47-0.73 ft.) vertical accuracy at a 95% confidence level (i.e. NSSDA) (Dewberry, 2011).

The Lidar raster layer was converted to contour lines at 1ft. intervals. The 1ft. contour line was extracted from the raster layer and a separate line feature was created for it. The 1ft. contour line corresponds to the landward boundary for tidal marshes previously calculated. This line feature was then referenced from feet to meters as NAD_1983_UTM_Zone18N.

To calculate slope, the 0.3048m (1ft.) contour was used as an input layer in DSAS. Using the contour line as a shoreline position input (i.e. landward tidal marsh boundary) in addition to the youngest digitized shoreline position (i.e. offshore tidal marsh boundary), the distance between these two lines was calculated. The distance was obtained by using the Net Shoreline Movement (NSM) as a statistics output.

Once the distance between the offshore and landward boundaries for tidal marshes was determined, the slope was calculated in degrees. The following equation was applied using ArcMap's Field Calculator tool:
where the factor $ATan$ is the tangent of the line’s angle, $dy$ is the elevation set as the landward boundary (i.e. 0.3048m) and $dx$ is the distance between boundaries in meters (i.e. NSM). Because the $ATan$ function returned radians, an additional factor (i.e. $\frac{180}{\pi}$) was included to convert slope to degrees.

For this Approach, land slope was defined by the distance between the shoreline position (i.e. offshore tidal marsh boundary) and the inland boundary in tidal marshes. Based on this, a low slope indicated a long distance between physical boundaries. This can be translated as a wide low-lying shoreline area. A low slope condition could also indicate higher probabilities of a tidal marsh to vertically accrete and migrate landward. However, this type of slope presents higher risks of inundation if marsh accretion is not as fast as sea level rise (Cahoon et al., 2009). A high slope indicates a shorter distance between boundaries and a relatively steeper shoreline area. In this scenario, the ability of a tidal marsh to migrate landward is reduced and higher marsh erosion rates are expected.

**Observed vs. Expected Inundated Lands**

To determine the influence of shoreline features on shoreline retreat and ultimately on land inundation, residual values were calculated. Residuals represented the difference between the observed net shoreline movement due to shoreline change and the expected net shoreline movement due to sea level rise at each shoreline unit. To calculate residuals the equation below was applied:

$$\text{Observed inundated lands (m)} - \text{(- Expected inundated lands (m))} \quad \text{Equation 4}$$

where the *observed inundated lands* is the net distance between the oldest and the youngest shoreline position. For this variable, instead of using EPR (m/yr) from DSAS statistics, the NSM or Net Shoreline Movement statistic was used. This statistic indicated
the net distance and the direction of the shoreline change (i.e. negative = erosion; positive = accretion). *Expected inundated lands* corresponds to the net expected horizontal movement in the shoreline position based on the total sea level rise experienced since historic times (i.e. since 1960s) at each locality and based on the slope gradient:

\[
\frac{\text{Increase in sea level rise (m)}}{\text{Slope gradient (m)}} \quad \text{Equation 5}
\]

where *increase in sea level rise* consists of the observed sea level rise in Mathews (0.17m) between 1968 and 2007 and for Hampton (0.23m) from 1963 to 2009 (Figures 2.3a-b). The sea level rise trend was obtained by using a linear regression method based on monthly average water surface elevation at the Gloucester Point/Yorktown, VA stations (Mathews) and at Sewells Point, VA (Hampton) (Cheng Liu and Ming Liu, 2014). The *slope gradient* was obtained by converting degrees to gradients in meters.

**APPROACH 3**

A third Approach was generated to simplify the modeling process and to help identify the best predictors for shoreline retreat in units with marsh presence. Shorelines with marsh presence are considered in this study to be driven mostly, if not completely by natural dynamics. By focusing on just eroding shorelines, this Approach provides additional information about the processes taking place in the shoreline systems where the largest changes are expected with future increasing sea levels. Beaches and defended shorelines were not considered in this approach due to the high shoreline change variability, either by seasonal variations and/or by management practices, identified in Approaches 1 and 2.

For this Approach, only the negative or erosive spectrum of shoreline change was considered. The shoreline units with marsh features and undergoing erosion (Table 2.7) as well as the shoreline components assessed in Approach 2 were also applied in Approach 3 (Table 2.5).
MODEL CALIBRATION

The model calibration process consisted of determining the best predictors for shoreline change based on shoreline component conditions in Mathews and Hampton. A calibration was individually generated for each approach and for each locality.

The calibration was mainly executed in R software by applying a script supplied by Isdell (2014). This script consisted of a univariate analysis model, a correlation analysis and an AIC analysis to determine the importance of the components and to generate a Global Model.

The calibration analysis considered shoreline change as the dependent variable and the component conditions as the independent variables or predictors. For Approaches 1 and 2, rates of shoreline change were scaled by subtracting the maximum accretion rate from each rate of change. This generated a data set with a value of zero representing the highest accretion rate and gradually increasing towards bigger positive values representing erosion rates. This particular scaling process was done to be able to identify the relation between predictors and shoreline change (i.e. negatively or positively correlated to shoreline increase). For Approach 3, shoreline retreat was binned based on -0.20m/yr intervals starting at -0.01m/yr. This generated 8 bins for Mathews and a total of 6 for Hampton. Due to the nature of the shoreline component datasets, including categorical and continuous variables, the model values for the components were scaled as well using R’s scaling command.

Due to the large number of shoreline components considered in each approach, two different statistical analyses were applied as part of the calibration procedure to reduce the number of predictors. The first analysis consisted of a univariate analysis model. A Gaussian distribution was used for Approaches 1 and 2 after statistically determining the normality of the shoreline change distribution for Mathews and Hampton. For Approach 3, a Poisson distribution was applied due to the binning of the dependent variable. For each of the univariate models generated, an Akaike Information Criterion (AIC) value was calculated. This value helps identify the models that explained most of the shoreline change that was observed (Leu et al., 2011). The models with an AIC lower than the null hypotheses were kept for the rest of the calibration process, but
the components with a higher value than the null were discarded. The models with an AIC score lower than the null value were the models closest to the "true" model.

A correlation analysis using Spearman's rank ($\geq 0.7$) was also applied to avoid the problem of collinearity (Leu et al., 2011). In cases where predictors were correlated, a priori knowledge was used to select a variable from the pair (Leu et al., 2011). After reducing the number of components, a Global Model (GM) was generated by applying a generalized linear model. To generate this model, all possible combinations of components that can explain shoreline change were created. The GM was calculated as a model-averaged composite where the top models with a cumulative weight of 95% confidence were selected. A GM was generated, for each approach and for each locality. Only the predictors included in the GM, the components' model coefficients and the intercepts were used during the verification process.

The model coefficients indicated the type of correlation identified between the shoreline components and shoreline change. A positive correlation indicated that the model value of the components increased as the shoreline erosion increased. A negative correlation showed that the model value of the components decreased as shoreline erosion increased.

MODEL VERIFICATION

The different GMs generated for each approach and for each locality were verified using an independent database generated for Gloucester County. A total of 120 randomly selected shoreline units were used to verify all three different approaches. These shoreline units were classified based on fetch and bank height conditions. The selection and classification of shoreline units was based on the methods under Approach 1 section. The same shoreline components assessed for all three approaches were also assessed in the independent database and the same categorical and model values were assigned. For this, a shoreline inventory for Gloucester County generated by the CCRM during 2008 was used. The additional components incorporated in Approach 1 and the
land slope component used in Approaches 2 and 3 were also assessed for Gloucester County using the methods previously discussed.

All selected shoreline units were used to verify Approach 1. For Approach 2, only shoreline units with presence of a marsh (i.e. \( n = 91 \)) or beach (i.e. \( n = 5 \)) were used. To validate Approach 3, only shoreline units with presence of eroding marshes were applied (\( n = 54 \)).

Shoreline change was calculated for all 120 shoreline units selected in Gloucester County. Rates of shoreline change were scaled using the same scaling method specific of each approach.

Each individual GM generated for each Approach and for each locality under the calibration procedure was verified. For each model that was verified, a predicted model value (PMV) was generated. This predicted value represented the predicted rate of shoreline change. The following equation was applied to calculate the PMV:

\[
\text{GM}_x = (c_1) \cdot (C_{1MV}) + (c_2) \cdot (C_{2MV}) + \ldots + (c_n) \cdot (C_{nMV}) + I = \text{PMV} \quad \text{Equation 6}
\]

where, \( \text{GM}_x \) is any model generated during the calibration process, \( c_x \) represents the model values given to each shoreline component included in the GM, \( C_{xMV} \) is the model coefficient generated for a specific shoreline component as part of the GM output and \( I \) is the intercept value for a specific GM. Ultimately, the predicted shoreline change values and the observed values were compared to determine the strength of the models.

**RESULTS AND DISCUSSION**

The three approaches considered in this study showed variations in shoreline change predictors between localities and among shoreline features. However, based on the models generated, fetch and land slope were the two most important components that consistently influenced shoreline change. Vegetation components were not as strong predictors as originally hypothesized. In addition, marshes at low slopes seemed to attenuate shoreline retreat and land inundation more efficiently than beaches and
managed shorelines in both localities. Based on the verification process, the models generated for each approach were not strong enough to predict shoreline change.

**APPROACH 1**

This Approach was generated to assess multiple physical, natural and land use predictors and to determine the type of influence these components have over shoreline change. Using an AIC analysis, the component’s effects were determined for over a hundred randomly selected sites in Mathews and Hampton. Preliminary AIC runs showed high variability between the models generated for Mathews and Hampton. The original model generated for Hampton indicated shoreline change was mainly controlled by anthropogenic activities contrary to what was observed in Mathews. A more in-depth review of the selected shoreline units for both localities indicated that 29% of the units in Hampton were characterized by shoreline armoring or beach nourishment. Only 12% of Mathews’ units showed these conditions. To reduce the effects from anthropogenically influenced shorelines in the analysis the assessed shoreline units were split in two groups: managed (i.e. presence of shoreline armoring or beach nourishment) and unmanaged (i.e. natural). An individual model was generated for each group of shoreline units and for each locality. Based on the models generated for this approach, both physical and vegetation components were important predictors.

**Mathews County**

**Unmanaged Shoreline Units: Mathews**

The GM generated for unmanaged units in Mathews showed a higher presence of natural components (i.e. beaches, riparian forested lands and vegetation composition) (Figure 2.4). Beaches were identified as the strongest predictor for unmanaged shorelines in Mathews (Table 2.8). This component was positively correlated with shoreline change.
indicating that the presence of a beach in a shoreline unit seemed to increase erosion rates in this locality. Because of the rural land use conditions observed in Mathews, most beaches were not managed. This suggests that most of the changes these features experienced were due to their natural dynamic behavior and constantly changing nature (Schlacher et al., 2007). As Rosen (1980) indicated, beaches with unconsolidated material can be the most erosive shoreline type.

Riparian land use was the second most important component in Mathews’ GM. A positive correlation with shoreline change suggested that higher rates of erosion were observed in natural shoreline units and lower rates where development was present. This particular pattern was not expected due to the well-known effects development can have in shoreline erosion. However, high erosion rates in natural lands could be due to scarce sediment sources present in these systems. Developed lands usually have additional sources of sediments such as surface runoff and nearby nourished shorelines that help provide a constant pool of sediments that ultimately get recycled within the system. This sediment pool is often not present in natural shorelines.

Fetch was identified with the lowest model coefficient indicating that physical conditions were not as important in unmanaged shorelines in Mathews County. This component showed a positive correlation with shoreline change suggesting that unmanaged shoreline units with long fetch (i.e. >2 miles = 3,218.69 m) conditions experienced higher erosion rates.

Interestingly, the presence of most natural components (i.e. beaches and riparian forested lands) triggered higher erosion rates. This indicates that natural components do not necessarily provide a strong buffer against erosional processes. In some cases, these natural conditions can promote the instability of a shoreline system. In addition, the identification of riparian land use as the second most important component and the identification of lower erosion rates in developed lands may indicate that vegetation components are not as important predictors in shoreline change as the need for a constant source of sediment material that could allow the shoreline to adjust to changes.
Managed Shoreline Units: Mathews

The number of managed shoreline units in Mathews was too small to apply an AIC analysis. For this reason, no model was generated for this group of shoreline units.

City of Hampton

Unmanaged Shoreline Units: Hampton

Shoreline changes in Hampton's unmanaged shorelines were largely driven by natural components (Figure 2.4 and Table 2.9). This coincided with Mathews' model. For Hampton, beaches, tidal marshes in addition to vegetation composition were the only components identified as predictors of shoreline change. A negative correlation was observed between all of the predictors and shoreline erosion. This suggests that high erosion rates were observed in shoreline units where beaches and marshes were absent and where vegetation composition was minimal or none. In other words, shoreline units that were not managed in Hampton were highly dependent on the protection shoreline features and vegetation offered from erosional processes.

Vegetation composition was considered the most important component in unmanaged shoreline units in Hampton. This could also indicate that vegetation diversity is possibly helping to retain sediments and to reduce the strength from physical forces. Another indication of the importance of vegetation in unmanaged shorelines is the negative correlation of tidal marshes with shoreline change. As was expected in this study and as previous studies have indicated, tidal marshes can provide protection from shoreline erosion and can be an important component defining shoreline dynamics.

Opposite to what was observed in Mathews, beaches in Hampton provided a protective buffer from erosion. However, unmanaged beaches included in Hampton's shoreline units were mainly located in low energy areas not adequate for the maintenance of these features. Based on aerial images, these beaches were located at the northern side of Hampton and could be influenced by the transport of sand material from a nearby man
made spit. This anthropogenic influence could have had the same effect in marshes present in this area.

Based on unmanaged models for Mathews and Hampton, vegetation components were more beneficial in Hampton for reducing shoreline retreat. This is the opposite of what was expected. However, it is a possibility that Hampton contains a bigger pool of sediments that is constantly in motion and being recycled along the City’s shoreline due to all the nourishment and armoring projects. The presence of sediment sources in this locality may increase the importance of vegetation for helping retain sediments in place.

**Managed Shoreline Units: Hampton**

Contrary to the unmanaged models for both localities, Hampton’s managed model showed mostly physical components related to shoreline change (Figure 2.4 and Table 2.10). However, riparian land use and tidal marshes showed the largest AIC coefficients and a positive correlation with shoreline change. Based on the model, higher erosion rates were observed where natural land use conditions and marshes were present. Interestingly this pattern in land use coincided with the unmanaged model for Mathews. Because the maintenance of defended shorelines and beach nourishment projects takes place mostly where public and private property could be at risk, natural lands and tidal marshes are not a priority to preserve with management plans. This leaves natural lands and tidal marshes more susceptible to higher erosion rates. Another possible explanation for higher erosion rates in natural lands and tidal marshes is based on the amount of change that occurred in the shoreline before placing a structure. Most developed areas under high erosion conditions were defended since the 1960s. However, natural areas were probably defended after major changes in shoreline position took place. Consequently these types of shorelines experienced most of the erosion. In addition, the location of hard or soft structures in reference to the position of a tidal marsh could also generate different outcomes. If an armored structure is located behind a tidal marsh, erosion rates could be higher due to the inability of the marsh to migrate inland or to vertically accrete and to adjust to changes in water levels.
Bank height and fetch were the second most important components in the model. Bank height was negatively correlated with shoreline change and fetch showed the opposite. Based on the predictor’s coefficients, this model indicated that higher erosion rates were found at the highest bank heights in Hampton (i.e. 1.5 - 9.1m) and at shorelines with long fetch conditions. Based on management practices armored structures and nourishment plans are usually located along shorelines with long fetch and low bank height conditions because of the higher risks of inundation. Because most of the attention in coastal management is focused on low elevation areas, shorelines with the highest bank heights in Hampton may be experiencing higher erosion rates. In addition, as indicated earlier, developed shoreline units with long fetch and low bank heights have been managed since the early 1960s, probably experiencing a lower shoreline change compared to moderate bank heights.

**General Findings: Approach 1**

This approach showed the importance in treating managed and unmanaged shorelines individually to better assess shoreline changes. While fetch conditions were considered important for managed and unmanaged shoreline units, the influence from vegetation components was stronger in unmanaged shorelines. However, vegetation’s influence over shoreline change appeared to be variable. Beach conditions, specifically in Hampton, may represent an artificial effect generated by anthropogenic influences. Due to the differences observed between managed and unmanaged shorelines, most of the attention in the next approaches is concentrated on unmanaged shoreline units.

**APPROACH 2**

In this approach hundreds of shoreline units with marshes, beaches or managed shorelines were assessed to determine the effect these features have on reducing or increasing land inundation. Due to the high variability beaches and managed shorelines showed in the previous approach, Approach 2 mainly focused on shoreline change
dynamics occurring in marshes and their importance in attenuating the effects from increasing water levels.

**Mathews County**

*Shoreline Change: Mathews*

Figure 2.5 shows a simple boxplot indicating rates of change (EPR) by feature. Mathews' marshes showed a lower rate of erosion compared to beaches and managed shorelines (Table 2.11). Even though Mathews County experienced an increase in sea level of 0.17m (4mm/yr) since 1968, marshes showed a higher resiliency to changes than the rest of the features. This rate of increase in water levels coincided with an average 4.02mm/yr rate of vertical accretion for marshes in Virginia (Glick et al., 2008). This suggests that Mathews’ marshes were able to keep pace with the rate of sea level rise experienced during the last 39 years. However, the expected increase in water levels (i.e., 1 - >2ft.) could deteriorate and drown most marshes if the local shoreline settings do not provide the necessary conditions to increase the rates of vertical accretion.

Beaches presented the opposite conditions with the highest erosion rates observed in Mathews. In addition, this feature presented a wide range of EPR values in the positive and negative realm of shoreline change. The EPR values for beaches in Figure 2.5, confirm the dynamic nature of this feature and the complex processes interacting in this type of environment. Beaches have a capacity to buffer storm impacts, but with future climate changes storms could become stronger and more erosive. This will reduce the ability of these features to restore their natural conditions if no plan to manage them is generated.

Managed shorelines were the second most erosive shoreline feature. Interestingly, marshes and managed shorelines showed similar medians suggesting a similar distribution shape (Table 2.11). However, managed shorelines were more negatively skewed. Based on the results, these soft and hard man-made structures are reducing shoreline retreat more efficiently than beaches, but not as effectively as
marshes. Although these features were expected to present an accretionary pattern by introducing a physical barrier that controls shoreline movement, Mathews land use management could be the main reason of their inefficiency. Land use determines management decisions in coastal areas. The large amount of open lands in this County and the general behavior of developing relatively far from the shoreline could be an indication of a more reactive than proactive coastal management. In other words, shoreline structures in Mathews seemed to be placed after major shoreline changes have occurred. This could explain most of the high erosion rates observed in managed shorelines in Mathews. Another possible explanation to the erosional pattern in managed shorelines is the poor quality or inefficiency of the types of structures applied in Mathew’s shorelines.

An initial Kruskal Wallis Test showed that the rates of change of at least two shoreline features were significantly different from each other (i.e. p-value<0.05). To determine the specific significance between features a Mann Whitney U Test was applied. This test concluded that EPR values for marshes were significantly different from beaches and managed shorelines (i.e. p-value <0.05). However, EPR values were not significantly different between beaches and managed shorelines. These tests support this study’s hypothesis indicating that shorelines with vegetation, in this case marsh presence, can influence shoreline retreat by reducing the magnitude of erosion compared to other shoreline features.

**Land Slope: Mathews**

Based on Figure 2.6, beaches and marshes showed similar low averaged slopes and similar medians suggesting both features presented a comparable slope distribution. Averaged slope and slope variance in managed shorelines were the highest registered indicating that structures are placed along a wide range of slopes, but mainly steep slopes. In addition, marshes and defended shorelines presented the same wide range in slope values which explains why these two are the most common features in Mathews.
Low slopes of 3.29-3.70° represent an approximate distance of 3.5m from the most landward boundary of the marshes to the shoreline position. These conditions could provide some necessary surface area for marshes and beaches to adapt to sea level rise by migrating landward or accreting vertically. However, low slopes could also represent a higher risk of inundation if these features cannot accrete vertically as fast as sea level increases or if conditions landward do not allow the features to migrate.

A higher averaged slope condition was identified for defended shorelines (Figure 2.6 and Table 2.11). Managed shorelines in high slope environments provide a barrier that could protect the riparian and upland zones from future water levels. However, if the sea level keeps increasing, the risks of shoreline erosion on-site and in adjacent areas could increase as well due to the presence of a non-movable structure.

City of Hampton

Shoreline Change: Hampton

Averaged shoreline rates of change for Hampton showed large differences between shoreline features (Figure 2.5). Opposite to what was observed in Mathews, beaches and defended shorelines accreted during the last decades. Marshes were the only feature to show signs of erosion in this locality. In addition, Hampton’s marshes showed slightly higher erosion rates than identified in Mathews (Table 2.11). Medians for these three features differed from each other with the widest range of EPR values observed in shorelines with marsh presence (Table 2.11). Even though the shoreline change pattern observed in Hampton was not similar to Mathews’, the results were not completely unexpected.

Hampton’s marshes are currently being squeezed by increasing water levels seaward and heavily developed lands landward. In addition, the rate of sea level rise experienced since 1963 of 5mm/yr is already above the average vertical accretion rate identified for VA (i.e. 4.02mm/yr) (Glick et al., 2008). Although sea level rise was higher for Hampton since 1960s compared to Mathews, similar shoreline retreat was
identified for marshes in both localities (Figure 2.5; Table 2.11). This could indicate that since 1963 Hampton’s marshes were able to keep pace with sea level rise even at a 5mm/yr rate and did not reach a threshold where most marshes will start drowning.

Currently most beaches in Hampton are being managed while others are indirectly nourished by nearby projects. Most changes in Hampton’s beaches were artificially driven specifically by anthropogenic influences explaining the shoreline accretion experienced since historic times.

The clear difference between the averaged shoreline change in Hampton’s and Mathews’ managed shorelines could be an indication of differences in coastal management. Mathews County seems to apply a more reactive approach for coastal management decisions. In contrast, the current socioeconomic conditions in Hampton have triggered a proactive coastal management mainly to protect the highly populated coastal zone. This could explain the more effective and positive influence of shoreline structures in Hampton.

**Land Slope: Hampton**

Hampton’s shoreline features showed average slope conditions similar to Mathews’ (Figure 2.6). Beaches and marshes were characterized by low averaged slopes (Table 2.11). Managed shorelines were predominant at higher slopes.

**Influence of Shoreline Features on Inundation**

Residual values were calculated for every shoreline unit in Mathews and the City of Hampton. These values represented the difference between the observed (i.e. net shoreline movement) and the expected horizontal displacement of the shoreline due to historic sea level rise and slope conditions (i.e. Equation 3). The residuals defined the influences beaches, marshes and defended shorelines have over shoreline inundation. Figures 2.7-2.9 show the distribution of residuals and observed values by slope and by shoreline feature. In these figures, a positive residual indicates a positive influence of a
shoreline feature by reducing the expected inundation. In this case, the observed shoreline retreat was less than the expected horizontal displacement due to increasing water levels. The opposite scenario is a negative residual where the conditions provided by the shoreline feature were not enough to reduce erosion and a larger than expected horizontal displacement of the shoreline was observed at the shoreline unit.

**Marshes**

Figures 2.7a-b indicate the overall influence of Mathews’ and Hampton’s marshes on shoreline retreat. Based on residual values, marshes showed the highest positive influence (i.e. positive values) at low slopes (i.e. 1-10° slopes) compared to beaches and defended shorelines. This suggests that marshes attenuated shoreline retreat more effectively than the other two features, especially at low land elevations. As a result, less inundation than expected was observed in these areas. These results indicate that marshes can provide protection from shoreline retreat and sea level rise by possibly vertically accreting faster than sea level rise. However, many marshes at low slopes showed a negative influence (i.e. negative values) by experiencing higher inundation levels than expected.

Interestingly, shoreline units above a 10° slope showed similar observed and residual values (Figure 2.7c). This indicates that values for the expected inundation coincided with the observed net movement. Based on the observed values, these high slopes were experiencing shoreline retreat. However, the expected horizontal change due to inundation was not as large as that registered for the low slopes.

The rate at which marshes will inundate depends on future rates of sea level rise and sediment availability. If sediment sources are limited, the ability of marshes to vertically accrete will diminish increasing the risk of inundation. Based on the rates of sea level rise since the 1960s for Mathews and Hampton, the methods applied in Approach 2 were able to determine that marshes are currently keeping up with changes in water levels. Interestingly, marshes at low land elevations are showing most of the resilience. This particular result shows the opposite of what was expected. Table 2.12
shows the difference between the observed and the expected values for both localities and for different slope intervals (i.e. 1-10°). The observed values were not that different between localities even though Hampton experienced a much faster increase in sea level. Interestingly, from slope intervals 5-10°, the observed values were higher than the expected for both localities. This information suggests that: 1) Hampton’s marshes inundated faster than Mathews’ marshes, however they must have experienced some vertical accretion to help reduce inundation to levels similar to Mathews specifically at slopes 1-4°; 2) on average, marshes in slopes <5° are experiencing inundation at a slower rate than marshes at slopes >5° in both localities. However, the large number of marshes with negative residuals at low slopes is also a reminder that not all marshes at low slopes have the shoreline settings necessary to keep pace with sea level rise.

Beaches

Shoreline units with beach presence showed the second highest positive residual values at low slopes (Figures 2.8a-c). As observed in shoreline units with marshes, a beach presence can also ameliorate effects from sea level rise at low slopes, but the variability in shoreline change is extremely high. However, different residual patterns were identified between localities. Beaches located in Mathews showed a larger number of shoreline units with negative residuals indicating that beaches were mostly retreating (Figure 2.7a). Mathew’s beaches were mainly driven by natural factors. This also explains the pattern of the residuals observed in Hampton where almost all residuals were positive (Figure 2.7b). This translates into mostly accreting beaches or beaches experiencing less shoreline retreat than expected. As indicated previously, the number of beach nourishment projects in Hampton and the effects on-site and nearby reduced the expected effects from sea level rise.

Beaches also presented similar observed and residual values above the 10° slope (Figure 2.8c). Most of the healthiest and managed beaches in Hampton and Mathews are exposed to the Bay’s waters, with high fetch and low slopes conditions. Beaches at
shoreline units above the 10° could be lacking a continuous source of unconsolidated material to maintain themselves.

**Managed Shorelines**

Defended shorelines influenced shoreline retreat in a similar way in both localities (Figures 2.9a-c). These shoreline features showed the lowest positive influence at low slopes compared to marshes and beaches suggesting that natural shoreline features provided a more efficient barrier against shoreline retreat. From a management perspective, defended shorelines are placed to help reduce shoreline retreat. Based on this, managed shorelines were expected to present mainly positive residuals indicating a lower shoreline retreat than expected. However, Figures 2.9a-b show mostly negative residuals for Mathews and a similar distribution between positive and negative residuals in Hampton. This may be due to two different reasons. The first possible explanation could be that most of the shoreline retreat occurred before placing the structure in the shoreline unit. After placing the structure, shoreline retreat was possibly reduced or controlled. The second possible reason for the negative residual values could be the location of the structure within the shoreline unit. A shoreline structure placed behind a tidal marsh can cause exacerbated rates of erosion. The inability of a marsh to migrate landward due to the presence of a physical barrier could rapidly erode and drown the marsh.

Similar observed and residual values for managed shorelines were identified mostly at slopes higher than 5° (Figure 2.9c). In most of these high slopes, structures are in fact controlling inundation. However, many of these structures are expected to keep experiencing erosion seaward and become inundated with future sea levels.
Influence of Physical and Vegetation Components in Marshes
Shoreline Change

Mathews

An AIC analysis including components in Table 2.5 confirmed that physical conditions dominated rates of change in Mathews’ marshes (Figure 2.10 and Table 2.13). Contrary to what was hypothesized, vegetation components under Approach 2 were not found as important predictors of shoreline change. Fetch showed a positive correlation with shoreline erosion and was identified with the highest coefficients in all GMs generated. The results suggest that shoreline erosion increases as fetch increases. As previous studies confirmed shorelines exposed to long fetch conditions experience higher erosion rates and are more variable due to the physical forces that characterized them (Hardaway and Byrne, 1999). This current study also confirms that vegetated shorelines, specifically marshes, are also susceptible to larger erosion rates under strong physical conditions.

Land slope was identified as the second most important component; however this variable presented a much lower model coefficient than fetch. Land slope showed a positive correlation suggesting erosion rates increase as slope increases. Based on how slope conditions were determined for this study, the results indicate that erosion was higher where land elevations were steeper. Additional vegetation and physical components in Mathews’ marshes model (i.e. tree fringe, canopy overhang, bank height) were identified as important by the AIC, but the coefficients were too low to be considered in subsequent analyses.

Hampton

Shoreline change in Hampton’s marshes were completely dominated and influenced by physical components (Figure 2.10 and Table 2.14). Fetch conditions showed the highest model coefficients followed by maximum wind direction and slope.
All three components were positively correlated with shoreline erosion indicating that high rates of change were observed in marshes with long fetch, maximum winds from the north and steep slope conditions. The same pattern for fetch and slope conditions was observed in Mathews’ model.

**Shoreline Change Variations in Marshes Based on Land Use Types**

Higher erosion rates were observed in natural lands in both localities. In Mathews, natural lands showed the highest erosion rates followed by units with developed lands and agricultural lands (Table 2.15). A one-way analysis of variance (ANOVA) confirmed that all rates of change were significantly different from each other (i.e. p-value<0.05).

Hampton showed a similar pattern to Mathews, but with higher erosion rates for each land use type. The highest erosion rates were observed in shoreline units characterized by natural lands. However, agricultural lands showed a slightly higher erosion rate than developed lands. Contrary to Mathews, rates of change for each land use type in Hampton were very similar and were not significantly different from each other (i.e. p-value> 0.05).

A two-way crossed ANOVA was used to determine differences in rates of shoreline change in marshes among land use types and between localities as well as interactions between these factors (i.e. land use type * Locality). The analysis suggested that marshes shoreline change was significantly different between land use types and between land use type per locality (i.e. p-value<0.05). However, no significance was observed between localities. This indicates that a difference exists in shoreline change dynamics between land use types, but marshes are experiencing similar shoreline change dynamics in these two localities. This also coincides with the averaged rates of change from Figure 2.5.

Higher averaged erosion rates in natural lands could be due to differences in sediment supply between land use types. Surface runoff, a process mainly observed in developed lands and agricultural lands is a source of sediment supply in these land use
types that ultimately delivers material to the shoreline and can consequently help reduce shoreline erosion.

To understand the differences in shoreline change by land use type a series of statistical analysis were generated to determine if these variations could be explained by physical components. Fetch and land slope, the two most important physical components identified in the AIC analysis were used. This analysis assessed the difference in physical condition between developed and natural lands. Based on this analysis, marshes in developed and natural lands showed significantly different conditions (i.e. p-value <0.05) for fetch and slope in both localities.

Marshes in natural lands showed longer fetch conditions than developed lands possibly explaining the higher rates of change previously indicated (Table 2.15). Mathews’ marshes were characterized by longer fetch conditions than Hampton. The difference in fetch conditions between localities could be due to the distribution of marshes along the localities’ shoreline. A large number of marshes in Mathews are located at the east side of the County and exposed to the Bay’s conditions. In Hampton, most marshes are along the Back River’s shorelines at the north of the City and protected from long fetch conditions.

Land slope conditions in both localities showed lower elevations for marshes in natural lands than in developed lands (Table 2.15). This relationship between slope and land use was expected. As explained in the Methods section, slope was calculated by determining the distance between the inland boundary of a marsh and the shoreline position. Based on this, natural lands are often characterized by extensive marshes that will present a low slope condition due to the long distance between the shoreline and the inland boundary. In developed lands fringe marshes are usually observed and anthropogenic activities take place closer to or at the riparian bank. Consequently this reduces the distance between physical boundaries and the slope is considered steeper.

Table 2.15 indicates fetch and land slope conditions in Mathews followed the same pattern observed for shoreline change by land use type: Agriculture < Developed < Natural. The lowest conditions for these two components were observed in agricultural lands and the highest conditions were observed under the natural lands. This may suggest
that fetch and slope conditions were the most important components generating most of the changes in marshes observed in Mathews County.

Hampton’s results were not as clear. This locality did show the same pattern observed in Mathews for fetch and slope (i.e. Agriculture < Developed< Natural) (Table 2.15). However, all land use types presented almost the exact same rate of change (i.e. -0.08 m/yr). Even though Mathews showed longer fetch conditions, Hampton’s shoreline rates of change were higher than observed in Mathews. The fact that Hampton’s results do not follow the clear pattern identified for Mathews, may indicate that sea level rise could be influencing shoreline change conditions by similarly eroding marshes along Hampton’s shoreline.

**General Findings Approach 2**

Marshes in Mathews and Hampton experienced shoreline retreat since the 1960s. Mathews’ marshes showed a slightly lower averaged erosion rate than Hampton and the rates were significantly different from each other. Even though marshes in these two localities experienced rates of sea level rise equal or higher than the averaged vertical accretion rate for VA of 4.02mm/yr, most marshes were able to adapt and kept pace with increasing water levels. However, with increasing sea levels the threshold that will trigger the drowning of most marshes could be reached.

Residual values showed the importance of marshes in shoreline systems, especially along shorelines with a land slope ≤5°, by attenuating land inundation more effectively than beaches and managed shorelines. The resilience observed in marshes at low land slopes indicated that marsh inundation will not be a uniform process and that vertical accretion will define the patterns in marsh submergence.

Marshes at shoreline units characterized by natural land use showed higher erosion rates in both localities. Natural land use showed longer fetch and lower slope conditions than developed lands. Mathews’ rates of change by land use type were significantly different and defined by fetch and land slope conditions. However, erosion rates by land use type in Hampton were not significantly different and were not
completely explained by fetch and land slope patterns. This suggests that sea level rise could have a bigger influence in Hampton’s shoreline dynamics.

**APPROACH 3**

This approach consisted of simplifying shoreline change prediction by determining the components controlling only shoreline erosion in marshes. For this analysis only shoreline units with marshes undergoing erosion were assessed. Based on the AIC analysis and similar to the results from Approach 2, fetch was the most important predictor of shoreline changes with the highest model coefficient for both localities (Figure 2.11 and Tables 2.16-2.17). Fetch was positively correlated with shoreline retreat indicating high erosion rates were observed at shoreline units with longer fetch conditions.

In Mathews, tree fringe and land slope were the most important predictors after fetch. These two components were negatively correlated indicative of higher erosion rates along shorelines with low presence or absence of tree fringe and in low land slopes. In this case, a vegetation component was identified to provide a positive influence to reduce erosion in marshes. In addition, marshes can adapt to changes and maintain a certain vertical and horizontal growth in coastal areas with fetch conditions ≤300m (Williams, 2001). Longer fetch conditions increase the risk of erosion in marshes and even more if the land slope is low. The combination of long fetch and low slope conditions could have speeded up erosion to a faster rate than marsh accretion can occur.

Hampton’s model only showed physical components influencing marsh retreat. Bank height was the second most important component after fetch. Height was positively correlated with shoreline change suggesting that higher erosion rates were identified at shoreline units with long fetch and low bank heights. Based on the GM, bank height represented the same pattern as land slope in Mathews County.

The models generated for Approach 2 and 3 identified fetch and land slope as the main predictors of shoreline change by shoreline feature. However, land slope presented different patterns. For Approach 2, erosion increased with increasing slope, but for
Approach 3 erosion was higher in shoreline units with low land slopes or low bank heights. Figure 2.12a shows all marshes in Mathews by land slope and rate of shoreline change and based on Approach 2. Shoreline units in low slopes showed equal presence of shorelines with accretion and erosion, but shoreline units above 5° in slope experienced mostly shoreline retreat. In Figure 2.12b the conditions for land slope against shoreline change based on Approach 3 are shown. This figure clearly indicates that most of the shoreline units experiencing most of the erosion were located at a land slope lower than 5°. These results also reiterated that even though marshes keeping up with increasing sea levels are found mainly at land slopes ≤5°, most of the marshes under the same land slope conditions are showing signs of retreat. This indicates that these marshes are under higher risks of being inundated. This also suggests that other local settings in shoreline units are controlling the response of marshes to sea level rise and land flooding will be a variable process along coastlines and not completely defined by land elevation and fetch conditions.

MODEL VERIFICATION

For the model verification process, an independent database for Gloucester County was used. This database was specifically generated for a 120 shoreline units to verify the models generated in all three approaches. Based on the observed versus predicted graphs, the models generated for each approach were not strong enough to predict shoreline change in Gloucester County (Figures 2.13a-b). Even though the databases incorporated in this analysis were based on highly resolved data, this could indicate that: 1) shoreline change conditions in these two localities cannot be generalized and are more complex than usually portrayed; 2) the total annualized error including the error from images resolution and the shoreline digitizing error does not allow proper assessment of shoreline changes below <0.15 m/yr (i.e. the most commonly observed magnitudes in shoreline change) over a short period of time (<46yr); 3) other short scale processes (e.g. storms) could be playing a more important role in shoreline change.
CONCLUSIONS

This study generated three different approaches with the objective of identifying the most important predictors of shoreline change in Mathews and Hampton. The ultimate goal was to create a model able to predict future shoreline changes in estuarine systems. The multiple models generated for each approach showed high variability by shoreline features and by locality in predictors and in the strength of their effects. This evidenced the complexity of the behavior and the dynamics that take place in shoreline systems.

Fetch was consistently identified as an important predictor in most models. This physical component showed a positive correlation with shoreline change indicating higher erosion rates with longer fetch conditions. Land slope was the second most important physical predictor in most models. Under Approach 2, this component was positively correlated to shoreline change in marshes indicating that shoreline retreat was common at shorelines with steep slopes. However, land slope showed the opposite correlation under Approach 3, indicative of higher marsh erosion rates in shorelines with low slopes. This is evidence of the complexity of marsh responses to shoreline change and increasing water levels. This also suggests that shoreline change will not occur uniformly along the shoreline.

Vegetation was not considered a constant and strong predictor of shoreline change. However, the inability to verify the models generated for each approach suggests that none of the models was strong enough to predict shoreline change. In other words, the components identified as important could not explain the dynamics driving most of the changes in shoreline systems. The model failure during verification could also be explained by the total error included in aerial images and shoreline digitization.

Approach 1 indicated the importance of analyzing unmanaged and managed shorelines individually due to the effects anthropogenically influenced shorelines can have in the models. This approach also determined that shoreline changes in unmanaged shorelines were mainly influenced by natural components and physical components controlled changes in managed shorelines.
Approach 2 showed all shoreline features in Mathews experiencing shoreline retreat since the 1960s. However, marshes showed the lowest averaged erosion rate compared to beaches and managed shorelines. These results implied that Mathews' marshes were more efficient at reducing shoreline retreat than managed shorelines. Hampton showed an accretion pattern specifically for managed shorelines and beaches. These two shoreline features showed the importance of anthropogenic influences in this locality and the indirect influence nearby managed shorelines have over beaches. Hampton's marshes were the only eroding feature in this locality and showed a higher averaged erosion rate than observed in Mathews. Even though Hampton experienced a higher rate of sea level rise than Mathews, the difference in averaged rates of change in marshes for these two localities was too small to identify a strong indication of sea level rise affecting Hampton's shorelines to a higher degree. This suggested that marshes were probably adapting to higher water levels by accreting at a rate similar to sea level rise.

Approach 3 identified fetch and land slope as the main predictors of shoreline erosion in marshes. However, based on the models, most eroding marshes are located at low land slopes contradicting what was observed under Approach 2. These results suggested that shoreline dynamics in marshes are highly complex.

With increasing development in coastal areas and the increase in threats from climate change, marshes are expected to reach a threshold where most of these features will be submerged and lost. Due to this reason, this study provided three different approaches to assess the components that are influencing shoreline change the most and to determine how different shoreline features are responding to the changes. However, more coastal observations and high quality data are necessary to determine with more precision the fate of shoreline features under future rates of sea level rise. This will allow the generation of effective management plans to proactively prepare for future changes, help protect the coastal population and to maintain as many estuarine ecosystems as possible.
TABLES AND FIGURES
Figure 2.1  a. Map of the Chesapeake Bay, USA indicating the location of  b. Mathews (M) and c. Hampton (H).
Approach I: Stratified Random Sampling
Unmanaged vs. Managed Shorelines

Approach II: Management Perspective Assessment at Local Scale
Marshes, Beaches and Managed Shorelines

Approach III: Shoreline Retreat Assessment for Marshes at Local Scale
Shoreline retreat in marshes

Figure 2.2 Diagram summarizing the three different approaches applied in this study. Each approach assessed shoreline change at different spatial scales and based on different shoreline types. A series of predictors were used per approach to statistically determine their influence over shoreline change.
<table>
<thead>
<tr>
<th>Uncertainty Source</th>
<th>Uncertainty for beach features</th>
<th>Uncertainty for short fetch marsh features</th>
<th>Uncertainty for long fetch marsh features</th>
<th>Uncertainty for managed shorelines</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pixel Error ((E_p)) (m):</td>
<td>0.264583</td>
<td>0.264583</td>
<td>0.264583</td>
<td>0.264583</td>
</tr>
<tr>
<td>Digitization Error ((E_d)) (m):</td>
<td>2.05</td>
<td>1.53</td>
<td>1.09</td>
<td>1.36</td>
</tr>
<tr>
<td>Rectification Error ((E_r)) (m):</td>
<td>5.4</td>
<td>5.4</td>
<td>5.4</td>
<td>5.4</td>
</tr>
<tr>
<td>Total Shoreline Positional Error ((E_{sp})) (m):</td>
<td>6.065286945</td>
<td>5.666445599</td>
<td>5.529313314</td>
<td>5.588810761</td>
</tr>
<tr>
<td>Mathews: Annualized Transect Error ((E_a)) (m/yr):</td>
<td>0.155520178</td>
<td>0.145293477</td>
<td>0.141777264</td>
<td>0.14330284</td>
</tr>
<tr>
<td>Hampton: Annualized Transect Error ((E_a)) (m/yr):</td>
<td>0.131854064</td>
<td>0.1231836</td>
<td>0.120202463</td>
<td>0.121495886</td>
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Table 2.1 Shoreline uncertainties for Mathews and Hampton by shoreline type.
<table>
<thead>
<tr>
<th>Component</th>
<th>Type of Data</th>
<th>Categorical Values/Model Values</th>
</tr>
</thead>
<tbody>
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<td>Maximum fetch</td>
<td>Categorical</td>
<td>Short (&lt;300m) / 1.00</td>
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<tr>
<td></td>
<td></td>
<td>Long (&gt;300m) / 0.50</td>
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<tr>
<td>Bathymetry</td>
<td>Categorical</td>
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<tr>
<td></td>
<td></td>
<td>Deep / 0.50</td>
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<tr>
<td>Defended shoreline</td>
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</tr>
<tr>
<td></td>
<td></td>
<td>Absent / 3.00</td>
</tr>
<tr>
<td>Bank height</td>
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<td>0-1.5m / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1.5-9.1m / 2.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>&gt;9.1m / 1.00</td>
</tr>
<tr>
<td>Bank stability</td>
<td>Categorical</td>
<td>Stable / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Undercut / 2.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Unstable / 1.00</td>
</tr>
<tr>
<td>Maximum wind direction</td>
<td>Categorical</td>
<td>West / 4.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>South / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>East / 2.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>North / 1.00</td>
</tr>
<tr>
<td>SAV</td>
<td>Categorical</td>
<td>Present / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Absent / 0.00</td>
</tr>
<tr>
<td>Phragmites</td>
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</tr>
<tr>
<td></td>
<td></td>
<td>Absent / 0.00</td>
</tr>
<tr>
<td>Beach</td>
<td>Categorical</td>
<td>Present / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Absent / 0.00</td>
</tr>
<tr>
<td>Mudflat</td>
<td>Categorical</td>
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<tr>
<td></td>
<td></td>
<td>Absent / 0.00</td>
</tr>
<tr>
<td>Tidal marsh</td>
<td>Categorical</td>
<td>Present / 3.00</td>
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<td></td>
<td></td>
<td>Absent / 0.00</td>
</tr>
<tr>
<td>Riparian forested lands</td>
<td>Categorical</td>
<td>Present / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Absent / 0.00</td>
</tr>
<tr>
<td>Riparian inland marsh</td>
<td>Categorical</td>
<td>Present / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Absent / 0.00</td>
</tr>
<tr>
<td>Vegetation cover</td>
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<td>Total (&gt;75%) / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Partial (25-75%) / 2.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Bare (&lt;25%) / 1.00</td>
</tr>
<tr>
<td>Vegetation composition</td>
<td>Categorical</td>
<td>High (3 or more types vegetation) / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Low (1 or 2 types vegetation) / 2.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>None / 0.00</td>
</tr>
<tr>
<td>Riparian land use</td>
<td>Categorical</td>
<td>Natural / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Agriculture / 2.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Developed / 1.00</td>
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<td>Present / 3.00</td>
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<td></td>
<td>Absent / 0.00</td>
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<td>Upland inland marsh</td>
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<td>Present / 3.00</td>
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<tr>
<td></td>
<td></td>
<td>Absent / 0.00</td>
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<tr>
<td>Upland land use</td>
<td>Categorical</td>
<td>Natural / 3.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Agriculture / 2.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Developed / 1.00</td>
</tr>
</tbody>
</table>

Table 2.2 Shoreline components assessed for Mathews and Hampton and specifically applied in Approach 1. Components included under the Database I were assessed by the CCRM. Database II was generated by the current study. The categorical values and model values are specified for each component.
<table>
<thead>
<tr>
<th>Component</th>
<th>Influence over shoreline change</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fetch</td>
<td>This coastal component is mainly applied as a simple measure of relative wave energy. It has been observed in previous studies that long fetch conditions trigger most of the shoreline erosion, especially during high energy storm events (Hardaway et al., 1992). Fetch is also highly correlated with marsh planting, a strategy promoted for shoreline protection and viable along low energy shorelines (Knutson et al., 1981). Several studies suggest that marsh creation or natural marsh areas did poorly in areas with fetch conditions exceeding 1,600m (Hardaway and Byrne, 1999).</td>
</tr>
<tr>
<td>Bathymetry</td>
<td>Shallow depths, like those observed in tidal flats and sand bars, can attenuate wave energy before reaching the shoreline more effectively than deeper waters (Hardaway et al., 1992).</td>
</tr>
<tr>
<td>Defended shoreline</td>
<td>Studies have found that structures alter hydrodynamics; wave regime; and sediment size, transport, and deposition (Runyan and Griggs, 2003; Martin et al., 2005; Dugan et al., 2011). Hard structures also function as barriers for marsh communities preventing landward migration.</td>
</tr>
<tr>
<td>Bank height</td>
<td>Based on a tool generated by the CCRM (2010), Decision Tree for Undeceived Shorelines and Those with Failed Structures, a failing high bank will erode large volumes of sediments and remove large amounts of vegetation if any are present. In low banks, the loss of sediments typically is less.</td>
</tr>
<tr>
<td>Bank stability</td>
<td>The instability of a bank is generated by different factors that can act individually or as an integrated unit. Some of the factors that promote instability are bank height, wave action, storm surge, rainfall impact, surface water runoff, groundwater seepage, sediment starvation, bank slope, bank vegetation cover and boat wakes (Hardaway et al., 1992). All these factors increase the probability of generating an unstable bank and consequent failure.</td>
</tr>
<tr>
<td>SAV</td>
<td>SAV modifies energy regimes and stabilizes sediments (Deaton et al., 2010; Fonseca and Calahan, 1992).</td>
</tr>
</tbody>
</table>

Table 2.3 Description of the type of influence shoreline components have over shoreline change. (Continuation below).
| **Phragmites** | Rooth et al. (2003) concluded that the *P. australis* community was associated with higher depositional patterns and faster increase in substrate elevation over relatively short periods compared to other marsh communities. |
| **Beach** | In sandy environments, beaches are typically gently sloping. Based on Rosen (1980), beaches can have the largest vertical buffer to the impact of storm surge and waves. |
| **Mudflat** | Low energy areas and high deposition rates of clay, silts, and biological detritus (Little, 2000). |
| **Tidal marsh** | Knutson et al. (1982) concluded that over 50% of wave energy was dissipated within the first 2.5m of marshes. This reduces erosion of the adjacent riparian and upland zones. As suggested by Rosen (1980) who identified marsh margins as the least erodible shorelines in Chesapeake Bay, the natural cohesive properties of the fine-grained sediment that comprise marshes make them more resilient to wave erosion than unconsolidated beach material. |
| **Land vegetation (forested lands, inland marsh, vegetation cover, composition, tree fringe, canopy overhang)** | Known for providing a buffer system that contributes to reduced effects from flooding events. They contribute small and large debris to the soil and nearby waters. In nearshore waters, large debris can provide roughness to the channel bed and bank toe-slopes; reducing water velocity and increasing deposition. (Dossey et al., 2010). |
| **Land use** | Developed shorelines show lower presence of vegetation and higher deposition of fine particles (Jennings et al., 2001). Presence of shoreline armoring increases with development interrupting the natural trends in sediment transport and deposition. |
| **Land slope** | Based on Gesch et al. (2009) and Cahoon (2009) low lying lands are the most vulnerable to inundation due to sea level rise. |

Table 2.3 (Continuation) Description of the type of influence shoreline components have over shoreline change.
<table>
<thead>
<tr>
<th>Shoreline Types</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Fetch = Long</strong></td>
</tr>
<tr>
<td>Bank height = 0-1.5m</td>
</tr>
<tr>
<td><strong>Fetch = Short</strong></td>
</tr>
<tr>
<td>Bank height = 0-1.5m</td>
</tr>
</tbody>
</table>

Table 2.4 Six different shoreline classes based on fetch (short = ≤300m; long = >300) and bank height classifications. These classes were used for Mathews' and Hampton's shorelines under Approach 1.
<table>
<thead>
<tr>
<th>Component</th>
<th>Type of Data</th>
<th>Categorical Values/Model Values</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maximum fetch</td>
<td>Continuous (m)</td>
<td>Short (&lt;804.67m; &lt;0.5 mile) Moderate (804.67 to 3218.69m; 0.5 - 2 miles) Long (&gt;3218.69; &gt;2 miles)</td>
</tr>
<tr>
<td>Bank height</td>
<td>Categorical</td>
<td>0-1.5m / 3.00 1.5-9.1m / 2.00 &gt;9.1m / 1.00</td>
</tr>
<tr>
<td>Maximum wind direction</td>
<td>Categorical</td>
<td>West / 4.00 South / 3.00 East / 2.00 North / 1.00</td>
</tr>
<tr>
<td>Tree fringe</td>
<td>Categorical</td>
<td>Present / 3.00 Absent / 0.00</td>
</tr>
<tr>
<td>Canopy overhang</td>
<td>Categorical</td>
<td>Present / 3.00 Absent / 0.00</td>
</tr>
</tbody>
</table>

| Database III               |                |                                                                     |
| Land slope                 | Continuous (°) | Mathews' range: 0- 88°; Hampton' range: 0-70°                     |

Table 2.5 Physical and vegetation components assessed for Approach 2 and Approach 3. Components included in the Database I were assessed for the CCRM's shoreline inventory. Database III was generated by the current study. The categorical values and model values are specified for each component.

<table>
<thead>
<tr>
<th>Locality</th>
<th>Feature</th>
<th>Total # of features</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mathews</td>
<td>Marsh</td>
<td>13,174</td>
</tr>
<tr>
<td></td>
<td>Beach</td>
<td>623</td>
</tr>
<tr>
<td></td>
<td>Defended</td>
<td>1,442</td>
</tr>
<tr>
<td>Hampton</td>
<td>Marsh</td>
<td>3,746</td>
</tr>
<tr>
<td></td>
<td>Beach</td>
<td>183</td>
</tr>
<tr>
<td></td>
<td>Defended</td>
<td>1,697</td>
</tr>
</tbody>
</table>

Table 2.6 Total number of shoreline units assessed per shoreline feature for Mathews and Hampton and for Approach 2.
Figures 2.3a-b a. Linear regression for the sea level rise trend at the Gloucester Point/Yorktown and at the b. Sewells Point stations, VA.
<table>
<thead>
<tr>
<th>Locality</th>
<th>Feature</th>
<th>Eroding Marshes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mathews</td>
<td>Marsh</td>
<td>8,266</td>
</tr>
<tr>
<td>Hampton</td>
<td>Marsh</td>
<td>2,818</td>
</tr>
</tbody>
</table>

Table 2.7 Total number of shoreline units with eroding marshes that were assessed for Mathews and Hampton and for Approach 3.
Figure 2.4 Conceptual model indicating predictors of shoreline change based on Approach 1. Plus and minus symbols next to the predictors indicate the type of correlation with shoreline change. Solid lines indicate the component showed a high model coefficient, a dashed line is a moderate model coefficient, and a dotted line is a low model coefficient.
### Mathews Global Model: Unmanaged Shoreline Units

<table>
<thead>
<tr>
<th>Intercept</th>
<th>Beach</th>
<th>Vegetation composition</th>
<th>Riparian forested lands</th>
<th>Fetch</th>
<th>Riparian land use</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.054</td>
<td>0.06433</td>
<td>-0.01735</td>
<td>0.023985</td>
<td>0.00992</td>
<td>0.041428</td>
</tr>
</tbody>
</table>

**Table 2.8** Global model (GM) for unmanaged shoreline units in Mathews. The table includes the components identified as predictors, the model coefficients and the model intercept.

### Hampton Global Model: Unmanaged Shoreline Units

<table>
<thead>
<tr>
<th>Intercept</th>
<th>Beach</th>
<th>Vegetation Composition</th>
<th>Tidal marsh</th>
</tr>
</thead>
<tbody>
<tr>
<td>3.228471</td>
<td>-0.00586</td>
<td>-0.18547</td>
<td>-0.03442</td>
</tr>
</tbody>
</table>

**Table 2.9** Global model (GM) for unmanaged shoreline units in Hampton. The table includes the components identified as predictors, the model coefficients and the model intercept.

### Hampton Global Model: Managed Shoreline Units

<table>
<thead>
<tr>
<th>Intercept</th>
<th>Fetch</th>
<th>Bank height</th>
<th>Maximum wind direction</th>
<th>Riparian land use</th>
<th>Tidal marsh</th>
<th>Beach</th>
<th>Vegetation composition</th>
</tr>
</thead>
<tbody>
<tr>
<td>2.897143</td>
<td>0.11741</td>
<td>-0.12728</td>
<td>-0.09051</td>
<td>0.20387</td>
<td>0.13169</td>
<td>-0.03162</td>
<td>0.002011</td>
</tr>
</tbody>
</table>

**Table 2.10** Global model (GM) for managed shoreline units in Hampton. The table includes the components identified as predictors, the model coefficients and the model intercept.
Figure 2.5 Box plot describing rates of shoreline change for beaches, marshes and defended (i.e. managed) shorelines in Mathews and Hampton. The values included in the graph are the averaged shoreline change per shoreline feature.

Table 2.11 General shoreline change and slope statistics for Mathews’ and Hampton’s shoreline features.
Figure 2.6 Box plot describing land slope for beaches, marshes and defended (i.e. managed) shorelines in Mathews and Hampton. The values included in the graph are the averaged slope in degrees per shoreline feature.
Figures 2.7a-c Residuals and observed values for marshes in a. Mathews and b. Hampton. c. Mathews’ and Hamptons’ residual values combined.
Figures 2.8a-c Residuals and observed values for beaches in a. Mathews and b. Hampton. c. Mathews' and Hampton's residual values combined.
Figures 2.9a-c Residuals and observed values for defended shorelines in a. Mathews and b. Hampton. c. Mathews' and Hamptons' residual values combined.
<table>
<thead>
<tr>
<th>Slope</th>
<th>Hampton Observed</th>
<th>Mathews Observed</th>
<th>Hampton Expected</th>
<th>Mathews Expected</th>
<th>Observed_Difference</th>
<th>Expected_Difference</th>
</tr>
</thead>
<tbody>
<tr>
<td>1-2°</td>
<td>-3.56</td>
<td>-2.16</td>
<td>-9.48</td>
<td>-6.98</td>
<td>-1.4</td>
<td>-2.5</td>
</tr>
<tr>
<td>3-4°</td>
<td>-3.48</td>
<td>-2.39</td>
<td>-3.82</td>
<td>-2.82</td>
<td>-1.09</td>
<td>-1</td>
</tr>
<tr>
<td>5-6°</td>
<td>-2.55</td>
<td>-2.15</td>
<td>-2.43</td>
<td>-1.78</td>
<td>-0.4</td>
<td>-0.65</td>
</tr>
<tr>
<td>7-8°</td>
<td>-3.92</td>
<td>-2.66</td>
<td>-1.77</td>
<td>-1.3</td>
<td>-1.26</td>
<td>-0.47</td>
</tr>
<tr>
<td>9-10°</td>
<td>-3.04</td>
<td>-2.91</td>
<td>-1.4</td>
<td>-1.02</td>
<td>-0.13</td>
<td>-0.38</td>
</tr>
</tbody>
</table>

Table 2.12 Observed and expected horizontal displacement of the shoreline per slope intervals. The table shows lower observed values than the expected values for land slopes under 5°. The opposite was observed for land slopes over 5°.
Figure 2.10 Conceptual model indicating predictors of shoreline change based on Approach 2. Plus and minus symbols next to the predictors indicate the type of correlation with shoreline change. Solid lines indicate the component showed a high model coefficient, a dashed line is a moderate model coefficient, and a dotted line is a low model coefficient.
Table 2.13 Global model (GM) for shoreline units with marsh presence in Mathews. The table includes the components identified as predictors, the model coefficients and the model intercept.

<table>
<thead>
<tr>
<th>Component</th>
<th>Coefficient</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>1.752618</td>
</tr>
<tr>
<td>Fetch</td>
<td>0.06398</td>
</tr>
<tr>
<td>Land slope</td>
<td>0.00967376</td>
</tr>
<tr>
<td>Bank height</td>
<td>-0.003796</td>
</tr>
<tr>
<td>Tree fringe</td>
<td>-0.0042181</td>
</tr>
<tr>
<td>Canopy overhang</td>
<td>0.0017609</td>
</tr>
</tbody>
</table>

Table 2.14 Global model (GM) for shoreline units with marsh presence in Hampton. The table includes the components identified as predictors, the model coefficients and the model intercept.

<table>
<thead>
<tr>
<th>Component</th>
<th>Coefficient</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>1.585216</td>
</tr>
<tr>
<td>Fetch</td>
<td>0.02246</td>
</tr>
<tr>
<td>Maximum wind direction</td>
<td>0.01468033</td>
</tr>
<tr>
<td>Land slope</td>
<td>0.013656922</td>
</tr>
<tr>
<td>Locality</td>
<td>Land Use Type</td>
</tr>
<tr>
<td>----------</td>
<td>---------------</td>
</tr>
<tr>
<td>Mathews</td>
<td>Natural</td>
</tr>
<tr>
<td></td>
<td>Developed</td>
</tr>
<tr>
<td></td>
<td>Agriculture</td>
</tr>
<tr>
<td>Hampton</td>
<td>Natural</td>
</tr>
<tr>
<td></td>
<td>Developed</td>
</tr>
<tr>
<td></td>
<td>Agriculture</td>
</tr>
</tbody>
</table>

Table 2.15 Differences in shoreline change (EPR), fetch and slope conditions per land use type in Mathews and Hampton.
Figure 2.11 Conceptual model indicating predictors of shoreline change based on Approach 3. Plus and minus symbols next to the predictors indicate the type of correlation with shoreline change. Solid lines indicate the component showed a high model coefficient, a dashed line is a moderate model coefficient, and a dotted line is a low model coefficient.
Mathews Global Model: Eroding Marshes

<table>
<thead>
<tr>
<th>Intercept</th>
<th>Fetch</th>
<th>Slope</th>
<th>Tree fringe</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.204388</td>
<td>0.060469333</td>
<td>-0.006594545</td>
<td>-0.01276596</td>
</tr>
</tbody>
</table>

Table 2.16 Global model (GM) for shoreline units with eroding marshes in Mathews. The table includes the components identified as predictors, the model coefficients and the model intercept.

Hampton Global Model: Eroding Marshes

<table>
<thead>
<tr>
<th>Intercept</th>
<th>Fetch</th>
<th>Bank height</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.570316</td>
<td>0.047975879</td>
<td>0.010688001</td>
</tr>
</tbody>
</table>

Table 2.17 Global model (GM) for shoreline units with eroding marshes in Hampton. The table includes the components identified as predictors, the model coefficients and the model intercept.
Figures 2.12a-b  Slope vs. shoreline change for marshes in Mathews County.  

a. Rates of shoreline change by land slope for all marshes from Approach 2.  
b. Rates of shoreline change by land slope for eroding marshes from Approach 3.
Figures 2.13a-b Model verification for a. Mathews' and b. Hampton's eroding marshes models from Approach 3. This is an example of the lack of strength observed in the models generated in this study.


(lead authors)]. U.S. Environmental Protection Agency, Washington DC, pp. 11-24.


Chapter 3

Influence of Sea Level Rise and Management Practices on Potential Capacity to Provide Habitat and Water Quality Services in Tidal Shorelines by 2050
ABSTRACT

Sea level rise is currently threatening tidal shorelines and ecosystems services in low lying lands in the Chesapeake Bay. By 2050, water levels are expected to increase between 0.35m to 0.80m jeopardizing the potential capacity of tidal shorelines to produce habitat and water quality services. Due to the high uncertainty included in sea level rise forecasts and due to high variability in shorelines response to land inundation, the identification of possible future impacts in ecosystems is becoming a challenge. To assess possible changes that the potential capacity of ecosystems may experience due to land inundation, this study generated the Shoreline Condition Integrated Model (SCIM). The SCIM integrates the Habitat and Water Quality Services Models to determine potential capacity of ecosystem services by 2050. Potential capacity was defined based on two different scenarios. For Scenario 1, an Inundation Model was generated to determine changes in potential capacity based on the total land area inundated by 2050 using two different rates of sea level rise. Scenario 2 assumed application of living shoreline management practices under the same sea level rise conditions.

This study found a drastic decrease in ecosystems potential capacity by 2050 in Mathews and Hampton with increasing sea level rise. Most shoreline units will be reduced to moderate capacity in Mathews and low capacity in Hampton. All vegetation and natural components experienced a large loss in area due to land inundation. Beaches, tidal marshes, inland marshes and trees experienced most of the area loss. Living shoreline methods increased potential capacity for more than 90% of the shoreline units assessed in this study indicating the importance of providing a stronger vegetation buffer in tidal shorelines. In addition, up to 65% of Hampton's shoreline and 78% of Mathews' shoreline presented suitable conditions for living shoreline projects. Based on these results, a larger effort to expand living shoreline projects along other coastal localities in the Chesapeake Bay is recommended to preserve ecosystem services in the face of sea level rise.
INTRODUCTION

With sea levels expected to rise in the near future between 0.35 to 0.80m, shoreline ecosystems will be vulnerable to inundation and many of their ecosystem services could be lost (Titus and Wang, 2008; Ramhstorf, 2007). Despite the growing body of research and literature on ecosystem services during the last couple of decades, it is still unknown what the possible conditions in potential capacity may be based on different sea level rise scenarios. This limits the opportunity to generate practical and coherent research that could provide guidance for the use of services in trade-off analysis and to provide proactive management decisions. Even though current projections cannot exactly predict sea level rise in the next decades, recent management practices present a solution that could help buffer some of the possible effects in shoreline ecosystems generated by increasing water levels.

The Chesapeake Bay region experienced critical changes in land use and shoreline change during the last decades mostly due to population growth in the coastal zone and climate change. These changes generated large impacts on shoreline components and consequently on habitat and water quality services. Currently, approximately 60% of Virginia’s population lives in 22% of the state’s coastal zone (Erdle et al., 2006). In addition, one third of the Bay’s shoreline is undergoing erosion and some areas lose between 20-40 cm of shoreline per year. With expected increases in developed lands, coastal population and sea level rise in the years to come, a larger effort to protect private and public infrastructure is already taking place. Shoreline management methods employed in the past will result in expanded structural shoreline armor such as riprap and bulkheads. These methods are known to change shoreline dynamics by removing vegetation buffers and altering shoreline elevation. Consequently these changes reduce water filtration and habitat space from the system.

Living shorelines and hybrid shoreline stabilization methods were recently identified as the preferred techniques to help reduce shoreline erosion and to maintain habitat and water quality services in shoreline systems. Currently, additional assessments are required to identify suitable shoreline sites for living shoreline projects based on
shoreline settings and expected rates of sea level rise. This type of information will facilitate decision making and a more effective management plan.

The main goal of Chapter 3 was to generate a categorical integrated model capable of determining ecosystem services by 2050 based on changes in land inundation due to sea level rise and local management practices. To generate this model a series of objectives were pursued:

- Estimate total loss of shoreline components based on two different sea level rise scenarios.
- Identify the most suitable locations to establish living shorelines and determine the possible future conditions of shoreline components if this practice is applied.
- Determine potential capacity to provide services by 2050 based on land inundation due to sea level rise, and management practices.

SEA LEVEL RISE

Increase in the concentration of carbon dioxide in the atmosphere has caused a general trend of warming of the planet. Growing evidence suggests that the increase in temperature is the cause of thermal expansion of the oceans and melting of polar ice, inducing a gradual rise in sea levels (Church et al., 2006). It is clear that since the 20th century the earth has experienced an overall upward tendency in global temperatures (Pyke et al., 2008; Stevenson et al., 2002). Scientists also agree that the decade of 1990s was characterized by record breaking warm temperatures around the globe. Among the different projections published, the Intergovernmental Panel on Climate Change (IPCC, 2007) generated an upper estimate of one meter increase in sea level over the next century. However, the IPCC has recognized that sea level rise by 2100 can be 0.10 to 0.20 meter higher than previously predicted. This possible increase in the predicted sea level rise is due to the uncertainty of ice sheet melt and glacier dynamics. Full
understanding of the factors causing sea level rise is still lacking and the uncertainty in future sea level is probably higher than already estimated (Rahmstorf, 2007).

The sea level trend is not uniform around the world. Some areas experience a much faster sea level rise than the global average trend of $3.0 \pm 0.4 \text{ mm/yr}$ based on satellite observations from 1993-2010 (Boon et al., 2010). Sea level trends are identified in two different ways: as eustatic (global) sea level change and a relative sea level change. The eustatic sea level rise or global sea level rise is currently increasing due to an overall increase in the volume of water in the oceans and seas. This increase in the volume of the world's oceans is driven by an increase in the water column temperature, thermal expansion, and by melting of ice sheets, ice caps, and glaciers. The second trend is the relative sea level. This trend indicates changes in the water level relative to the land surface. Processes such as subsidence or sinking of the land, emergence or uplift of the land, tectonics, groundwater extraction, and soil compaction are drivers of relative sea level change at local and regional scales.

Coastal areas are highly vulnerable to variations in sea level. Coastal flooding, erosion, inundation, and saltwater intrusion can all cause significant biophysical impacts and may become much more severe in the future (Brown, 2006). The impacts of sea level rise interact with current coastal stressors, such as development and pollution, severely compromising the overall capacity and resilience of coastal ecosystems (Neumann et al., 2010). It is possible for ecosystems to adapt to sea level rise in coastal areas where human influence is minimal. Habitats in areas with limited human disturbances can migrate landward or accrete vertically in response to sea level rise (Mcleod et al., 2010).

Sea-level rise poses a potentially greater long-term threat, depending on its rate, because the effects of inundation and a more persistent salinity regime could cause widespread changes or loss of coastal ecosystems. Hence, ecosystem services all over the world will be affected due to sea level rise in the near future putting society and the provision of goods, such as fishery resources, in higher risk (Callaway et al., 2007; Titus, 1991). The design of coastal climate change policies will require an evaluation of potential future impacts on coastal communities and ecosystems. The inclusion of these
impacts in policy discussions may help reduce future costs and will help inform decisions to balance investments in mitigation and adaption (Neumann et al., 2010).

MANAGEMENT PRACTICES: LIVING SHORELINES

The benefits of shoreline and riparian vegetation buffers are well documented in the scientific literature (e.g. Dosskey et al., 2010; Klapproth and Johnson, 2001). Vegetated buffers are considered a main line of defense against pollution by the filtering function provided by the vegetation’s root system. At the same time, these vegetated areas provide the habitat space required for multiple aquatic and terrestrial organisms. For example, Mattheus et al. (2010) showed tidal and non-tidal marshes provide essential services to shoreline ecosystems by improving water quality and by dampening wave and tidal energies, facilitating sedimentation. These plant communities provide shelter, nursery habitat, and feeding grounds for aquatic and terrestrial organisms. However, anthropogenic activities can alter sedimentation patterns in the nearshore influencing shoreline erosion. Other modifications such as shoreline armoring can increase erosion in marsh communities by disrupting the along-shore sediment transport and/or wave refraction and generate adverse effects in habitats and organisms (Dugan et al., 2011). In addition, these structures inhibit the potential of to migrate inland.

Living shoreline is the most recent type of shoreline stabilization method that promotes the use of a variety of natural features such as deeply rooted vegetation, marshes, and sand beaches (Duhring, 2006). Opposite to this natural practice, the term shoreline armoring defines the use of physical barriers such as breakwaters and bulkheads with the main purpose of controlling shoreline erosion. However, the placement of this type of structure in shoreline systems generates a permanent loss in nearby vegetation (National Research Council, 2007). A hybrid method is another shoreline stabilization technique that incorporates both non-structural (i.e. living shoreline) and structural (i.e. armoring) methods. Depending on the shoreline settings (i.e. fetch conditions), the placement of physical barriers allows the creation of natural buffers providing erosion protection, habitat and water quality services.
In 2011 the Senate Bill 964 became law in Virginia stating that living shorelines are the preferred method for stabilizing tidal shorelines in the Commonwealth (CCRM, 2012). The law defines living shorelines as "...a shoreline management practice that provides erosion control and water quality benefits; protects, restores or enhances natural shoreline habitat; and maintains coastal process through the strategic placement of plants, stone, sand fill, and other structural and organic materials." (Code of Virginia §28.2-104.1).

Several living shoreline and hybrid methods are currently being used in the state of Virginia (from Duhring, 2006):

- **Enhance, maintain and/or widen marsh:** Tidal marsh enhancement includes adding new marsh plants to barren or sparsely vegetated marsh areas. Sand fill can be added to a marsh surface to maintain its position in the tide range or to increase its width for more protection. Replacing marsh plants washed out during storms also fits into this category. Less mowing of wetland vegetation can also enhance the stabilizing and habitat features of a tidal marsh. Shorelines with existing marshes or where marshes are known to have occurred in the recent past may be suitable for this treatment. Water depth and the amount of sunlight available are key factors to consider. A wide, gently sloping intertidal area with minimal wave action also indicates suitability.

- **Plant marsh with sill:** Tidal marsh creation can be applied where a natural marsh does not exist. Non-vegetated intertidal areas can be converted to a tidal marsh by planting on the existing substrate. Because a wide marsh is needed for effective stabilization, this method normally requires either grading the riparian area landward or filling channelward into the subtidal area for a wider intertidal zone. The plant species will depend on the local salinity range plus the depth and duration of tidal flooding. Two common tidal marsh grasses used for this purpose are *Spartina alterniflora* and *S. patens*. The most suitable shorelines for tidal marsh creation have wide, gradual slopes from the upland bank to the subtidal waters, a sandy substrate without anaerobic conditions, and plenty of sunlight.
Extensive tree removal in the riparian buffer just to create suitable growing conditions for a tidal marsh should be avoided, especially if the forested bank is relatively stable. Salt marsh plants have a limited tolerance for wave action. The wave climate and the frequency and size of boat wakes must also be considered. Marsh sills are a similar type of low stone structure, but they are used where no existing marsh is present. Sills are usually located near the low tide line, then backfilled with clean sand to create a suitable elevation and slope for planted tidal marsh vegetation. Like marsh toe revetments, the height of the sill should be near the mean high water elevation to minimize interruption of tidal exchange. Eroding banks without a tidal marsh present are candidate sites for marsh sills, particularly if marshes exist in the general vicinity. However, the physical alterations needed to create suitable planting elevations and growing conditions should not require major disturbance to desirable shoreline habitats, such as mature forested riparian buffers or valuable shallow water habitats (e.g., shellfish beds, submerged aquatic vegetation). If bank grading is appropriate to create target slopes, then the bank material can possibly be used to backfill a marsh sill if it is mostly coarse-grained sand. Sand fill can also be imported from an upland source.

- Enhance and/or maintain beach or beach nourishment: Beach nourishment is the addition of sand to a beach to raise its elevation and increase its width to enhance its ability to buffer the upland from wave action. The use of structural methods can be applied when necessary. Beach stabilization may require plants usually after a beach nourishment event. Common plant species for Chesapeake Bay beaches and dunes include *Ammophila breviligulata*, *Panicum amarum*, and *Spartinapatens*. Beach and bank erosion may still occur during storms. Periodic replenishment is usually needed to maintain the desired beach profile. This method may not provide sufficient protection where no beach currently exists or where tidal currents and wave action remove sand rapidly.

- Enhance and/or maintain riparian buffer: Activities to enhance the density or species diversity of stabilizing bank vegetation are referred to collectively as riparian vegetation management. These actions include trimming tree branches
overhanging a marsh to increase sunlight, selectively choosing desirable plants for
natural regeneration, or planting additional landscape material to increase cover or
diversity. Using vegetation buffers to intercept stormwater runoff from developed
areas and controlling invasive species that degrade habitat quality and
stabilization effectiveness are also included. Most tidal shorelines are suitable for
some type of riparian vegetation management and enhancement activities.

An assessment of Virginia's shoreline indicated that more than half of the state’s
shoreline presents the conditions necessary (i.e. fetch ≤ 2 miles) for the success of living
shoreline projects (CCRM, 2012). This indicates that this management practice could be
part of the solution to reduce the risks inland from sea level rise and to maintain shoreline
ecosystem services.

STUDY SITES

CHESAPEAKE BAY

The Chesapeake Bay is located in the Mid-Atlantic coast of the United States and
is one of the largest estuaries in the world (Figure 3.1a). The strong interactions found in
this estuary between land surface, fresh, and saltwater provide the conditions for a variety
of ecotones, high biodiversity, and high productivity. However, a recent State of the Bay
(2012) indicates poor water quality, a reduction in natural habitats, and compromised
conditions of many coastal resources and organisms. These circumstances jeopardize the
health and quality of the ecosystem services generated in the estuary.

The unpredictability of climate change also increases the challenge in restoration
of the Bay’s conditions. Based on a 35 year database from 10 tide gauges from Norfolk,
VA and Baltimore, MD the relative rates of sea level in the Chesapeake Bay range from
2.91 to 5.80mm per year. These rates are higher than the rates observed in many other
areas in the U.S. East Coast (Boon et al., 2010). Ramhstorf (2007) predictions indicate
that the Chesapeake Bay will be experiencing an increase of 0.7m (700mm) to 1.6m
(1,600mm) in sea level by 2100.
This study focused on the shorelines along Mathews County and City of Hampton in the state of Virginia (Figures 3.1b-c). The socioeconomic characteristics differ between localities with more rural lands observed in Mathews and a highly developed landscape in Hampton. However, these localities share similar physical coastal conditions (i.e. mean tidal range, coastal slope, rate of relative sea level rise, shoreline erosion and accretion rates, mean wave height, geomorphology) (Boruff et al., 2005). More importantly, both localities coastal area lies below the 6m elevation contour (Titus and Wang, 2008). This implies future greater risks of inundation for developed coastal areas and the loss of shoreline features.

MATHEWS COUNTY

Mathews County is located on the middle peninsula of the state of Virginia. The county is bordered by Mobjack Bay to the south, Chesapeake Bay to the east, North River to the west, the Piankatank River to the north, and Gloucester County at the northwest (Figure 3.1b). According to the U.S. Census Bureau, the county has a total area of 652.68 km² of which 222.74 km² is land, 429.94 km² is water, with 559.04 kilometers of shoreline.

Mathew’s shoreline types vary along the County’s coast. Wave climate conditions range from fetch-limited creeks to open Bay high fetch. Most of the tidal shorelines in Mathews County are found in narrow, small creeks and rivers with low wave energy (Hardaway et al., 2010).

The intertidal zone is mainly characterized by the presence of marshes, wetlands, maritime forests, high and low energy shorelines, beaches, and dunes. These coastal components are currently providing habitat for different aquatic and terrestrial species, reducing wave energy and erosion, and stabilizing shoreline sediments. The North River is characterized by having very low uplands and marsh coasts. The eastern part of the coast has very high energy barrier beaches and marshes. High uplands are commonly
observed along the Piankatank River. For 2010, about 80 kilometers of Mathews’ 559.04 kilometers of shoreline were already defended (Hardaway et al., 2010). From these 80 km, 27 miles were built in the last ten years and this amount is expected to increase greatly in the years to come.

Historically, shoreline change rates varied from 0ft/yr to over ±2.44 m/yr for both erosion and accretion along the Bay’s coast (Byrne and Anderson, 1978). A recent study from Hardaway et al. (2005a) calculated shoreline rates of change from 1937 to 2002 that varies from 0.88 m/yr to –3.17 m/yr. Strange et al. (2008) concluded that an increase of 2mm in water levels will transform marshes in the Mobjack Bay area to marginal marshes. With future increasing water levels it is expected that some marshes and unnourished beaches will be completely lost in the Piankatank River due to bank elevations greater than 3m. Beaches facing the Chesapeake Bay are currently showing signs of high erosion rates. Marshes and beaches with sufficient sediments to accrete and keep pace with a 7 to 16mm/yr increase of sea level are likely to continue migrating inland, but most marshes are likely to be lost with a predicted 7mm per year of sea level increase.

CITY OF HAMPTON

The City of Hampton is one of the seven major cities in Hampton Roads metropolitan area. It is located on the southeastern end of the Virginia Peninsula. The City shares physical boundaries with Newport News and York County to the northeast and it is contiguous to the Back River to the north, Chesapeake Bay waters to the west and the James River to the south (Figure 3.1c). Based on the U.S. Census Bureau, the City has a total area of 352.76 km², of which 134.16 km² is land, and 218.60 km² is water (Hardaway et al., 2005b; CCRM, 2011). The City has a total of 234.13 kilometers of shoreline that includes 12.07 kilometers of tidal shoreline extend along the James River, 12.87 kilometers along the Chesapeake Bay, and 8.05 km along the Back River.

Shorelines are characterized by a wave climate defined by a large fetch exposure mainly to the northeast and east across the Chesapeake Bay (Hardaway et al., 2005).
Most of the shorelines along Hampton River are bulkheaded. The bayfront shorelines and lowland areas prone to tidal flooding are occupied by extensive marshes and surrounded by heavy development in the upland zone.

Hampton’s shorelines have experienced strong impacts in the past due to coastal flooding during hurricanes and nor’easters (Boon et al., 2010). In addition, the combination of effects from sea level rise and land subsidence in this City will expose many shorelines and coastal communities to greater risks from sea level rise in the future. Observations already confirmed the inundation of marsh areas, converting these to tidal flats and then open water (Strange et al., 2008).

Historically, shoreline rate of change for Hampton shorelines varied between 0 to 1.37 m/yr for both shoreline retreat and accretion (Byrne and Anderson, 1978). Hardaway et al. (2005b), calculated similar rates between 1937-2002 of 0 to -1.25 m/yr. Based on the expected future increase in sea level, planners indicate that the developed portion of the City is almost certain to be protected by defended shorelines while other areas east of the city are already experiencing shoreline erosion (Strange et al., 2008).

METHODS

To determine potential capacity to provide habitat and water quality services by 2050, the Shoreline Condition Integrated Model (SCIM) was generated. The SCIM is a categorical model based on the Habitat and the Water Quality Services Models from Chapter 1. These two models were used to determine capacity to provide services. To determine the effects from land inundation and management practices on the capacity to provide services, the SCIM assessed the impacts from two accelerated sea level rise scenarios and the influence from multiple living shoreline methods. Ultimately, the capacity was determined and classified for the same shoreline units specified in Chapter 1 (Tables 3.1-3.2 and Figures 3.2a-b).

This study defined future capacity to provide services based on two different scenarios. Scenario 1 considered impacts from sea level rise by 2050 by generating the Inundation Model. This scenario assumed changes generated in shoreline components by
2050 will only be generated by sea level rise and land elevation. Sea level rise will follow its course without human interventions. For Scenario 1, management practices and development will remain the same through time. Scenario 1 also assumed that current management practices, specifically the Chesapeake Bay Preservation Act (CBPA), will be rigorously implemented. As part of the CBPA regulations, a 100 foot wide buffer area is required as the landward component of the Resource Protection Area (RPA) (JLARC, 2003; Baird and Wetmore, 2006). The Act defines RPA as “…that component of the Chesapeake Bay Preservation Area comprised of lands adjacent to water bodies with perennial flow that have an intrinsic water quality value due to the ecological and biological processes they perform or are sensitive to impacts which may result in significant degradation to the quality of state waters.” This assumes that no additional development will occur within 100m from the shoreline position.

Scenario 2 was based on identifying the most suitable locations for living shoreline projects and their effect on shoreline components based on future land inundation. Scenario 2 assumed sea level rise will generate changes in shoreline components due to land inundation, but the appropriate location and type of living shoreline and/or hybrid management practices will reduce impacts from inundation preserving most of the shoreline components. Consequently, these methods are assumed to maintain habitat and water quality services in most shoreline units.

ECOSYSTEM CAPACITY: DRIVERS OF CHANGE

Future Land Inundation due to Sea Level Rise: Scenario 1

The goal of Scenario 1 was to generate a single-value surface model or a bathtub model to project sea level conditions by 2050 for Mathews County and City of Hampton. A bathtub model only includes two variables, the sea level and the land elevation (Schmid et al., 2013). The forecast of sea level was generated for two different accelerated scenarios selected from Pyke et al. (2008).
Future shoreline position was forecasted based on a simple Inundation Model (IM). The assumption is that shorelines will simply move to the appropriate upland contour depending on how much sea level rises (Nüñez, 2010). For example, if the sea level increases by two feet, it is assumed that the shoreline position will move to the location of the current plus 2 foot contour. In addition, the IM assumes that no other processes will occur that might affect shoreline position. Erosion and accretion processes are assumed to be absent.

Lidar elevation data collected by the USGS and under College of William & Mary domain was used for Mathews (2010) and the City of Hampton (2011) (www.wm.edu/as/cga/VALIDAR/). The original raster was referenced to NAD_1983_HARN_StatePlane_Virginia_South_FIPS_4502_Feet. This data layer was referenced to MHW using a script provided by NOAA. Lidar data provided high quality elevation data, high vertical and high spatial resolution (Gesch, 2009). These layers had a 0.47-0.73ft. vertical accuracy at a 95% confidence level.

Shoreline position by 2050 was determined by applying two accelerated sea level rise scenarios. The first accelerated scenario (ASC1) indicated a sea level increase of 700mm (or 0.35m by 2050 = 1.15ft.) by 2100 (Pyke et al., 2010). The second accelerated scenario (ASC2) forecasted an increase in sea level of 1600mm (or 0.8m by 2050 = 2.62ft.) by 2100. The scenarios were calculated using Lidar data and the Raster Calculator in ArcGIS. Each surface generated for each scenario was considered an IM. These surfaces were converted to a vector format as polygons. Using the 60m assessment buffer defined in Chapter 1, the IMs polygons were clipped to include just the inundation layer from each shoreline unit that was assessed.

The shoreline components digitized in Chapter 1 (Table 3.3) for shoreline units in Mathews and Hampton using current aerial photographs (i.e. 2007 and 2009) were used in this scenario (Refer to the Methods section in Chapter 1 for more information). The components at each shoreline unit were superimposed with the IMs and the area from shoreline components overlapping with the IMs were clipped off and deleted from the analysis. The loss in area from shoreline components represented the effects of land inundation generated by the two selected sea level rise scenarios. Shoreline components that were completely inundated at a site were not included for that particular site. Only
the area of the shoreline components that did not overlap with the IMs was used to run the Habitat and Water Quality Services Models. Ultimately, capacity was classified as described in Chapter 1 and as indicated in Tables 3.1-3.2.

**Management Practices: Scenario 2**

The goal for *Scenario 2* was to: 1) determine the most suitable locations for living shoreline and hybrid methods based on sea level rise scenarios and 2) determine the effects from living shoreline methods over shoreline components under future land inundation. To accomplish the first goal, shoreline units located at bank heights ≤1.5m were reassessed to identify units that are expected to be completely inundated under both scenarios. If both the Sc1 and Sc2 inundated the entire 60m assessment buffer that comprises the shoreline unit, then this site was classified as “Inundated” by 2050. Sites classified this way were considered unsuitable for living shoreline projects due to the low potential shoreline components will have to migrate landward as fast as sea level rises.

To accomplish the second goal for *Scenario 2*, the Shoreline Management Model from the CCRM (2012a, 2012b) was used. This model indicates the Shoreline Best Management Practice (BMP) along the entire shoreline of Mathews and Hampton for the time the model was generated. The BMP reflects the preferred method of the Commonwealth for shoreline stabilization using mostly natural habitats or living shorelines. The classification of the shoreline based on the BMP took into consideration sites characteristics such as bank height, fetch conditions, presence of armor structures, development, among others (Decision Tree, CCRM). Using this information, the current study was able to determine the best living shoreline method(s) suitable for each shoreline unit that was assessed. Based on the benefits or characteristics each living shoreline method offers, the effects on shoreline components were determined and specified in a summary table.
RESULTS AND DISCUSSION

POTENTIAL CAPACITY TO PROVIDE ECOSYSTEM SERVICES BY 2050: INFLUENCE OF LAND INUNDATION AND MANAGEMENT PRACTICES

**Mathews County: Scenario 1**

Based on the SCIM’s output the capacity to provide habitat and water quality services in Mathews County is expected to experience an important decrease by 2050 due to increasing water levels. Habitat and water quality services will be reduced in both sea level rise scenarios (Figures 3.3a-c and 3.4a-c). The number of shoreline units with moderate capacity will predominate in both scenarios and low capacity sites will increase with increasing sea level. Moderate and low capacity sites under Sc1 were characterized by high bank height conditions. Under Sc2 moderate capacity sites also showed mostly high bank heights, but low capacity sites were predominantly low bank heights. This indicates that with an increase of 0.35m (Sc1), most shoreline components such as tidal marshes and beaches at banks >1.5m in height will be inundated possibly due to their inability to migrate inland, but these same components may be able to adjust at low bank heights. However, under Sc2 the impacts will be inverted by reducing inundation at banks >1.5m in height due to the effects of land elevation, but low bank heights will undergo most of the impacts. High capacity sites will be almost absent under the Sc1 and completely absent under Sc2. Sites with the highest capacity by 2050 showed completely natural conditions mainly dominated by extensive marshes, forested lands and low bank heights.

Most of the impacts in capacity in both sea level rise scenarios will be observed at the east side and southern areas of the county where most of the lowest elevations and long fetch conditions are observed (Figures 3.5a-d). Areas with moderate capacity will be mostly found along the west and northwest regions of Mathews where land elevations are higher and fetch conditions are shorter.
By 2050 all natural and anthropogenic components are expected to experience a decrease in area based on the Sc1 and an even larger decrease under the Sc2 (Table 3.4). The loss in area from 2007 to 2050 for vegetation and other natural components was 39% on average under Sc1 and 56% on average under Sc2. More importantly, beaches, trees, tidal marshes and inland marshes showed the largest loss in area due to sea level rise. These components are essential for the provision of habitat and water quality services. This suggests that the loss of these components could define most of the changes in ecosystems capacity by 2050 if no action is taken to reduce their impacts.

**City of Hampton: Scenario 1**

The methods applied in the SCIM identified a similar trend, as observed in Mathews, in the impacts sea level rise will generate in Hampton's ecosystems capacity to provide services. By 2050, most ecosystems along Hampton's shoreline will experience a decrease in capacity to provide habitat and water quality. Most shoreline units will present low capacity to provide services under both sea level rise scenarios (Figures 3.6a-c and 3.7a-c). Even though shoreline units with high capacity were scarce or absent in Mathews under Sc1 and Sc2, most of the shoreline units were classified as moderate. In comparison to Hampton's capacity conditions by 2050, this indicates that future conditions in capacity are very likely to be higher in Mathews County.

Based on the model's output, shoreline units with high and moderate capacity were characterized by low bank heights primarily dominated by extensive marshes and forested lands. Low capacity sites were mainly units with high bank heights (i.e. 1.5-9.1m). The predominance of low capacity in shoreline units with high banks could be due to the loss of mainly intertidal and riparian vegetation and other natural components and an upland zone mainly dominated by anthropogenic activities.

As observed in Figures 3.8a-d, low capacity sites will be uniformly distributed along Hampton's shoreline under Sc1 and Sc2 and all high capacity sites will be located at the northern region of the City where most of the natural lands are located.
As determined for Mathews, all natural components in Hampton will lose thousands of square meters by 2050 (Table 3.5). This loss in area, especially in vegetation components, will trigger a decrease in ecosystems and ultimately in the capacity to provide services. In average, Hampton will experience a 47% loss in natural components under the Sc1 and a 60% loss under the Sc2. This averaged loss in shoreline components by sea level rise scenario is higher than determined for Mathews and reflects the low elevation conditions commonly observed in Hampton. Trees, tidal marshes and inland marshes are expected to experience most of the land inundation in the City. However, living shoreline methods could provide a solution to preserve most of these components.

Scenario 2: Best Management Practices

Studies have confirmed that the creation of living shoreline projects along estuarine shorelines can help reduce impacts from shoreline erosion and ultimately from sea level rise (Erdle et al., 2006). Based on this, the current study assumed that the placement of this type of management practice along Mathews’ and Hampton’s shoreline can be a solution to the expected land inundation and the loss in ecosystems capacity. Figures 3.9a-b indicates the different living shoreline methods per location necessary in Mathews and Hampton to minimize land inundation and to increase the sustainability of ecosystem services during the next decades. Most of the methods, if not all, will help improve or enhance the conditions specifically for beaches, marshes and riparian vegetation. As indicated earlier these components will be the most affected under Sc1 and Sc2 in these two localities. In addition, these are some of the most important components in providing services. Ultimately, the placement of the methods indicated in Figures 3.9a-b could provide the adequate conditions in shoreline systems to reduce land inundation or at least to allow ecosystems to adapt faster to changes in water level and to provide a higher capacity to provide services at each shoreline unit. In addition, based on the nature of land elevation and the conditions expected in sea level rise, only 12 shoreline units in Mathews and 15 shoreline units in Hampton were identified with
unsuitable conditions to place a living shoreline project (Figure 3.9a-b). These sites classified as "Inundated" (i.e. red dots) showed extremely low elevations and will be completely inundated under both sea level rise scenarios. In these specific cases, capacity will be absent by 2050. A total of 12 to 15 shoreline units by 2050 is a much lower number of units with no capacity than observed in Figures 3.3-3.4 and 3.6-3.7 or under Scenario 1, where no living shoreline method was considered. This suggests that by integrating living shoreline management practices in these two localities, approximately 90% to 93% of the shoreline units could have the potential to present moderate to high capacity to provide services by 2050. If living shorelines are not considered in future management plans it is expected that 32% to 35% of the units will mainly present a moderate capacity for habitat and water quality services.

A larger effort to create living shoreline projects along Mathews and Hampton’s shoreline should be considered. According to data available for these two localities, marshes are found along 270.8 miles of Mathews’ shoreline and 94.23 miles of Hampton’s shoreline from a total shoreline length of shoreline of 347 miles and 145 miles respectively (Tables 3.6a-b). This indicates that 78% to 65% of these localities’ shoreline is adequate for the placement of at least some type of living shoreline method that could help enhance or maintain marshes. These methods can benefit ecosystems along the entire shoreline in Mathews and Hampton generating a continuous flow of habitat and water quality services that could provide protection to coastal population and infrastructure located inland.

CONCLUSIONS

Capacity to provide habitat and water quality services will be reduced with a projected increase in sea level of 0.35m or 0.80m by 2050. Shorelines in Mathews County will be characterized by moderate to low capacity and Hampton’s ecosystems will mainly present low capacity for services. Both localities are expected to lose a considerable amount of vegetation and other natural components due to land inundation. A comparison of the losses between localities indicated that Hampton’s ecosystems will
experience a larger loss in area than expected for Mathews. This could be possibly due to the predominance of low elevation lands in Hampton. This trend suggests that capacity for habitat and water quality services by 2050 will be highly compromised mainly in the City of Hampton.

Beaches, marshes and forested lands were identified by the SCIM as the most impacted components due to land inundation under both sea level rise scenarios. These components provide essential services that not only sustain biodiversity and the filtration function, but also provide a buffer against shoreline change and protection to population and structures inland. Due to the importance of these natural features in shoreline systems, implementing the use of living shorelines in place of traditional shoreline armoring is becoming essential for shoreline stabilization and to preserve and maintain ecosystems capacity to provide services. This study’s analysis suggests that living shoreline practices could enhance ecosystem’s capacity for more than 90% of the shoreline units that were assessed.

Currently, shoreline armoring is used along many miles in Mathews and Hampton. However, these structures are expected to generate a larger adverse effect than a positive one in shoreline ecosystems. With the foreseen impacts in the Chesapeake Bay due to sea level rise, population growth and development expansion, a solution to the loss of ecosystem services is imperative. Living shorelines provide an alternative that needs to be considered at a county scale. By enhancing and maintaining ecosystems at a county level, shoreline ecosystems could provide a continuous flow of services and a stronger buffering mechanism to effectively dissipate effects from climate change, specifically from sea level rise, in the years to come.
Figures 3.1a-c  a. Map of the Chesapeake Bay, USA indicating the location of  b. Mathews (M) and  c. Hampton (H).
### Habitat Model Capacity Classification

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<thead>
<tr>
<th>Capacity</th>
<th>Maximum Score</th>
<th>Minimum Score</th>
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</thead>
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<tr>
<td>Moderate</td>
<td>22.07</td>
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<tr>
<td>Low</td>
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<td>5.00</td>
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</table>

*Table 3.1 HSM capacity classifications. Capacity classes were generated applying Jenks Natural Breaks method in GIS. The maximum and minimum scores represent the range of values for each capacity class.*

### Water Quality Model Capacity Classification

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<th>Capacity</th>
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<th>Minimum Score</th>
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<tr>
<td>Moderate</td>
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<td>Low</td>
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<td>7.00</td>
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*Table 3.2 Capacity classifications for the Water Quality Model generated by applying Jenks Natural Breaks method in GIS. The maximum and minimum scores represent the range of values for each capacity class.*
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<th>Assessment Zone</th>
<th>Component</th>
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<th>Water Quality Model</th>
<th>Categorical Values/Model Scores</th>
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<th>Minimum Score</th>
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</thead>
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<td>✓</td>
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<td>✓</td>
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<td>Long / 0.50</td>
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<td>Partial (25-75%)/2.00</td>
<td>Bare (&lt;25%)/1.00</td>
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<td>Vegetation composition</td>
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<td>✓</td>
<td>High (3 or more types vegetation) / 3.00</td>
<td>Low (1 or 2 types vegetation) / 2.00</td>
<td>None / 0.00</td>
</tr>
<tr>
<td>Upland Zone</td>
<td>Riparian land use</td>
<td>✓</td>
<td>✓</td>
<td>Natural / 3.00</td>
<td>Agriculture / 2.00</td>
<td>Developed / 1.00</td>
</tr>
<tr>
<td></td>
<td>Upland forested lands</td>
<td>✓</td>
<td>✓</td>
<td>Present / 3.00</td>
<td>Absent / 0.00</td>
<td>3.00</td>
</tr>
<tr>
<td></td>
<td>Upland forested marsh</td>
<td>✓</td>
<td>✓</td>
<td>Present / 3.00</td>
<td>Absent / 0.00</td>
<td>3.00</td>
</tr>
<tr>
<td></td>
<td>Upland land use</td>
<td>✓</td>
<td>✓</td>
<td>Natural / 3.00</td>
<td>Agriculture / 2.00</td>
<td>Developed / 1.00</td>
</tr>
</tbody>
</table>

Table 3.3 Shoreline components per assessment zone. The categorical values and model scores are specified for each component assessed for the HSM and/or WQM. The total maximum and minimum model score that each component can receive are indicated in the last two columns. The two bottom rows at the right indicate the possible maximum and minimum total model scores that shoreline units can receive by model.
Capacity To Provide Habitat Services (2050): Sc1

Habitat: Number of Sites Per Capacity (1968-2050)

- High
- Moderate
- Low

a. Capacity based on Sc1 for sea level rise
b. Capacity based on Sc2 for sea level rise

c. Graph indicating changes in the number of shoreline units per capacity from 1968 to 2050. The percent change is based on the difference in sites between 2007 and the scenarios. Map source: 2007 VBMP Imagery.
Figures 3.4a-c  Capacity of ecosystem services to provide water quality services by 2050 in Mathews County.  a. Capacity based on Sc1 for sea level rise b. Capacity based on Sc2 for sea level rise.  Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity.  c. Graph indicating changes in the number of shoreline units per capacity from 1968 to 2050.  The percent change is based on the difference in sites between 2007 and the scenarios.  Map source: 2007 VBMP Imagery.
Figures 3.5a-d  Prediction surfaces indicating capacity scores for habitat services in Mathews by 2050 under a. Sc1 and b. Sc2. c. Prediction surface indicating water quality services under Sc1 and d. Sc2.
### Table 3.4 Mathews. Changes in area (m²) and/or amount in shoreline components in the HSM and WQM for 2050 based on forecasted land inundation from Sc1 and Sc2. Changes are displayed by assessment zones and in percent change.

<table>
<thead>
<tr>
<th>Shoreline Component</th>
<th>2007</th>
<th>2008 Sc1</th>
<th>2008 Sc2</th>
<th>Δ in Area (m²)</th>
<th>Δ in Areas (m²)</th>
<th>Sc1 % Change</th>
<th>Sc2 % Change</th>
<th>Management methods to increase components' sustainability by 2050</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>BAV (m²)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>63,894.76</td>
<td>0</td>
<td>0</td>
<td>-63,894.76</td>
<td>-100.00</td>
<td>-45.80</td>
<td>-100.00</td>
<td>Likely absent due to increase in water depth.</td>
</tr>
<tr>
<td></td>
<td>Deffined shoreline (% of structure)</td>
<td>45.00</td>
<td>&gt;45.00</td>
<td>&gt;45.00</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Expected to increase with lower impacts through hybrid methods.</td>
</tr>
<tr>
<td></td>
<td>Beach (m²)</td>
<td>26,309.48</td>
<td>17,706.00</td>
<td>5,305.76</td>
<td>-9,602.76</td>
<td>-32.70</td>
<td>-31,103.02</td>
<td>Beach maintenance, sedimentation through plants and riparian buffers, including structures such as breakwaters.</td>
</tr>
<tr>
<td></td>
<td>Total marsh (m²)</td>
<td>116,248.17</td>
<td>54,280.13</td>
<td>61,967.45</td>
<td>-59,739.64</td>
<td>-51.41</td>
<td>-110,144.72</td>
<td>Enhance, maintain, and restore structures such as breakwaters with all.</td>
</tr>
<tr>
<td></td>
<td>Sediment (m²)</td>
<td>147.45</td>
<td>20.11</td>
<td>0.00</td>
<td>-128.35</td>
<td>-85.50</td>
<td>-447.50</td>
<td>-100.00</td>
</tr>
<tr>
<td></td>
<td>Vegetation composition (m²)</td>
<td>170,465.33</td>
<td>123,759.78</td>
<td>46,705.81</td>
<td>-66,723.75</td>
<td>-27.41</td>
<td>-85,054.32</td>
<td>Enhance and maintain vegetation cover through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Trees (m²)</td>
<td>1,234.64</td>
<td>83.00</td>
<td>216.00</td>
<td>-1,150.64</td>
<td>-92.70</td>
<td>-1,117.88</td>
<td>Enhance and maintain vegetation cover through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Sarah-shrubs (m²)</td>
<td>8,138.58</td>
<td>7,645.65</td>
<td>492.93</td>
<td>-1,532.63</td>
<td>-17.15</td>
<td>-4,718.93</td>
<td>Enhance and maintain vegetation cover through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Grass (m²)</td>
<td>23,222.44</td>
<td>23,641.72</td>
<td>127,607.97</td>
<td>-1,446.82</td>
<td>-6.58</td>
<td>-9,257.23</td>
<td>Enhance and maintain vegetation cover through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Intertidal marsh (m²)</td>
<td>20,599.79</td>
<td>2,454.92</td>
<td>4,044.77</td>
<td>-18,392.77</td>
<td>-86.90</td>
<td>-25,531.42</td>
<td>Enhance and maintain vegetation cover through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Riparian forested lands (m²)</td>
<td>153,825.48</td>
<td>90,541.20</td>
<td>64,284.84</td>
<td>-23,990.28</td>
<td>-22.79</td>
<td>-49,129.00</td>
<td>Enhance and maintain riparian buffer through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Riparian vegetation cover (m²)</td>
<td>170,465.33</td>
<td>123,759.78</td>
<td>46,705.81</td>
<td>-66,723.75</td>
<td>-27.41</td>
<td>-85,054.32</td>
<td>Enhance and maintain vegetation cover through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Riparian land use</td>
<td>726.00</td>
<td>724.10</td>
<td>1.90</td>
<td>-1.90</td>
<td>-0.27</td>
<td>-11.11</td>
<td>-1.27</td>
</tr>
<tr>
<td></td>
<td>Natural (m²)</td>
<td>170,665.33</td>
<td>123,759.78</td>
<td>46,905.81</td>
<td>-66,723.75</td>
<td>-27.41</td>
<td>-85,054.32</td>
<td>Enhance and maintain vegetation cover through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Agriculture (m²)</td>
<td>726.00</td>
<td>724.10</td>
<td>1.90</td>
<td>-1.90</td>
<td>-0.27</td>
<td>-11.11</td>
<td>-1.27</td>
</tr>
<tr>
<td></td>
<td>Developed (m²)</td>
<td>15,216.84</td>
<td>10,952.82</td>
<td>5,264.82</td>
<td>-5,964.12</td>
<td>-28.49</td>
<td>-7,286.13</td>
<td>-47.97</td>
</tr>
<tr>
<td></td>
<td>Vegetation composition (m²)</td>
<td>453,481.68</td>
<td>418,666.51</td>
<td>378,816.55</td>
<td>-34,835.17</td>
<td>-7.68</td>
<td>-80,463.25</td>
<td>Enhance and maintain riparian buffer through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Trees (m²)</td>
<td>2,655.30</td>
<td>71.10</td>
<td>94.70</td>
<td>-2,484.40</td>
<td>-97.22</td>
<td>-2,495.74</td>
<td>Enhance and maintain riparian buffer through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Sarah-shrubs (m²)</td>
<td>10,677.98</td>
<td>10,697.28</td>
<td>7,121.19</td>
<td>-10.70</td>
<td>-0.10</td>
<td>-3,356.79</td>
<td>Enhance and maintain riparian buffer through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Grass (m²)</td>
<td>123,814.30</td>
<td>123,667.33</td>
<td>110,283.71</td>
<td>-248.97</td>
<td>-0.20</td>
<td>-12,630.99</td>
<td>Enhance and maintain riparian buffer through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Intertidal marsh (m²)</td>
<td>40,603.54</td>
<td>9,249.27</td>
<td>1,749.76</td>
<td>-3,463.87</td>
<td>-77.27</td>
<td>-18,899.48</td>
<td>Enhance and maintain riparian buffer through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Upland forested lands (m²)</td>
<td>276,640.86</td>
<td>275,991.43</td>
<td>253,311.13</td>
<td>-949.43</td>
<td>-0.23</td>
<td>-23,328.71</td>
<td>Enhance and maintain riparian buffer through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Upland vegetation cover (m²)</td>
<td>453,481.91</td>
<td>418,666.51</td>
<td>378,816.55</td>
<td>-34,835.40</td>
<td>-7.68</td>
<td>-80,463.26</td>
<td>Enhance and maintain riparian buffer through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Upland land use</td>
<td>11,246.50</td>
<td>11,246.50</td>
<td>10,849.29</td>
<td>-3.91</td>
<td>0.00</td>
<td>-2,515.21</td>
<td>-18.82</td>
</tr>
<tr>
<td></td>
<td>Natural (m²)</td>
<td>453,481.91</td>
<td>418,666.51</td>
<td>378,816.55</td>
<td>-34,835.40</td>
<td>-7.68</td>
<td>-80,463.26</td>
<td>Enhance and maintain riparian buffer through planting of additional vegetation to increase vegetation cover and diversity.</td>
</tr>
<tr>
<td></td>
<td>Agriculture (m²)</td>
<td>11,246.50</td>
<td>11,246.50</td>
<td>10,849.29</td>
<td>-3.91</td>
<td>0.00</td>
<td>-2,515.21</td>
<td>-18.82</td>
</tr>
<tr>
<td></td>
<td>Developed (m²)</td>
<td>45,022.41</td>
<td>45,099.00</td>
<td>42,630.56</td>
<td>-122.71</td>
<td>-0.24</td>
<td>-3,201.75</td>
<td>-7.17</td>
</tr>
</tbody>
</table>
Figures 3.6a-c  Capacity of ecosystem services to provide habitat services by 2050 in Hampton. a. Capacity based on Sc1 for sea level rise b. Capacity based on Sc2 for sea level rise. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity. c. Graph indicating changes in the number of shoreline units per capacity from 1963 to 2050. The percent change is based on the difference in sites between 2009 and the scenarios. Map source: 2009 VBMP Imagery.
Figures 3.7a-c Capacity of ecosystem services to provide water quality services by 2050 in Hampton. a. Capacity based on Sc1 for sea level rise b. Capacity based on Sc2 for sea level rise. Green circles indicate high capacity, yellow is moderate capacity and red circles are sites with low capacity. c. Graph indicating changes in the number of shoreline units per capacity from 1963 to 2050. The percent change is based on the difference in sites between 2009 and the scenarios. Map source: 2009 VBMP Imagery.
Figures 3.8a-d Prediction surfaces indicating capacity scores for habitat services in Hampton by 2050 under a. Sc1 and b. Sc2. c. Prediction surface indicating water quality services under Sc1 and d. Sc2.
<table>
<thead>
<tr>
<th>City of Hampton</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Shoreline Component</strong></td>
</tr>
<tr>
<td><strong>Subaqueous Zone</strong></td>
</tr>
<tr>
<td>SAV (m²)</td>
</tr>
<tr>
<td>Defended shoreline (% of structure)</td>
</tr>
<tr>
<td>Beach (m³)</td>
</tr>
<tr>
<td>Tidal marsh (m³)</td>
</tr>
<tr>
<td>Saltmarsh (m³)</td>
</tr>
<tr>
<td><strong>Intertidal Zone</strong></td>
</tr>
<tr>
<td>Vegetation composition (m²)</td>
</tr>
<tr>
<td>Forest (m²)</td>
</tr>
<tr>
<td>Scrub-shrub (m²)</td>
</tr>
<tr>
<td>Grass (m²)</td>
</tr>
<tr>
<td>Animal marsh (m²)</td>
</tr>
<tr>
<td>Riparian forested lands (m²)</td>
</tr>
<tr>
<td>Riparian vegetation cover (m²)</td>
</tr>
<tr>
<td>Riparian land use</td>
</tr>
<tr>
<td>Natural (m²)</td>
</tr>
<tr>
<td>Agriculture (m²)</td>
</tr>
<tr>
<td>Developed (m²)</td>
</tr>
<tr>
<td><strong>Upland Zone</strong></td>
</tr>
<tr>
<td>Vegetation composition (m²)</td>
</tr>
<tr>
<td>Forest (m²)</td>
</tr>
<tr>
<td>Scrub-shrub (m²)</td>
</tr>
<tr>
<td>Grass (m²)</td>
</tr>
<tr>
<td>Animal marsh (m²)</td>
</tr>
<tr>
<td>Upland forested lands (m²)</td>
</tr>
<tr>
<td>Upland vegetation cover (m²)</td>
</tr>
<tr>
<td>Upland land use</td>
</tr>
<tr>
<td>Natural (m²)</td>
</tr>
<tr>
<td>Agriculture (m²)</td>
</tr>
<tr>
<td>Developed (m²)</td>
</tr>
</tbody>
</table>

**Table 3.5 Hampton. Changes in area (m²) and/or amount in shoreline components in the HSM and WQM for 2050 based on forecasted land inundation from Sc1 and Sc2. Changes are displayed by assessment zones and in percent change.**
Figures 3.9a-b Location and type of management practices suitable for each shoreline unit in a. Mathews and b. Hampton. Legend: I (Inundated), E/MM (Enhance/Maintain Marsh), WM (Widen Marsh), PMS (Plant Marsh with Sill), WM/EB (Widen Marsh/Enhance Buffer), ER/MB (Enhance Riparian/Marsh Buffer), ER/MB/BN (Enhance Riparian/Marsh Buffer or Beach Nourishment), E/MRB (Enhance/Maintain Buffer), E/MB (Enhance/Maintain Beach), MB/OBBN (Maintain Beach or Offshore Breakwaters with Beach Nourishment).
### Distribution of Shoreline Components in Mathews

<table>
<thead>
<tr>
<th>Component</th>
<th>Total kilometers</th>
<th>% of shoreline length</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marshes</td>
<td>434.52</td>
<td>78</td>
</tr>
<tr>
<td>Beach</td>
<td>46.67</td>
<td>8</td>
</tr>
<tr>
<td>Riparian buffer</td>
<td>268.34</td>
<td>48</td>
</tr>
</tbody>
</table>

*Mathews’ shoreline total length = 559.04 km

Table 3.6a-b Total shoreline length where marshes, beaches and riparian buffer (i.e. forested lands and scrub-shrubs) are present in a. Mathews and b. Hampton.

### Distribution of Shoreline Components in Hampton

<table>
<thead>
<tr>
<th>Component</th>
<th>Total kilometers</th>
<th>% of shoreline length</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marshes</td>
<td>151.65</td>
<td>65</td>
</tr>
<tr>
<td>Beach</td>
<td>19.02</td>
<td>8</td>
</tr>
<tr>
<td>Riparian buffer</td>
<td>71.42</td>
<td>31</td>
</tr>
</tbody>
</table>

*Hampton’s shoreline total length = 234.13 km
LITERATURE CITED


VITA

Cielomar Rodríguez-Calderón