ABSTRACT

Title of Dissertation:	EVALUATING RESTORATION POTENTIAL AND STORM SURGE ATTENUATION IN DITCHED AND UNDITCHED COASTAL MARSHES
	Dorothea June Lundberg, Doctor of Philosophy, 2016
Dissertation directed by:	Associate Professor, Karen Prestegaard, Department of Geology

The effects of ditching on the hydrological regime and ecosystem services of ditched coastal marshes—as well as the effects of hydrologic restoration of these systems—have yet to be extensively studied. The goals of this project were (1) to determine differences between ecohydrological processes in Ditched and Unditched coastal marshes, (2) to determine the effects of ditch plugging restoration projects on Atlantic Coast and Chesapeake Bay marsh hydrology, and (3) to evaluate Hurricane Sandy storm surge in the coastal marshes. Two separate pairs of Ditched and Unditched marshes were used in this study. The paired sites were adjacent, with similar topography, vegetation, and tidal patterns. Data collection included hydrological properties such as ditch density, tidal stage, water table fluctuations; as well as soil properties. Soil properties were similar in Ditched and Unditched marshes, while ditched marshes had lower water table elevations than Unditched marshes. Ditch

plugging restoration partially restored the hydrological regime. A comparison of Chesapeake and Atlantic coastal marshes during Hurricane Sandy indicated similar storm surge elevations, but shorter durations of inundation at the Chesapeake Bay marshes when compared with the Atlantic marshes.

EVALUATING RESTORATION POTENTIAL AND STORM SURGE ATTENUATION IN DITCHED AND UNDITCHED COASTAL MARSHES

by

Dorothea June Lundberg

Dissertation submitted to the Faculty of the Graduate School of the
University of Maryland, College Park, in partial fulfillment
of the requirements for the degree of
Doctor of Philosophy
2016

Advisory Committee:
Dr. Karen Prestegaard, Chair
Dr. Kaye Brubaker
Dr. Keith Eshleman
Dr. Gary Felton
Dr. Sujay Kaushal
Dr. Brian Needelman (non-voting committee member)

Dedication

This Dissertation is dedicated to my parents, Douglas Sr. and Carol Lundberg, who have always been by my side, my biggest supporters throughout my life, and through graduate school's ups and downs. Thank you for your unwavering support, love, and encouragement over the years.

Acknowledgements

I would like to express my deep appreciation and gratitude to my advisor, Karen Prestegaard, for the guidance and mentorship she has provided to me from my Master's project to the completion of this Ph.D. degree in the Marine Estuarine Environmental Science Program. Dr. Prestegaard's intelligence is matched only by her passion for hydrology and geomorphology, her dedication to educating amazing scientists, and her support of women in the scientific field. I am truly fortunate to have had the opportunity to work with her, and I am proud to follow in her footsteps as a hydrologist and geomorphologist.

I would also like to thank my committee members Drs. Kaye Brubaker, Keith Eshleman, and Gary Felton for the encouragement, guidance, thought-provoking suggestions, and general collegiality that each of them offered to me over the years. You all have gone above and beyond, participating more than most committees, and I greatly appreciate it. I have learned so much from all of you, and felt like my committee was always invested in my success. I would also like to thank Brian Needelman for the opportunity to work on such a ground breaking project in hydrology, and supporting me my first few years of the project. Lastly, I would like to thank Dr. Sujay Kaushal for coming onto the project so late in the process and being a helpful member of my defense committee.

Table of Contents

Dedication ii
Acknowledgements iii
Table of Contents iv
List of Tables vii
List of Figures viii
List of Abbreviations xiii
Chapter 1: Introduction and Project Overview with Site Descriptions 1
Statement of the problem: Hydrologic consequences of Ditch-Drained Marshes 1
Objectives 3
Previous Research 4
Coastal Marshes 4
Marsh Alterations 6
Ditched Marshes 7
Ditch networks and their consequences to plant communities 9
Ditch Marsh Restoration 11
Climate Change and Storm Surges 19
Project Design 21
Selection of Study Marshes and Site Descriptions 23
Chesapeake Sites 23
Atlantic Coast Marshes 23
Approach: Hydrological Data collection to address research questions 26
Organization of the dissertation 27
Chapter 2: Soil hydraulic properties and tidal amplitude attenuation in ditched and un-ditched coastal marsh peat aquifers 28
Objectives and Hypotheses: 31
Study Sites and Methods 32
Hydrological Monitoring 33
Marsh sediment characterization 38
Statistical Analyses 40
Results: 41
Chesapeake Sites 41
Soil Texture and Hydraulic Properties 47
Atlantic Sites 56
Soil Hydraulic Properties 62
Statistical Comparison of soil properties among ditched and unditched marshes at both Atlantic and Chesapeake sites 70
Discussion 71
Chapter 3: Impact of ditches on Eastern Shore Maryland Salt Marsh Groundwater levels and Plant Species Composition 75
Introduction 75
Objectives and Hypotheses: 76
Methods 76
Vegetation Cover 77
Table 3.1: 78

Hydrological Monitoring 78
Results: 79
Chesapeake Sites 79
Atlantic Sites 103
Salinity characteristics 106
Hydrologic Behavior of Atlantic Ditched and Unditched marshes 109
Statistical Comparison of salinity between ditched, unditched, Atlantic and Chesapeake Marshes 118
Discussion 119
Chapter 4: Changes in marsh hydrology due to Hurricane Sandy Storm surges in Ditched and Unditched Atlantic and Chesapeake Coastal Marshes 124
Introduction 124
Purpose of this paper 127
Storm Track of Hurricane Sandy 127
Study sites 130
Atlantic Coast Sites- 130
Chesapeake Bay Sites- 131
Methods: 131
Results 134
Atlantic Coast Sites: 134
Storm Track, rainfall, wind directions, and tidal surge during Hurricane Sandy 134
Response of Marsh Water Levels during Sandy 137
Chesapeake Bay Sites: 143
Storm Track, rainfall, wind directions, and tidal surge during Hurricane Sandy 143
Marsh Water Levels during Sandy 146
Discussion 152
Chapter 5: Evaluation of hydrological consequences of ditch plugging restoration projects 154
Introduction 154
Ditch Plugging as a restoration practice: 156
Methods and Study Sites 157
Hydrological Monitoring 157
Results: 160
Chesapeake Bay Marshes 160
Salinity characteristics 162
Hydrologic Behavior of restored ditched and unditched marshes 163
Atlantic Sites 167
Salinity characteristics 169
Hydrologic Behavior of restored ditched and unditched marshes 170
Summary: 174
Comparison of Atlantic and Chesapeake Marshes 174
Discussion: 176
Conclusion: 178
Implications: 179

Appendices.. 182
Well Construction:.. 185
Well Calibration:... 186
References.. 187

List of Tables

Table 1.1: Marsh sample sites information table. 37
Table 1.2: Timeline of sampling events 44
Table 2.1: Mean tidal range (cm) for Chesapeake Bay Sites 63
Table 2.2: Chesapeake Sites BD and K 67
Table 2.3: Summary of Tidal range (cm) for Atlantic Bay Sites 78
Table 2.4: Summary of Tidal Ranges and Amplitudes 79
Table 2.5: Atlantic Sites BD and K 86
Table 2.6: Statistical Analysis – Soil Properties 89
Table 3.1: Vegetation Composition top three species 96
Table 3.2: Vertical Hydraulic Gradients for Chesapeake Ditched and Unditched Site 99
Table 3.3: Vegetation Composition top three species 106
Table 3.4: Percent time at or above ground surface for each well for ditched and unditched sites with respect to location. 123
Table 3.7: Some previous research - effects from ditching in marshes 136
Table 4.1: Annual Climatological Summary, National Centers for Environmental Information. 140
Table 4.2: Storm Surge Duration at maximum (hours) 162
Table 4.3: Annual Climatological Summary, National Centers for Environmental Information. 169
Table 4.4: Duration at storm surge maximum (hours) NOTE: Transect 1 is closest to tidal source 172
Table 5.1: Pre- (2013, 2014) and Post- (2014) restoration tidal range 185
Table 5.2: Post-restoration tidal range Atlantic sites 193
Table 5.3: Site and parameter 200

List of Figures

Figure 1.1: Photo of a linear, narrow ditch typical of ditches in Maryland coastal marshes, E.A. Vaughn Wildlife Management Area 19
Figure 1.2: Ditch located at the Deal Island Wildlife Management Area 25
Figure 1.3: Photo showing a recently installed ditch plug. View of the tidal creek in back of photo. Photo courtesy of Donald Webster 27
Figure 1.4: Marsh vegetation changes at Wertheim National Wildlife Refuge, NY, from before hydrologic restoration (2004) and 4 years post hydrologic restoration (2008). The overall percent cover (statistically significant) of native species increased, while the invasive species *Phragmites australis* declined (Rochlin 2012). 28
Figure 1.5: Water table levels and surface levels before ditching (late 1930s) and after ditch restoration (1938) in Delaware (Stearns et al. 1940). 31
Figure 1.6: (a) Water level elevations of unrestricted marsh versus tide-restricted marsh before restoration during one tidal cycle (b) Water elevation of unrestricted marsh versus tide restricted marsh after restoration during several tidal cycles. Marshes located Sachuest Point Salt Marsh (Middletown, RI, U.S.A.) (Roman 2002). 32
Figure 1.7: Water table level, relative to the marsh surface (at 0.0cm) along a transect comparing a free flowing ditch and a plugged ditch. (Adamowicz 2002) 34
Figure 1.8: (Top) Deal Island ditched site showing common reed, black needlerush, and cordgrasses; (Bottom) E.A. Vaughn Wildlife Management area showing cordgrasses. 39
Figure 1.9: (Top) Site map showing the location of the study sites in Maryland; (Middle) Chesapeake Sites with ditched and unditched sites (Bottom) Atlantic sites ditched and unditched sites. Wells are shown as solid (marsh) and open (ditches) circles. Site maps show ditch tracks and their spacing. 42
Figure 2.1: Conceptual diagram showing the attenuation of tidal range in an unconfined aquifer (from Todd, 1980) 48
Figure 2.2: Array of transects and well locations at the Atlantic ditched (top) and Atlantic unditched sites (bottom). A similar layout was used at the Chesapeake sites. 52
Figure 2.3: Transect at Chesapeake site showing monitoring well (front) and a set of 3 nested short-interval wells. Tall pole at the back is the midpoint of a circular vegetation sampling plot. 53
Figure 2.4: Conceptual drawing of monitoring well and mini wells layout 56
Figure 2.5: Example of falling head data used to determine hydraulic conductivity. (a) Depth to water vs time, (b) H/H0 versus time, T_0, is defined as time when H/H0 = 0.37 59
Figure 2.6: An example of a 15-day time period showing the variations in tidal amplitude associated with spring (red) and neap (blue) tides. 61
Figure 2.7: Figure 2.7: Tidal range, cm, for the 2 year period (2012-2014) for: (A) Chesapeake unditched site; (B) Chesapeake ditched site 64

Figure 2.8: (A) Tidal range, cm, plotted against distance away from the tidal creek (m) for Chesapeake ditched site and unditched site (B) Interior marsh wells tidal range, cm, plot against distance away from ditch 65
Figure 2.9: Soil Stratigraphy for Chesapeake Ditched and Unditched sites. Data indicate higher peat depths in the unditched sites than the ditched sites 66
Figure 2.10: Bulk Density for Chesapeake Bay Site Ditched and Unditched by site and by depth 68
Figure 2.11: Probability density curves for bulk density for Chesapeake Bay Site Ditched and Unditched by depth 69
Figure 2.12: Bulk Density for Chesapeake Bay Site vs Distance and Elevation. Bulk density does not vary with distance from the creek or site elevation 70
Figure 2.13: Hydraulic conductivity for Chesapeake Ditched site. (A) By Site, (b) By Depth. Data indicates higher hydraulic conductivity closer to the ground surface 72
Figure 2.14: Hydraulic conductivity for Chesapeake Ditched site versus distance and elevation 73
Figure 2.15: Tidal range for 2 years during pre-restoration period for (a) Atlantic Unditched site and (b) Atlantic Ditched site, (c) NOAA Ocean City Tide gage 75
Figure 2.16: (A) Tidal range, cm, plotted against distance away from the tidal creek (m) for Atlantic ditched site and unditched site (B) Interior marsh wells tidal range, cm, plot against distance away from ditch 77
Figure 2.17: Example of a soil core extracted from the site. 80
Figure 2.18: Soil Stratigraphy for Atlantic Ditched and Unditched sites. Data indicate that the upper 75 cm of the marsh sediment is predominately peat 81
Figure 2.19: Bulk Density for Atlantic Bay Site Ditched and Unditched by site and by depth 83
Figure 2.20: Probability density curves for bulk density for Atlantic Bay Site Ditched and Unditched marsh soils separated by depth intervals 84
Figure 2.21: Bulk Density for Atlantic Bay Site vs Distance and Elevation. Bulk density does not vary with distance from the creek or site elevation 85
Figure 2.22: Hydraulic Conductivity for Atlantic Bay Sites. (A) K by site and (B) K by depth 87
Figure 3.1: Vegetation plot layout and sampling area 96
Figure 3.2: Percent cover of the dominant species at the Chesapeake Ditched and Unditched sites versus distance from tidal creek and ground elevation 98
Figure 3.3: Salinity by site and by sample date for Chesapeake Ditched and Unditched Sites 101
Figure 3.4: Salinity for Chesapeake Ditched and Unditched Sites versus distance from tidal creek and ground elevation 102
Figure 3.5: Precipitation data for the study period: upper diagram is the cumulative precipitation, which indicates precipitation distributed throughout the year, but a significant step for Hurricane Sandy. Lower diagram indicates individual storms. Note the larger storm size for the summer months: July through November 104
Figure 3.6: Time series of Unditched site (left) time frame of 2/1/2012 to 9/1/2013 and Ditched site (right). No distinct seasonal pattern observed at either site. Datum is depth to ground (DTG). Transect 3 is the back of the site; Transect 1 is located closest to the tidal creek 109

Figure 3.7: (A) Unditched site showing slight seasonal difference in well 2 (unditched well); (B) Chesapeake Ditched site showing slight seasonal difference in wells 3, 6, and 25 for the ditched site. Boxplots using depth to ground surface as the datum indicate whether the marshes were inundated. FW= Fall/Winter; SS = Spring/Summer; # = well number 111
Figure 3.8: (A) Chesapeake Unditched Site; (B) Ditched site. Boxplots constructed using NAVD88 datum indicated relative groundwater elevation data, which control groundwater flow directions. FW= Fall/Winter; SS = Spring/Summer; # = well number 112
Figure 3.9 Head data for the ditched site. A: Hydraulic head data for the center ditch and interior marsh B) Head difference values between the center ditch, which indicates continuous groundwater flow towards the ditch 113
Figure 3.10: Head difference between center ditch along transect 1 (closest to tidal creek) and marsh interiors on random, selected dates. These data also indicate drainage towards the ditch. 114
Figure 3.11: Cumulative curves showing percent time at or below the ground surface (indicated as 0), for (A) Unditched (B) Ditched sites. Negative values indicates water levels are below ground. 117
Figure 3.12: Percent of time that water level is at or below the surface as a function of a) a tidal creek and b) the proximal ditch. 118
Figure 3.13: 10th and 90th percentile on cumulative probability distributions water levels versus distance (m) 119
Figure 3.14: 10th and 90th percentile on cumulative probability distributions water levels versus ground elevation (m) 120
Figure 3.15A: Percent cover of the dominant species at the Chesapeake Ditched and Unditched sites versus percent saturation 121
Figure 3.15B: Percent cover of the dominant species at the Chesapeake Ditched and Unditched sites versus salinity 121
Figure 3.16: Percent cover of the dominant species at the Atlantic Ditched and Unditched sites versus distance from tidal creek and ground elevation 124
Figure 3.17: Salinity by site and by sample date for Atlantic Ditched and Unditched Sites 126
Figure 3.18: Salinity for Atlantic Ditched and Unditched Sites versus distance from tidal creek and ground elevation 127
Figure 3.19: Time series of Unditched site (A) time frame of 4/1/2012 to 12/31/2013 and Ditched site (B) time frame of 12/12/2011 to 12/31/2013. No distinct seasonal pattern 131
Figure 3.20A: Boxplots showing seasonal differences between wells and sites. (A) Unditched Site (B) Ditched Site on the Atlantic Coast. Relative to ground is to determine flooding and saturation. SS= spring/summer; FW = fall/winter 132
Figure 3.20B: Boxplots showing seasonal differences between wells and sites. Water level relative to datum is to determine groundwater flow directions SS= spring/summer; FW = fall/winter 132
Figure 3.21: Head difference between center ditch and marsh interior. (B) Head difference between the mars interior and the tidal creek. Data indicate drainage head distributions except for during major storm events. 134

Figure 3.22: Water level cumulative probability curves showing percent time at or above the ground surface at 0 135
Figure 3.23: Percent cover of the dominant species at the Atlantic Ditched and Unditched sites versus percent saturation 137
Figure 3.24: Percent cover of the dominant species at the Atlantic Ditched and Unditched sites versus salinity 138
Figure 4.1: (A) Hurricane Sandy Track. Source: National Hurricane Center (NHC) Sandy Report (B) Satellite image of Hurricane Sandy Source: NOAA. Black star shows location of field sites 150
Figure 4.2: Site maps showing ditched and unditched sites at the Chesapeake Bay sites. Note well locations, ditch spacing, orientation, and tidal creek 154
Figure 4.3: Comparison of NOAA tide gage at Ocean City Maryland versus USGS tide gage at Chincoteague. Research Site location is in between both gages on Chincoteague Bay (Figure 1.9) 155
Figure 4.4: (a) Precipitation. Total accumulation over 3 days was 22.5 cm, (b) Maximum wind was 65 km/hr; wind direction shifted near the peak storm; (c) Tidal stage at Ocean City compared to predicted gauge height. Storm Surge shown in red (peak is 122 cm greater than the expected tidal stage) 158
Figure 4.5: (A) Ditched Site- Time series showing water levels in wells located in the ditches and in the interior portions of the marsh. (B) Unditched Site- Time series showing water levels in wells located in the unditched wells and the tidal creeks. Note the duration at maximums and differences in the maximums levels 161
Figure 4.6: Time series showing water levels in wells located in the ditches and in the interior portions of the marsh for the Ditched Site, and time series showing water levels in wells located in the unditched wells and tidal creeks for a lunar high tide and Hurricane Sandy 164
Figure 4.7: Hysteresis graphs for Ditched and Unditched Sites. See figure in appendix for comparison between sandy, full moon, and second biggest storm 165
Figure 4.8: Probability curve water levels at or above the ground surface for unditched site for three time periods: Annual 2012 dataset with Sandy, Annual 2012 when Sandy subset data was removed, and the Sandy subset data 166
Figure 4.9: (a) Precipitation. Total accumulation over 3 days was 25 cm, (b) Maximum wind was 65 km/hr; wind direction shifted near the peak storm; (c) Tidal stage at Bishop Head, NOAA tide gage 168
Figure 4.10: Time series of sites and transects. (A) Unditched Site- Time series showing water levels in wells located in the unditched wells and the tidal creeks. (B) Ditched Site- Time series showing water levels in wells located in the ditches and in the interior portions of the marsh…. 171
Figure 4.11: Maximum water levels on the NOAA tide gage and maximum marsh water levels for Ditched and Unditched Sites. 173
Figure 4.12: Hysteresis graphs for Ditched and Unditched Sites. 174
Figure 5.1: (A) Chesapeake Site (B) Atlantic Site with locations of wells and ditches 182
Figure 5.2: Photo of a ditch plug at the Atlantic site facing the tidal creek 183
Figure 5.3: Tidal range, NAVD88 datum during Post-restoration period for Chesapeake Unditched site and Chesapeake Ditched site 185

Figure 5.4: Salinities during post-restoration period indicate similar average values for the restored ditched and unditched marshes. 187
Figure 5.5: Time series of Post-restoration water level data for ditched site and unditched Chesapeake Bay sites 189
Figure 5.6: Boxplots showing pre- and post-restoration differences between wells and sites in ditched and unditched sites. Unditched sites are wells labeled with a "U" on x axis. 190
Figure 5.7: Water level cumulative probability graphs showing post-restoration (left) and pre-restoration (right) cumulative probability distributions of ditched and unditched wells 191
Figure 5.8: Tidal range, NAVD88 datum during Post-restoration period for Atlantic Unditched site and Atlantic restored Ditched site 192
Figure 5.9: Salinity during post-restoration period for the restored ditched site and the unditched reference site. Salinity collected on 6/18/2014 and 7/1/2014 combined for each site. 194
Figure 5.10: Time series of pre- and post-restoration water level data for ditched site and unditched Atlantic sites 196
Figure 5.11: Boxplots showing pre- and post-restoration water levels wells and sites in ditched and unditched sites. Unditched sites are wells labeled with a "U" on x axis. 197
Figure 5.12: Cumulative curves showing pre- and post-restoration differences between wells and sites in ditched and unditched sites. 198

List of Abbreviations

EAV = EA Vaughn Wildlife Management Area = Atlantic Sites

DEAL = Deal Island Wildlife Management Area = Chesapeake Sites

K = Hydraulic conductivity

m = meters

NOAA = National Oceanic Atmospheric Administration

OMWM = Open Water Marsh Management

r = radius of the well screen

L = length of well screen

T_0 = time required for the water level to rise to 37% of the initial change

Sy = Specific yield

Chapter 1: Introduction and Project Overview with Site Descriptions

Statement of the problem: Hydrologic consequences of Ditch-Drained Marshes

Hydrology is a primary control on the spatial organization of plant species, and associated ecological functions in wetland systems (Foti et al. 2012). Tolerance for saturated and/or reducing conditions varies widely among plant species, therefore, modifications to the hydrologic regime can significantly affect plant species composition and changes in marsh ecological functions. Ditches are a wide-spread coastal marsh modification that can modify hydrological processes (Figure 1.1). Ditching of coastal marshes in the U.S. began in the 1700s with the goal of increasing *Spartina patens* acreage for salt marsh hay (Daiber 1986). In 1912, extensive ditching of coastal marshes began in New Jersey to drain intermittent pools, which are the preferred breeding habitat for salt marsh mosquitoes. Marsh ditching was originally restricted to low-lying metropolitan areas until 1933, when it was expanded using relief labor during the Great Depression to provide acreage for salt marsh hay. By 1938, mosquito control ditching programs comparable to the New Jersey program had begun in other coastal states, bringing the total area of marshlands ditched along the Atlantic Coast to approximately 560,000 acres, which represents 90% of the original marsh area between Maine and Virginia. Although ditch programs were often initiated to drain mosquito ponds, early studies indicated limited evidence that ditches were effective in reducing mosquito production (Bourn and Cottam 1950). However, these ponds were able to support small fish.

In addition to their ecological importance, coastal marshes also provide protection of coastlines and human structures from inundation and erosion during extreme storms by attenuating storm surges (Temmerman et al. 2013). Storm surges are among the most damaging and dangerous phenomena in coastal communities, causing significant erosion, morphological change, and disturbances within these ecosystems. The effects of hurricane-associated damage on coastal ecosystems can be extensive even with the mitigating effect of coastal wetlands (Sheikh 2005). Damage to the surrounding marshes produced by hurricanes also depends on storm surge heights and associated water depths at the time of maximum wind stress and storm inundation (Morton and Barras 2011).

Although there have been studies of the effects of ditches on salt marshes, few of these studies have focused on changes in marsh hydrology. Studies of the hydrological effects of ditches were primarily performed immediately following ditch construction. In addition, there have been few field studies of the effects of storm surges on ditched marshes in comparison to unditched marshes. The research project presented here was part of a larger project designed to measure and compare hydrological, soil, mosquito, nekton, and vegetation characteristics in unditched and ditched marshes before and after ditch plugging restoration projects were conducted.

Figure 1.1: Photo of a linear, narrow ditch typical of ditches in Maryland coastal marshes, E. A. Vaughn Wildlife Management Area.

Objectives

This work focused on marsh hydrological processes through a comparison of soil hydraulic characteristics, salinity, and marsh water levels relative to both drainage ditches and the ground surface in ditched and unditched marshes. An additional goal was to determine the response of marsh water levels to meteorological, tidal, and storm surge events.

The main objectives of this research were:

1) To determine if there were differences in marsh hydraulic characteristics (hydraulic conductivity, bulk density, and specific yield) in ditched and unditched microtidal coastal marshes.
2) To determine whether there were significant differences in marsh water levels between ditched and unditched marshes. If differences were found, a sub-objective was to determine whether these differences were driven by drainage or other mechanisms. The relationships of marsh water levels to plant species composition were also examined.
3) To determine the effect of storm surges on marsh hydrology in ditched and unditched marshes. This evaluation included a comparison of the magnitude and duration of storm tide elevations and duration in ditched and unditched marshes located on Chesapeake Bay and along the Atlantic Coast.
4) To determine the effects of ditch plugging restoration on marsh water levels.

Previous Research

Coastal Marshes

Salt marsh habitats are found at nearly all latitudes (Costa & Davy 1992). Coastal marshes are productive and valuable habitats along the Atlantic Coast, covering approximately 5.8 million acres on the East Coast from Maine to Florida (Stedman 2008). These coastal marshes represent the interface between terrestrial and marine environments, therefore, their ecosystem functions are at risk due to changes associated with rising sea levels that may include erosion, sea-water intrusion, and submergence of coastal marshes.

Brackish water and salt marshes perform essential ecosystem services and functions valuable to society and the surrounding ecosystem such as regulation, habitat, and production (Daily et al. 1997; de Groot et al. 2002). Marsh functions relate to the ecosystem's ability to regulate essential ecological processes such as biogeochemical cycles (de Groot et al. 2002), while habitat functions include providing refuge and breeding grounds for plants and animals. These functions contribute to the conservation of biological and genetic diversity and evolutionary processes within coastal ecosystems (de Groot et al. 2002). Production functions in coastal marshes consist of photosynthesis and nutrient uptake by autotrophs, which then convert energy, carbon dioxide, water, and nutrients into a wide variety of carbohydrate structures. These structures are used by secondary producers to create an even larger variety of living biomass (de Groot et al. 2002).

Hydrologically, marshes act as buffers for the mainland by slowing and absorbing storm surges, thereby reducing coastline erosion. They provide valuable habitat for hunting, crabbing, fishing, and heritage. They are a major producer of detritus, and provide nursery grounds for numerous commercially and recreationally valued species. In addition, brackish marshes serve to remediate and filter nutrients, sediment, and toxins.

Marsh Alterations

Review of the literature indicates that salt marsh alterations can be documented as far back as the 1600s, when marshlands were used for cattle ranching. The first documented examples of marsh ditching date back to the 1700s (Shisler 1990). Coastal marshes have been directly altered through diking, constructing impoundments, ditching, dredging, and Open Marsh Water Management (OMWM). In addition, marshes can be indirectly altered through hydrologic changes within adjacent watersheds (i.e. increase in urbanization or impervious surfaces). These structural changes can affect the marsh hydroperiod (as measured by water depths, flood durations, flood frequencies, and the spatial extent of open water), while hydrological changes can impact plant communities, ecological structure, and sediment processes (e.g., erosion and deposition of sediment and biogeochemical processes that may include acid sulfate soil formation, organic carbon oxidation, and other processes that affect water quality). Changes in hydrological and water quality can, in turn, affect nekton, and semiaquatic invertebrates (Bourn and Cottam 1950; Roman et al. 1984; Portnoy 1991; Allen et al. 1994; Turner 1997; Anisfeld & Benoit 1997; Anisfeld et al. 1999; Portnoy 1999; Raposa & Roman 2001; Gedan et al. 2009). Hydrologic alterations that restrict lateral movement of surface and/or groundwater or prevent natural marsh flooding regimes may stress wet-adapted species and limit the flux of resources into and out of marshes, thereby limiting the marsh function and ecosystem structure (Swenson and Turner 1987; Reed et al. 1997). Marsh degradation and loss through submergence is a serious concern for people living and working in coastal communities, as well as adjacent terrestrial watersheds.

Ditched Marshes

Of the various human modifications, the most common and extensive anthropogenic change is salt marsh ditching. It has been estimated that 90% of the Atlantic Coast marshes (approximately 560,000 acres) between Maine and Virginia are ditched (Bourn and Cottam 1950). Ditches were designed to drain ponds that were the habitat of the salt marsh mosquito, *Ochlerotatus sollicitans* (also known as *Aedes sollicitans*), which, according to Smith 1904, was such a fierce biting mosquito that it is said to have stalled development along the Atlantic Coast. There is, however, limited evidence that ditches reduce mosquito production. Many sites have also been hydrologically altered with Open Marsh Water Management (OMWM), which is a system designed to decrease mosquito production while maintaining marsh condition. Open Marsh Water Management began in New Jersey during the 1960s (Weis and Butler 2009). The OMWM method consisted of installation of small, shallow ponds and inter-connecting ditches in known mosquito-breeding habitats. These new larger ponds, in combination with the elimination of pothole breeding habitats, created permanent water habitats that are unattractive for mosquito egg deposition while simultaneously improving habitats for mosquito-eating larvivorous fishes.

In general, ditched marshes have more channel edge area, less interior marsh area, and fewer shallow ponds than unaltered marshes. Shallow ponds are critical habitats for ducks, wading birds, shellfish, and fish—including those that eat mosquito larvae. Plant species composition has been documented to change in response to ditching (e.g. Bourn and Cottam 1950; Roman 1995). Previous studies suggest that ditches can lower marsh water tables (Stearns 1940; Bourn and Cottam 1950; Adamowicz 2005; Gedan 2009), but recent studies suggest that long-term water level lowering does not occur in all settings (Vincent 2013). The study by Vincent (2013) indicates that moderate ditching (>30 m between ditches) has minor long-term impact on marsh water levels, soil characteristics, and marsh surface elevations. Vincent (2013) study also found that soil accretion processes in ditched marshes were comparable to those observed in unditched marshes. He noted, however, that ditch spacing is an important parameter, and subsidence rates and pore-water retention may be more significantly altered in marshes with more closely spaced grid ditches (Vincent 2013).

These studies suggest that the size and spacing of ditches may affect hydrological consequence. The ditches originally dug by hand in the Mid-Atlantic region have been reported to be approximately 0.4 m wide and 0.6 m deep (Pincus 1938; Williamson 1951). Currently, these ditches average 1-1.5 meter wide by 1.2 - 1.8 meters deep (Figure 1.2). Chesapeake Bay marsh sites, like other sites in the northeast, have a slight berm from the ditch spoil being deposited immediately adjacent to the ditches (Miller and Egler 1950). In many regions, there has been no ditch maintenance following ditch construction.

Figure 1.2: Ditch located at the Deal Island Wildlife Management Area.

Ditch networks and their consequences to plant communities

Marsh drainage ditches were typically hand dug, extending from a main tidal creek into the high marsh. In the Mid-Atlantic region, ditches are spaced approximately 40 meters apart, typically measuring 0.4 meters wide and 0.6 meters deep (Pincus 1938). Although ditching does not destroy the salt marshes, it does change the abundance of certain fauna and flora. Plant species composition was shown to change in response to ditching (e.g. Bourn and Cottam 1950; Niering and Warren 1980) and the associated restriction of tidal flow (e.g. Roman et al. 1984, 1995). Ditches can effectively increase the drainage capacity of marshes—which may reduce the duration of inundation in interior portions—by containing surface water and groundwater through an area of increased channel density. In some ditched marshes, the marsh becomes drier, which allows less salt- or flood-tolerant species to flourish. Restriction of tidal flow often results in conversion of Spartina-dominated marsh areas to *Phragmites australis* (Burdick et al. 1997). In other cases, ditches alter flooding and drainage patterns that cause marshes to become more frequently flooded than unditched marshes, leading to an increase in salinity and increasing species tolerance of these conditions. In some wetter high marshes where pannes were often dominated by the native flora, *Spartina alterniflora,* it was replaced by *Spartina patens*. Marsh pannes occur as very shallow, wet depressions embedded within marshes and often isolated from tidal creeks. Studies indicate a decline in waterfowl, shorebirds, and wading birds along the Atlantic Coast as their preferred shallow water habitats decreased or disappeared completely (Wilson et al. 1987).

By changing hydrological flow paths, nutrient rich water transport, sediment transport, and sedimentation can also be affected in altered marshes. The ditches also appear to act as sediment traps due to disconnect between the hydrologic source and the interior marsh. It has been observed that mosquito ditches sometimes fill with sediment and vegetation (Jewett, 1949; Redfield 1972; personal observation). Redfield, 1972, hypothesized that the coastal marsh mosquito ditches over drain salt marshes. Since water spends more time in ditches and less time on the marsh interior, it is possible that heavily ditched areas accrete more slowly, as a larger fraction of the total sediment supply settles in the ditches and not in the interior.

Ditch Marsh Restoration

Ditch plugging is currently being used as a restoration practice, but it is not known whether plugging restores normal tidal flow and natural ecological functions (Figure 1.3). Ditch plugs are small dams inserted in the ditch close to the tidal source. Their design is variable, because they have to fit the site topography, soils, location, availability of backfill materials, and embankment fill heights and slopes. Plugs are designed to block the conduit (ditch) water flow from entering and leaving the marsh, and intended to raise ditch water levels and thereby reduce drainage into the ditches from the interior portions of the marsh.

Previous work indicates that restoration of ditched marshes can produce different outcomes in different marsh settings (Adamowicz 2005; Vincent 2013). Increasing shallow pond areas through ditch plugging could result in beneficial ecological changes. Successful ditch marsh plugging includes hydrologic restoration and restored fish and wildlife habitat functions while still maintaining mosquito abatement. Ditches are frequently plugged near the tidal source, where the ditch joins a natural creek or larger ditch. Ditch surface water is impounded upstream of the plug, resulting in flooding in the ditch and surrounding surface area.

Figure 1.3: Photo showing a recently installed ditch plug. View of the tidal creek in back of photo. Photo courtesy of Donald Webster

The most successful restoration and improvement projects in the United States have been completed on marsh systems (Turner 1997; Kennish 2002). The main objective of current projects of salt marsh restoration is to reestablish natural hydrologic flow. Alteration of the historically changed hydroperiod could be all that is necessary for successful restoration (Shisler 1990). Reestablishing and reconnecting tidal connectivity and flow facilitates reestablishment of normal sediment fluxes, patterns, and accretion rates, thus supporting the growth of native salt marsh vegetation while reducing the cover of invasive plants such as *Phragmites australis* (Figure 1.4) (Roman et al. 1984; Raposa & Roman 2002, 2003; Buchsbaum et al. 2006; Konisky et al. 2006; Raposa 2008; Rochlin 2012). Increasing and establishing more natural tidal exchanges (degree of flooding, duration, and frequency) from previous tidally impaired marshes often results in restored ecological functions similar to typical marsh hydro-ecological systems (Stearns 1940; Sinicrope et al. 1990; Peck et al. 1994; Roman et al. 1995; Burdick et al. 1997; Dionne et al. 1999; Warren et al. 2002; Raposa 2002; Roman 2002). A few cases studies are summarized below.

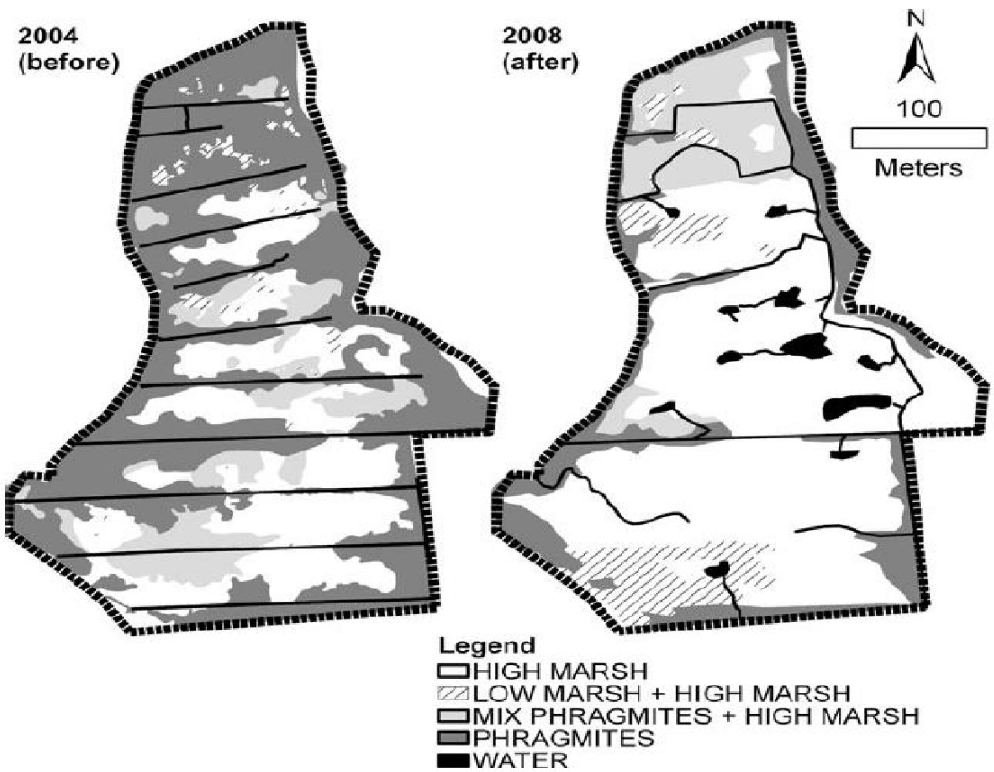


Figure 1.4: Marsh vegetation changes at Wertheim National Wildlife Refuge, NY, from before hydrologic restoration (2004) and 4 years post hydrologic restoration (2008). The overall percent cover (statistically significant) of native species increased, while the invasive species *Phragmites australis* declined (Rochlin 2012).

Stearns et al. (1940) researched the changes in the water table and surface levels following construction of mosquito ditches in Delaware in the 1930's. This is one of the first documented ditched marsh restoration projects in the United States which occurred in response to complaints about declines in muskrat harvesting. After marsh hydrology was reestablished and reconnected by ditch filling, trends in water table and ground level change were reversed (Figure 1.5). Stearns et al. (1940) also confirmed an invasion of shrubs, a change in soil pH, and a negative impact on the muskrat population due to the lowering of the water table and the surface levels. Muskrats were observed abandoning the ditched area within one year but returning the year after the ditches were filled and restored. Reestablishment of hydrologic conditions altered by grid ditching often initiates a change back to typical marsh vegetation (Burdick et al. 1997).

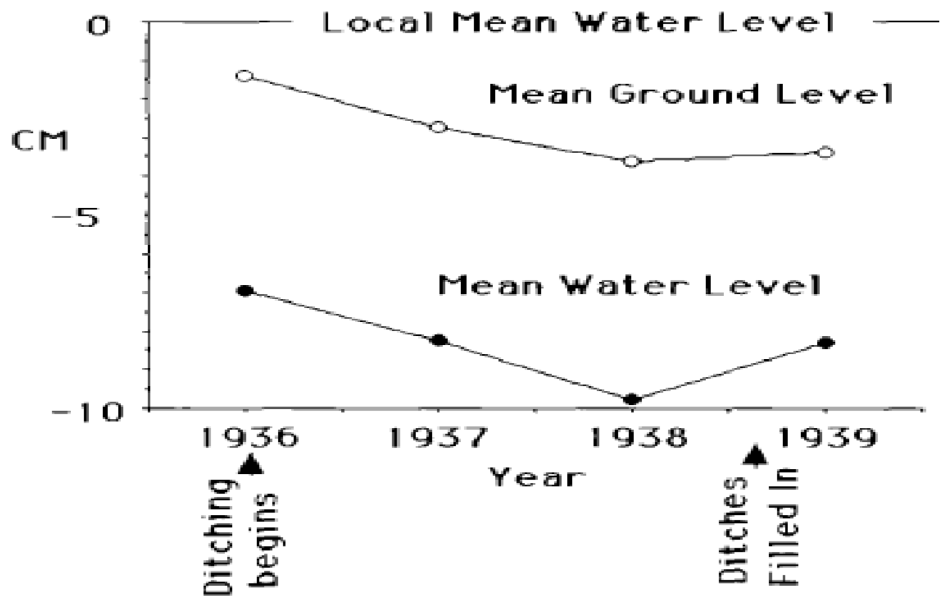


Figure 1.5: Water table levels and surface levels before ditching (late 1930s) and after ditch restoration (1938) in Delaware (Stearns et al. 1940).

Tidally restricted marshes show similar patterns compared to ditch marshes. Roman's (2002) research in 1998 studied tidal flow restoration to a restricted marsh by installing two 76 cm diameter culverts adjacent to the 51 cm culvert. This installation allowed for the tidal range (vertical difference between the high tide and the following low tide) in their unrestricted control marsh and tide-restored marsh to reach equivalency (Figure 1.6). They found after 2 years of restored tidal exchange, vegetation of the tide-restored marsh developed characteristic patterns of a southern New England marsh such as an increase in *Spartina patens* and *Spartina alterniflora* abundance and a corresponding decrease in *Phragmites australis* abundance. After 2 years, vegetation in the tide-restored marsh remained different from that of the unrestricted control marsh, but it demonstrated a trajectory towards similar communities of the unrestricted marshes. In addition to hydrologic and vegetation corrections, one year after restoration, the density, species richness, and community composition of fishes and decapods in the tide-restored marsh were similar to the unrestricted control marsh.

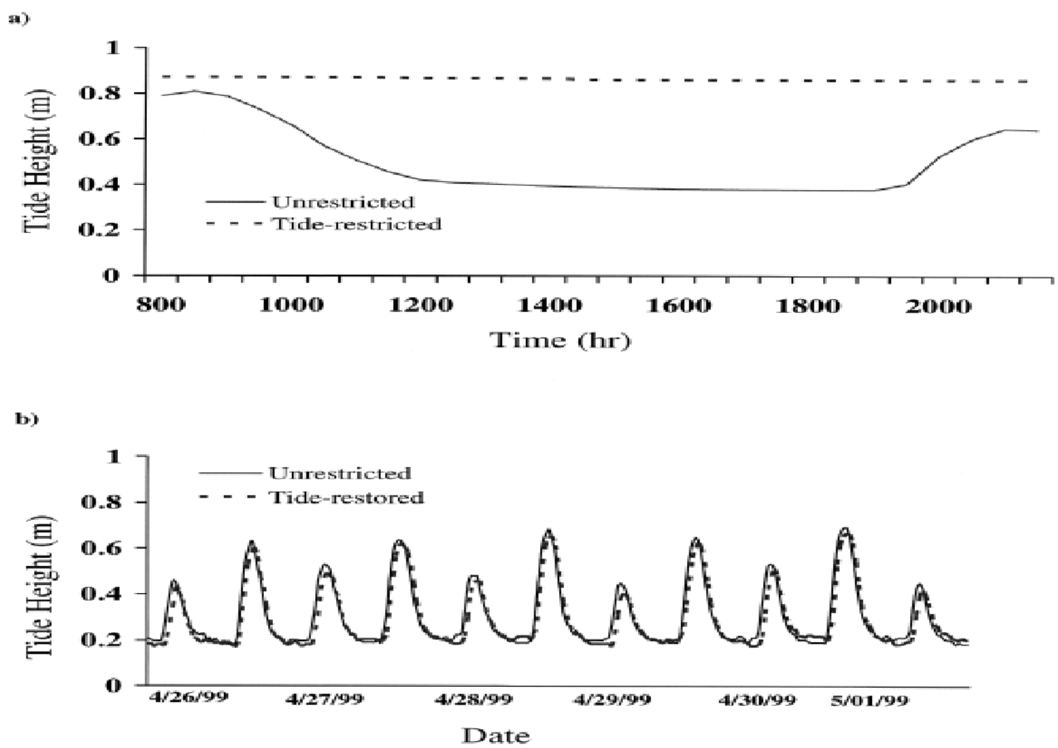


Figure 1.6: (a) Water level elevations of unrestricted marsh and tide-restricted marsh before restoration during one tidal cycle (b) Water elevation of unrestricted marsh and tide restricted marsh after restoration during several tidal cycles. Marshes located Sachuest Point Salt Marsh (Middletown, RI, U.S.A.) (Roman 2002).

Adamowicz (2002) showed that water table levels increased significantly (water table became closer to the surface) following ditch plugging, whereas the unditched marsh did not change (Figure 1.7). With the new wetter hydrologic conditions, vegetation changed from high marsh species (e.g. *Spartina patens*) to species more tolerant of flooded conditions (e.g. *Spartina alterniflora*). When compared to the control marsh, vegetation of the plugged site showed relatively lower cover of high marsh species and a corresponding higher cover for *Spartina alterniflora*. Vegetation response to ditch plugging was rapid, showing a significant change one growing season after restoration. When Adamowicz (2002) compared the pre-plug site with post-plug site (same site 1-2 years later), a decline in the high marsh species *Spartina patens* was the most significant observation contrast between pre- and post-plug conditions.

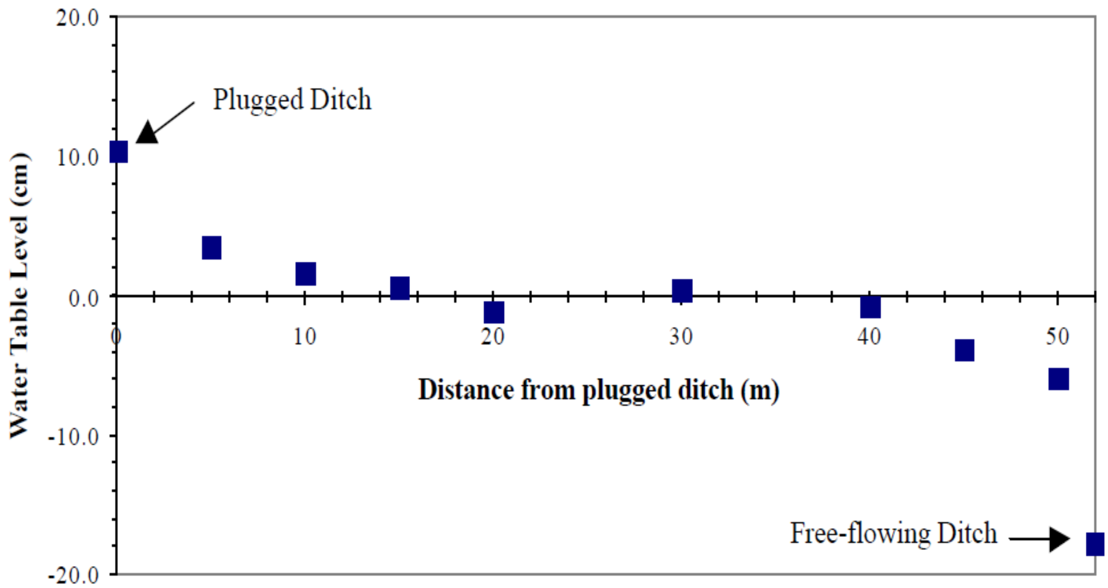


Figure 1.7: Water table level, relative to the marsh surface (at 0.0cm) along a transect comparing a free flowing ditch and a plugged ditch. (Adamowicz 2002)

Morris et al. (2002) determined that flooding can have positive or negative effects on primary production and marsh accretion depending on whether or not flooding exceeds optimal levels. Vincent (2013) showed that hydrologic regime differences were distinct for ditch-plugged compared to ditch sites, and showed that in ditch-plugged systems altered tidal ranges and flooding thresholds were severe enough to have dramatic impacts on the surrounding habitat, based off on Morris (2002). Similarly to Roman's (2002) research on tidally restricted marshes, species richness and nekton density were greater in the plugged marshes than the controls; nekton species richness, total fish density, total decapod density, and nekton community structure remained unchanged following ditch plugging (Adamowicz 2002).

Climate Change and Storm Surges

Tidal marshes are vulnerable to climate change, such as accelerated sea-level rise (SLR). Climate change has the ability to increase inundation and erosion, as well as cause salt water intrusion into freshwater aquifers used by nearby communities. Inundation and intrusion will be affected by the increases in the rate of sea level rise. Eustatic sea level (sea level change due to an adjustment in the volume of water in the oceans) is expected to increase 1–2 m by 2100 (Pfeffer et al. 2008). The vulnerability of tidal marshes depends on geologic factors and geomorphological conditions which buffer shorelines from SLR, and subsidence, which accelerates it. Tidal range also affects marsh vulnerability, as microtidal (< 2 m) marshes are most susceptible to SLR followed by mesotidal (2–4 m), and macrotidal (> 4 m) (Stevenson and Kearney 1986). Rising sea level may result in tidal marsh submergence and habitat migration, as salt marshes move inland (Park et al. 1991). As sea level rises, these marshes are in danger of becoming submerged if they are unable to accrete (soil accumulation) at a sufficient pace to keep up with sea level rise.

Mid-Atlantic coastal marshes commonly have extensive ditch networks that can partially drain or alter flow directions. It is unknown whether ditching affects the ability of a marsh to offer protection from storm surges and wave action. One study from Loder et al., 2009, suggests that small tidal channels have relatively little effect on the storm surge mitigation; indicating that the large tidal channels conveyed storm surges within marshes, but that the effects of ditches or channels decreased with channel size.

Storm surges are among the most damaging and dangerous phenomena in coastal region communities and can cause morphological change and disturbance within ecosystems (NOAA 2014). The effects of hurricane associated damage to coastal ecosystems can be extensive despite the mitigating effect of coastal wetlands (Sheikh 2005). Damage to the surrounding marshes produced by hurricanes depends partly on the water depths at the time of maximum wind stress and water inundation (Morton and Barras 2011).

Climate change may alter the intensity and/or frequency of tropical storms and hurricanes that can generate significant heavy rainfall and/or storm surges along the Maryland Coast (NOAA 2014). Storms can have both long-term and short term effects on marsh hydrology or salinity. Short term effects of storm surges include increasing marsh hydrologic gradients, rising/lowering salinities, and temporarily shifting of plant distributions (Gedan 2009). Marsh surface flooding can vary in duration and magnitude within marshes and it can displace or drown nekton, invertebrates, mammals, and ground-nesting birds, which may result in localized population shifts/declines (Michener et al. 1997).

Storm-induced sediment transport can be a significant part of marsh sediment budgets. Increases in elevation may occur due to sediment deposition and stimulation of root growth; decreases in marsh elevation may result from erosion and compaction or decomposition of marsh soils (Cahoon 2006; Turner et al. 2006). The contribution of storm-supplied sediments to overall marsh accretion likely varies due to marsh type, position, and storm frequency. In marshes with altered hydrology, such as ditched marshes, net losses in elevation due to storms are possible (Cahoon 2006). Many coastal marshes can adjust to sea-level rise associated with climate change, but they may not be able to adjust to both sea level rise and changes in storm magnitude and frequency. It is possible that ditched marshes with altered hydrology may be preferentially affected by these multiple stressors (Day 2008).

Project Design

The project was conducted on the Maryland portion of the Delmarva Peninsula. Marsh sites were selected at the Deal Island Wildlife Management Area and in the EA Vaughn Wildlife Management Area. Site selection was in consultation with Maryland Department of Natural Resources (MDNR) with an overall project goal of providing guidance to MDNR on marsh management (Table 1.1). In each of these marshes, pairs of ditched and nearby unditched (i.e., reference) sites were selected for study. The Deal Island sites are located along the western shore of the Delmarva mainland to the Chesapeake Bay; the EA Vaughn site is located along the eastern (Atlantic) shore on the mainland side of Chincoteague Bay. Deal Island sites will also be referred to as the Chesapeake sites and EA Vaughn sites will be referred to as the Atlantic sites (Figure 1.8).

Table 1.1: Marsh sample sites information table.

Site	**Latitude**	**Longitude**	**Site Area (km^2)**	**Mean ditch length (m)**	**Mean Ditch Spacing (m)**
Chesapeake Unditched	38° 11’08.78”	75° 54’33.27”	0.047	No ditches	No ditches
Chesapeake Ditched	38° 11’02.21”	75° 54’23.02”	0.031	378.8	37.3
Atlantic Ditched	38° 04’34.75”	75° 22’33.09”	0.039	205.6	45.4
Atlantic Unditched	38° 04’40.99”	75° 22’55.44”	0.008	No ditches	No ditches

Figure 1.8: (Top) Deal Island ditched site showing common reed, black needlerush, and cordgrasses; (Bottom) E. A. Vaughn Wildlife Management area showing cordgrass.

Selection of Study Marshes and Site Descriptions

Chesapeake Sites

Two marsh study sites were selected to represent ditch and unditched marshes within the state of Maryland Deal Island Management Area (Somerset County). These sites will be referred to as Chesapeake Bay Sites (Figure 1.9). The Chesapeake Unditched site is located at N38° 11' 08.78” W75° 54' 33.2" and the Chesapeake Ditched site at N38° 11' 02.21" W75° 54'23.02" (Table 1.1).

Abundance of ditches in the Chesapeake Bay area is extensive; within a 20 ha marsh area, ditches consist of a total linear distance of about 6 km, providing a drainage density of 30/km. They are oriented perpendicular to the tidal creeks and run parallel to each other, extending to upland areas. Lateral ditches were also installed, running parallel to the tidal source. The ditches in the Chesapeake Bay sites are spaced at roughly every 37.3 meters. When dug by hand in the 1930s, average salt marsh ditches were approximately 0.4 m wide and 0.6 m deep (Pincus 1938; Williamson 1951). The Chesapeake Bay sites ditches average 1 - 1.5 meter wide by 1.2 - 1.8 meters deep. The Chesapeake Bay sites ditches have a slight berm from the ditch spoil being deposited along the ditch edges (Miller and Egler 1950). Most sites have not been maintained since they were dug.

Atlantic Coast Marshes

Two marsh study sites were also selected within the EA Vaughn Wildlife Management Area (Worcester County, Maryland (Figure 1.9); sites will be referred to as Atlantic Coast Marshes (Figure 1.9). The Atlantic unditched site is located at N38°

04'40.99", W75° 22' 55.44" and the Atlantic ditched site at N38° 04'34.75" W75° 22' 33.09". The nearby bay tides are semidiurnal with a mean range of 5 cm.

Ditches in the Atlantic Coast area are extensive; within a 23 ha marsh area, ditches extend a total linear distance of about 5 km, generating a drainage density of 22/km, slightly lower than the Chesapeake marshes (30/km). Ditches are primarily oriented perpendicular to the tidal creeks, extending to upland areas. Some lateral ditches were also installed parallel to the tidal creek. The ditches in the Atlantic Coast area are spaced on average every 45.4 meters. Back in 1930s ditches were dug by hand, approximately 0.4 m wide and 0.6 m deep (Pincus 1938; Williamson 1951). Currently, ditches average 1 - 1.5 meter wide by 1.2 - 1.8 meters deep. Possible expansion of the ditches could be a result of deterioration of the sites. Unlike the Chesapeake Bay sites and other sites in the northeast, Atlantic Coast sites do not have a slight berm or levee constructed from the ditch spoil along the edges of the ditches (Miller and Egler 1950). There has been no ditch maintenance since they were dug.

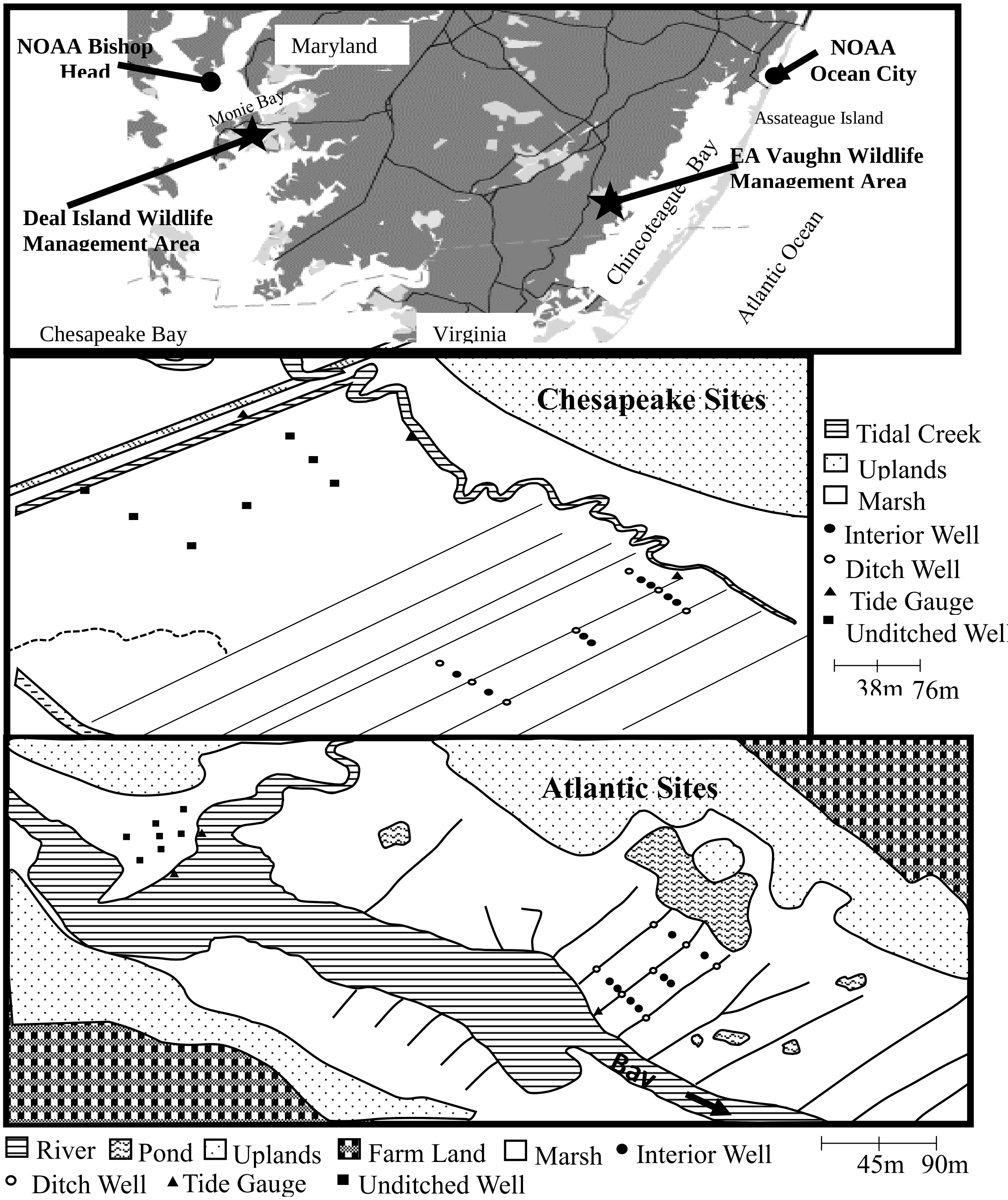


Figure 1.9: (Top) Site map showing the location of the study sites in Maryland; (Middle) Chesapeake Sites (Bottom) Atlantic sites. Wells are shown as solid (marsh) and open (ditches) circles. Site maps show ditch tracks and their spacing.

Approach: Hydrological Data collection to address research questions

Hydrological monitoring networks were established to monitor precipitation, wind speed, and relative humidity, along with creek, ditch, and interior marsh groundwater water levels for a period of four years. At the ditched sites, pre-restoration monitoring was designed to cover a two-year period, to be followed by 2 years of post-restoration monitoring. Equipment installation was initiated in the fall of 2011 and all sites were operational in spring 2012 (Table 1.2). A major hurricane ("Sandy") in the fall of 2012 damaged or destroyed most of the instrumentation at the marsh sites. Therefore, the pre-restoration period was restricted to a one-year period, while the post-restoration piezometric monitoring was based on a smaller number of wells that survived the storm. Since the response of marsh groundwater hydrology to extreme hurricane storm surges has not been well studied, an analysis of the effects of Hurricane Sandy was added to the study objectives.

In addition to the hydrological monitoring, vegetation and soil hydraulic characteristics were sampled and analyzed. Soil data collected included soil horizon characteristics and field measurements of hydraulic conductivity. Meteorological components include rainfall, wind speed and direction, and temperature. Synoptic event-sampling was used to determine salinity and to provide data on hand-measured mini wells. During these synoptic sampling days, salinity, pH, and water levels in mini wells were measured. These synoptic data were used to provide seasonal data on marsh salinity (shown in results) and to determine vertical components of groundwater flow. These data were also integrated with data from the continuous data obtained from water table loggers. Detailed descriptions of methods are found in each chapter.

Table 1.2: Timeline of sampling events

Activity	Spring 2012	Summer	Fall 2012	Spring 2013	Summer	Fall 2013	Winter 2013	Spring 2014	Summer	Fall 2014	Spring 2014	Summer	Fall 2015
Water level- data logging	X	X	X	X	X	X	X	X	X	X	X	X	X
Mini wells /Salinity					X				X			X	
Weather monitoring		X	X	X	X	X	X	X	X	X	X	X	X
Restoration								X					
Vegetation cover		X			X				X			X	
Soil sampling		X			X				X			X	

Organization of the dissertation

In addition to this chapter, the dissertation includes four results chapters (2-5). Chapter 2 - 5 provided a description and pair-wise (ditched vs. unditched) comparison of the soil and hydrological behavior of the marshes. These chapters also includes a detailed description of the hydrological monitoring system, methods of field data collection, and the hydraulic characteristics of marsh sediments. Chapter 4 focuses on the effects (water levels, magnitudes, durations, etc.) of the Hurricane Sandy storm surge on ditched and unditched marshes. In Chapter 5, a comparison of hydrological characteristics (time series) for pre- and post- restoration in the restored ditched marshes is discussed. Pre- and post-restoration data are used along with the Hurricane Sandy data to assess whether plugging ditches in marshes is a successful management technique for restoring marsh water levels and improving resiliency to storm surges.

Chapter 2: Soil hydraulic properties and tidal amplitude attenuation in ditched and un-ditched coastal marsh peat aquifers

Tidal marshes provide ecological services such as wildlife habitat, shoreline protection, organic carbon and nitrogen sequestration, and they often maintain stable elevations relative to sea level due to sediment and organic matter accumulation (Barbier et al. 2011). Tidal marshes serve as carbon sinks in the global carbon cycle; they store an estimated 4.8–87.2 Tg carbon per year (Mcleod et al. 2011) due to high plant productivity, low decomposition rates, and burial of organic matter (Bridgham et al. 2006). The accumulation of organic matter generates peat soils, which are low bulk density, porous soils rich in organic matter that can effectively hold and transmit water if they are not compacted. Along the northeastern and mid-Atlantic coast of the U.S., salt marshes have been extensively ditched with parallel or gridded arrays of ditches. Although ditching began in the 1700s to increase *Spartina paten*s farming of salt hay (Shisler 1990; Daiber 1986; Sebold 1992), concerns about mosquito control in the early 1900's expanded ditching efforts along coastal regions adjacent to New York City (Richards 1938) and coastal New Jersey (Smith 1904). Most of the extensive ditching of NE and Mid-Atlantic marshes, however, occurred during the Great Depression, as part of the national effort to combat high unemployment (Glasgow 1938). The massive restructuring of marsh environments through ditching can have significant ecological impacts (e.g. Daiber 1986). The purpose of this chapter is to examine the effects of ditches on soil hydraulic properties (bulk density and hydraulic conductivity) and on the attenuation or amplification of tidal fluctuations in marsh groundwater.

Soil hydraulic properties of peaty, low bulk density soils could be affected by drainage provided by ditch networks. Although soil drainage could compact peaty soils, the amount of soil drainage may be variable at ditched sites, and the compaction caused by drainage may be offset by organic matter accretion and dilation caused by root growth. Most studies that compared ditched and unditched marshes found fewer ponds and less open water in ditched marshes (Reinert et al. 1981; Merriam 1983; Lathrop et al. 2000, Adamowicz and Roman 2005). Studies of the effects of ditches on marsh groundwater drainage indicate mixed results, with some studies documenting the lowering of water levels (Bradbury 1938; Corkran 1938; Daigh et aln. 1938; Taylor 1938, Redfield 1972; Adamowicz and Roman 2005) and other studies indicating little variation in water levels between ditched and unditched marshes (Vincent 2013). Others studies indicate that marsh ground water tables only drain in regions adjacent (within 15 to 25 m) to ditches or tidal creeks (Provost 1977; Hemond and Fifield 1982; Agosta 1985; Nuttle 1988; Montalto et al. 2006).

The response of marsh groundwater gradients and flow directions to ditching may be significantly influenced by the propagation of tidal fluctuations into the marsh groundwater. Tidal range is the vertical distance between high tide and low tide. Ditched marshes along the East Coast are found in a wide range of tidal ranges. Many previous studies of marsh groundwater have been conducted in macrotidal settings (e.g. Hemond and Fifield 1982; Nuttle 1988). If tidal ranges exceed 1 m (a typical ditch depth), then ditches connected to tidal creeks are likely to totally drain at low tide (Tonjes, 1993). When the tidal range is less than 1 meter, ditches will likely retain water at low tide (Adamowicz and Roman 2005). The head differences between marsh interiors and ditches or tidal creeks that drive drainage would likely be smaller for marsh systems with small tidal ranges (<1m) but these if head differences are persistent, they could lead to partial drainage and the lowering of marsh water tables.

Groundwater discharge from unconfined marsh aquifers to ditches or tidal creeks can be significantly influenced by tides (Robinson et al. 1998; Robinson and Gallagher, 1999). Therefore, an understanding of groundwater discharge process in tidal marshes must consider the tidal boundary conditions (tidal amplitude and period) and their effects on groundwater heads. Tidal amplitude typically attenuates with distance into the marsh and attenuation is most pronounced in unconfined aquifers with low hydraulic conductivity, K, values (Figure 2.1).

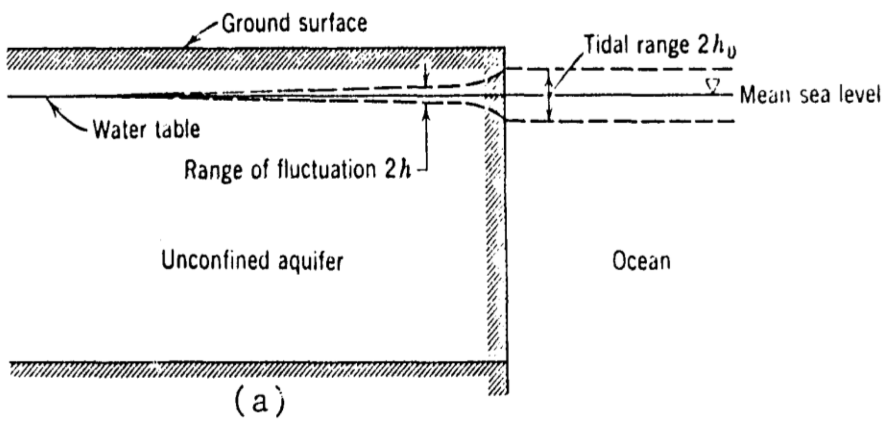


Figure 2.1: Conceptual diagram showing the attenuation of tidal range in an unconfined aquifer (from Todd, 1980)

Hydraulic diffusivity from amplitude attenuation, D_{amp} (m^2/day), can be estimated from direct measurements of tidal amplitude in both creeks and in marsh groundwater. For an idealized marsh (e.g., Fig. 2.1), the degree of attenuation can be related to aquifer properties, tidal characteristics, and the distance from the ocean, tidal creek, or ditch (Rotzoll et al., 2013):

$$D_{amp} = Kb/Sy = (x^2\pi)/(\ln A)^2\tau$$

where A is the amplitude attenuation factor, which is the ratio of the tidal amplitude (half of the tidal range) in an observation well to the tidal amplitude in the tidal creek, K is hydraulic conductivity (m/day) b is aquifer thickness (m), S_y is specific yield, x is the distance (m) from the tidal creek, and τ is tidal period (days). Thus, tidal amplitude would be most effectively attenuated in marsh groundwater aquifers with high values of hydraulic conductivity (K), aquifer thickness (b), or both. Conversely, measured values of D_{amp} could be used to estimate specific yield, S_y, if K and b are known. Horizontal hydraulic head gradients toward tidal creeks or ditches may be more effectively established when tidal amplitudes are strongly attenuated, for example in an unconfined aquifer with a high K. Alternatively, low values of K have also been associated with poor drainage in some ditched marshes (Montalto et al. 2006).

Objectives and Hypotheses:

In this chapter, I compare hydraulic soil properties and the attenuation of tidal amplitude in ditched marshes with properties in adjacent unditched marshes. The analysis is conducted with field monitoring data of tidal stage and marsh groundwater levels along with field sampling and analysis of marsh soil horizons, soil bulk density, and hydraulic conductivity. It was hypothesized that ditches would drain and compact soils, which would cause the upper soil layers in the ditched marsh site to have higher bulk density values and lower hydraulic conductivity values than those in unditched marshes. It is also hypothesized that these lower values of hydraulic conductivity in the ditched marshes would lead to less attenuation of tidal amplitude in marsh groundwater than observed in unditched marshes. If ditched marshes have higher bulk density values due to compaction, they would also likely have lower specific yield (Sy) values due to a decrease in porosity (Sy = porosity – specific retention); where specific retention is the amount of pore water that does not drain readily under gravity (Fetter 1994).

Study Sites and Methods

Measurements were made at paired marshes (one ditched, the other unditched) at two locations (Chesapeake Bay and Atlantic Coastal Bays). Both the Chesapeake Bay and Atlantic sites are located in marshes that are connected to shallow coastal bays and experience microtidal (less than 0.5 m), semi-diurnal tides. Three types of measurements were made to facilitate a comparison of the ditched marshes with unditched marshes for soil properties and tidal range: (1) marsh soil samples were collected and analyzed to determine soil horizons and bulk density. (2) *In situ* measurements were conducted in mini wells to estimate hydraulic conductivity; and (3) hydrological monitoring networks were established to monitor tidal fluctuations with distance and elevation. Measurement techniques are described in the following sections and in the section on statistical analyses.

Hydrological Monitoring

Each of the two paired marshes was instrumented with well networks that consisted of three transects of wells arrayed perpendicular to the ditches and parallel to the tidal creek (Figure 1.9). Wells were installed laterally from a dominant ditch. The first transect at the ditched sites, located closest to the tidal creek, contains seven wells, the second transect has three wells, and the far transect has five wells (Figure 2.2). Wells were located along transects at midpoints, quarter points, and within the ditches. Well installation at unditched sites were also arrayed in three transects of wells parallel to the main tidal channel, with the first transect composed of three wells; the second transect with one well, and the far transect with three wells (Figure 2.3). Each well and mini wells top and ground elevation data were determined by a survey conducted by Maryland DNR.

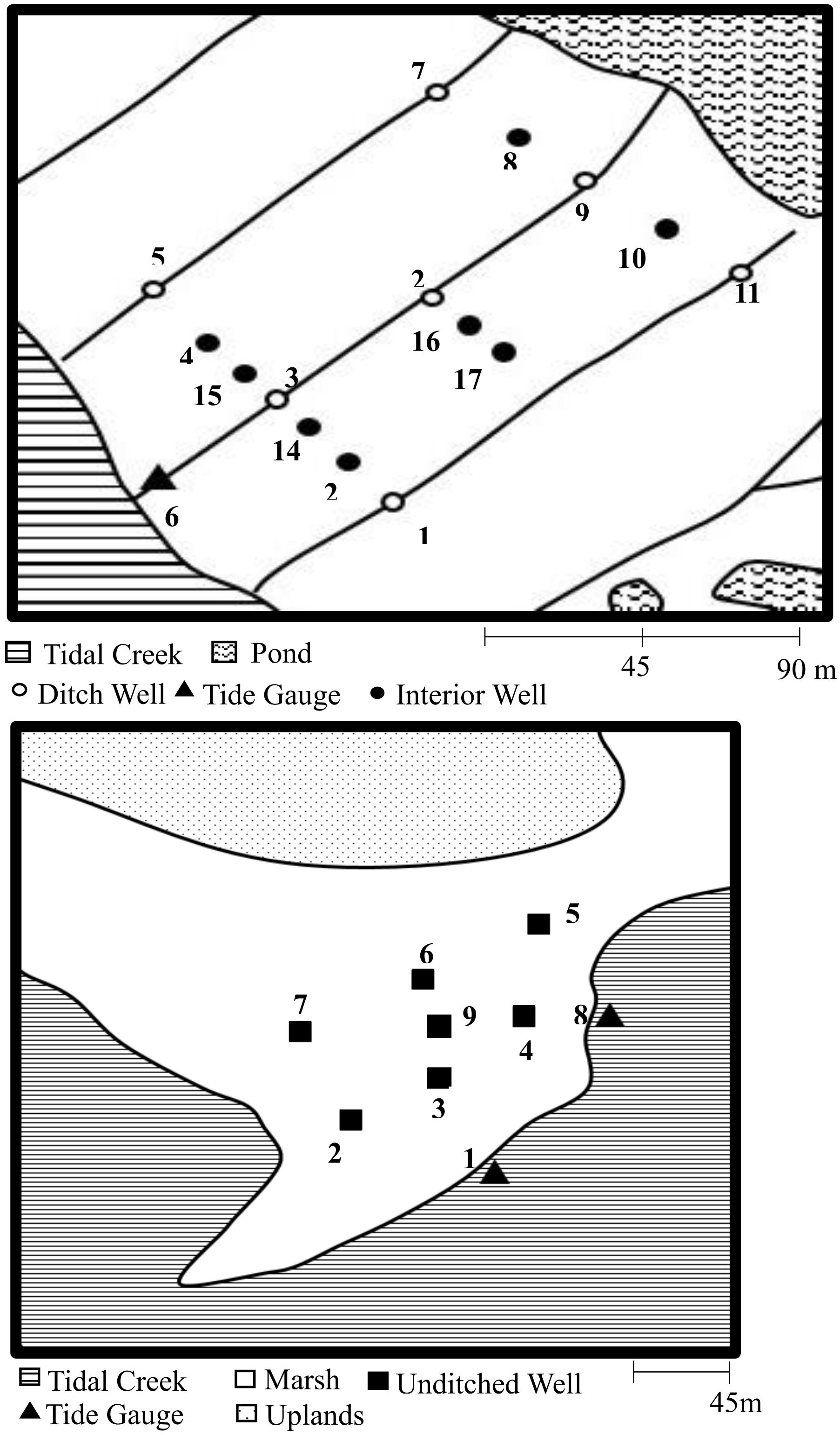


Figure 2.2: Examples of transects and well locations at the Atlantic ditched (top) and unditched sites (bottom). The same layout was used at the Chesapeake sites.

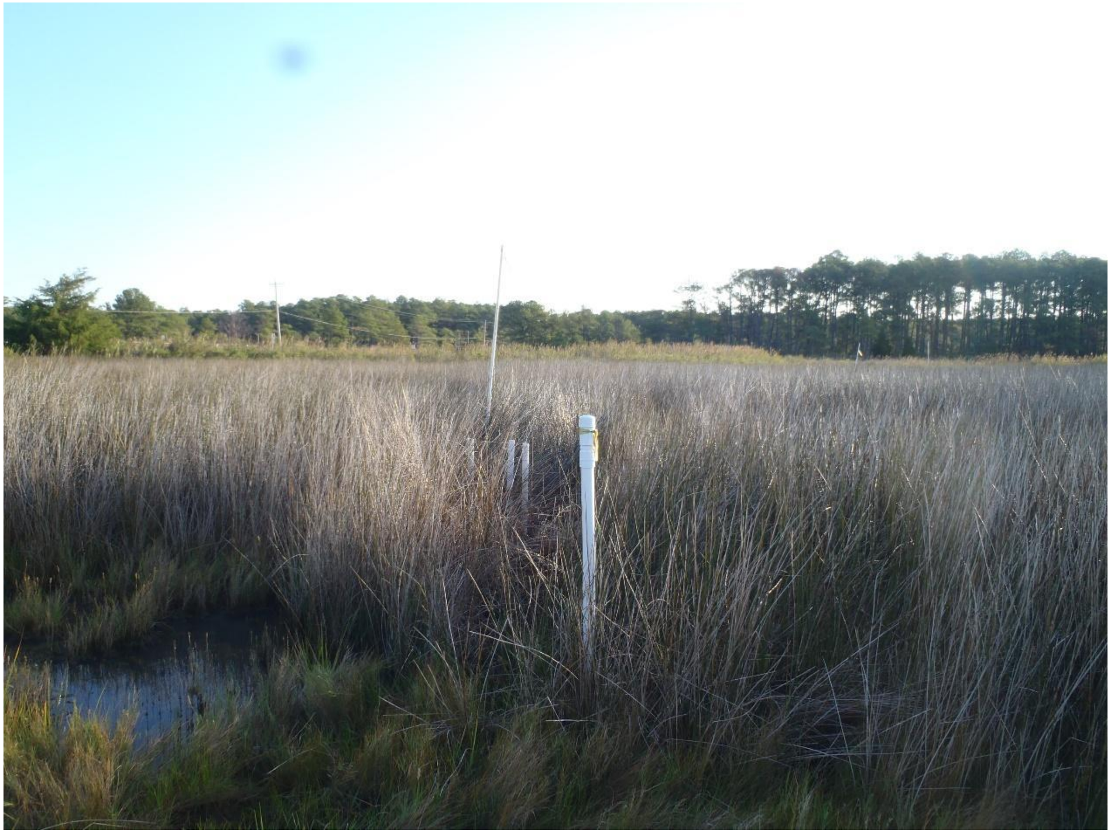


Figure 2.3: Transect at Chesapeake site showing monitoring well (front) and a set of 3 nested short-interval wells. Tall pole at the back is the midpoint of a circular vegetation sampling plot.

Wells were installed 2.2 meters deep by hand-auguring. Well casings were constructed of 3.05 m long sections of Schedule 40 PVC pipe (pipe diameter = 3.8 cm). The bottom 2.15 m of each casing section was machined slotted (0.010 cm slots with 0.50 cm spacing between slots). A PVC point was attached at the bottom of the well to allow better driving of casing into the subsurface and a screen covered the slotted portion of the well to prevent coarse sediments from entering (Figure 2.4). Wells were equipped with Odyssey data doggers. The data loggers were calibrated in the lab (See appendix on calibration); water levels were periodically field-checked over a range of conditions to generate calibration curves that were used to correct the data logger data (when necessary). The data loggers were programmed to record water levels in each well at 15 minute intervals. More detailed calibration methods can be found in the appendix.

Wells with short screened intervals (mini wells) were installed to different depths in nests of three using hand augers. These short interval wells were used to determine the vertical component of the hydraulic head gradient (i.e., dh/dz) in the marsh subsurface. Mini wells casings were constructed from the same materials as the wells, except the slotted sections extended for 15 or 30 cm intervals. Slotted intervals were placed at depths of 5-20, 20-50, and 50-80 cm below the soil surface (Figure 2.4). These short-interval wells were used for making hydraulic head measurements using hand (dipstick methods). The short-interval wells were also used for *in*-situ hydraulic conductivity measurements.

Data collected from the groundwater wells were used to calculate tidal range (2x tidal amplitude) in the tidal creeks, ditches, and marsh. These data were used to examine attenuation or amplification in unditched in comparison to ditched marshes. Tidal range is the vertical difference between the high tide and the successive low tide (Figure 2.1). Tides are defined as the rise and fall of sea levels caused by the combined effects of the gravitational forces exerted by the moon and the sun with the rotation of the Earth. The tidal range is not constant; it changes depending on the location of the sun and the moon and it can be amplified or attenuated by coastal morphology.

In addition to field monitoring at the site, tide gauge data were obtained from National Oceanic Atmospheric Administration (NOAA) at their Ocean City (Station ID: 8570283) and Bishop Head (Station ID: 8571421) gages shown in Figure 1.9.

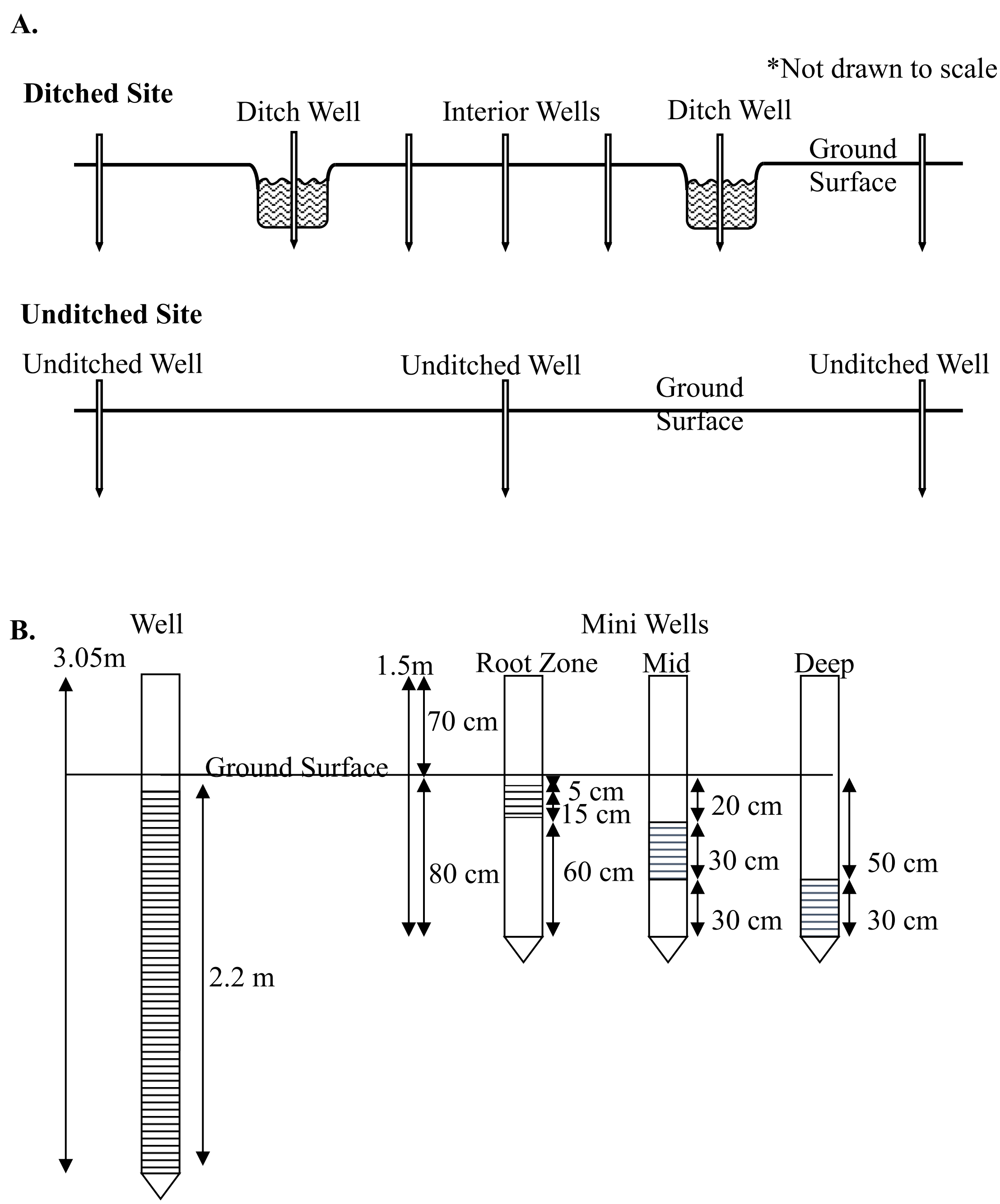


Figure 2.4: Conceptual drawing of monitoring well and mini wells layout.

Marsh sediment characterization

Soil cores were collected using an Eijkelkamp Peat Sampler in the summers of 2012 and 2013 selected at locations in both ditched and unditched marsh sites. Sampling locations were established near wells and mini wells locations. Depending on location, there were 15 to 20 soil sampling plots at each site. At each plot, a core was removed to approximately 90-100 cm deep, which was then examined in the field by color and hand textured by rubbing between two fingers to identify the interface between adjacent soil horizons. Cores were segmented by horizons and returned to the lab for bulk density analyses. Soils in the plots had 4-6 horizons.

In the lab, each horizon was textured using methods modified from Thien (1979). A small subsample was placed in the palm of the hand and kneaded. It was tested for holding its shape in a ball, then tested to determine if a ribbon could be made and how durable the ribbon was. The soils were tested for grittiness or smoothness, and based on these soil properties a soil type was assigned to the sample.

Each horizon soil sample was also subsampled into segments of known volume, weighed, dried in the oven for 48 hours at 105°C, and reweighed; the data were used to compute bulk density (SSSA 1998). The soil bulk density (*BD*) is the weight of dry soil divided by the total soil volume (g/cm^3). The total soil volume is the combined volume of solids and pores which may contain air, water, or both (McKenzie 2004). Horizon subsample results for BD were grouped into specific depths of 0 – 20 cm (root zone), 20 – 50 cm (mid), and 50 – 90 cm (deep) depths to correspond with the intervals of the mini wells that were used for hydraulic conductivity analyses.

Hydraulic conductivity, K (m/min), was determined by conducting *in situ* falling head slug tests (i.e., Ferris and Knowles 1963). A falling-head test is conducted by rapidly raising the water level ("head") in the well (using a "slug" of water) and subsequently measuring the falling water level as it approaches its original ("static") position. Head versus time measurements were made with timed dipstick readings. Head measurements were expressed as the ratio of H/H_0, where H_0 is the adjusted head measurement after the head was raised. These data are graphed against time (with a logarithmic H/H_0 axis). Using the Hvorslev method, graphs of H/H_0 versus time were constructed, with the H/H_0 axis logarithmic (Fetter 2001). The value of T_0, defined as the time when the value of H/H_0 = 0.37, was determined when fitted to a linear trend on the semi-log plot. The Hvorslev equation was used to determine hydraulic conductivity, K:

$$K\ \left(\frac{m}{\mathrm{min}}\right) = r^2\,\frac{\frac{LnL}{R}}{2T_0L}$$

where r is the radius of the well (m), K is the hydraulic conductivity (m/day), R is the radius of well screen (m), and L is the length of well screen (m). T_0 is the time (min) required for the water level to fall to 37% of the initial change in head, obtained from graphs, and particularly using the blue line (Figure 2.1). Hydraulic conductivity values were calculated in m/min, but converted to m/day for subsequent analyses.

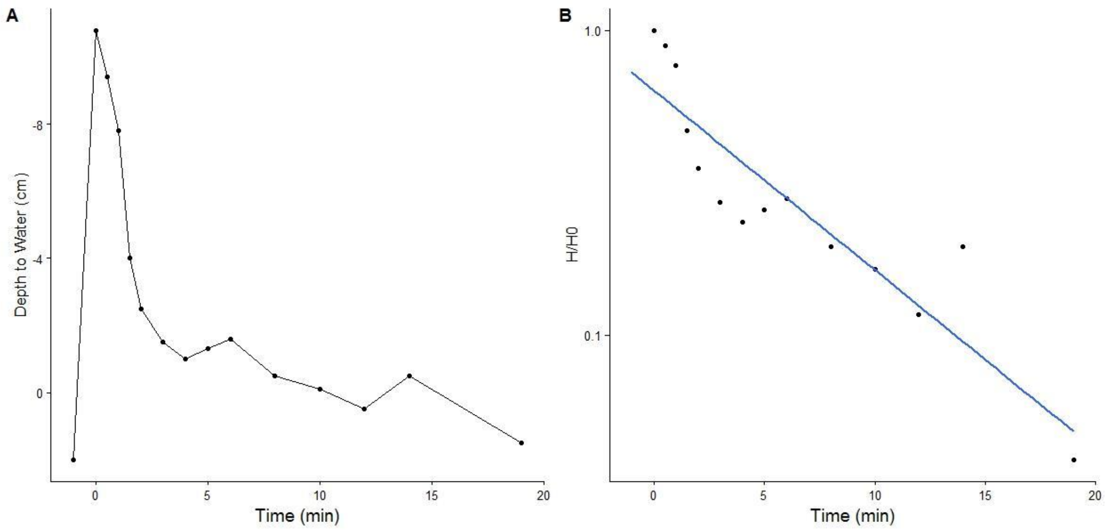


Figure 2.5: Example of falling head data used to determine hydraulic conductivity. (a) Depth to water vs time, (b) H/H_0 versus time, T_o, is defined as time when $H/H_0 = 0.37$

Statistical Analyses

Student t-tests were used to compare the means of two groups under the assumption that both samples are random, independent, and drawn from normally distributed populations. In this analysis, a *p* greater than 0.05 indicated that the means of two groups were statistically similar. The t-test assumes that the mean would be a good measure of central tendency for the data being tested. When samples did not fit a normal distribution, the Wilcoxon rank sum test, a nonparametric test, was used to determine if the sites were statistically different. Statistical tests were also used to test differences between soil depths and salinity. T-tests were performed in Microsoft Excel with unknown variances and two tails. Wilcoxon rank sum tests were performed in R. Hydraulic conductivity can vary over multiple orders of magnitudes and are often log-normally distributed. Therefore, hydraulic conductivity data were log-transformed before conducting the t-test.

Results:

In the presentation of results, the comparisons of ditched and unditched marshes at the Chesapeake and Atlantic sites will be presented separately.

Chesapeake Sites

Tidal Range Data:

Tidal range is the difference between low and high tide during a tidal cycle, It affects the hydraulic gradients between marsh interiors and tidal creeks or ditches. Tidal range was measured in the wells located in the tidal creeks, in the ditches, and the interior portions of ditched and unditched marshes. Tidal range was calculated by day (max-min), and daily values were averaged by month. An example of tidal amplitude data shown over short time scales is shown in the figure below. On this diagram, red indicates the spring tides at the ocean source, while blue indicates neap tides. Spring and neap tides within these coastal embayments and associated marshes are delayed from the ocean spring and neap tides.

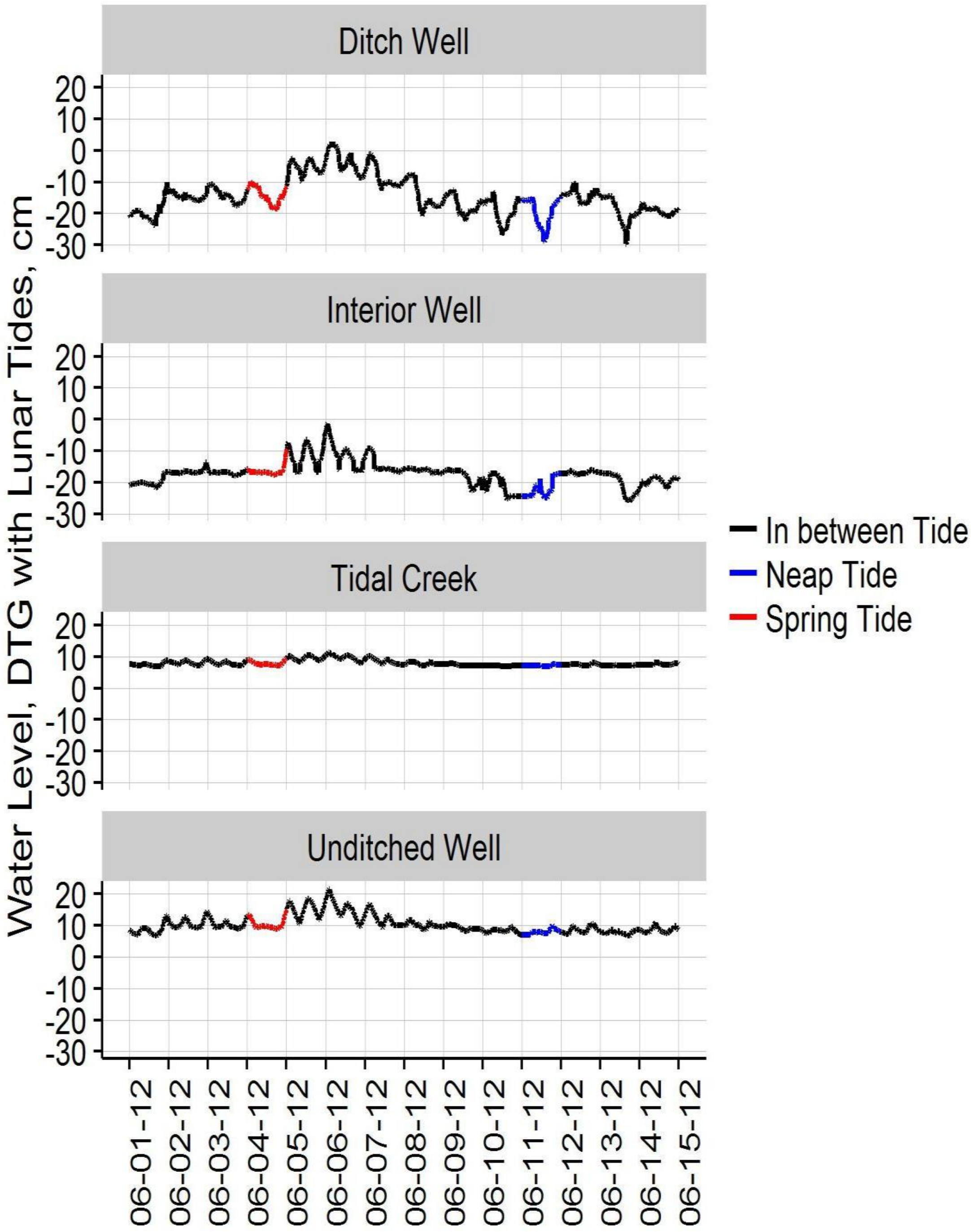


Figure 2.6: An example of a 15-day time period showing the variations in tidal amplitude associated with spring (red) and neap (blue) tides.

The average tidal range for the 2 year period (2012-2014) in the tidal creek near the unditched marsh was 18.4 cm. The average tidal range in the tidal creek near the ditched sites was 10.8 cm, but the tidal range in the nearby ditches was higher, with an average value of 15.3 cm—suggesting local amplification of tidal range in the ditches (Table 2.1). Comparison of the open bay NOAA tide gage and the tidal creek data indicates attenuation of tidal range by 3.5 to 6.5 cm between the open Bay NOAA tide gauge and the marsh tidal creek gauges located near the marsh sites.

Marsh interior wells indicated significant attenuation of the tidal signal compared to either the tidal creek or the ditch. Average tidal range in the marsh interiors was 7.0 cm at the unditched site and 8.2 cm at the ditched site. Site-averaged values of tidal range was 11.4 cm at the ditched site and 9.8 cm at the unditched (Table 2.1). The largest values of tidal range in the tidal creeks occurred between February and July 2012 at both ditched and unditched sites (Figure 2.7). Figure 2.7 also shows that ditched interior and unditched marsh groundwater tidal range did not fluctuate systematically during the year.

The marsh groundwater at the unditched site indicated that tidal range decreased with distance from the tidal creek (Figure 2.8). The average of the values plotted on the graphs indicated an average decline in tidal range from ~ 20 cm to ~ 5 cm over a distance of 150 m. Because the ditches had higher tidal range values than the adjacent creeks, the change in tidal range relative to the tidal creek indicated amplification at some locations rather than attenuation. For the Chesapeake site, Transect 2 increased in tidal range with distance from the tidal creek. Transect 3 (back of site) showed the largest range.

Table 2.1: Mean tidal range (cm) for Chesapeake Bay Sites

	Chesapeake unditched	Chesapeake ditched	NOAA
Site Averages	9.8	11.4	66.5
Interior Well	-	8.2	-
Ditch Well	-	15.3	-
Unditched Well	7.0	-	-
Tidal Creek	18.4	10.8	-

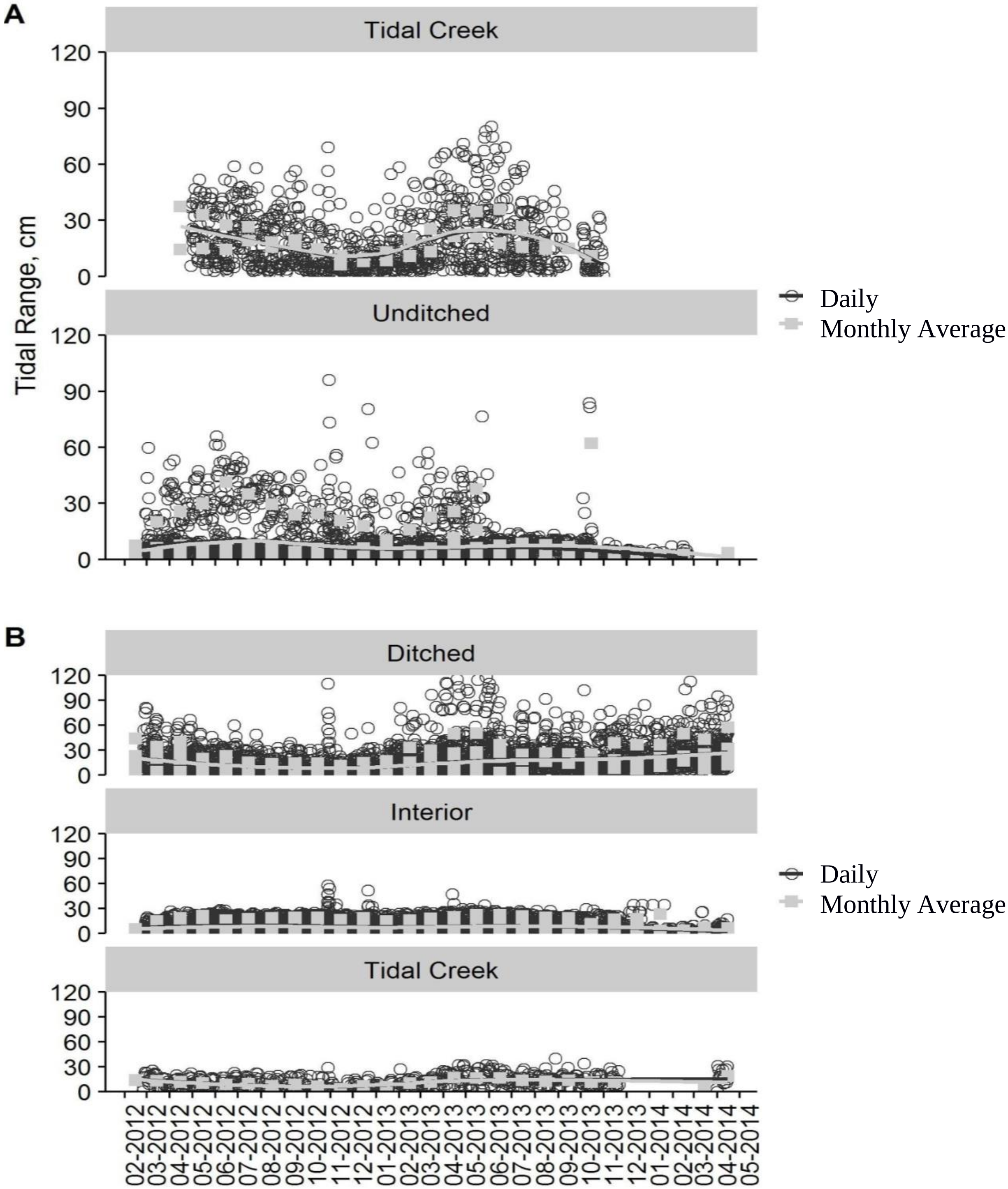


Figure 2.7: Figure 2.7: Tidal range, cm, for the 2 year period (2012-2014) for: (A) Chesapeake unditched site; (B) Chesapeake ditched site.

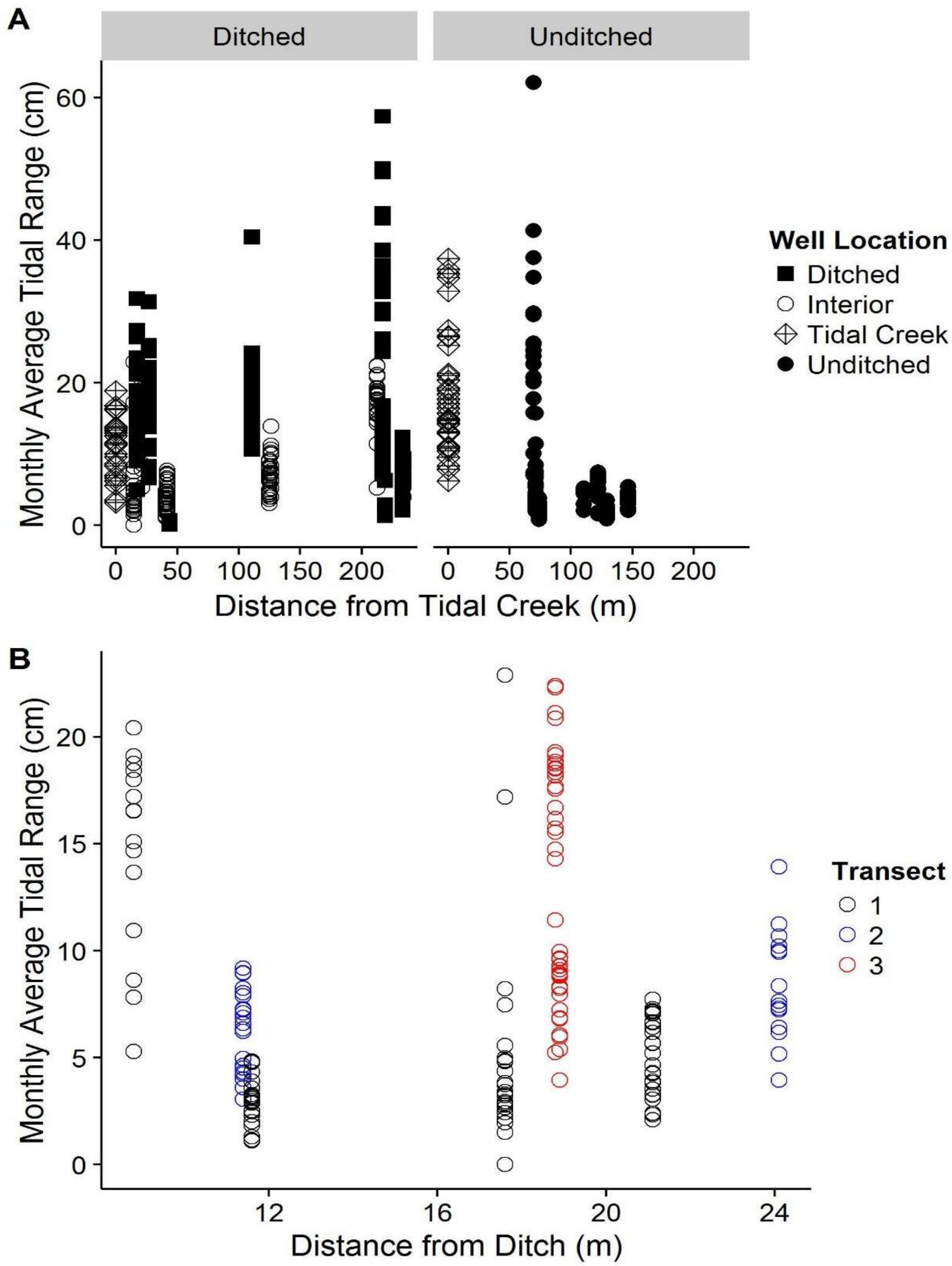


Figure 2.8: (A) Tidal range (cm) plotted against distance away from the tidal creek (m) for Chesapeake ditched site and unditched site; (B) Interior marsh wells tidal range (cm) plot distance away from ditch. Legend describes well locations.

Soil Texture and Hydraulic Properties

Soil core data indicate that the marsh sediment at both ditched and unditched was primarily peat up to depths of 65 cm with an underlying loam/silt loam/sandy loam layer (Figure 2.9). This was determined in the field and lab from soil texturing analysis, and supported by the bulk density data.

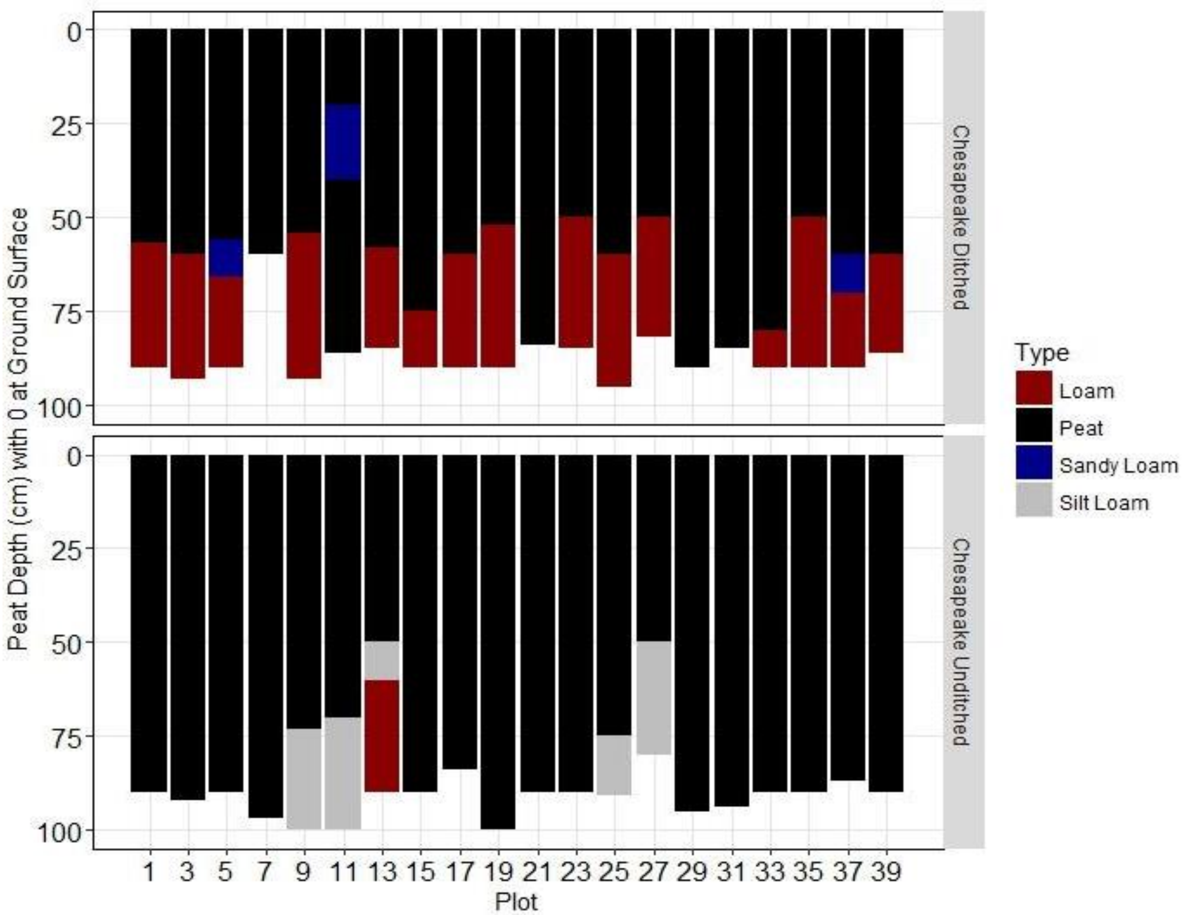


Figure 2.9: Soil Stratigraphy for Chesapeake Ditched and Unditched sites. Data indicate higher peat depths in the unditched sites than the ditched sites.

The soil bulk density (BD) showed a wide range of values at both the Chesapeake ditched and unditched sites. Unditched site bulk density values ranged from 0.02 to 1.6 g/cm^3, and ditched sites values ranged from 0.04 to 2.2 g/cm^3 (Figure 2.10). Density plots were used to show and compare the distribution of data. Based on boxplots and histograms, bulk density was examined as a function of depth for each site (Figure 2.10 and 2.11). The rooting zone (5-20cm) had the lowest bulk density values at both sites and the deeper cores (50-100 cm) had the highest bulk density values (Figure 2.10 and 2.11). Bulk density was higher with depth and showed a significant range with depth, but did not show a relationship with marsh surface elevation. In addition, bulk density did not vary with distance from tidal creeks, except in the > 50 cm cores that had higher bulk density values further from the creek (Figure 2.12).

The comparison of bulk density values between the ditched and unditched marsh sites indicates that there was no statistical difference in bulk density between these sites. Mean bulk density was 0.50 g/cm^3 for the ditched site and 0.23 g/cm^3 for the unditched site, based on data from all of the depth intervals in the core sections (Table 2.2).

Table 2.2: Chesapeake Sites BD and K

	Bulk Density (g/cm^3)		Hydraulic Conductivity (m/day)	
Averages	**Ditched**	**Unditched**	**Ditched**	**Unditched**
Site (all)	0.50 (0.5)	0.23 (0.3)	2.5 (1.6)	-
Root Zone	0.12 (0.04)	0.11 (0.05)	2.59 (0.3)	-
Mid	0.16 (0.1)	0.14 (0.05)	3.44 (1.5)	-
Deep	0.93 (0.6)	0.34 (0.4)	0.8 (0.6)	-

() standard deviation

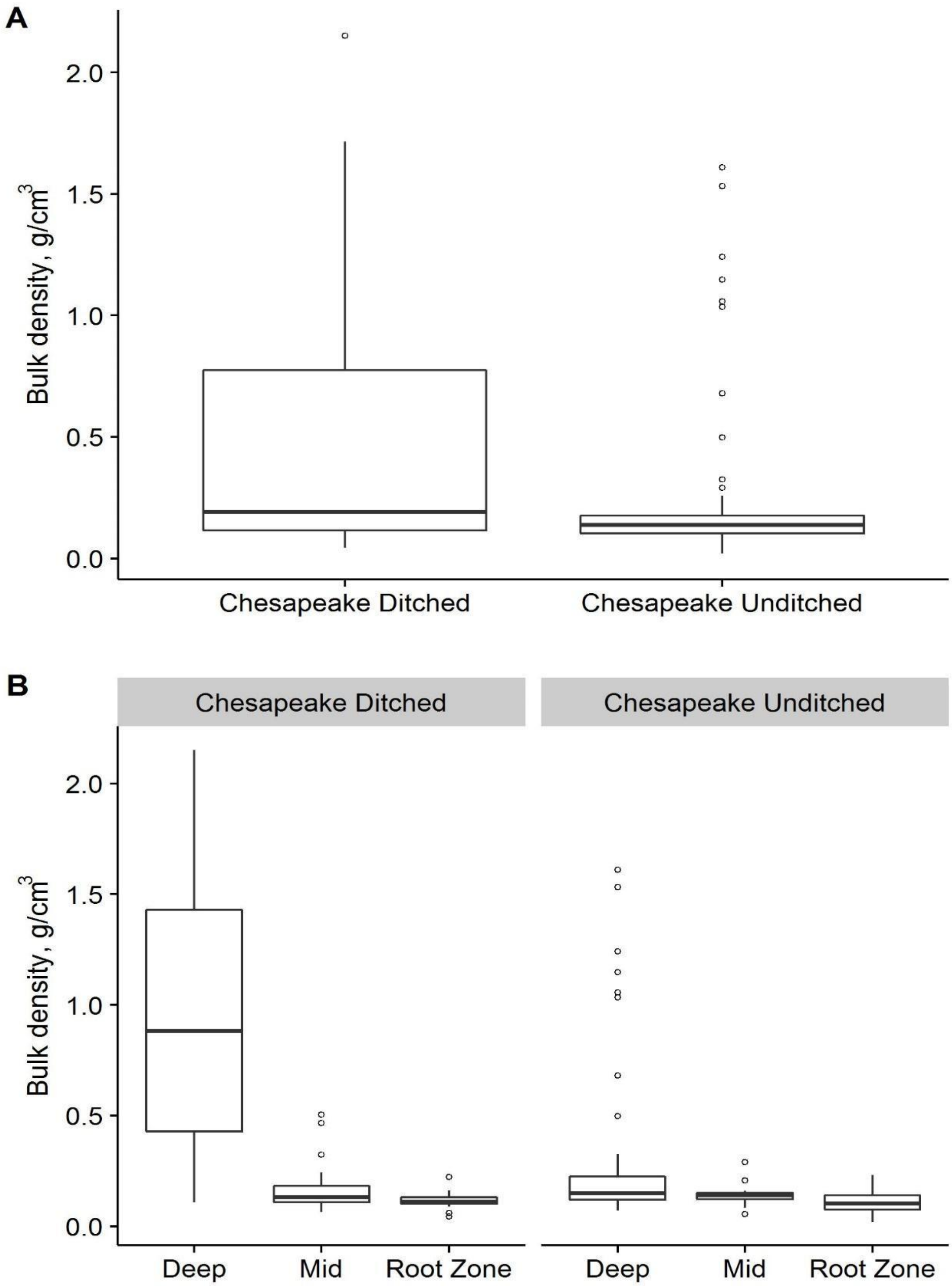


Figure 2.10: Bulk Density for Chesapeake Bay Site Ditched and Unditched by site and by depth. Box plot symbolism: center line = median; box boundaries = 1st and 3rd quartile; circles = outliers.

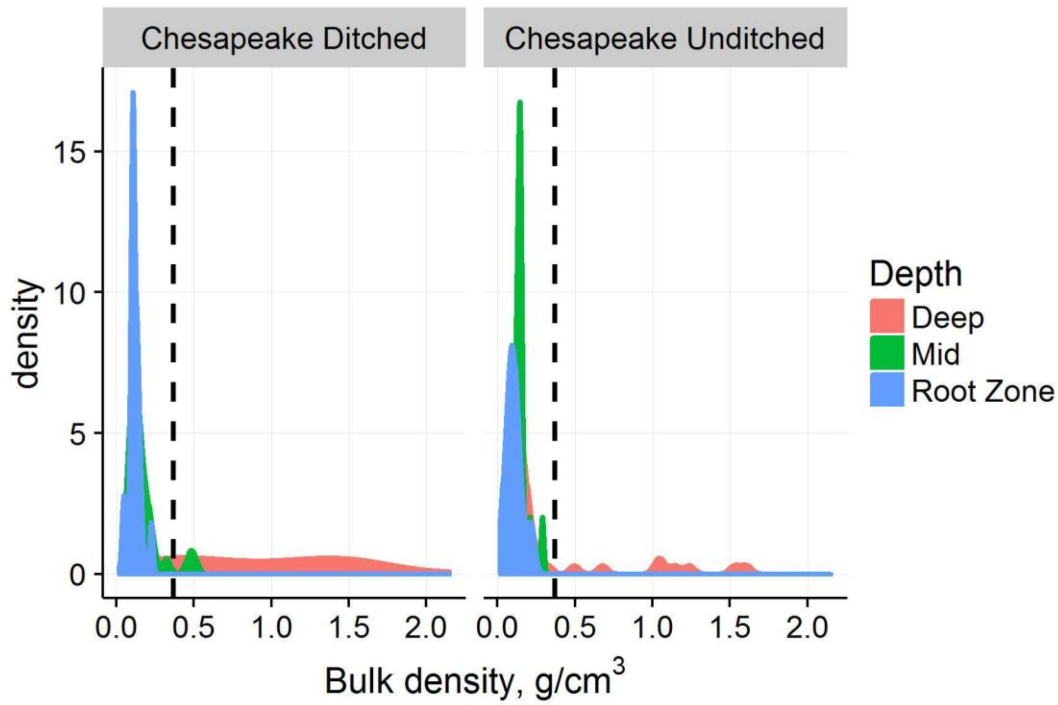


Figure 2.11: Probability density curves for bulk density for Chesapeake Bay Site Ditched and Unditched by depth.

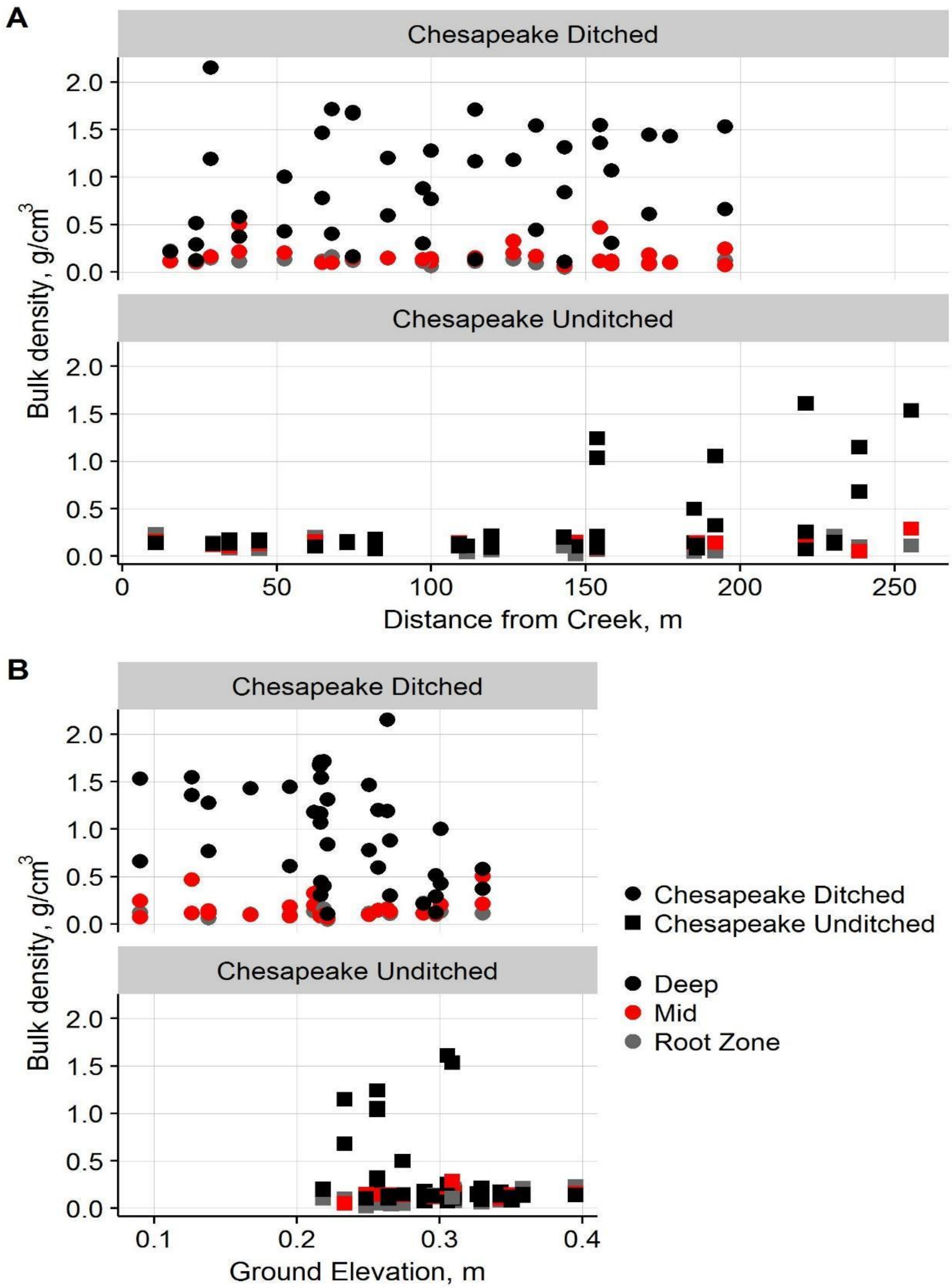


Figure 2.12: Bulk Density for Chesapeake Bay Site vs Distance and Elevation. Bulk density does not vary with distance from the creek or site elevation.

Hydraulic conductivity (K) was measured by *in-situ* slug tests for the ditched site (Figure 2.13) using the mini wells at three depths; 0-20, 20-50, and 50-80 cm below ground. The biggest differences in K were observed when data was separated by depth intervals. The upper two horizons (0-20 cm and 20-50 cm) at the ditched site have higher K values than the 50-30 cm intervals (Table 2.2), which is consistent with the low bulk density values observed at shallow depth and the higher bulk density data at depth. Hydraulic conductivity decreased with depth. Hydraulic conductivity values also decreased with distance from the creek in the 50-80 cm intervals, whereas the wells at 20-50 cm depths indicated an increase in K with distance (Figure 2.14). No relationship between marsh elevation and hydraulic conductivity was observed.

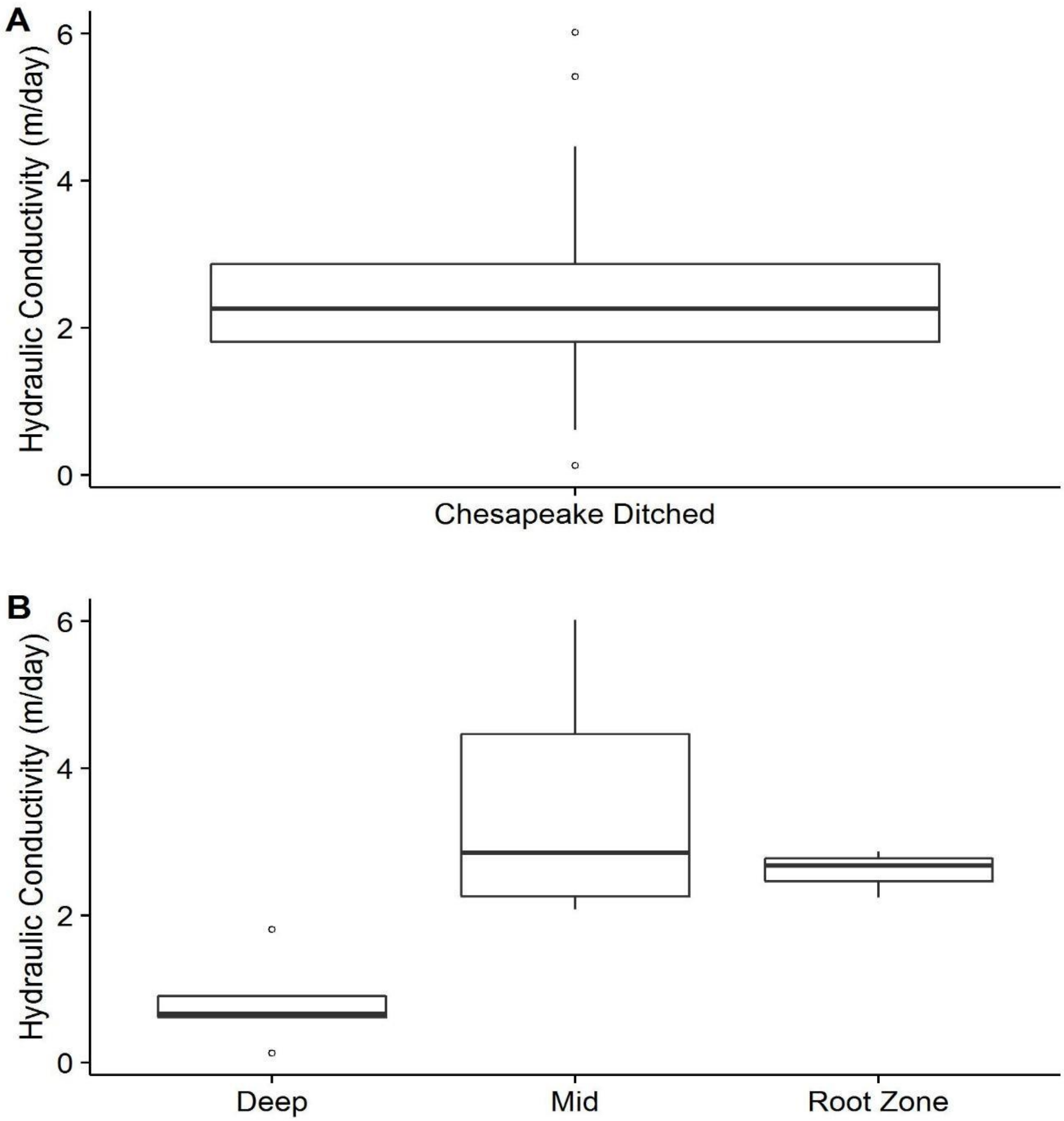


Figure 2.13: Hydraulic conductivity for Chesapeake Ditched site. (A) By Site, (b) By Depth. Data indicated higher hydraulic conductivity closer to the ground surface.

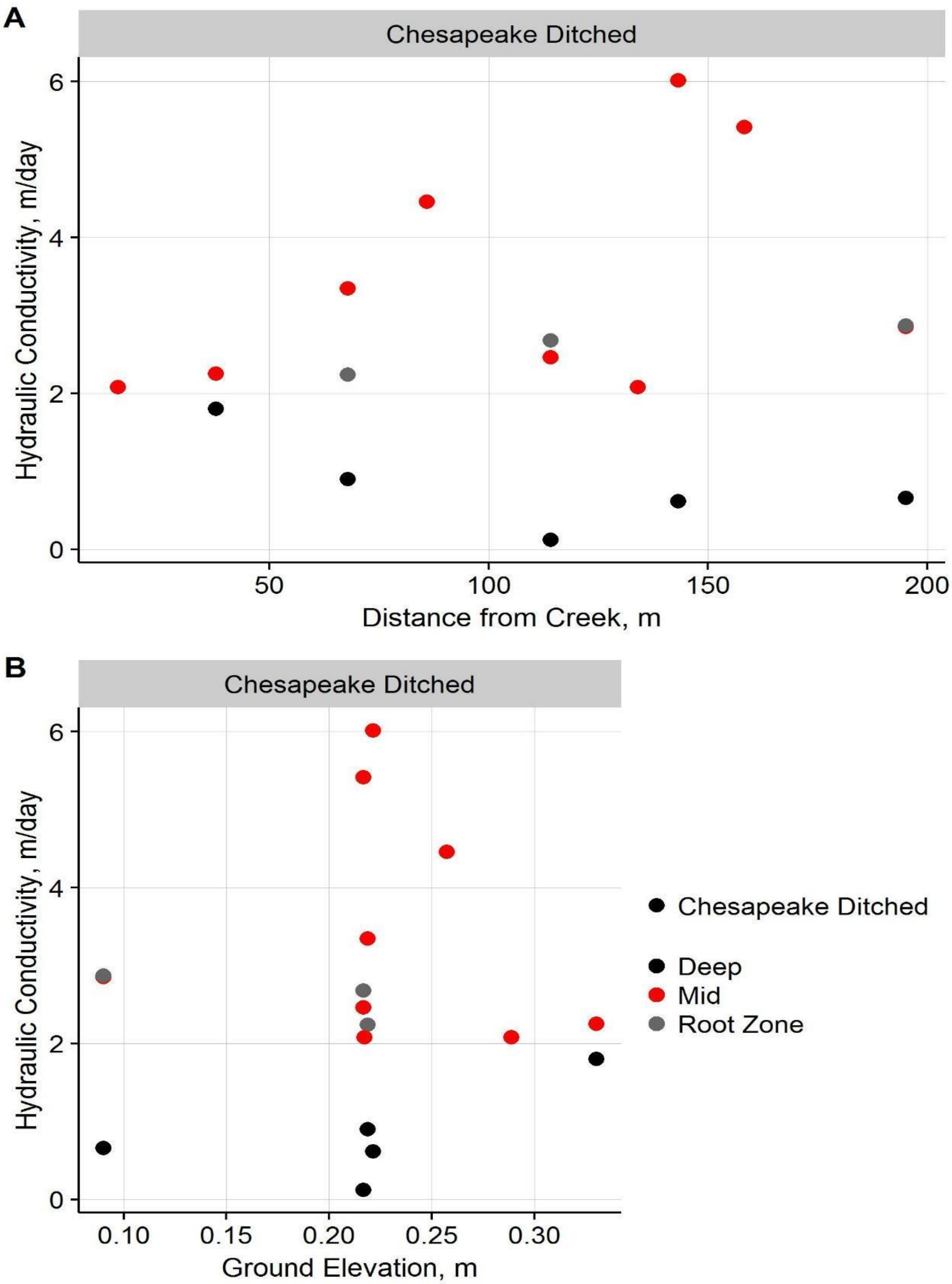


Figure 2.14: Hydraulic conductivity for Chesapeake Ditched site versus (A) distance from the tidal creek and (B) elevation.

Atlantic Sites

Tidal Range Data:

For the Atlantic sites, the closest tide gauge was the NOAA tide gage located at Ocean City, Maryland (Figure 1.9). The NOAA Tides and Currents webpage indicates that this station is located at 38° 19.7' N, 75° 5.5'W. It had a maximum water level referenced to Mean High High Water (MHHW) of 1.1 meters on February 05, 1998 and minimum water level referenced to MHHW of -0.73 meters on February 17, 2007. Hurricane Sandy did not produce the highest tidal range at this station, although it did at the Chesapeake Bay sites on record. This station has a mean tidal range of 0.64 meters.

The tidal range at marsh sites was determined using tide stage gages and interior groundwater wells at the Atlantic Coast sites. The average tidal range in the unditched tidal creek was 12.0 cm. The tidal creek near the ditched sites averaged 8.4 cm, but the tidal range in the ditches averaged 7.1 cm (Table 2.3); ditches have lower tidal ranges than adjacent tidal creeks, which is a different pattern than observed at the Chesapeake sites. The tidal range in the ditches at the ditched site show a tidal range of 7.1 cm. Comparison of these data to the NOAA Ocean City tide gauge tidal range of 66.6 cm produces an attenuation factor of 5.5 to 9.5 (Figure 2.15).

Marsh interior wells indicated further attenuation of the tidal signal. The marsh interior wells at the ditched site indicated an average tidal range of 4.2 cm. In comparison, the interior marsh wells at the unditched site indicated an average tidal range of 2.2 cm. Average tidal range of all water level monitors at the Ditched was 5.7 cm for the project time period and unditched averaged 4.5 cm (Table 2.3). At the 0.05

significance level, these data indicate that the tidal range of ditched and unditched sites came from non-identical populations.

Tidal range time series data (Figure 2.15) shows that regardless of site (Ditched vs Unditched), there is no distinct change in tidal range during the year. This graph also shows that regardless of well location (Ditched, Interior, Unditched, Tidal creek) tidal range does not fluctuate significantly during the year with the exception of Unditched tidal creek tide gage.

Comparing Atlantic Ditched site interior wells to Atlantic Unditched site interior wells indicates that ditched sites have higher daily tidal ranges, which extend to above 25 cm than their unditched counterpart wells but their averages remain similar (4.23 vs 2.3 cm).

In the ditched site, interior wells showed smaller ranges than wells located in the ditches. The unditched site showed tidal range decreased as distance from tidal creek increased (Figure 2.16). The average of the values plotted on the graphs decline from ~ 15 cm to ~ 8 cm over a distance of 100 m. Comparing interior marsh tidal range to distance from the ditch, Transect showed less larger tidal ranges. Transect 3 (back of site) showed the smallest range.

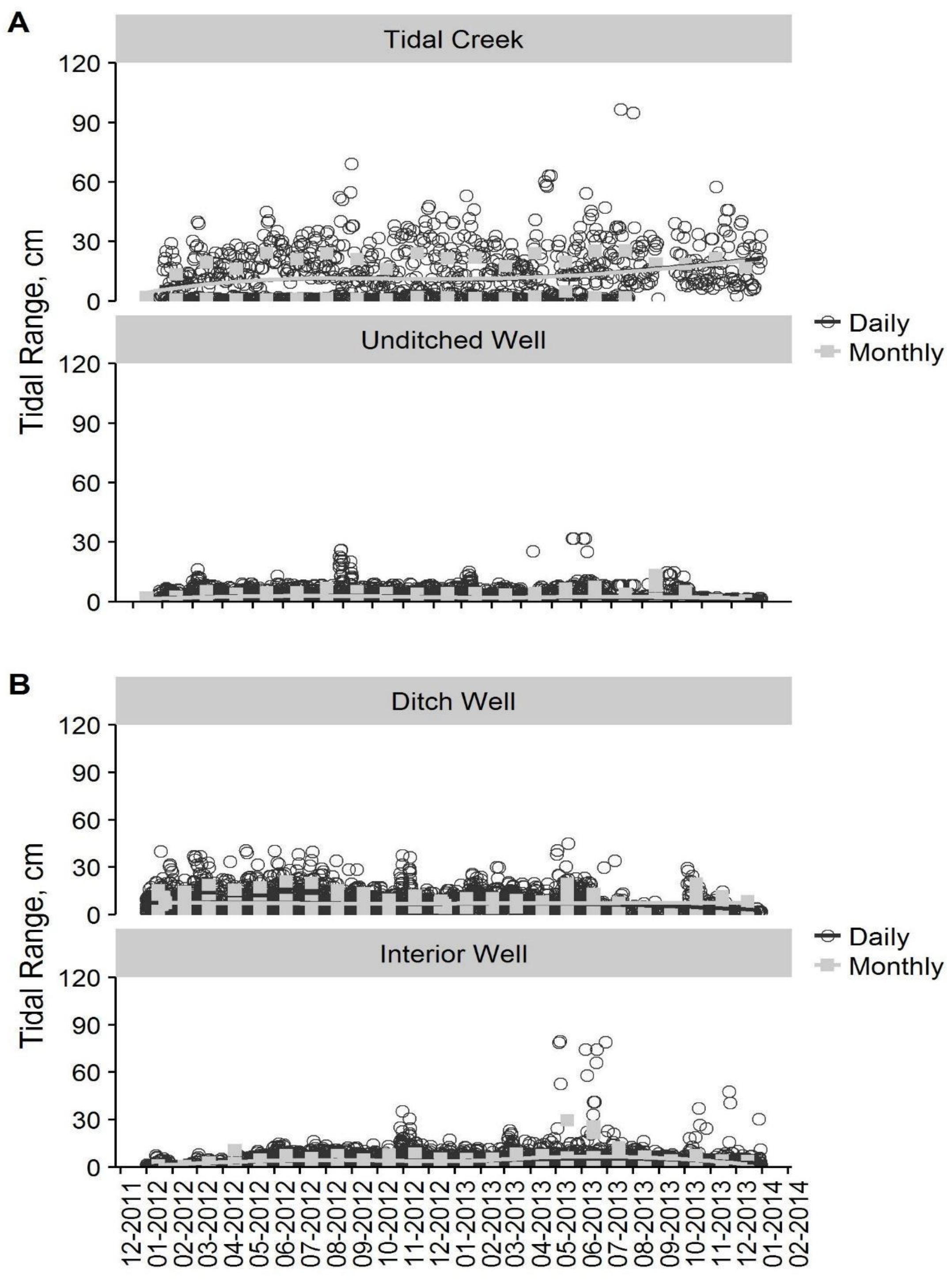


Figure 2.15: Tidal range for 2 years during pre-restoration period for (A) Atlantic Unditched site and (B) Atlantic Ditched site

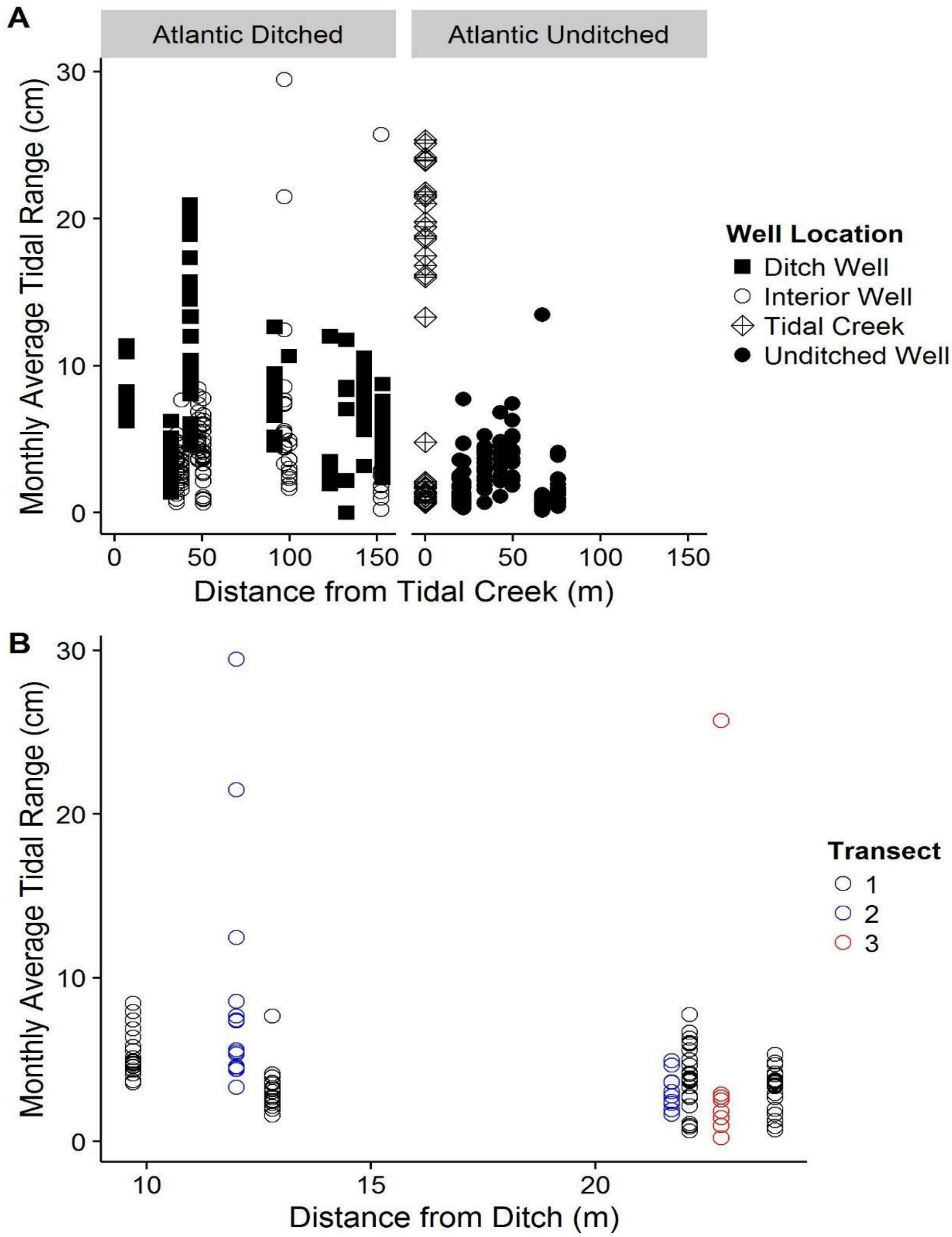


Figure 2.16: (A) Tidal range, cm, plotted against distance away from the tidal creek (m) for Atlantic ditched site and unditched site (B) Interior marsh wells tidal range, cm, plotted against distance away from ditch

Table 2.3: Summary of Tidal range (cm) for Atlantic Bay Sites

	Atlantic Unditched	Atlantic Ditched	NOAA
Site	4.5	5.7	66.7
Interior Well	-	4.2	-
Ditch Well	-	7.1	-
Unditched Well	2.2	-	-
Tidal Creek	12.0	8.4	-

Values of specific yield (Sy) were obtained by manipulation of tidal attenuation equation presented earlier: Sy = Damp/Kb, where b is the aquifer thickness in contact with the tidal creek or ditch. The amplification of the tidal amplitude in the ditches and the higher tidal range in ditched marsh interiors suggests that the ditch tidal amplitude should be used to calculate Damp and Sy. The value of b is not known for the marshes, but it is likely in the range of 1-4 m for the ditches and 3-10 m for the tidal creeks. Calculations of Sy using these boundaries indicates that Sy is in the range of 0.01 to 0.15. An example of a calculation is shown in table 2.4. Previous studies of specific yield range from 0.9 in *Sphagnum* peat to 0.05-0.15 for sedge peats. The data from these coastal marshes suggests that they have low Sy similar to sedge peats. These low values of Sy indicate the water holding capacity of the peat and suggest that small amounts of precipitation could generate significant increases in water table elevation in these marshes (Table 2.4).

Table 2.4 Summary of Tidal Ranges and Amplitudes

Well	Well Location	Distance From Creek (m)	Distance From Ditch (m)	Amplitude (m)	Amplitude attenuation factor, A From Creek	Amplitude attenuation factor, A From Ditch	Damp From Creek	Damp From Ditch	Specific Yield From Creek	Specific Yield From Ditch
		Ditched Site								
6	Tidal Creek	7	-	0.042						
1	Ditch Well	43.1	-	0.087						
2	Interior	50.6	22.1	0.020	0.47	0.49	27370.41	5690.30	0.001	0.004
3	Ditch Well	43.6	-	0.041						
4	Interior	35.6	24	0.016	0.39	0.40	8630.52	4200.06	0.003	0.006
5	Ditch Well	32.4	-	0.019						
14	Interior	47.8	9.7	0.027	0.64	0.66	70777.76	3382.17	0.000	0.007
15	Interior	38.3	12.8	0.016	0.37	0.39	9201.46	1097.39	0.003	0.021
16	Interior	97	12	0.044	1.03	1.08	51097254.6	133083.8	0.000	0.000
17	Interior	99.7	21.7	0.015	0.37	0.38	59537.47	3107.43	0.000	0.007
211	Ditch Well	91.5	-	0.040						
7	Ditch Well	123.2	-	0.016						
9	Ditch Well	142.6	-	0.039						
10	Interior	152.3	22.8	0.022	0.53	0.58	344086.00	10476.4	0.000	0.002
11	Ditch Well	153.2	-	0.026						
		Unditched Site								
1	Tidal Creek	0	-	0.008						
8	Tidal Creek	0	-	0.102	Using Tide Gage #1					
2	Unditched Well	42.8	-	0.018	2.31		15855.77		0.0020	
3	Unditched Well	33.9	-	0.015	2.00		14415.71		0.0022	
4	Unditched Well	19.6	-	0.007	0.89		159484.08		0.0002	
9	Unditched Well	49.8	-	0.021	2.68		15378.32		0.0021	
5	Unditched Well	21.8	-	0.010	1.29		44114.95		0.0007	
6	Unditched Well	66.7	-	0.006	0.82		652410.56		0.0000	
7	Unditched Well	75.8	-	0.008	1.05		13517937.3		0.0000	

Site Averages Sy	
Ditched From Creek	0.0010
Ditched From Ditch	0.0068
Unditched	0.0011

Soil Hydraulic Properties

Soil cores indicated that the marsh sediment at both ditched and unditched Atlantic site was primarily peat up to depths of 90 cm with an underlying sandy loam layer from soil texturing analysis (Figure 2.17). The soil horizons were determined in the field and lab from soil texture analysis (Figure 2.18). At two plots, located farthest from the tidal creek, a sandy loam was found at deeper depths. Sandy loam was also encountered during well installation at deeper depths below 100 cm at all plots, although not formally surveyed (Figure 2.18).

Figure 2.17: Example of a soil core extracted from the site.

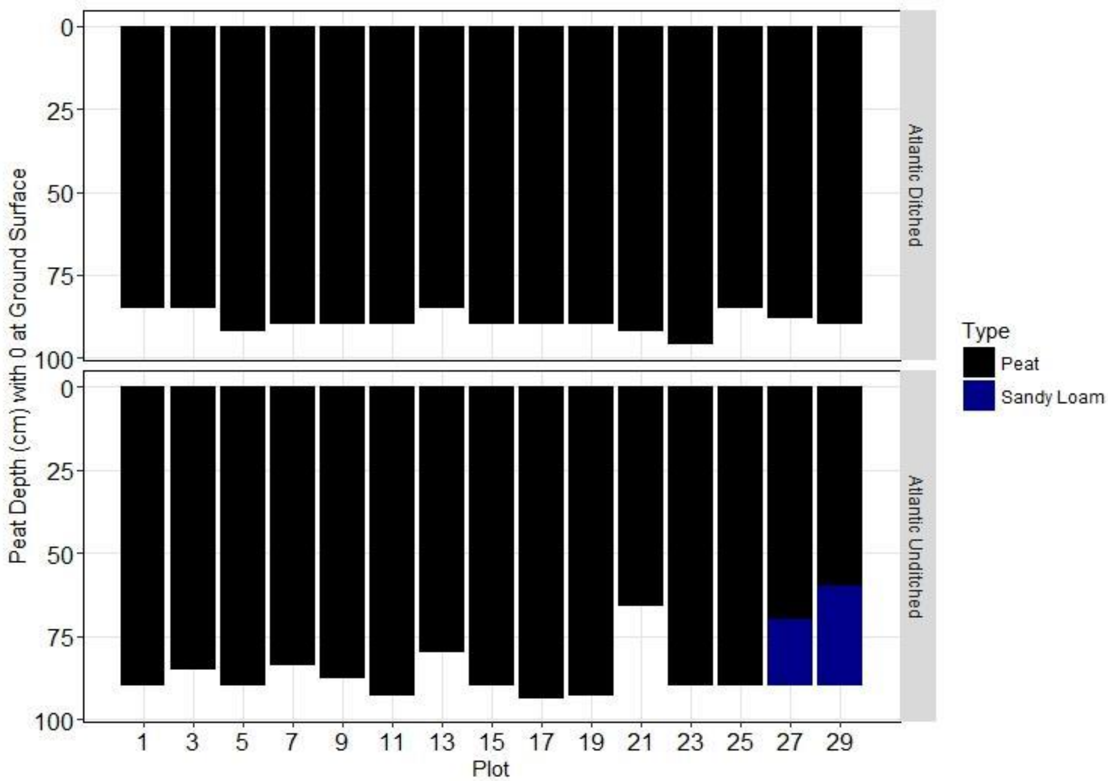


Figure 2.18: Soil Stratigraphy for Atlantic Ditched and Unditched sites. Data indicate that the upper 75 cm of the marsh sediment is predominately peat.

Bulk density values were similar for the ditched and unditched sites. Bulk density values exhibited a wide range of values, ranging 0.01 to 1.2 g/cm^3 for the unditched marsh sites and 0.04 to 0.9 g/cm^3 for the ditched site. The root zone (5-20cm) had the lowest bulk density values in both ditched and unditched marshes. The deepest soil cores (50-100 cm) had the highest bulk density values (Figure 2.19). The unditched site showed a larger range for bulk density values at each depth compared to values obtained from the ditched site. A comparison of the two sites indicates similar values for the root zones. Analysis indicates that the bulk density was not statistically different between the ditched and unditched sites. Average bulk density was 0.18 g/cm^3 for the ditched site and 0.23 g/cm^3 for the unditched site (Table 2.5). The variations in bulk density with position within the marsh were also examined. Bulk density was also highest closest to the tidal creek in the ditched site, but lower closer to the tidal creek at the unditched site (Figure 2.20 and Figure 2.21).

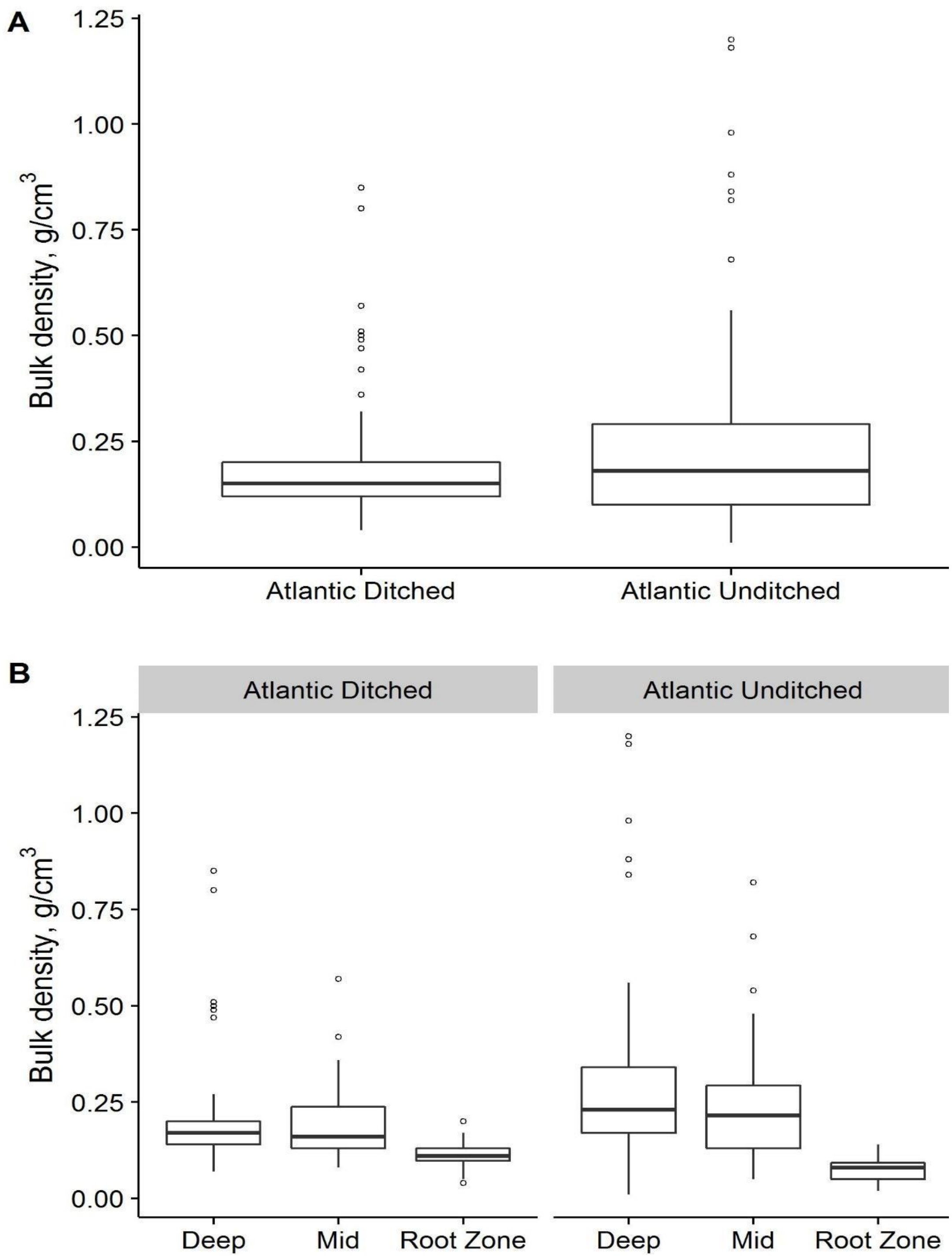


Figure 2.19: Bulk Density for Atlantic Bay Site Ditched and Unditched by site and by depth.

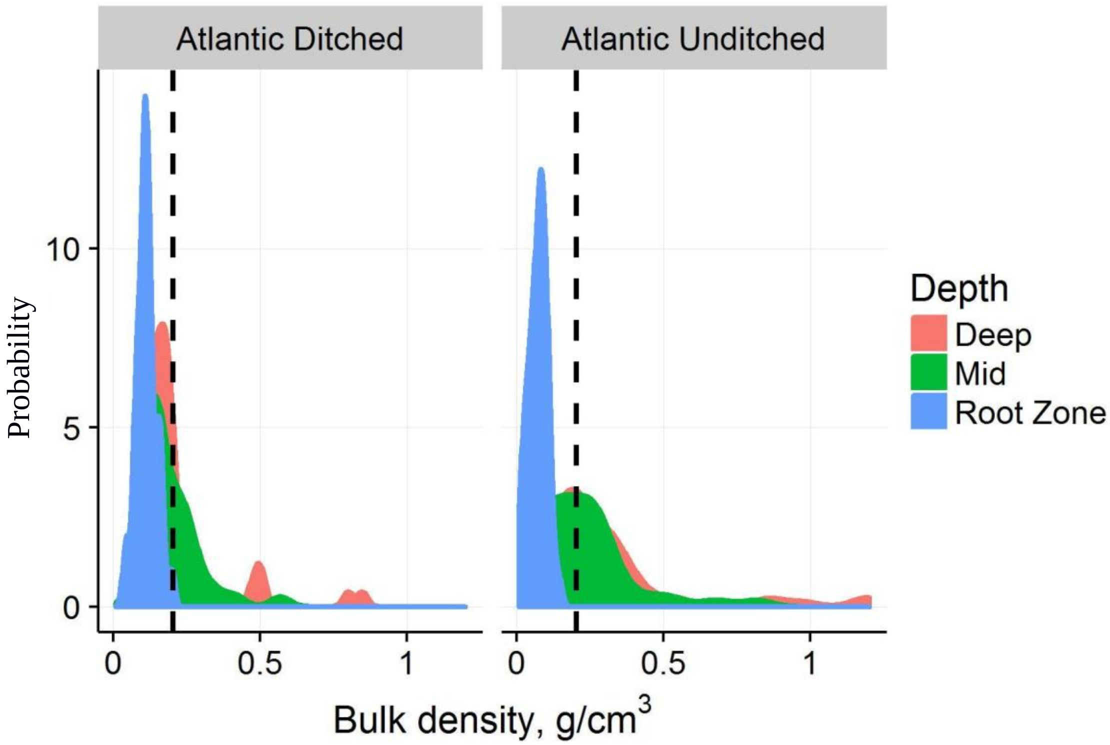


Figure 2.20: Probability density curves for bulk density for Atlantic Bay Site Ditched and Unditched marsh soils separated by depth intervals.

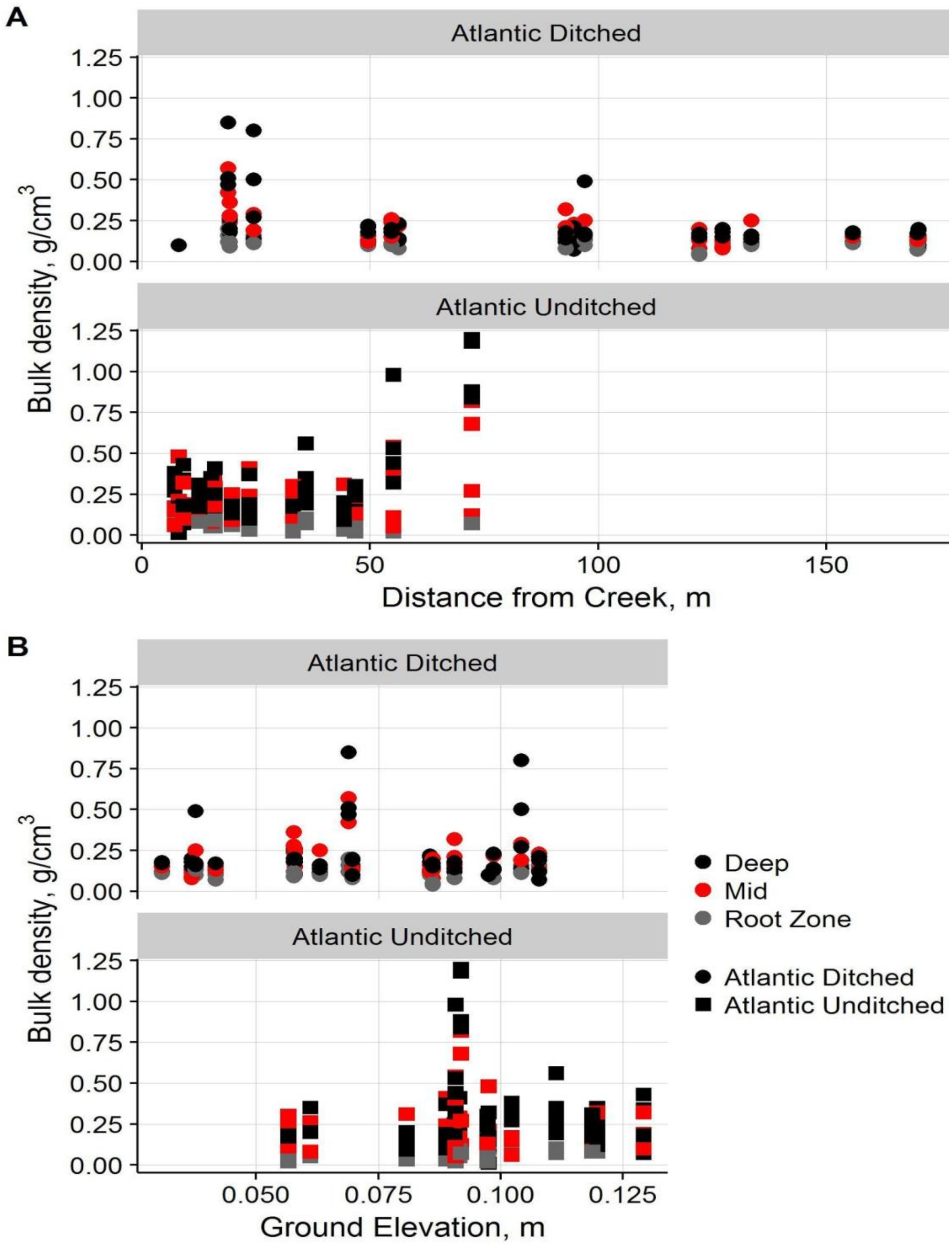


Figure 2.21: Bulk Density for Atlantic Bay Site vs Distance and Elevation. Bulk density does not vary with distance from the creek or site elevation.

Hydraulic conductivity was determined from *in*-situ slug tests at both ditched and unditched marsh sites. Comparing the ditched and unditched sites, these data indicated similar averages and ranges of hydraulic conductivity values, K, at both ditched and unditched sites. As was seen with the bulk density data, the largest contrasts in K were observed when K data were separated by depth intervals. The upper 2 soil horizons (0-20 cm and 20-50 cm) in the unditched sites have higher K values than the deepest horizon (Table 2.5). The spatial variation of K in the marsh was also examined. K was highest in the unditched site closest to the tidal creek (which also had the lowest bulk density values), but did not exhibit spatial variation at the ditched sites (Figure 2.22). Summary data comparing both bulk density and hydraulic conductivity at both ditched and unditched sites in the Atlantic sites are shown in Table 2.5. Standard deviations are shown in parentheses.

Table 2.5: Atlantic Sites BD and K	**Bulk Density (g/cm^3)**		**Hydraulic Conductivity (m/day)**	
	Ditched	**Unditched**	**Ditched**	**Unditched**
Site	0.18 (0.1)	0.23 (0.2)	1.16 (0.9)	1.62 (1.8)
Root Zone	0.11 (0.03)	0.07 (0.03)	2.02 (0.8)	1.62 (1.2)
Mid	0.19 (0.1)	0.24 (0.2)	0.95 (0.6)	2.07 (0.9)
Deep	0.21 (0.2)	0.30 (0.2)	0.47 (0.5)	0.51(1.4)

( ) standard deviation

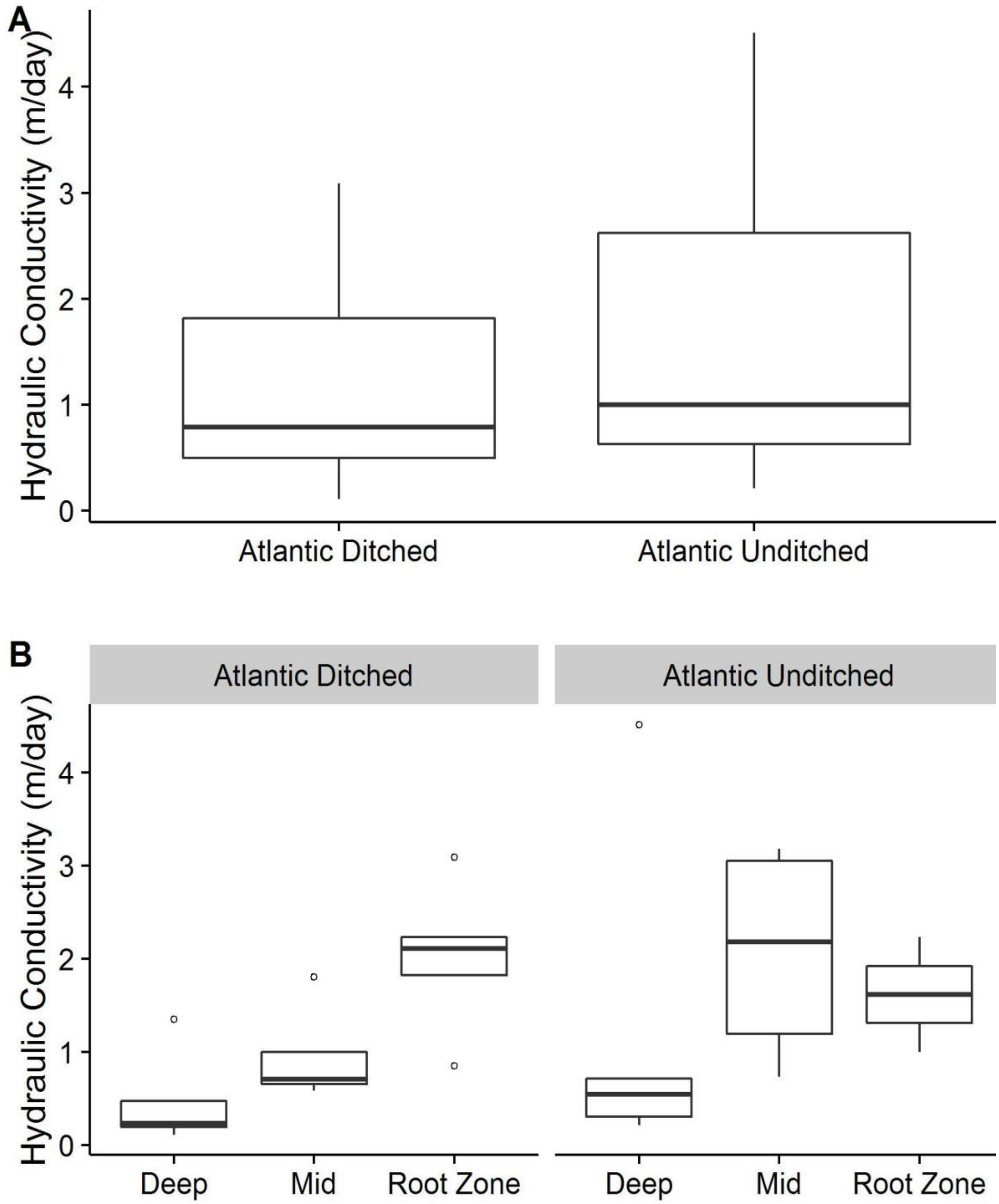


Figure 2.22: Hydraulic Conductivity for Atlantic Bay Sites. (A) K by site and (B) K by depth

Statistical Comparison of soil properties among ditched and unditched marshes at both Atlantic and Chesapeake sites

In this study, marsh tidal ranges and soil properties were examined in pairs of ditched and unditched marshes at both Atlantic Bay and Chesapeake Bay sites. Statistical t-Tests were used to compare the means of two groups under the assumption that both samples are random, independent, and come from normally distributed populations. In this analysis, p-value greater than 0.05 indicated that the averages of two groups are statistically similar. Hydraulic conductivity spanned across multiple magnitudes so a log transform was completed before the t-test.

Statistical analyses are shown in Appendix. Statistical analyses of bulk density indicated statistically different values for deep cores compared to root zone cores at all unditched and ditched sites (at both locations- Chesapeake and Atlantic). A comparison of Atlantic Unditched and Chesapeake Unditched sites t-test analysis indicates that these sites were statistically similar for both deep and shallow bulk density values.

The difference in hydraulic Conductivity between root zones (0-20 cm) and deep (50 – 80 cm) zones were statistically significant at both the Atlantic and Chesapeake Ditched sites. The Chesapeake Unditched site did not indicate a statistically significant difference between these two zones. Comparison of hydraulic conductivity between Atlantic and Chesapeake sites for specific depth intervals indicates no statistically significant difference among the sites (Table 2.6). As will be discussed later in Chapter 3, these similar K values occurs even though plant composition is different among the Atlantic and Chesapeake sites.

Table 2.6: Statistical Analysis – Soil Properties

	Hydraulic Conductivity t-test p=	**Bulk Density t-test p=**
Chesapeake Ditched Root vs Deep	**0.007**	**6.3 e-09**
Chesapeake Unditched Root vs Deep	**-**	**1.6 e-07**
Atlantic Ditched root vs deep	**0.013**	**0.0001**
Atlantic Unditched root vs deep	0.173	**3.3 e-07**
Atlantic Ditched root vs Chesapeake root	0.168	0.5997
Atlantic Ditched deep vs Chesapeake deep	0.318	1.0
Atlantic Unditched root vs Chesapeake root	-	0.999
Atlantic Unditched deep vs Chesapeake deep	-	0.69

Note: Values in Bold are statistically different at 5% level if significance

Discussion

This goal of this study was to compare soil hydraulic properties and tidal range characteristics in ditched and unditched coastal marshes and between Chesapeake and Atlantic marshes. The study marshes were chosen based on future ditch plugging restoration plans for the marshes by the Maryland Department of Natural Resources. It was hypothesized that the ditched marshes would have higher bulk density and lower hydraulic conductivity as a result of drainage and compaction. A literature review indicates that ditches in coastal marshes have substantially changed many marsh characteristics (Daiber 1986; Roman et al. 2000), but many of the previous studies are at marsh sites with significant creek tidal ranges. The sites chosen for this study have small tidal ranges (< 25 cm). In addition, significant attenuation of the tidal range was expected in marsh interiors due to the high hydraulic conductivity values and the unconfined nature of the marsh aquifers.

At the Atlantic Coast site, the upper 90 cm of soil in both ditched and unditched marshes is composed primarily of peat. Chesapeake Bay marshes had more mineral sediment, but soils were primarily peat to depths of 50 cm depth in both unditched and ditched marshes. Deeper Chesapeake soils were primarily loam soils. Both the Atlantic and Chesapeake Bay unditched marsh soil exhibited a range of bulk density values with most of the variation occurring as a function of depth. At both the Chesapeake and Atlantic sites, the root zone (0-20 cm) had the lowest bulk density. Higher values of bulk density were fund at depth. It was determined that bulk densities were statistically different between deep zones and upper shallow zones. The results obtained from this analysis support other studies that indicate an increase in bulk density with depth in peat soils.

In this study, ditched marshes did not have statistically significant differences in either bulk density or hydraulic conductivity compared to unditched marshes at the same site. The comparison indicated that differences exist between marshes Atlantic and Chesapeake sites. The Atlantic Ditched marshes had a lower bulk density and higher peat contents than the ditched marshes at the Chesapeake Bay sites. These differences may be related to the history of mineral soil deposition, or other factors that affect marsh organic matter and sediment accretion in these two marshes. Bulk density greater than 1.6 g/cm^3 tend to restrict root growth (McKenzie *et al.*, 2004). Sandy soils have higher bulk densities (1.3–1.7 g/cm^3) than fine silts and clays (1.1 – 1.6 g/cm^3) because of the larger, but fewer, pore spaces (NRCS). In clay soils with good soil structure, there is a greater amount of pore space because the particles are very small, and many small pore spaces fit between them.

Marsh drainage can cause collapse of pore space, and loss of organic matter, resulting in an increase in bulk density (Portnoy 1997). The data from this study suggesting that these processes have either not occurred in these marshes or that the marshes have recovered from the initial alteration. Previous works suggested that some restricted marshes can recover and establish an equilibrium characterized by low bulk densities (Ansfield 1999). Paquette et al (2004) indicates that for altered marshes that do have higher bulk densities compared to natural counterparts, restoration of tidal flow should lead to a decrease in bulk density through swelling of pore spaces with an increase in pore water pressure and an increase in the percentage of organic matter.

Both the Chesapeake and Atlantic marsh sites showed small tidal ranges and little tidal range fluctuation during the 2-year study period compared to NOAA tide gages located at Open Ocean and open Chesapeake Bay sites. Tidal range has been suggested as a factor that can affect marsh vulnerability to sea-level rise. It has been suggested that microtidal marshes are most vulnerable to sea level rise (Steavenson & Kearny 1986).

Marsh water levels did not exhibit significant variations that corresponded with tidal fluctuations or seasonal variations. These responses might be due to the strongly attenuated tidal signals in the marshes along with nearly constant inputs of precipitation and losses due to evapotranspiration and drainage. The main hydrological change exhibited in the marshes was the response to Hurricane Sandy, which will be described in more detail in the Chapter 4. Differences in tidal range for the creek data for the ditched and unditched sites suggests that the Chesapeake and Atlantic pairs have somewhat different characteristics. Ditches have higher tidal ranges than adjacent tidal creeks, suggesting that ditches enhance tidal range, primarily by generating lower low values (this is observed in the time series data – Chapter 3). Tidal range values in the interior portions of the marsh follow the pattern and values of the ditches. The unditched sites indicated heterogeneity in unditched wells farthest from the tidal creek.

Higher interior tidal range has been associated with marsh building processes and the lessening of stressful soil conditions and is directly related to marsh surface elevation (Chmura et al. 2001). Morris et al. (2002) determined that flooding in South Carolina marshes had positive effects on primary production and marsh accretion, yet flooding beyond optimal levels negatively impacted plants and marsh accretion processes. This study showed ditched marshes had increased tidal ranges.

Chapter 3: Impact of ditches on Eastern Shore Maryland Salt Marsh Groundwater levels and Plant Species Composition

Introduction

Coastal marshes have a long history of anthropogenic modifications (dikes, impoundments, ditching, open water management systems, etc.). These modifications have caused extensive changes to the structure and function of coastal ecosystems (Daiber 1986). In the 1930's, extensive digging of ditches to control mosquito populations redefined most marsh landscapes and ecosystems along the East Coast. It is estimated that 90% of marshes from Maine to Virginia have been ditched (Bourn and Cottam 1950).

Ditch networks can change the hydrological, geomorphological, and ecological processes within coastal marshes. Ditches constructed for mosquito control are spaced approximately 40 m apart and are reported to lower the marsh water table (Stearns 1940; Bourn and Cottam 1950; Turner 1997; Adamowicz 2005; Gedan 2009); other studies have shown that ditching did not result in lower water levels relative to the surface (Vincent 2013). Stearns et al. (1940) showed that the water table 2 years after ditches were installed was lowered by about 4 cm Delaware tide water marsh.

Ditches also have been shown to drain and decrease the prevalence of natural pools on marsh surfaces, which are important to fish and waterfowl (Adamowicz 2005; Lathrop, Cole, and Showalter 2000); and ditches replace tidal creek functions, leading to a decreased density of natural creeks (Adamowicz and Roman 2005). These changes in marsh hydrology caused by ditches are associated with changes in vegetation patterns (Bourn and Cottam 1950; Daiber 1986; Miller and Egler 1950; Niering and

Warren 1980; Clark et al. 1984; Raposa and Roman 2001) and decreased bird habitat within ditched salt marshes.

Efforts are underway to restore the hydrology and ecological functions of ditched salt marshes by removal of ditches through filling or plugging. To provide restoration guidance, it is important to examine the current relationships between marsh plant species distribution and marsh hydrological, geomorphological, and ecological processes of ditched coastal marsh systems in comparison to unditched marsh systems.

Objectives and Hypotheses:

The dissertation chapter sought to compare hydrological processes in ditched marshes with adjacent unditched marshes using (1) groundwater data to determine the responses to seasonal, tidal, and storm influences (2) salinity, and (3) vegetation species composition data. It was hypothesized that marsh ditches are draining the marsh interiors (as opposed to supplying them) and that water is contained in the ditches at relatively constant levels. It was also hypothesized that marsh water level duration at or the near the surface would be significantly lower in ditched marshes.

Methods

Two types of measurements were made in order to facilitate a comparison of ditched and unditched marshes: (1) Vegetation cover was determined from field sampling, (2) hydrological monitoring networks were established to generate time series of precipitation; wind speed; creek, ditch, and marsh water levels; and salinity for the time period 2011-2013. The hydrological monitoring networks are described in Chapter 2. Measurement techniques are described in the following sections.

Vegetation Cover

Vegetation base maps of study sites were developed using field data and imagery analysis and include coverages of vegetated, barren, open water, and SAV in ponds. Vegetation monitoring was conducted in July 2012 and 2013. Plant communities are described at two spatial scales: site and plot. At the site scale, broad plant communities will be delineated based on aerial photos and ground observations.

For the plot scale, marked sampling plots were setup along well transects. Plots were randomly selected by walking parallel to the ditches (perpendicular to the main tidal channel) followed by a random number distance perpendicular to the ditch. The unditched sites mimicked this plot setup, but with the initial transect established perpendicular to the main tidal channel followed by a random number location perpendicular to the transect. A center pole was installed into the ground. From the center pole an attached rope was used to determine the quadrant (Figure 3.1). At each sampling location, vegetation cover classes were assigned in a marked 5 meter diameter circle. Methods following Peet et al. (1998) were used. This method involves assigning one of 10 cover classes (Table 3.1) for each species in the designated area. The upper limit of species compostion (MAX %) was recorded followed by the species type.

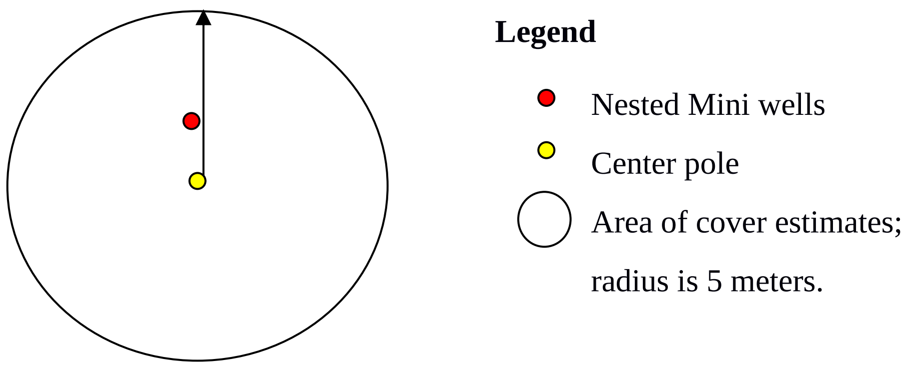


Figure 3.1: Vegetation plot layout and sampling area

Table 3.1: Cover Class breakdown and percentages

MAX%	Cover classes	Midpoints %	Range of cover %
Trace	1	0	TR
1	2	0.5	0-1
2	3	1.5	1-2
5	4	3.5	2-5
10	5	7.5	5-10
25	6	17.5	10-25
50	7	37.5	25-50
75	8	62.5	50-75
95	9	85	75-95
100	10	97.5	95-100

Hydrological Monitoring

Detailed methods of hydrological monitoring equipment can be found in Chapter 2. In addition to difference in hydraulic head, temperature, pH, and salinity. Salinity, temperature, and pH was measured using handheld multiprobes.

Data on precipitation, relative humidity, and wind speed and direction were also obtained from the weather stations located in the middle of each ditched site. Weather data were also downloaded from wunderground from a nearby station (reference). Measurements were recorded every 15 minutes. Tide gauge data were obtained from National Oceanic Atmospheric Administration (NOAA) at their Ocean City (Station ID: 8570283) and Bishop Head (Station ID: 8571421) gages shown in Figure 1.9.

Results:

Chesapeake and Atlantic site results will be presented separately.

Chesapeake Sites

Vegetation:

The plant species cover at the marsh sites was determined from the species sampling. Comparison of vegetation species composition (%) with distance from the tidal creek (m) and ground elevation (m) showed no relationship regardless of site (Ditched/Unditched) and regardless of species (Figure 3.2).

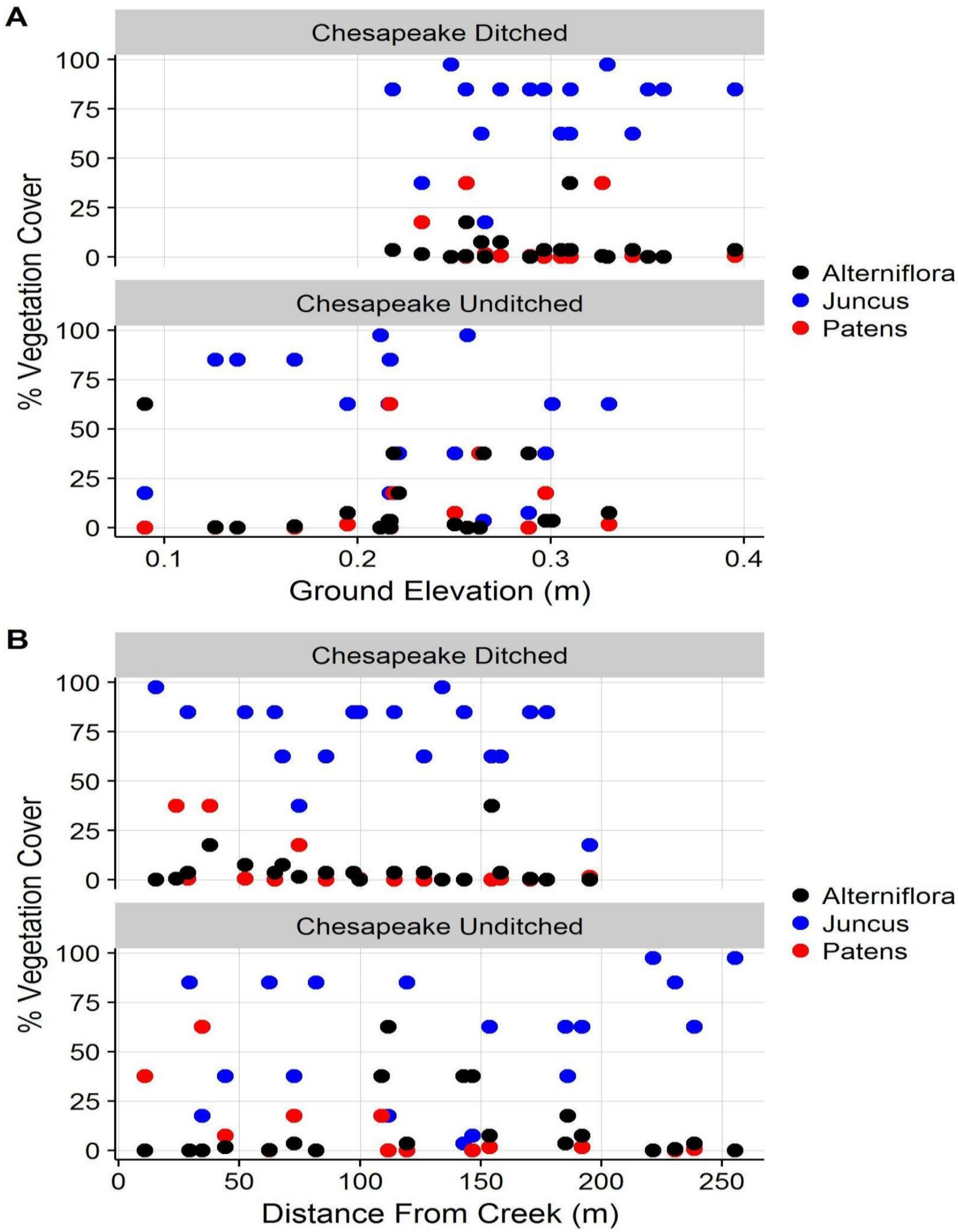


Figure 3.2: Percent cover of the dominant species at the Chesapeake Ditched and Unditched sites versus distance from tidal creek and ground elevation.

The main plant species are as follows: *Juncus roemerianus* (black needlerush), which provided 52.3% of the plant cover at the unditched sites and 70.1% of the plant cover at the ditched sites; *Spartina patens* (marsh hay cordgrass), which provided 10.2% of the cover at the unditched sites and 5.4% at the ditched sites; and *Spartina alterniflora* (smooth cordgrass), which provided 11.2% of the plant cover at the ditched site and 4.9% at the unditched sites. These data indicate that the dominant plant species at the Chesapeake Bay site is *Juncus roemerianus,* because it has the highest percent cover with 16% more area at the ditched sites. These data also conclude that *Spartina patens* is higher in the unditched site. Conducted two sided t-tests, however, showed that the Unditched and Ditched sites, for all species, were statistically similar (Table 3.2).

Table 3.2 Vegetation Composition top three species

	Spartina Alterniflora	***Spartina Patens***	***Juncus roemerianus***
Chesapeake Ditched	5.4 %	4.9 %	70.1 %
Chesapeake Unditched	10.2 %	11.2 %	52.3 %
t-test p values	0.166	0.303	0.057

Salinity characteristics

Salinity was sampled synoptically on four dates in the spring and summer of 2013 and 2014 using multimeter probes. The lowest salinity values occurred in March, and the highest values were observed in August. The Chesapeake unditched site showed higher salinity than its ditched counterpart sitewide and for each sample date. Salinity values from all wells within each site, the ditched site salinity values ranged from 8-18 ppt; the unditched site was higher, around 10-21 ppt (Figure 3.3). Figure 3.4

scatter plot shows that salinity in the ditched site decreased with increase in distance from tidal creek and increased with increase in elevation for all sample dates. No relationship was found in the unditched site with respect to ground elevation or distance from the tidal creek.

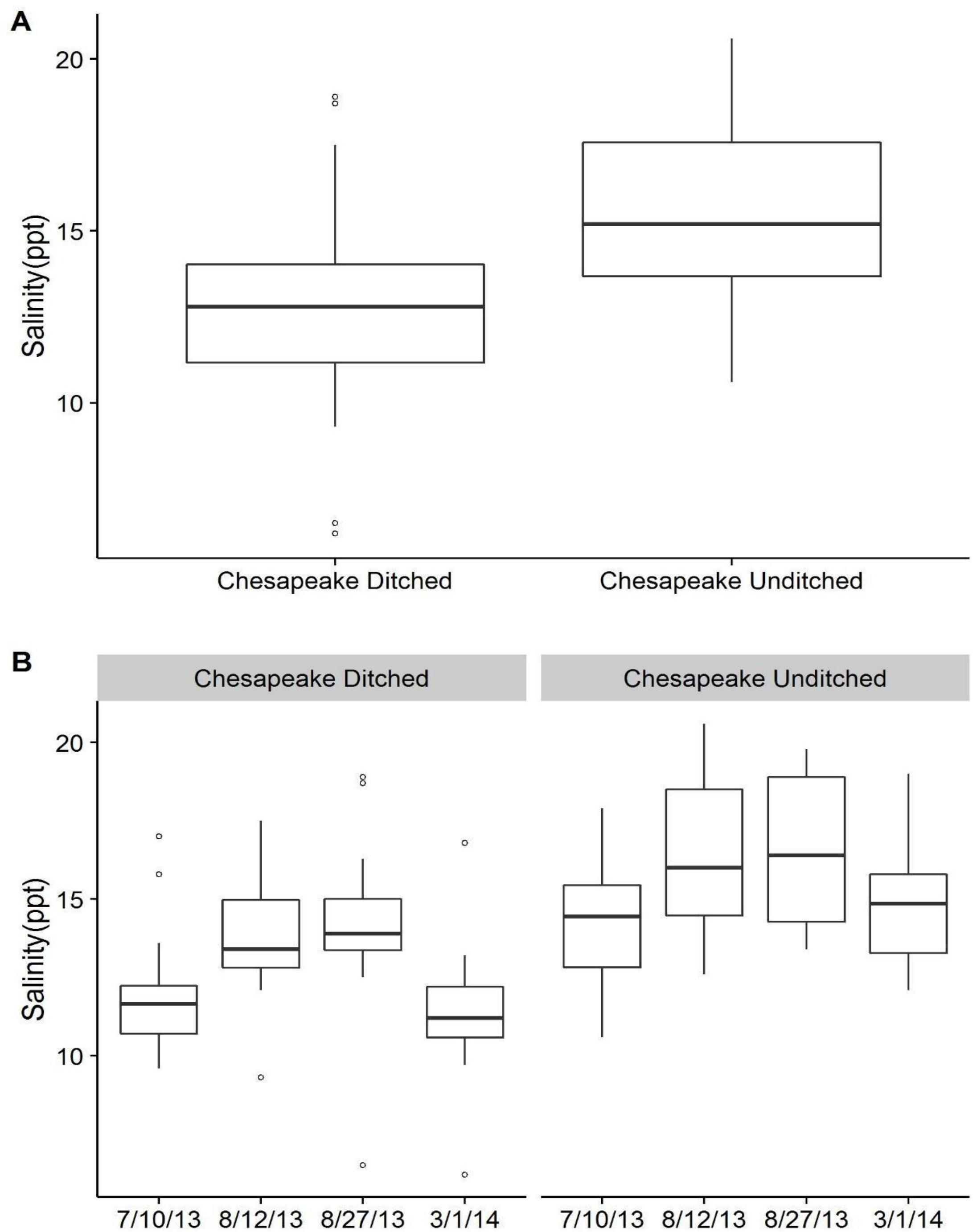


Figure 3.3: Salinity by site and by sample date for Chesapeake Ditched and Unditched Sites.

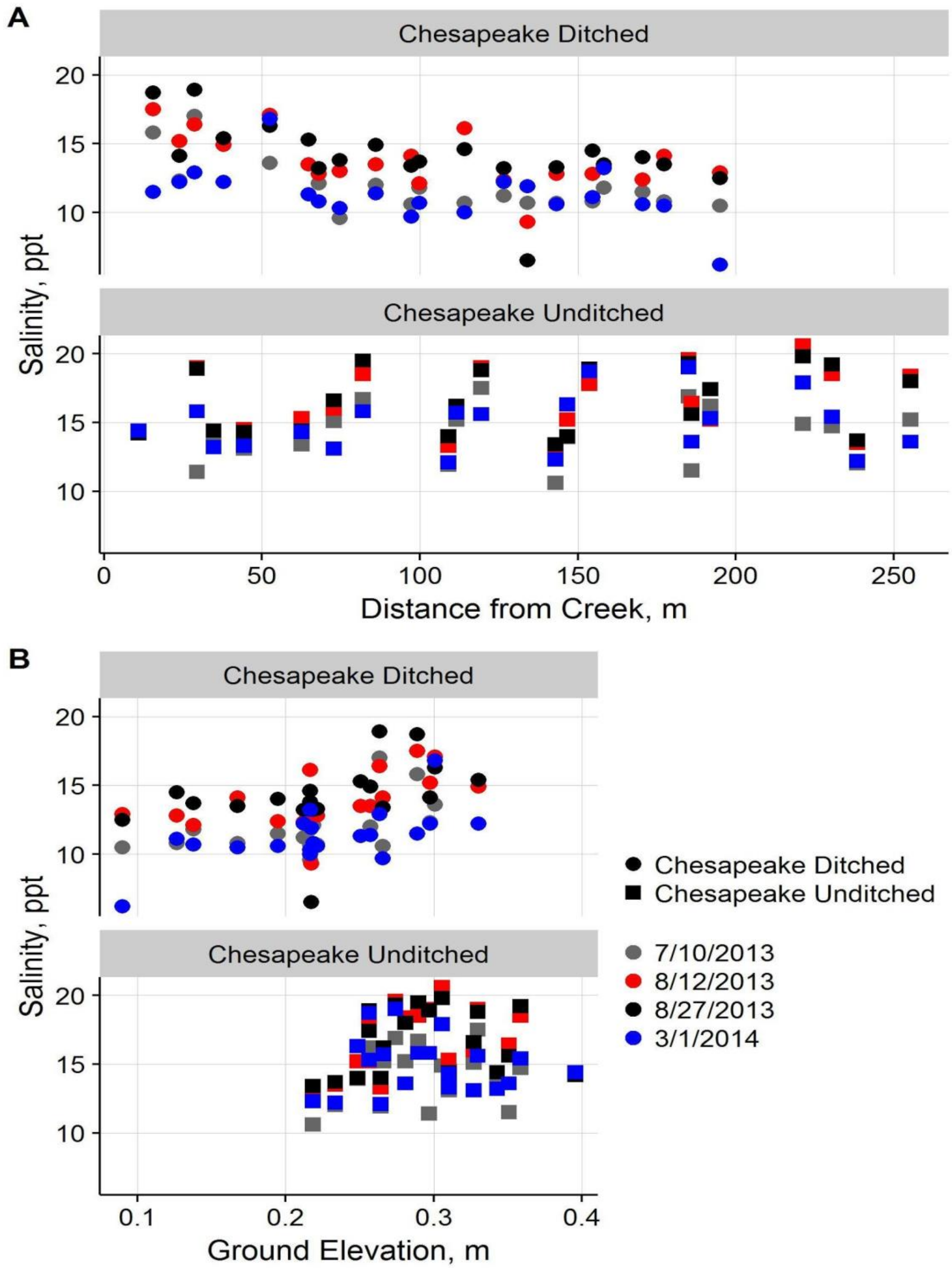


Figure 3.4: Salinity for Chesapeake Ditched and Unditched Sites versus distance from tidal creek and ground elevation.

Time series data of precipitation data

Precipitation data at the study site indicates that precipitation occurs throughout the year without a pronounced seasonal pattern (Figure 3.5). The consistent slope of cumulative precipitation indicates no major seasonality of precipitation. The major step in the cumulative diagram is associated with precipitation from Hurricane Sandy, which is addressed in Chapter 4 (Figure 3.5). Although precipitation depth is not seasonally distributed, summer storm events have larger magnitudes and more intense than winter storms (Figure 3.5).

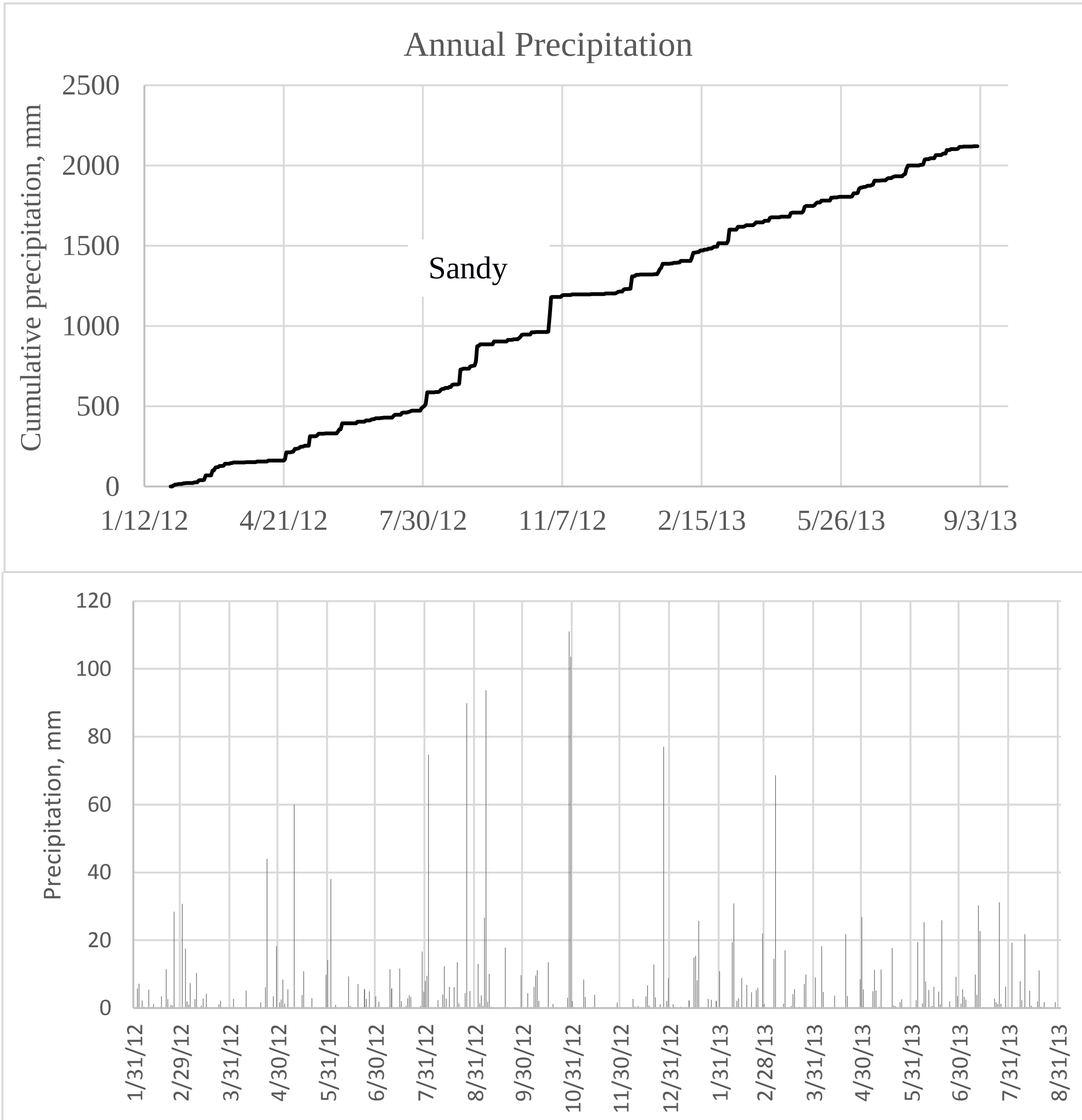


Figure 3.5: Precipitation data for the study period: upper diagram is the cumulative precipitation, which indicates precipitation distributed throughout the year, but a significant step for Hurricane Sandy. Lower diagram indicates individual storms. Note the larger storm size for the summer months: July through November in 2012.

Groundwater Piezometric Data

Hydraulic heads in nested mini wells were measured by hand to determine vertical hydraulic gradient on individual dates, which were primarily during the spring and summer months. Table 3.3 shows hydraulic gradient is most often slightly downwards (infiltrating) or hydrostatic. Infiltrating vertical gradients could result from either overbank flows or precipitation events. The unditched site had nonzero hydrostatic gradients more frequently than the ditched site, but all of the head difference data are small, so it is not known whether these data are significant. During hydrostatic conditions, flow will be primarily horizontal if horizontal gradients are present.

Table 3.3: Vertical Hydraulic Gradients for Chesapeake Ditched Site

Plot	P#	5/26/13	6/12/13	7/14/13	8/12/13	Average	Condition
1	1	-0.141	-0.040	-0.202	0.131	-0.06	**Downwelling**
1	3						
3	1	-0.023	-0.036	0.061	-0.048	-0.01	**Hydrostatic**
3	3						
5	1	-0.236	-0.055	0.025	-0.042	-0.08	**Downwelling**
5	3						
7	1	-0.042	-0.011	0.131	-0.112	-0.01	**Hydrostatic**
7	3						
9	1	-0.194	-0.090	-0.145	0.046	-0.10	**Downwelling**
9	3						
11	1	-0.141	-0.145	-0.192	0.124	-0.09	**Downwelling**
11	3						
13	1	-0.105	-0.204	-0.187	0.198	-0.07	**Downwelling**
13	3						
15	1	-0.187	0.023	-0.030	-0.057	-0.06	**Downwelling**
15	3						
17	1	-0.114	-0.040	-0.147	0.122	-0.04	**Downwelling**
17	3						
19	1	-0.059	-0.183	-0.288	0.019	-0.13	**Downwelling**
19	3						
21	1	-0.122	-0.040	-0.059	0.091	-0.03	**Downwelling**
21	3						
23	1	-0.061	-0.219	-0.029	0.048	-0.07	**Downwelling**
23	3						
25	1	-0.069	-0.128	-0.090	0.128	-0.04	**Downwelling**
25	3						
27	1	0.025	-0.105	-0.204	0.002	-0.07	**Downwelling**
27	3						
29	1	-0.027	-0.430	-0.008	0.004	-0.12	**Downwelling**
29	3						
31	1	0.080	0.086	0.013	-0.078	0.03	**Upwelling**
31	3						
33	1	-0.166	0.059	-0.027	-0.112	-0.06	**Downwelling**
33	3						
35	1	0.040	0.061	-0.006	-0.059	0.01	**Hydrostatic**
35	3						
37	1	-0.242	-0.042	-0.185	-0.042	-0.13	**Downwelling**
37	3						
39	1	-0.099	0.072	0.070	0.013	0.01	**Hydrostatic**
39	3						

Table 3.3: Vertical Hydraulic Gradient for Chesapeake Unditched

Plot	P#	5/26/13	6/12/13	7/14/13	8/12/13	Average	Condition
1	1	0.004	-0.284	0.013	0.210	-0.01	**Hydrostatic**
1	3						
3	1	-0.065	-0.297	0.036	0.173	-0.04	**Downwelling**
3	3						
5	1	-0.110	-0.065	-0.103	0.152	-0.03	**Downwelling**
5	3						
7	1	0.013	0.002	-0.072	-0.013	-0.02	**Downwelling**
7	3						
9	1	-0.051	-0.029	0.080	0.065	0.02	**Upwelling**
9	3						
11	1	0.002	-0.099	0.013	0.069	0.00	**Hydrostatic**
11	3						
13	1	-0.097	-0.183	0.158	0.036	-0.02	**Downwelling**
13	3						
15	1	-0.019	-0.183	0.030	0.095	-0.02	**Downwelling**
15	3						
17	1	0.168	-0.023	-0.284	0.171	0.01	**Hydrostatic**
17	3						
19	1	-0.133	0.032	0.055	0.078	0.01	**Hydrostatic**
19	3						
21	1	0.048	-0.175	-0.208	0.110	-0.06	**Downwelling**
21	3						
23	1	-0.170	-0.019	-0.110	0.036	-0.07	**Downwelling**
23	3						
25	1	-0.027	-0.070	-0.267	0.095	-0.07	**Downwelling**
25	3						
27	1	-0.149	0.204	-0.091	0.004	-0.01	**Hydrostatic**
27	3						
29	1	0.076	0.023	0.034	-0.070	0.02	**Upwelling**
29	3						
31	1	-0.069	-0.013	-0.032	0.017	-0.02	**Downwelling**
31	3						
33	1	-0.093	-0.070	-0.057	0.030	-0.05	**Downwelling**
33	3						
35	1	-0.048	-0.015	-0.057	0.050	-0.02	**Downwelling**
35	3						
37	1	0.114	0.029	0.088	0.038	0.07	**Upwelling**
37	3						
39	1	-0.046	-0.156	-0.002	0.088	-0.03	**Downwelling**
39	3						

Interior marsh, creek, ditch, and marsh groundwater levels

Water levels at the Chesapeake ditched marsh site were observed to be at or near the ground surface, whereas the unditched site water level was predominantly below ground (Figure 3.6). No distinct seasonal patterns were observed in the tidal creek, ditch, or marsh interior water level data in terms of maximum water levels, though there do appear to be distinct time periods where low water levels were at a minimum. The time series data indicate periods of low minima in the time intervals associated with the largest precipitation events. Although the mechanism for the pattern of low minima is unknown, one possibility is that terrestrial runoff into these tidal embayments elevates the creek or ditch water levels, resulting in less drainage from these features at low tide. Water-level fluctuations in the ditches are less regular than those in the interiors or in the unditched marsh. Other evidence of sensitivity to precipitation events is also observed; the numerous high water level events appear to correspond to storm events. The largest event shown is Hurricane Sandy, which occurred on 10/30/2012. Water levels are generally higher during fall/winter months, and lower in the spring/summer (Figure 3.6).

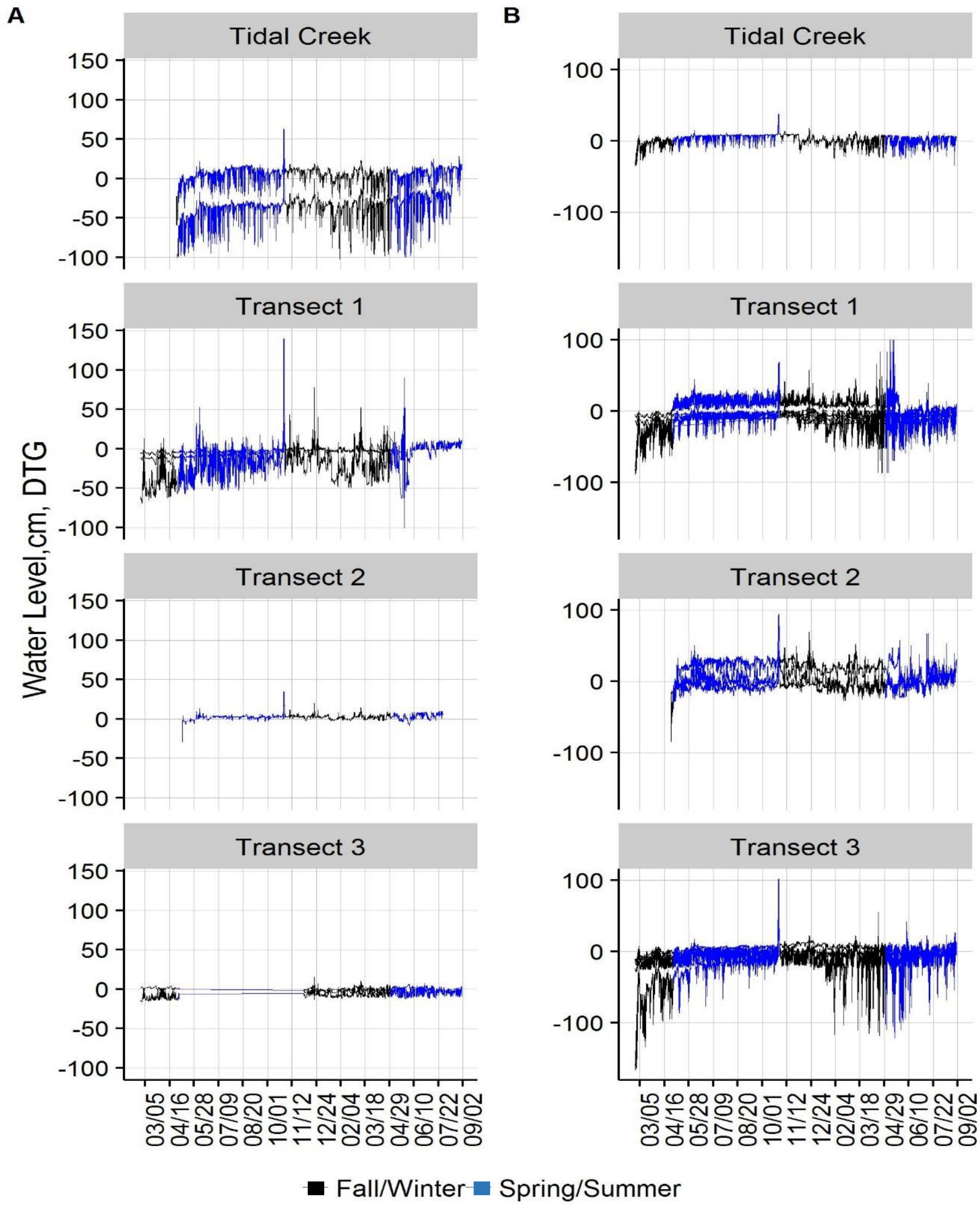


Figure 3.6: Time series of Unditched (left) and Ditched site (right) from 2/1/2012 to 9/1/2013. No distinct seasonal pattern observed at either site. Datum is ground; Depth to Ground (DTG) is positive if water level above ground. Transect 3 is the back of the site; Transect 1 is located closest to the tidal creek.

Boxplots of marsh water levels indicate a slight seasonal difference in well 2 (unditched well) at the unditched site and in wells 3, 6, and 25 for the ditched site, which are all ditch wells (Figure 3.7). Boxplots using depth to ground surface as the vertical coordinate, indicated whether the marshes were inundated. Boxplots constructed using NAVD88 as the datum indicate groundwater total head data, which controls groundwater flow directions (Figure 3.8). In the unditched Chesapeake marsh, head data indicate that head gradients produce drainage toward the marsh at some locations and time periods, and toward the creek at other locations. The ditched site indicated small changes in head in marsh interiors and drainage toward the ditch, particularly along transect 1, which is closest to the tidal creek. Wells in the middle of the marsh (transect 2), however, suggest that the ditches recharge the marsh (drainage away from the ditch).

To examine whether there were time periods of flow reversal in the ditched sites, detailed analysis of distribution and head difference was conducted using time series data of hydraulic head. Horizontal head difference was calculated by taking the difference between marsh interior wells and the center ditch well at the Ditched site. Figure 3.9 shows head in the interiors for Transect 1. All of the interior marsh head data are higher than the head in the tidal creek during 11/29/2012 to 5/25/2013. The head difference diagram indicates that the head difference—and thus hydraulic gradient—is small, but toward the tidal creek in the marsh interiors. At Transect 1, there were no events or seasons that resulted in changes in flow directions.

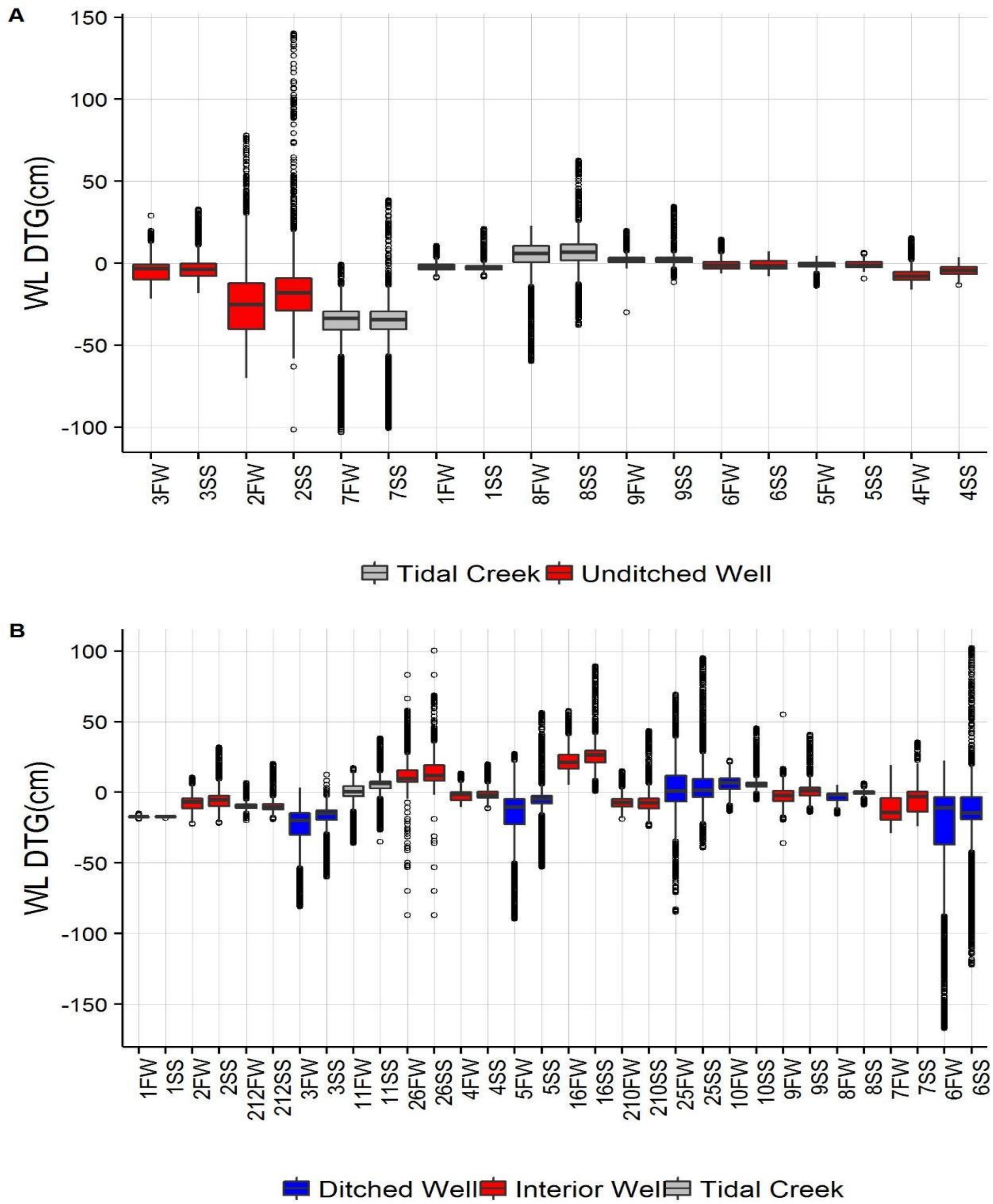


Figure 3.7: Depth to ground (DTG) (A) Unditched site showing slight seasonal difference in well 2 (unditched well); (B) Chesapeake Ditched site showing slight seasonal difference in wells 3, 6, and 25 for the ditched site. Boxplots using depth to ground surface datum indicate whether the marshes were inundated. FW= Fall/Winter; SS = Spring/Summer; # = well number. Box plot symbolism: center line = median; box boundaries = 1st and 3rd quartile; circles = outliers.

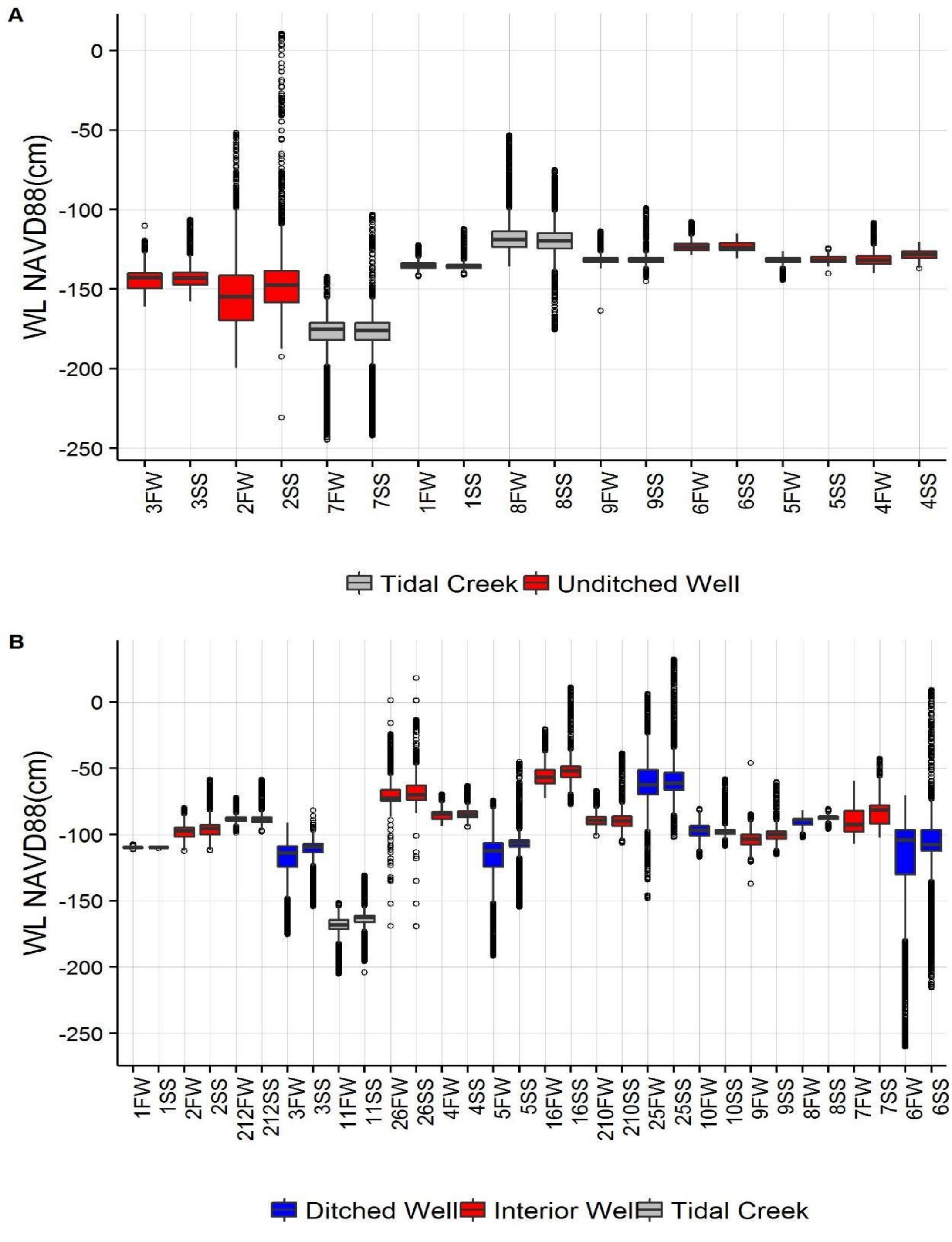


Figure 3.8: Groundwater elevation above NAVD88 Datum: (A) Chesapeake Unditched Site; (B) Ditched site. FW= Fall/Winter; SS = Spring/Summer; # = well number

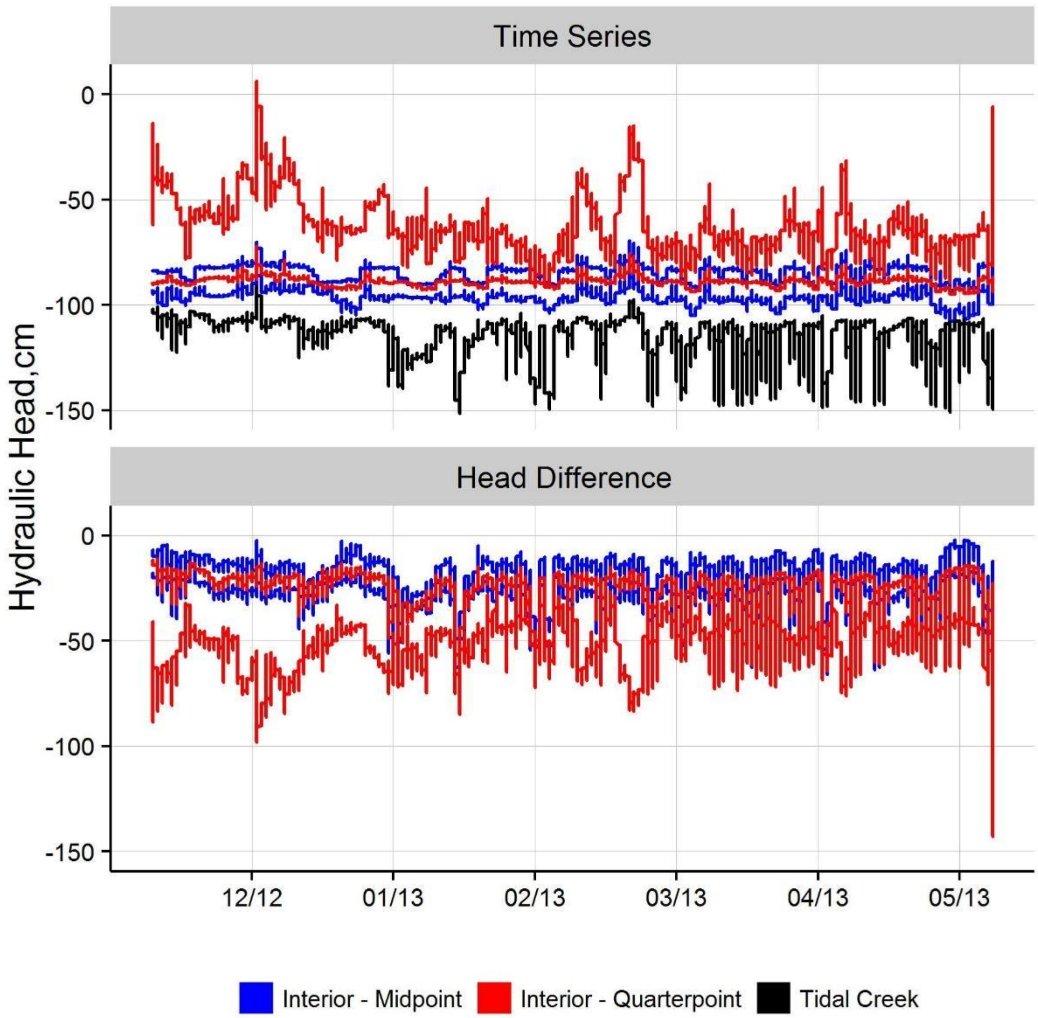


Figure 3.9: Head data for the ditched site. A: Hydraulic head data for the center ditch and interior marsh B) Head difference values between the center ditch and interior wells, which indicates continuous groundwater flow towards the ditch. Midpoints and quarter points lie on each side of the main ditch.

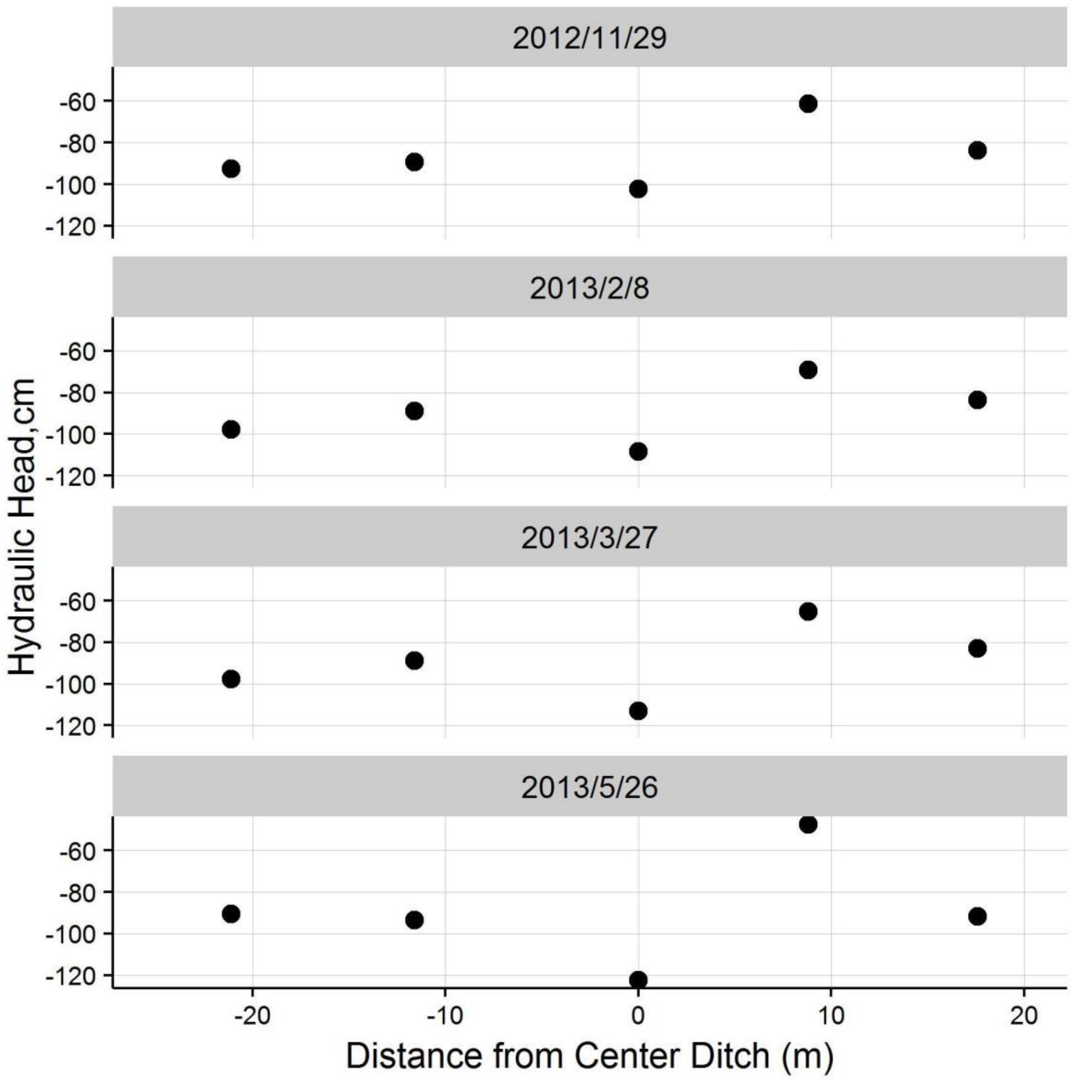


Figure 3.10: Hydraulic head at center ditch (0) along transect 1 (closest to tidal creek) and marsh interiors on random, selected dates. These data also indicate drainage from the midpoints towards the ditch and towards the outer points.

The data from the marsh water level time series were also summarized as groundwater cumulative distributions curves to determine the percentage of time the water level was at or above the surface. The water level exceedance curves displayed a range of distributions indicating almost constant water levels (low curvature) in marsh interiors to continuous change over the range of water levels for tidal creeks and some ditches (Figure 3.11). The exceedance diagrams indicate a larger range of water levels within the ditched sites, especially on transects farthest from the tidal creek (Figure 3.11). In the ditched site, maximum water levels for an interior well reached 93 cm above ground, while sites near ditches reached 92 cm above the ground. Maximum water level in the unditched marsh was 148 cm above ground with a below ground low of 95 cm. Unditched site water levels are at or above the ground surface ~50% of time. Ditched interior wells are at or above the ground surface 20-40% of the time. These data suggest a significant reduction in above-ground events at the ditched site compared to the unditched site in this paired study, even though the ditches provided for an amplification of tidal range (Chapter 2). This suggests that the ditches may have altered marsh water levels through drainage. It is also possible that higher percentages of brushy vegetation (with higher evapotranspiration demand) are responsible for the lower water levels in the ditched marsh. These possibilities will be discussed in the next section.

Marsh water level for various percentile values in the exceedance diagrams was examined with respect to distance from tidal creeks and ground elevation data. These data indicate that the Ditched sites showed no relationship between ground elevation and the percent time water is at ground surface (Figure 3.12). Ditched sites also showed no relationship between distance from tidal creek or distance from ditch and the percent time water was at ground surface (Figure 3.12). There was a relationship between ground elevation and the 10^{th}-percentile water level for ditched sites. The 10^{th} percentile could be primarily during draining conditions at low tides (Figure 3.13). The

90th percentile (wet conditions) also showed no relationship to ground elevation (Figure 3.14).

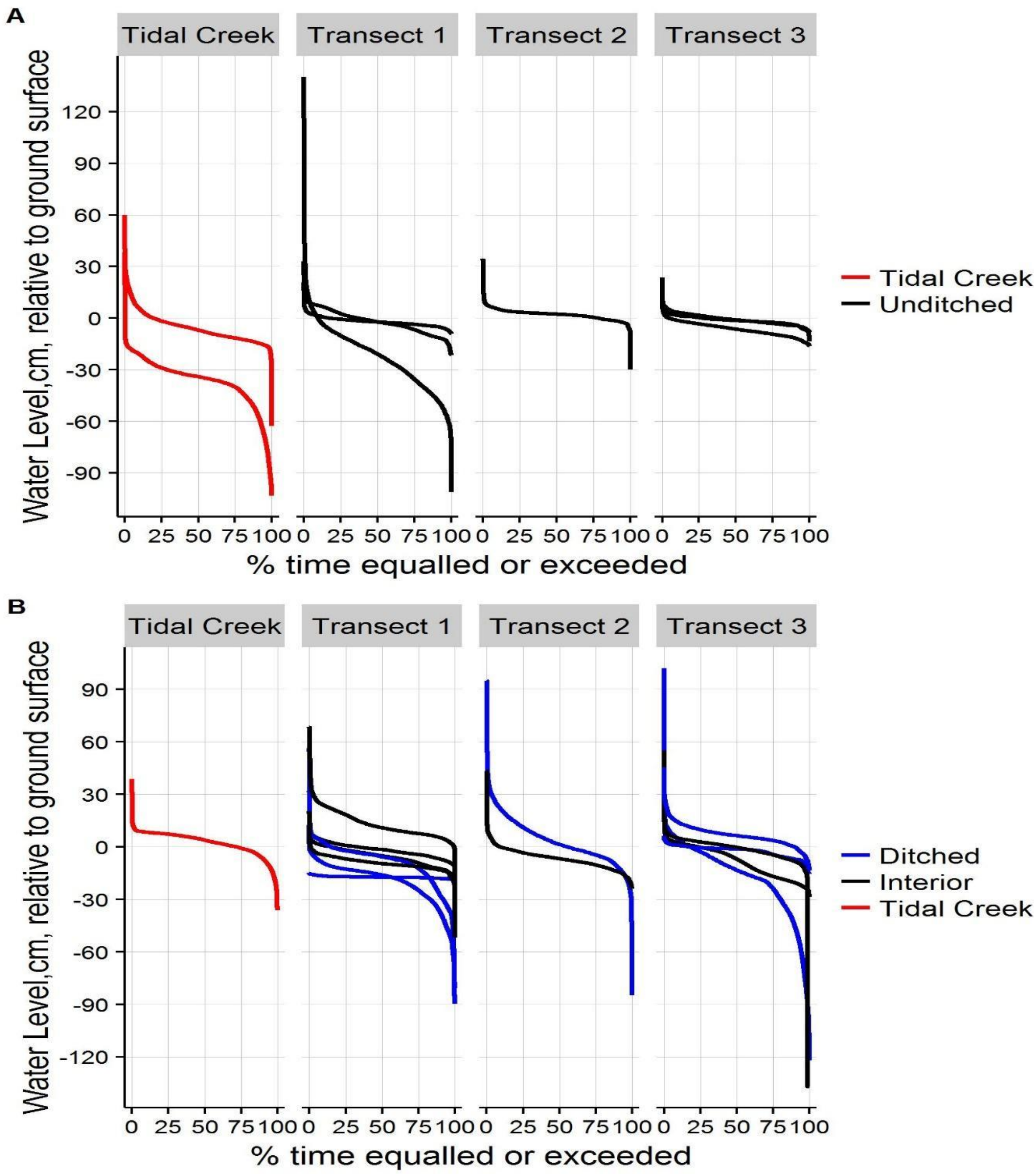


Figure 3.11: Cumulative curves showing percent time at or below the ground surface (indicated as 0), for (A) Unditched (B) Ditched sites. Negative values indicate water levels are below ground.

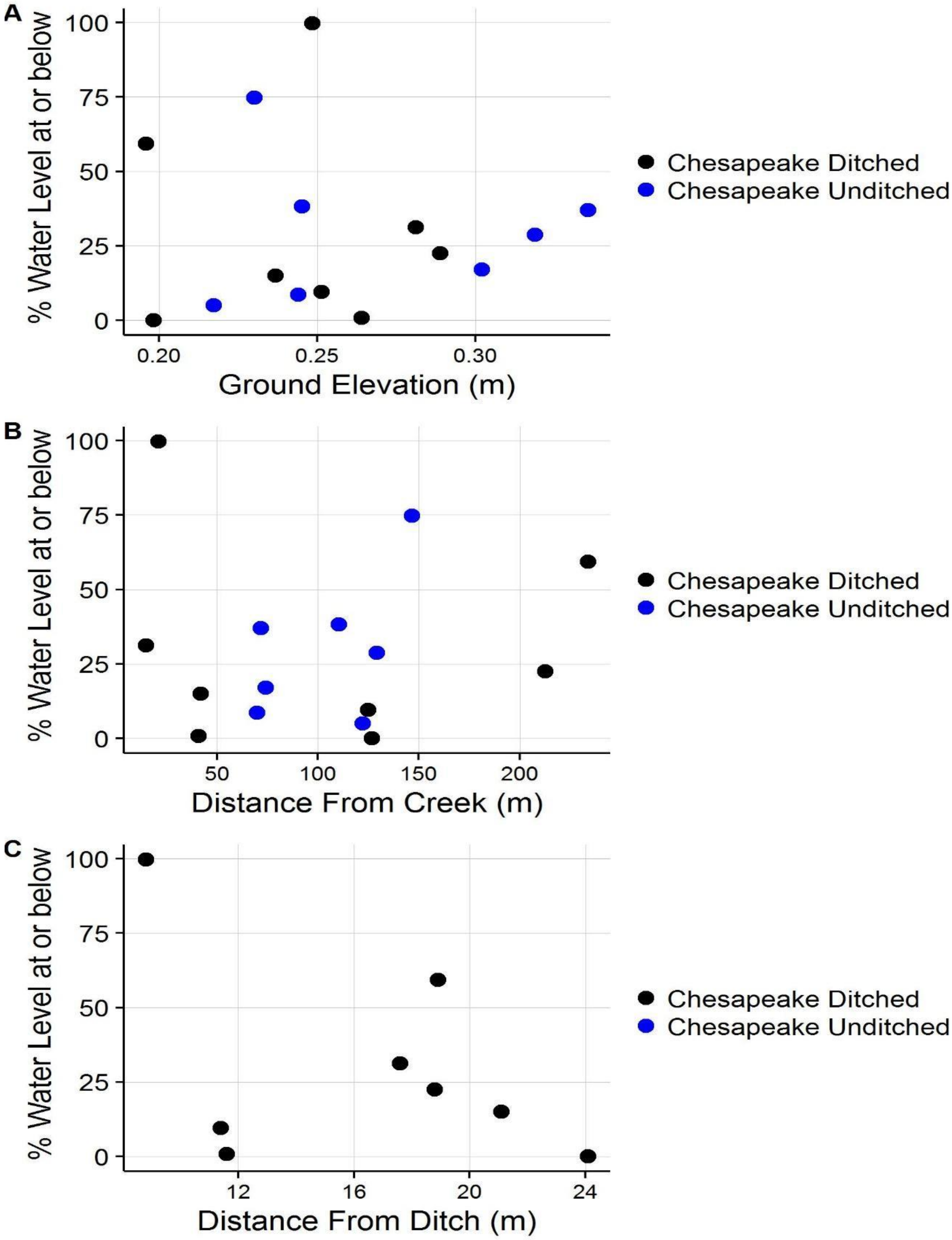
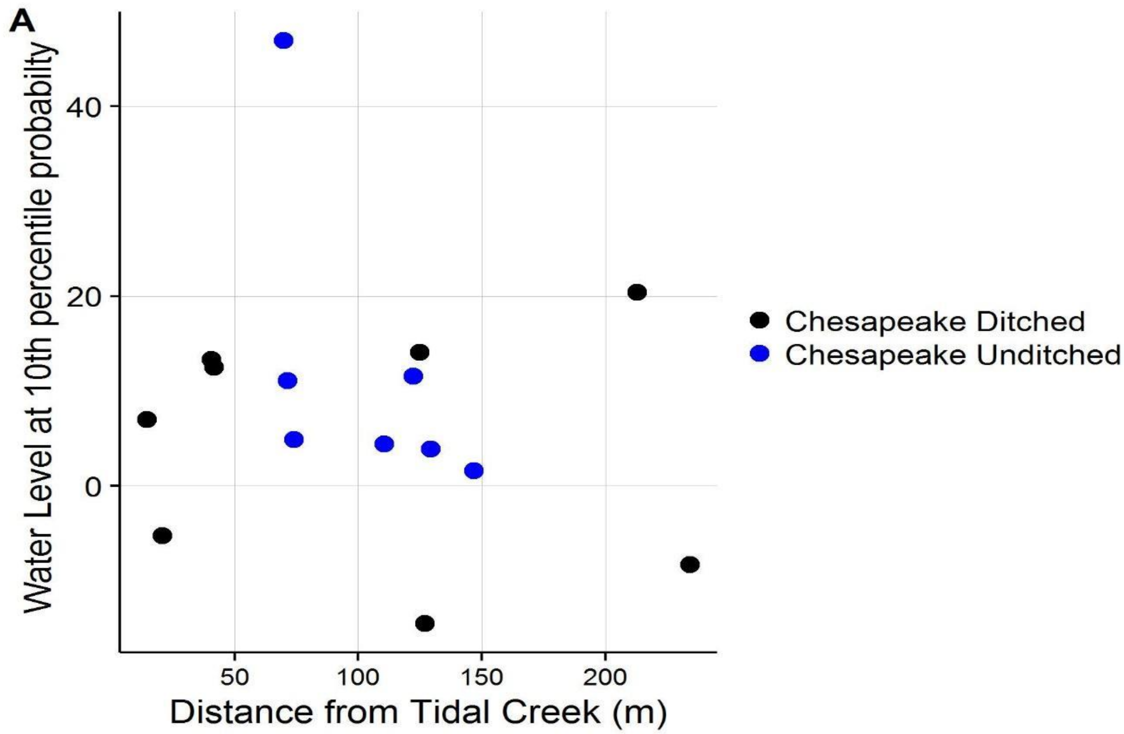


Figure 3.12: Percent of time that water level is at or below the surface as a function of elevation (a) and distance from b) a tidal creek and c) the proximal ditch.



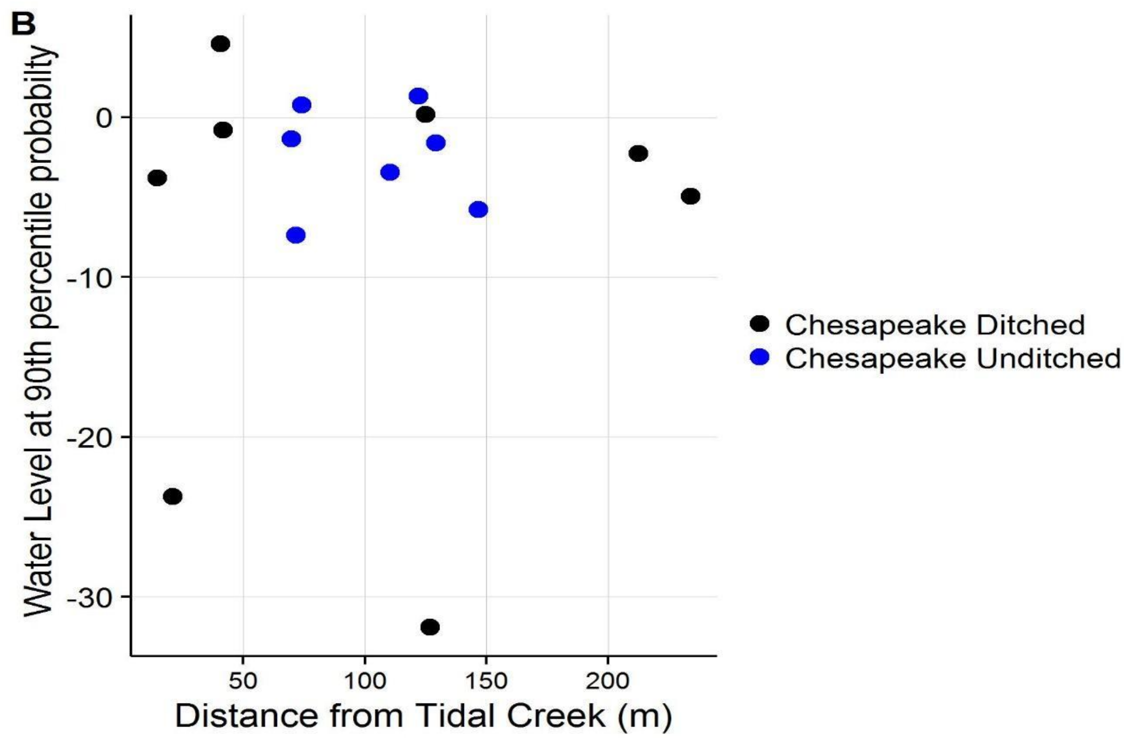


Figure 3.13: 10^{th} and 90^{th} percentile on water levels versus distance (m)

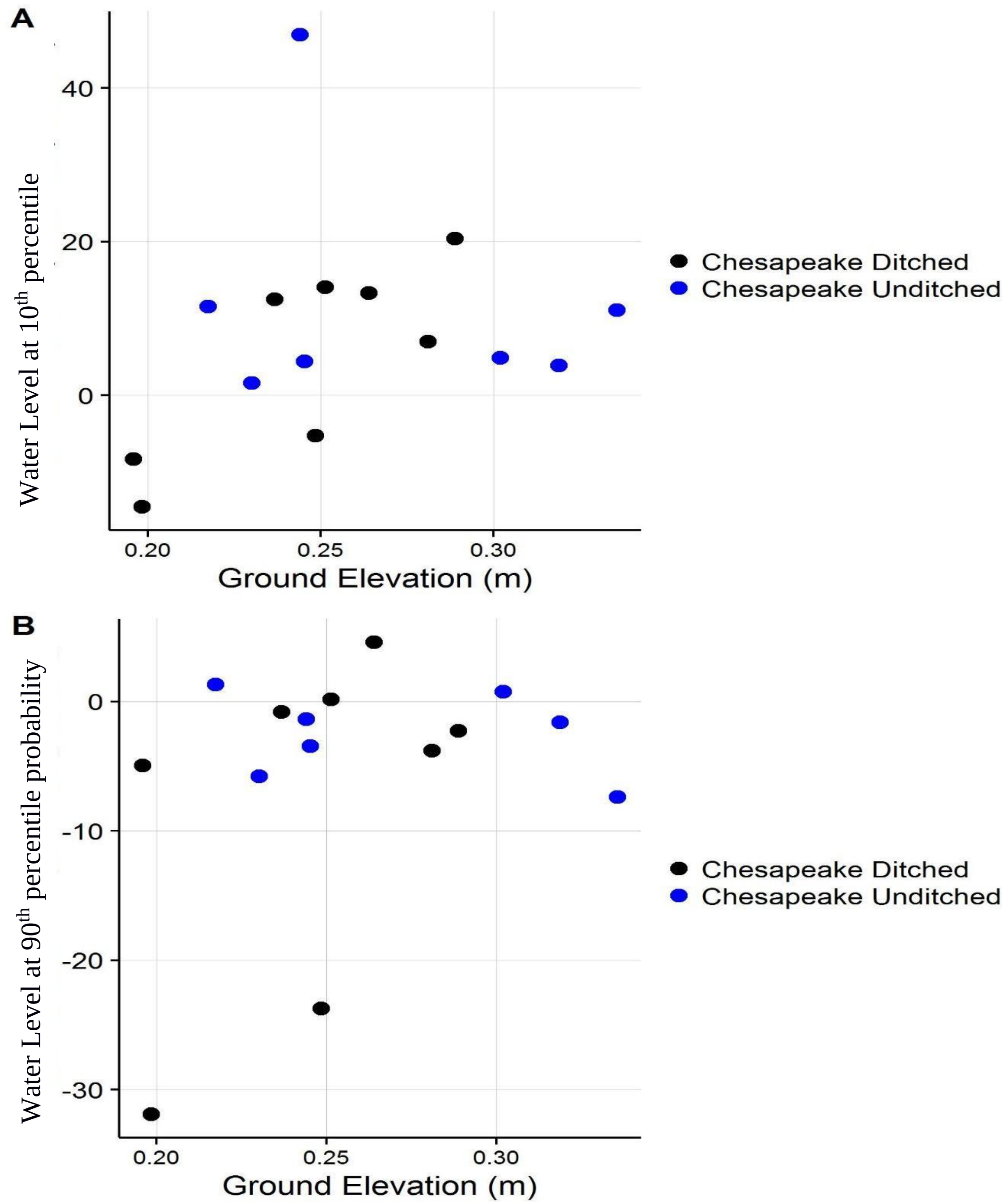


Figure 3.14: 10th and 90th percentile on water levels versus ground elevation (m)

Percent saturation is defined as percent of time water level is at or above the ground surface. In comparing vegetation species composition (%) with saturation (%), *Spartina* species did not show a relationship, while *Juncus roemerianus* declined as saturation increased—but only in the Ditched site (Figure 3.15A). There was no relationship between any species and salinity (Figure 3.15B).

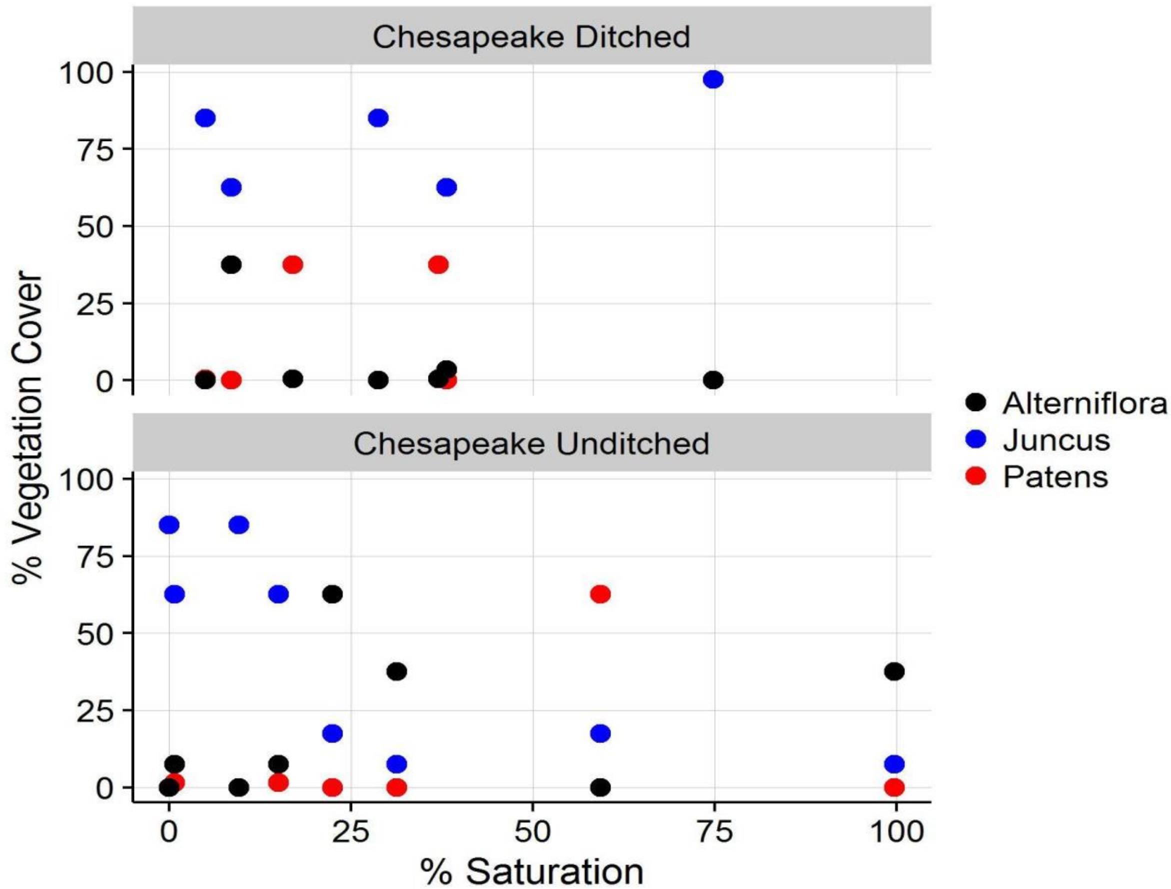


Figure 3.15A: Percent cover of the dominant species at the Chesapeake Ditched and Unditched sites versus percent saturation.

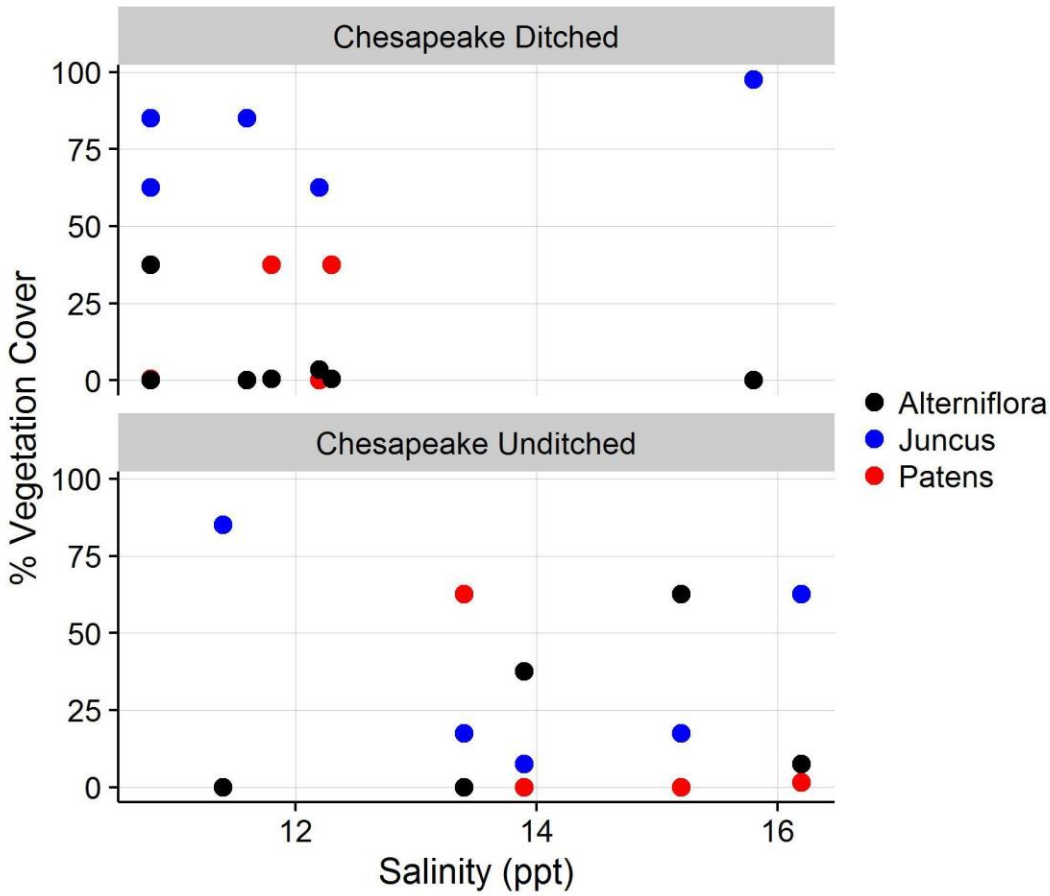


Figure 3.15B: Percent cover of the dominant species at the Chesapeake Ditched and Unditched sites versus salinity.

Atlantic Sites

Vegetation

The Atlantic Coast marshes were predominantly covered by the following species: *Spartina alterniflora* covered 45.7% of the ditched marsh and 31.5% of the unditched marsh, while *Spartina patens* (marsh hay cordgrass) covered 28.9% of the ditched marsh and 62% of the Unditched marsh. These data also indicated that *Spartina patens* was higher in the Unditched site but lower in the Ditched site. Conducted t-tests showed that at the Unditched and Ditched sites, *Spartina alterniflora* were statistically similar but *Spartina patens* was statistically different (p=0.003) (Table 3.4).

Table 3.4 Vegetation Composition top three species

	Spartina Alterniflora	***Spartina Patens***
Chesapeake Ditched	45.7 %	28.9 %
Chesapeake Unditched	31.5 %	62 %
t-test p values	0.146	0.003

Comparison of vegetation species composition (%) with distance from the tidal creek (m) and ground elevation (m) showed no relationship regardless of site (Ditched/Unditched) and regardless of species (Figure 3.16).

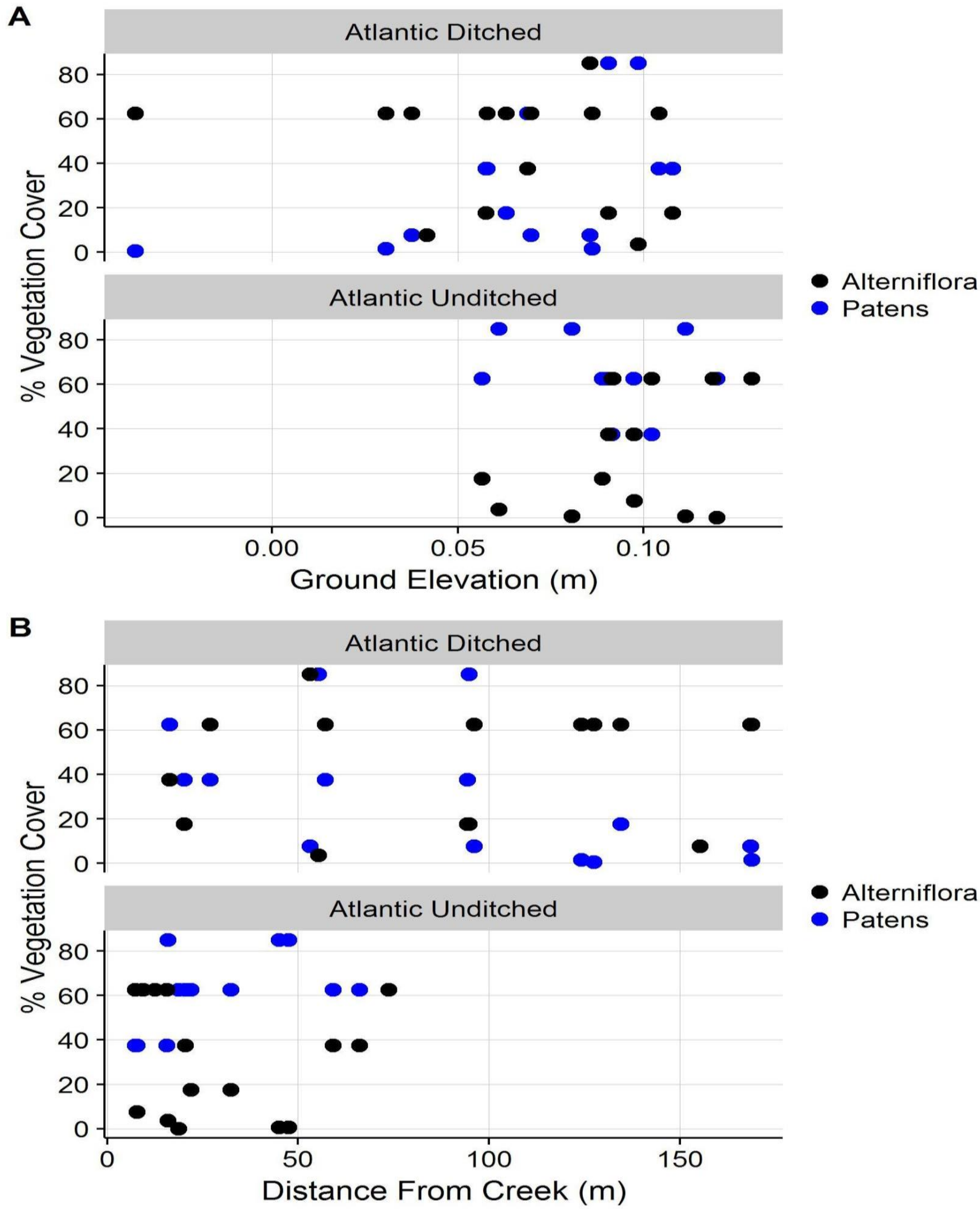


Figure 3.16: Percent cover of the dominant species at the Atlantic Ditched and Unditched sites versus (b) distance from tidal creek and (a) ground elevation.

Salinity characteristics

In the Atlantic coastal marshes, the ditched sites showed higher salinity values than their unditched counterparts (Opposite of the Chesapeake sites). The range of salinity values was 10-24 ppt at the Ditched site and 1-15 ppt at the Unditched site, which is located further up the main tidal creek than the ditched site (Figure 3.17). Separating the back of site (transect 3) values from the other data indicate that the ditched back transect is similar to the less saline unditched site salinity values, whereas the ditched marsh has higher salinity values than the unditched or interior ditched marsh sites. Figure 3.18 scatter plot shows that salinity in the unditched site decreased in salinity with an increase in distance from tidal creek and increased with increase in elevation for all sample dates. No relationship was found in the ditched site with respect to ground elevation or distance from the tidal creek.

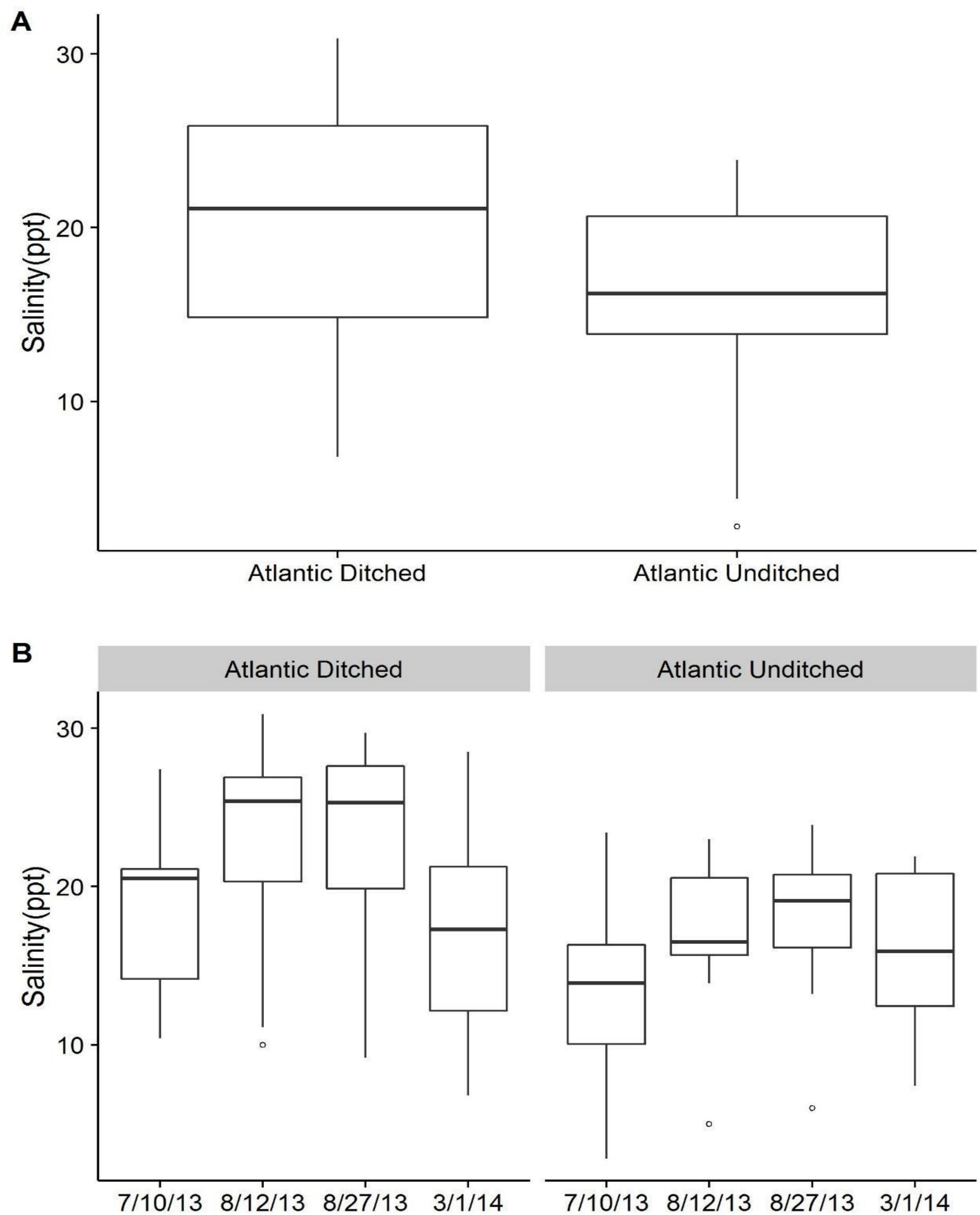


Figure 3.17: Salinity by site and by sample date for Atlantic Ditched and Unditched Sites. Box plot symbolism: center line = median; box boundaries = 1st and 3rd quartile; circles = outliers.

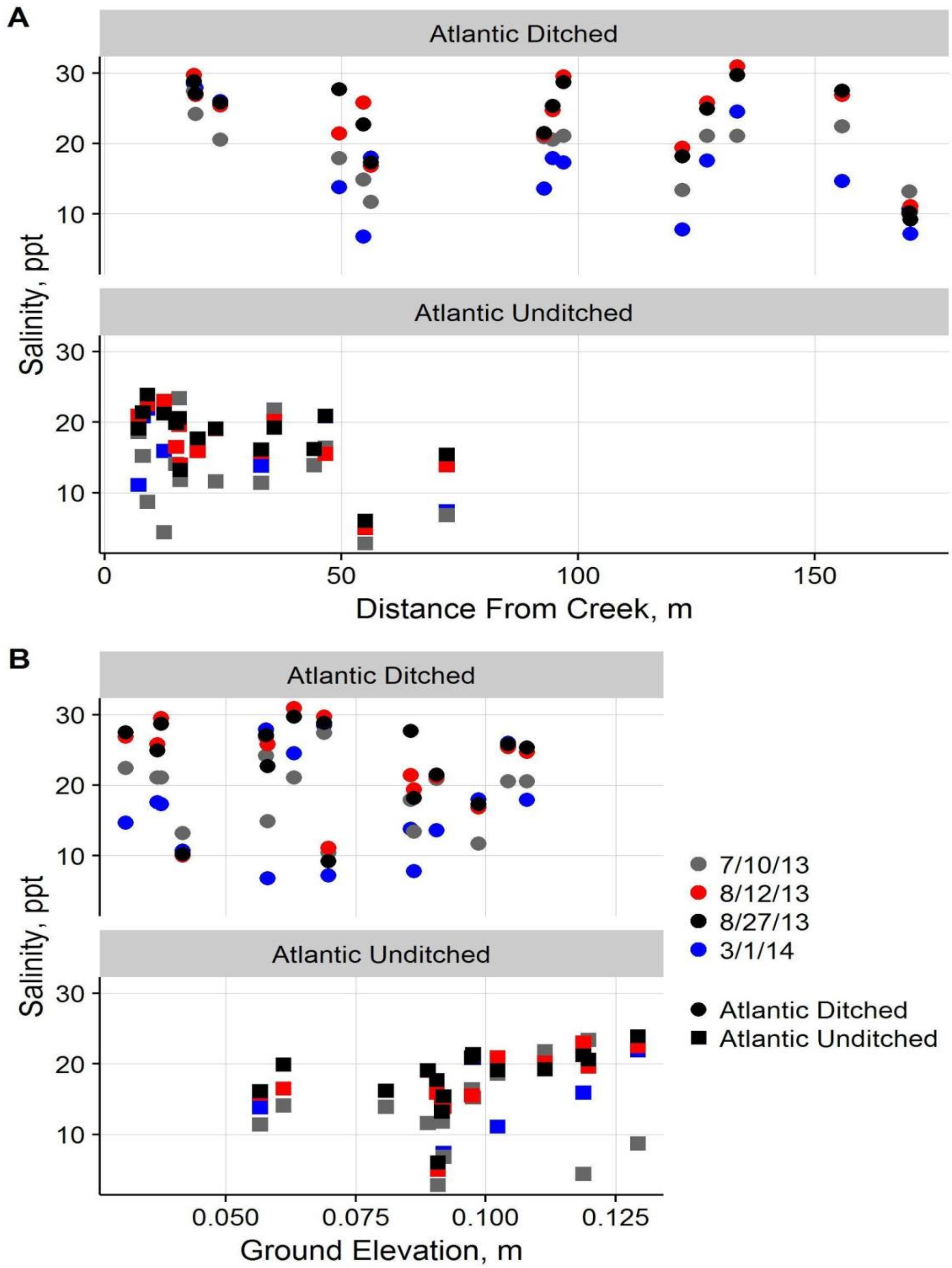


Figure 3.18: Salinity for Atlantic Ditched and Unditched Sites versus distance from tidal creek and ground elevation.

Hydrologic Behavior of Atlantic Ditched and Unditched marshes

Time series data of marsh water levels

Tidal creek and marsh groundwater levels are shown in several figures, beginning with 3.19. Water levels at the unditched site varied as a function of location. Water table elevations at sites located closest to the tidal creek revolved around ground surface, whereas at the ditched site the water table was predominantly below the ground (Figure 3.19). Based on the time series data, there is no obvious seasonal pattern to the water levels. There are numerous high water level events due to storm events. The largest event shown is Hurricane Sandy, which occurred on 10/30/2012. Some of the marsh time series data show water levels that are lower during the spring/summer months with higher water levels in the fall, possibly illustrating evapotranspiration in summer followed by a fall recharge period (Figure 3.19). Hydraulic responses to tidal fluctuations decreased with distance from the bank of tidal creeks at both ditched and unditched sites. Water-level fluctuations near and in the ditches are less regular than those in the interiors or at the unditched marsh, and do not show the seasonal pattern.

Boxplots show a slight seasonal difference present in wells 2 and 8 at the unditched site, and present in wells 1, 3, 211, 9 for the ditched site which are all located within a ditch. The unditched site showed seasonal difference in one of the smaller of the two tidal creeks and one of the unditched wells furthest from the tidal creek (Figure 3.20). Boxplots using depth to ground vertical coordinate shows inundation of the marshes, whereas NAVD88 datum indicates levels relative to tidal creeks and thus flow directions. In the unditched Atlantic site, the tidal creek appears to be a source for the marsh water levels during both fall/winter and spring/summer. There doesn’t seem to be any seasonal reversal of flow directions. At the ditched site, water level comparisons indicate that ditch water levels are lower than marsh interiors, which may result in local drainage of the marsh.

A time series of hydraulic head difference [cm] along Transect 1 (closest to tidal creek) at the Ditched site was calculated between marsh interior wells and the center ditch well. Figure 3.21 shows head 5/4/2012 to 6/25/2013. Positive difference in head shows marsh interior locations are draining into the center ditch.

The groundwater cumulative probability distributions displayed a range of patterns from low curvature (little variation over the entire period of record to continuous change [response to tidal variations—see Fig. 3.22]). Marsh groundwater cumulative probability distributions indicate a large range of water levels for the ditched sites, especially for the sites that are closer to the tidal creek (Figure 3.22). The highest water levels in the ditched site were in interior wells at 67 cm above ground, and 60 cm above ground in the ditch wells. Maximum water level in the unditched marsh was 45 cm above ground. Unditched site wells spend ~50% of time above ground, versus the ditched interior wells 10-25% of the time at or above the surface. This shows a substantial reduction in above-ground events for the ditched site compared to an adjacent unaltered site. Similarly, duration curves (Figure 3.22) showed the largest range within the tidal creek more similar to the wells closest to the tidal creek in the marsh. Breaking down the wells individually showed that interior wells had longer and more frequent times above ground (Table 3.5).

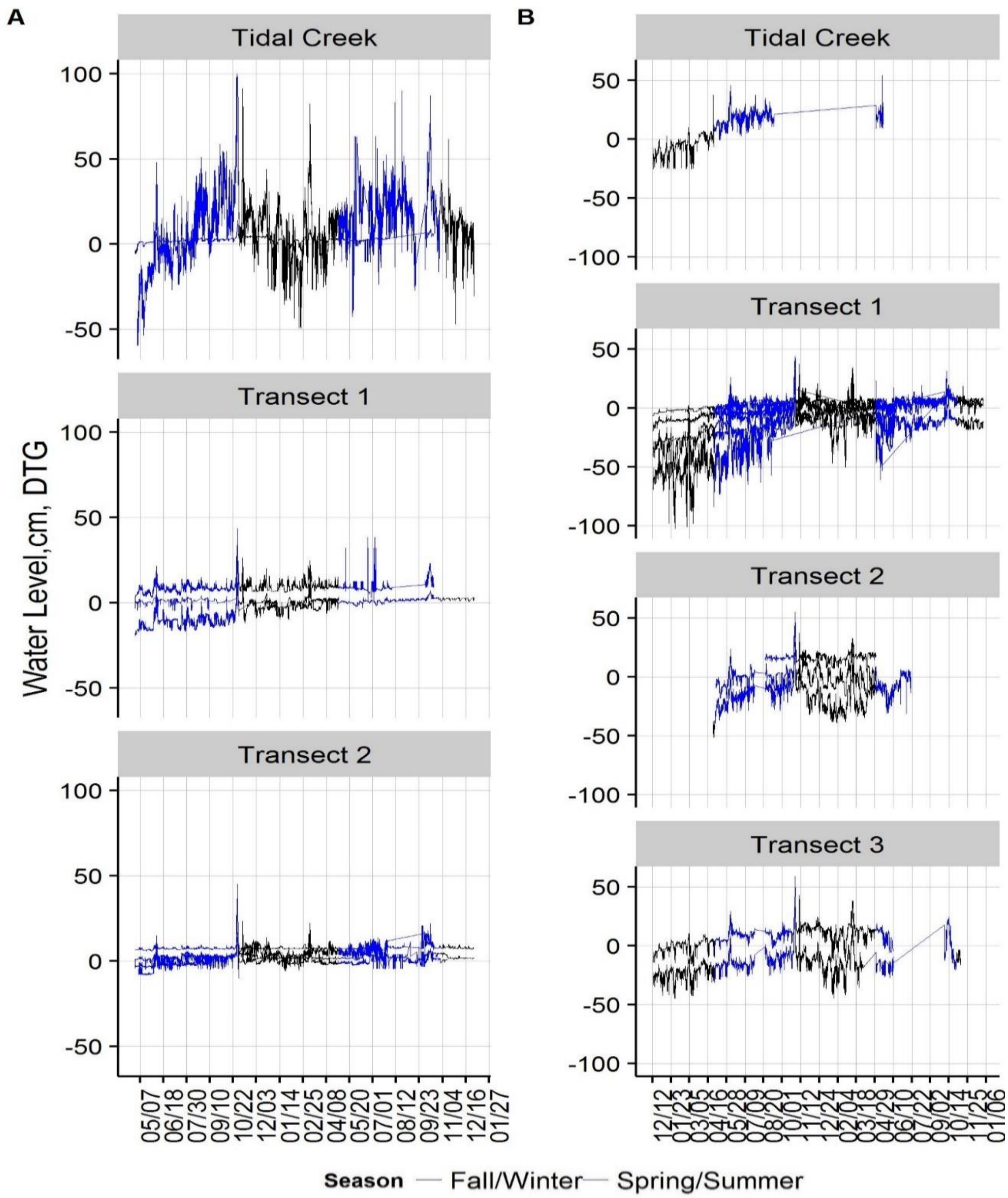


Figure 3.19: Time series of (A) Unditched site, 4/1/2012 to 12/31/2013 and (B) Ditched site, 12/12/2011 to 12/31/2013. No distinct seasonal pattern.

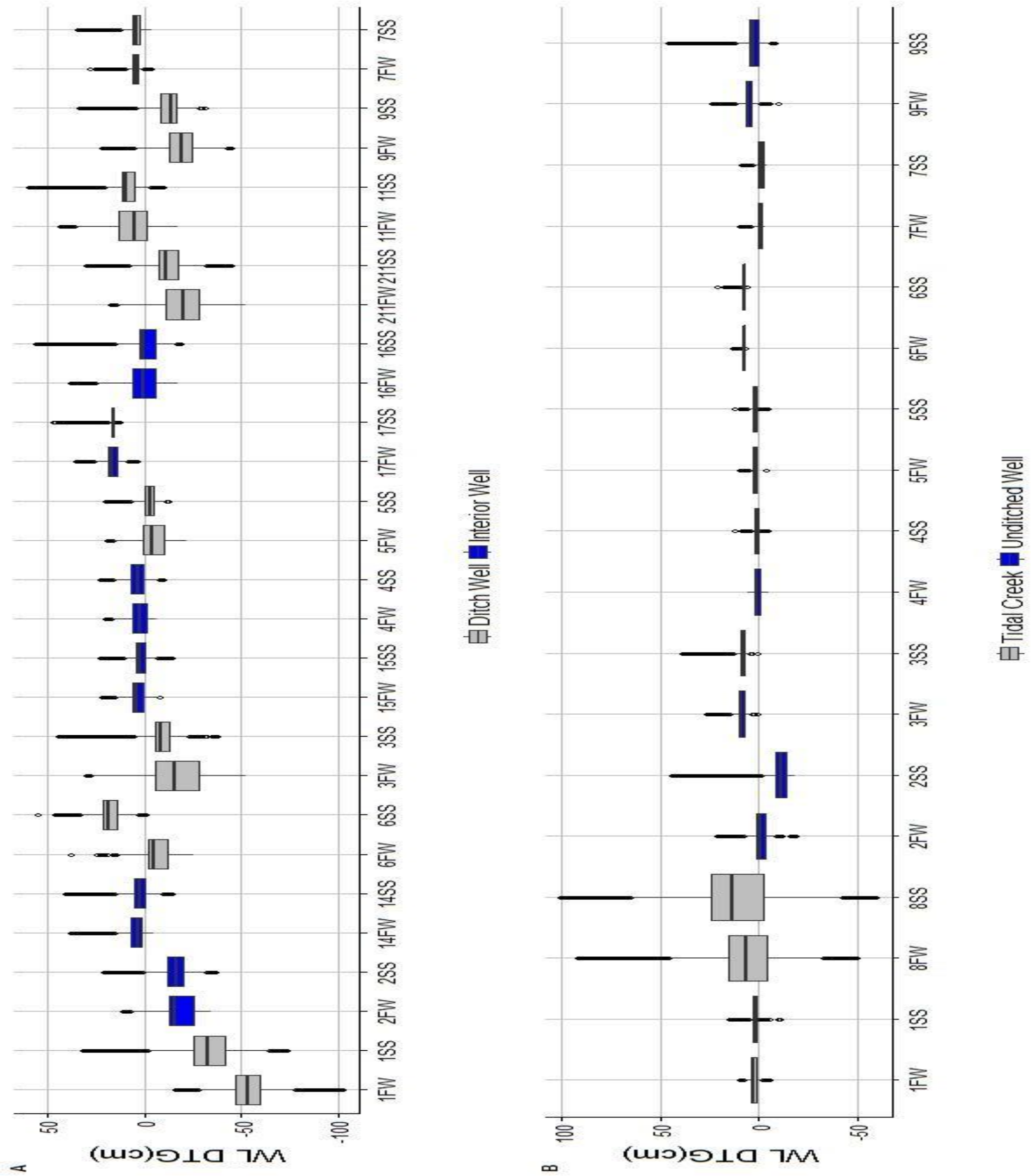


Figure 3.20A: Boxplots showing seasonal differences between wells and sites. (A) Unditched Site (B) Ditched Site on the Atlantic Coast. Relative to ground is to determine flooding and saturation. SS= spring/summer; FW = fall/winter. Box plot symbolism: center line = median; box boundaries = 1st and 3rd quartile; circles = outliers.

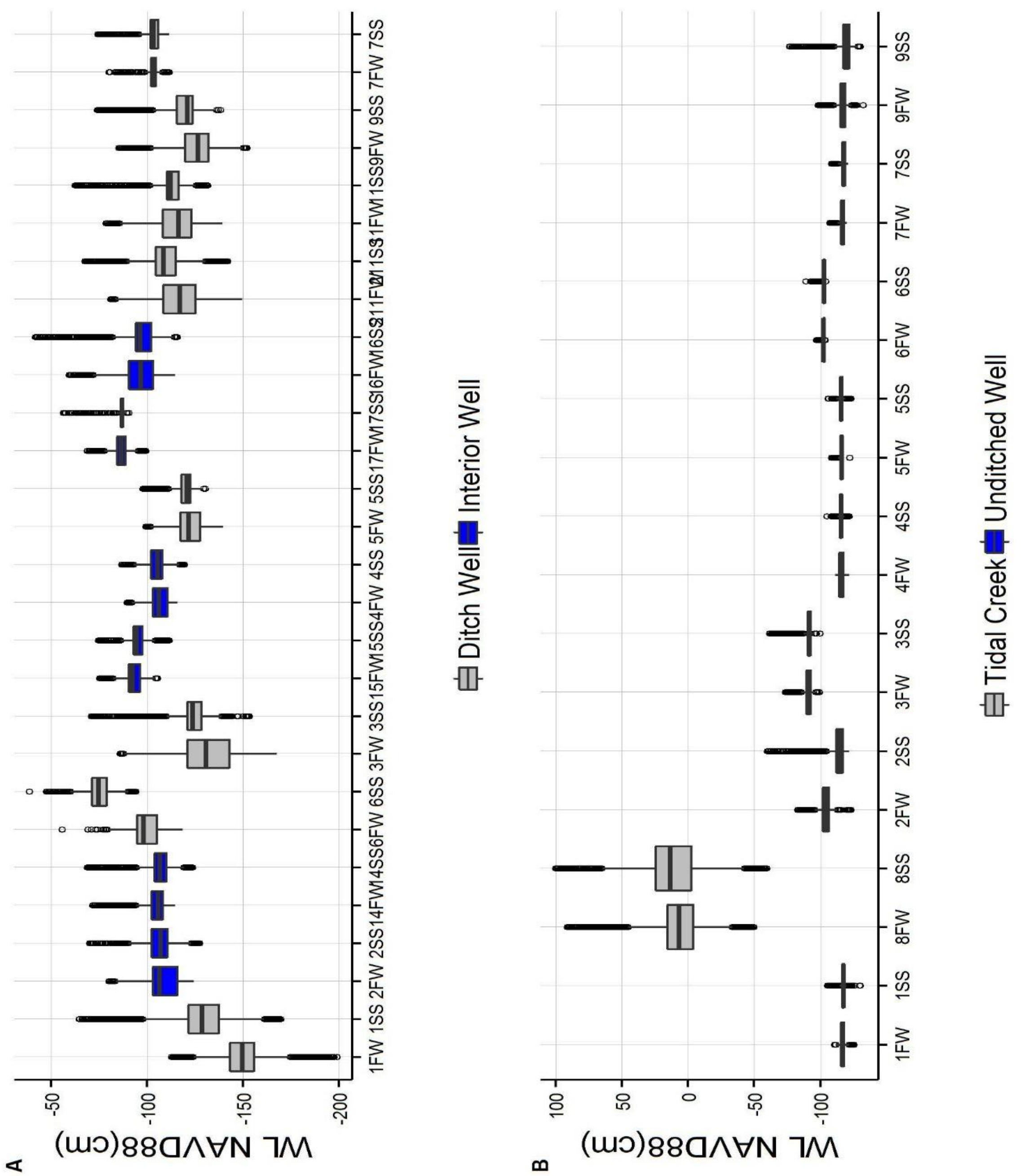


Figure 3.20B: Boxplots showing seasonal differences between wells and sites. Water level relative to datum is to determine groundwater flow directions SS= spring/summer; FW = fall/winter. Box plot symbolism: center line = median; box boundaries = 1st and 3rd quartile; circles = outliers.

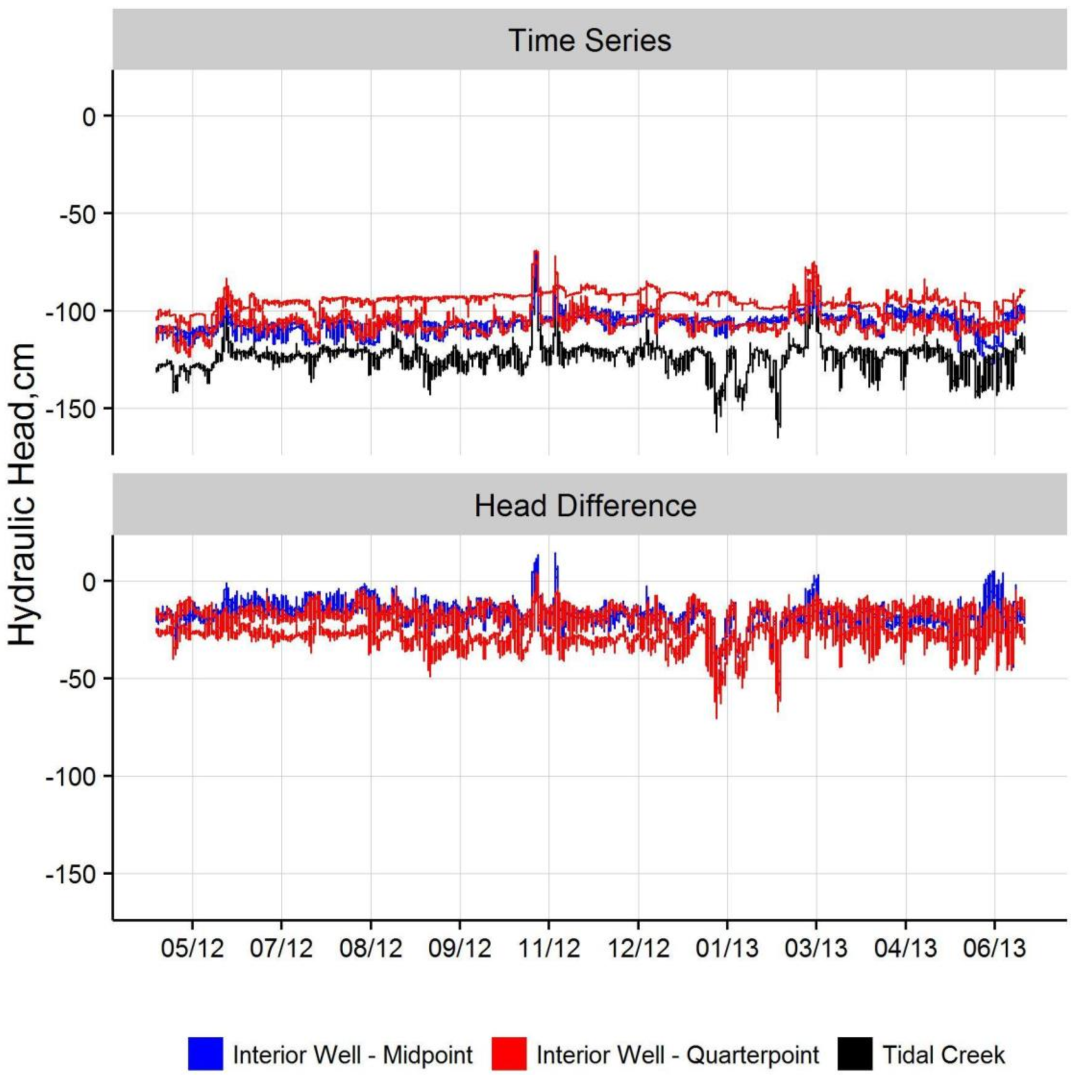


Figure 3.21: (A) Time series of hydraulic head at center ditch and marsh interior. (B) Head difference between the marsh interiors and the tidal creek. Data indicate drainage head distributions except for during major storm events.

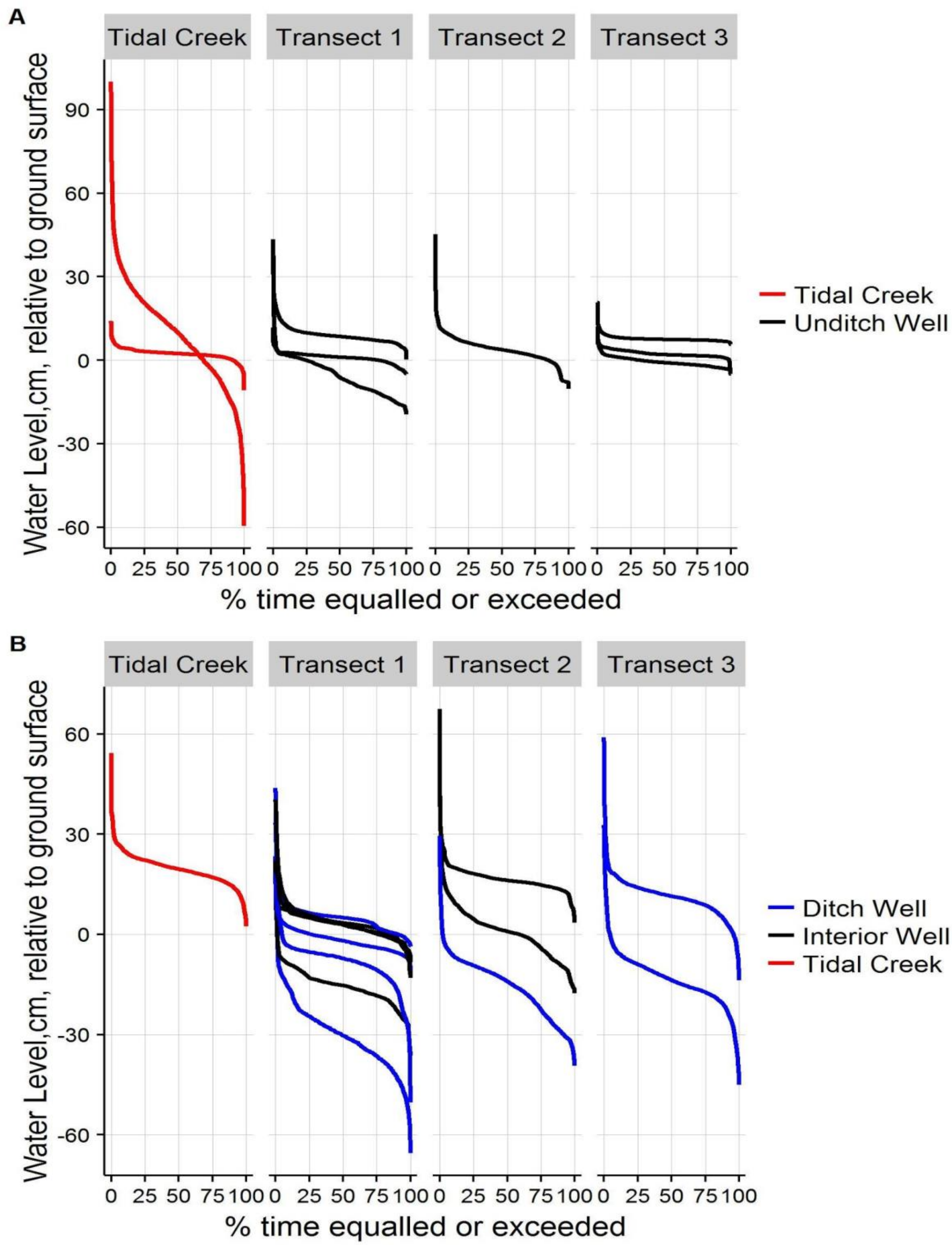


Figure 3.22: Water level cumulative probability curves showing percent time at or above levels measured relative to the ground surface at 0. Above ground > 0.

Table 3.5: Percent time at or above ground surface for each well for ditched and unditched sites with respect to location.

Well	Site	Type	Location	% time at or above ground surface
6	Ditched	Ditch Well	Close to creek- Tide Gage	100
1	Ditched	Ditch Well	Close to creek	1.2
2	Ditched	Interior Well	Close to creek	1.5
14	Ditched	Interior Well	Close to creek	76
3	Ditched	Ditch Well	Close to creek	4.9
15	Ditched	Interior Well	Close to creek	81
4	Ditched	Interior Well	Close to creek	87.2
5	Ditched	Ditch Well	Close to creek	26.5
16	Ditched	Interior Well	Middle of site	59
17	Ditched	Interior Well	Middle of site	100
211	Ditched	Ditch Well	Middle of site	2.2
7	Ditched	Ditch Well	Farthest from creek	90.7
8	Ditched	Interior Well	Farthest from creek	-
9	Ditched	Ditch Well	Farthest from creek	4.5
11	Ditched	Ditch Well	Farthest from creek	95.9
1	Unditched	Tidal Creek	Tidal Creek	91.4
2	Unditched	Unditched Well	Close to creek	24.9
3	Unditched	Unditched Well	Close to creek	100
4	Unditched	Unditched Well	Close to creek	78.9
8	Unditched	Tidal Creek	Tidal Creek	69.3
5	Unditched	Unditched Well	Close to creek	98.8
9	Unditched	Unditched Well	Middle of site	84.8
6	Unditched	Unditched Well	Farthest from creek	100
7	Unditched	Unditched Well	Farthest from creek	31.1

Comparing plant species composition (% cover) with saturation (%), both *Spartina* species did not show a relationship (Figure 3.23). Figure 3.24 shows that *Spartina Alterniflora* decreased with increase in salinity in the unditched site but not in the ditched site.

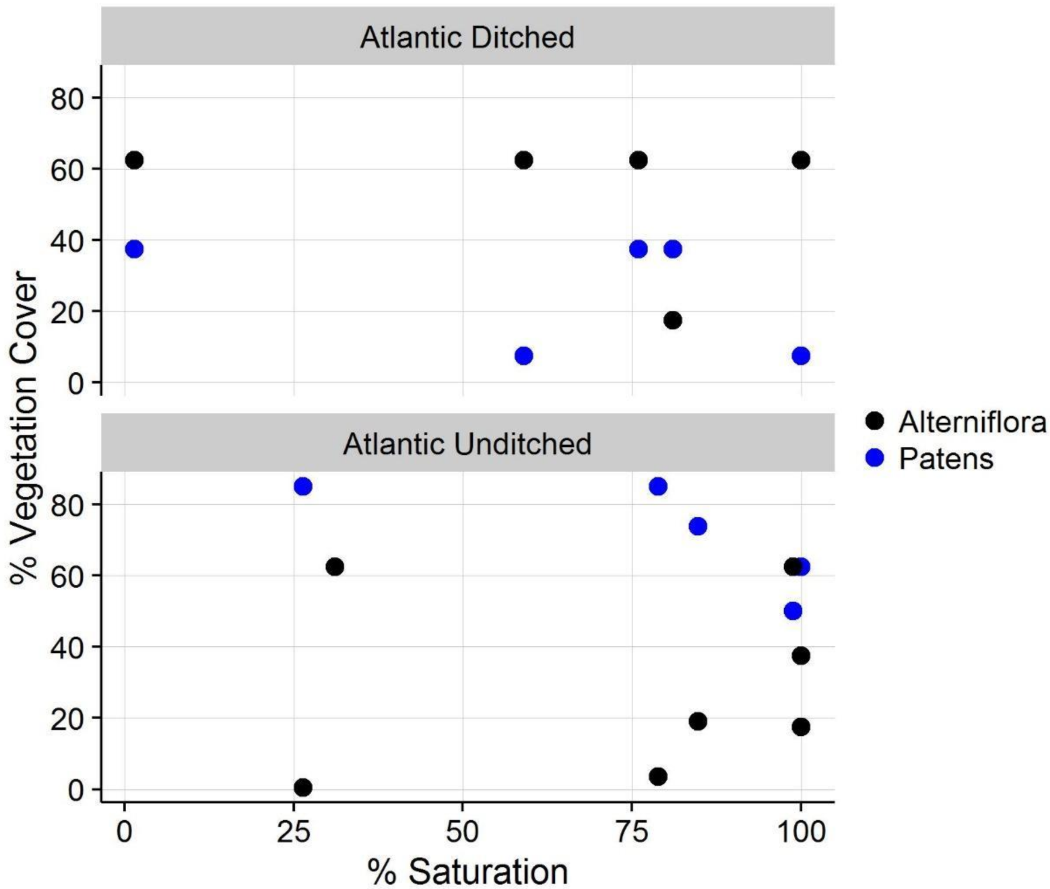


Figure 3.23: Percent cover of the dominant species at the Atlantic Ditched and Unditched sites versus percent saturation.

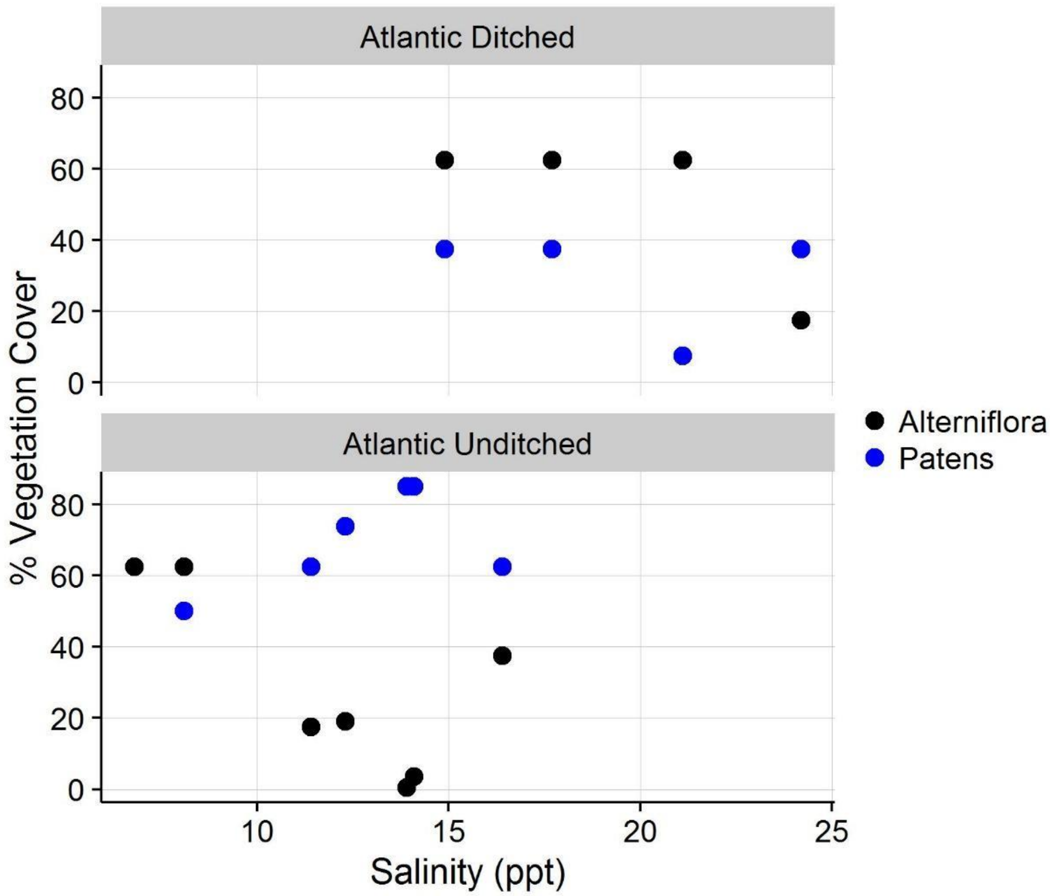


Figure 3.24: Percent cover of the dominant species at the Atlantic Ditched and Unditched sites versus salinity.

Statistical Comparison of salinity between ditched, unditched, Atlantic and Chesapeake Marshes

In this study, marsh water levels were examined in pairs of ditched and unditched marshes at both Atlantic Bay and Chesapeake Bay sites. Statistical two sided t-tests were used to compare the means of two groups under the assumption that both samples are random, independent, and come from normally distributed populations. In this analysis, p-value greater than 0.05 indicated that the averages of two groups are statistically similar.

Salinity was found to be statistically different between the Atlantic Ditched and Unditched sites (p-value = 0.0014) and between the Atlantic Ditched vs Chesapeake Ditched sites (p-value = 5.034e-14). Analysis did not indicate significant differences in salinity between the Chesapeake Unditched and Ditched sites. There was also no statistical difference between the Atlantic and Chesapeake Unditched sites (Appendix).

Discussion

This goal of this chapter was to compare groundwater levels, saturation, and vegetation composition in ditched and unditched coastal marshes. The study marshes were chosen based on future ditch plugging restoration plans for the marshes by the Maryland Department of Natural Resources. One of the sources of physical energy in coastal marshes is the hydraulic gradients. The marsh ecosystem is driven by the interaction of tidal and interior marsh hydrology and soil hydraulics that determine water depth and duration of inundation. The movement of water throughout the marshes also establishes local water quality such as salinity, temperature, and dissolved oxygen.

A literature review indicates that ditches in coastal marshes have been substantially changed by ditching (Daiber 1986; Roman et al. 2000) (Table 3.6). The sites chosen for this study have small tidal ranges (< 12 cm), and the significant attenuation of the tidal range is likely due to marsh morphology and flow resistance.

Many tidal marshes have modified hydrologic processes due to marsh ditching, which altered habitat and ecological interactions due to changes in water levels and salinity levels (e.g., Roman et al., 1984).

Table 3.6: Some previous research - effects from ditching in marshes		
Feature	**Effect from Ditching**	**Source**
Water Table	Lower	Taylor 1938; Bourn 1950; Redfield 1972; Adamowicz 2005
	Increase	Vincent 2012
Surface Ponding	Decline	Reinert 1981; Merriam 1983; Lathrop 2000; Adamowicz 2005
Waterway stability	Ditches fill in or widen	Miller and Egler 1950
Spartina alterniflora	Decreases	Bourn 1950; Reinert 1981
	Increases	Taylor 1938; Miller 1950; Niering 1980; Kennish 200
Spartina patens	Increases	Reinert 1981
Phragmites australis	Ditch edges increases	Bart 2006
Woody Vegetation	Increases	Daigh 1938/1939; Miller 1950; Kuenzler 1973; Shisler 1973; Chapman 1974; Cooper 1974; Clarke 1984
Waterbirds	Decreases	Cottam 1938; Reinert 1981; Nixon 1982; Clarke 1985; Daiber 1986; Dreyer 1995;
Reduced nitrogen	Increases	Koch 2009
Muskrats	Decreases	Bourn 1950
Elevation	Loss in 50 meter wide ditches; Increases in spoil piles	Bourn 1950
Sediments	More Aerated, higher sulfide concentrations	Chambers 2003
Mosquito populations	Decline	Smith 1904; Taylor 1938; Daigh 1938; Dreyer 1995
	Increases	Shisler 1973; Kuenzler 1973; Cowan 1986

Analysis of vegetative cover indicated differences between plant species at ditched and unditched sites. Ditched sites had lower percentages of *S. patens* compared to unditched sites at both the Chesapeake and Atlantic sites (Chesapeake ditched 10%, unditched 19%; Atlantic ditched 29%, unditched 62%). This is opposite of the objective of ditching, which was often to increase the coverage of *S. patens* (marsh hay). For hundreds of years, ditches were dug to support salt hay farming (Daiber 1986; Mitsch et al. 1994; Dreyer and Niering 1995; Bart 1997; Phillipp 2005), but there are few literature references to salt hay (*S. patens*) increases post mosquito ditching (except Taylor 1938) or findings of *S. patens* being greater in ditched areas compared with unditched areas (except Merriam 1983). Montalto et al. (2006) found consistently "high" water tables (~10 cm from ground surface to water) where *S. patens* was found. This project found no relationship between percent *S. Patens* and ground elevation, percent time at saturation, and distance from tidal creek.

The Chesapeake Bay site data indicated higher coverage of woody high marsh vegetation at the ditched sites compared to the unditched sites. Woody, upland-type vegetation has been found in other studies as well in the interior marshes after ditching due to the spoilage from hand digging creating higher elevations alongside the ditches (Daigh et al. 1938; Daigh and Stearns 1939; Miller and Egler 1950; Kuenzler and Marshall 1973; Shisler 1973; Chapman 1974; Cooper 1974; Clarke et al. 1984). *Spartina alterniflora* (smooth cordgrass) forms dense, monospecific stands with mid to high tide levels (Duncan 1987). It dominates with the average water table 10.2 cm above ground level. *Spartina alterniflora* tolerates being inundated with salt water for up to 20 hours per day. Unlike most other marsh plants, the salt-tolerance of cordgrass is directly proportional to water depth (Allen 1950). It thrives in anoxic, low marsh habitats. A possible cause for the zonation between low marsh (an *S. alterniflora* domination) and high marsh (usually dominated by *S. patens*) could be the frequency of tidal inundations. *S. alterniflora* has a higher ability to cope with root zone anoxia from constant flooding.

Marsh water levels did not exhibit significant variations that corresponded with tidal fluctuations or seasonal variations. These muted responses might be due to the strongly attenuated tidal signals in the marshes along with nearly constant inputs of precipitation and losses due to evapotranspiration and drainage. The main hydrological change exhibited in the marshes was the response to Hurricane Sandy, which will be described in more detail in the next chapter.

This study showed that hydrologic regimes in ditched marshes differed from unditched sites. Atlantic Ditched sites showed a relationship between ground elevation and percent time water is at ground surface, but no relationship was found in the Chesapeake Ditched sites. The unditched plots were shown to be all above the ditched trend, indicating that the unditched site has more saturated conditions. The relationship between ground elevation and percent time water is at the surface was improved looking only at low water levels (10th percentile), but the 90th percentile showed no relationship at both locations. Water levels at the Unditched sites were above ground 50% of the time, and at the Ditched sites 15-35% of the time. These values indicate that the ditches are preventing frequent flooding and increase in saturation found in an unaltered marsh. These results support previous studies that suggested ditches decrease soil pore-water levels (Bourn and Cottam 1950; Lesser 1982; Adamowicz and Roman 2005), in which ditching did result in lower water levels relative to the surface.

Chapter 4: Changes in marsh hydrology due to Hurricane Sandy Storm surges in Ditched and Unditched Atlantic and Chesapeake Coastal Marshes

Introduction

Coastal salt marshes are productive and valuable habitats along the Atlantic Coast, covering approximately 5.8 million acres just on the East Coast (Stedman 2008). These coastal marshes are the interface between terrestrial and marine environments. Ecosystem functions in coastal marshes are at risk due to rising sea levels if marsh accretion does not keep pace with sea-level rise. Storm surges in particular may cause erosion, sea-water intrusion, and submergence of coastal marshes.

Coastal marshes are also important during extreme events because they can buffer the mainland from inundation and erosion by attenuating storm surges (Temmerman et al. 2013). Storm surges are among the most damaging and dangerous phenomena in coastal region communities, causing morphological change and disturbance within ecosystems (NOAA 2014). The effects of hurricane associated damage on coastal ecosystems can be extensive despite the mitigating effect of coastal wetlands (Sheikh 2005). Damage to the surrounding marshes produced by hurricanes depends partly on the water depths at the time of maximum wind stress and water storm inundation (Morton and Barras 2011).

Climate change may change the intensity and/or frequency of tropical storms and hurricanes that can generate significant heavy rainfall and/or storm surges along the Maryland Coast (NOAA 2014). Storms can have both long-term and short term effects on marsh hydrology and/or salinity. Short term effects of storm surges include increasing marsh hydrologic gradients, rising/lowering salinities, and temporarily shifting plant distributions (Gedan 2009). Marsh surface flooding can vary in duration, and magnitude within marshes and it can displace or drown nekton, invertebrates, mammals, and ground-nesting birds, which may result in localized population shifts/declines (Michener et al. 1997). Storm-induced sediment transport can be a significant part of marsh sediment budgets. Increases in elevation may occur due to sediment deposition and stimulation of root growth; decreases in marsh elevation may result from erosion and compaction or decomposition of marsh soils (Cahoon 2006; Turner et al. 2006). The contribution of storm-supplied sediments to overall marsh accretion likely varies due to marsh type, position, and storm frequency; and marshes with altered hydrology, such as ditched marshes, can have a net loss in elevation due to storms (Cahoon 2006). Many coastal marshes can adjust to sea-level rises associated with climate change, but may not be able to adjust to both sea level rises and changes in storm magnitude and frequency. It is possible that ditched marshes with altered hydrology may be more affected by these multiple stressors (Day 2008).

Mid-Atlantic coastal marshes commonly have extensive ditch networks that can partially drain or alter flow direction, and thus affect the ability of these coastal marshes to offer protection from storm surge and wave action. Currently 90% of the Atlantic Coast marshes between Maine and Virginia are ditched (Bourn and Cottam 1950). Many sites have also been hydrologically altered with Open Marsh Water Management (OMWM), which is a system designed to decrease mosquito production while maintaining marsh conditions. Open Marsh Water Management began in New Jersey during the 1960s (Weis and Butler 2009). Ditches are spaced typically 40 meters apart and usually arrayed perpendicular to tidal creeks (Figure 1.9). Large marshes often have additional orthogonal ditches that create a grid pattern. Ditches were placed to drain the marsh, decrease the duration of ponded water, and lower the water table (Bourn and Cottam 1950; Vincent 2013).

Storm surge behavior in ditched marshes likely varies with ditch size, orientation, and storm surge characteristics. Previous studies indicate that moderate storm surges with peak water depths less than 2 to 3 meters above the marsh are effectively attenuated by coastal marshes, whereas high storm surges (with depths > 5 m) are attenuated less (Loder et al. 2009; Wamsley et al. 2010). A modeling study by Wamsley et al. 2010 suggests that the reduction in storm surges by marshes is dependent on geomorphology (bathymetry, orientation of channels), on marsh plant height and flow resistance, and on storm characteristics (size, speed, storm track, and intensity). Comparisons of storm surges with paired vegetated & non-vegetated sites indicate that marsh vegetation reduces near-bed water velocity and increases drag (Neumeier and Ciavola 2004). Water velocities within vegetation are greatly reduced, and the flow often diverts around areas of dense growth. Vegetation characteristics such as stem height, diameter, flexibility, and impeded flow area also affect velocity profiles within and over vegetated marshes (Nepf 2004). Field-based observations of storm surges traversing wetlands often indicate a considerable decrease in storm surge height as a function of distance into the marsh (Lovelace 1994; Day et al. 2007; Krauss et al 2009; Wamsley et al. 2010). Reported storm surge attenuation rates range from 4.4 cm/km during Hurricane Andrew (Lovelace 1994) to 15.8 cm/km over coastal wetlands during Hurricane Charley (Krauss et al. 2009).

Purpose of this paper

Hurricane Sandy provided a unique opportunity to assess the magnitude and duration of the effects of storm surge for a pair of marshes with different management, ditched and unditched. Storm tides within marshes are affected by marsh narrowing, bed friction, and ditch size, orientation and spacing. It is not known whether ditched marshes respond similarly to unditched marshes and whether they significantly differ in attenuation or amplification of storm surges. The goal of the paper is to (1) Compare storm tide elevations and storm surge duration of ditched and unditched marshes, (2) Determine the effects of local morphology on attenuation/amplification of storm surges, and (3) Compare Atlantic Coast marsh and Chesapeake Bay marshes during Hurricane Sandy.

Storm Track of Hurricane Sandy

Hurricane Sandy originated as a tropical wave off the west coast of Africa on October 11, 2012, and it began its travel across the Atlantic. Around October 19, near east-central Caribbean Sea, the environment was becoming more conducive for storm development with falling pressures. On October 21 high pressure strengthened over the Gulf of Mexico and the southwestern Atlantic Ocean circulation of the low became well defined. On October 22, a strong band of deep convection formed, marking the start of a tropical depression. Sandy was initially slow but increased the rate of formation by the next day, October 23, began to head north northeast. On October 24, Sandy transitioned from tropical storm to a hurricane with a prominent eye. The hurricane weakened over Jamaica and Cuba, the slowing of the hurricane caused a shift towards northwest around October 26. The northward movement of the storm is shown in Figure 4.1. On October 28, Hurricane Sandy passed North Carolina a few miles off the coast. The storm center made landfall near Brigantine, New Jersey, just northeast of Atlantic City. After landfall, a weakened Sandy moved west-northwest while the center of the storm continued to move through southern New Jersey, northern Delaware and parts of southern Pennsylvania. (Figure 4.1) (NOAA 2013).

In the United States, most of the rain from Sandy fell south and west of landfall location, right where the field sites are located. The heaviest rainfall was reported in eastern Maryland and Virginia, southern Delaware and extreme southern New Jersey, with a widespread area of 12.7-17.8 cm of rain, and a peak amount of 32.6 cm in Bellevue, Maryland (Blake 2012). The research sites received 25 cm of accumulated precipitation.

The highest storm surge recorded by gauges along the Delmarva Peninsula was in Lewes, Delaware, with a surge of 1.63 meters. On the ocean side of the Maryland coast, the gauge at Ocean City Inlet measured a storm surge of 1.31 meters. The maximum storm surge measured in Virginia was 1.50 meters at Wachapreague on the Eastern Shore (Blake, 2012).

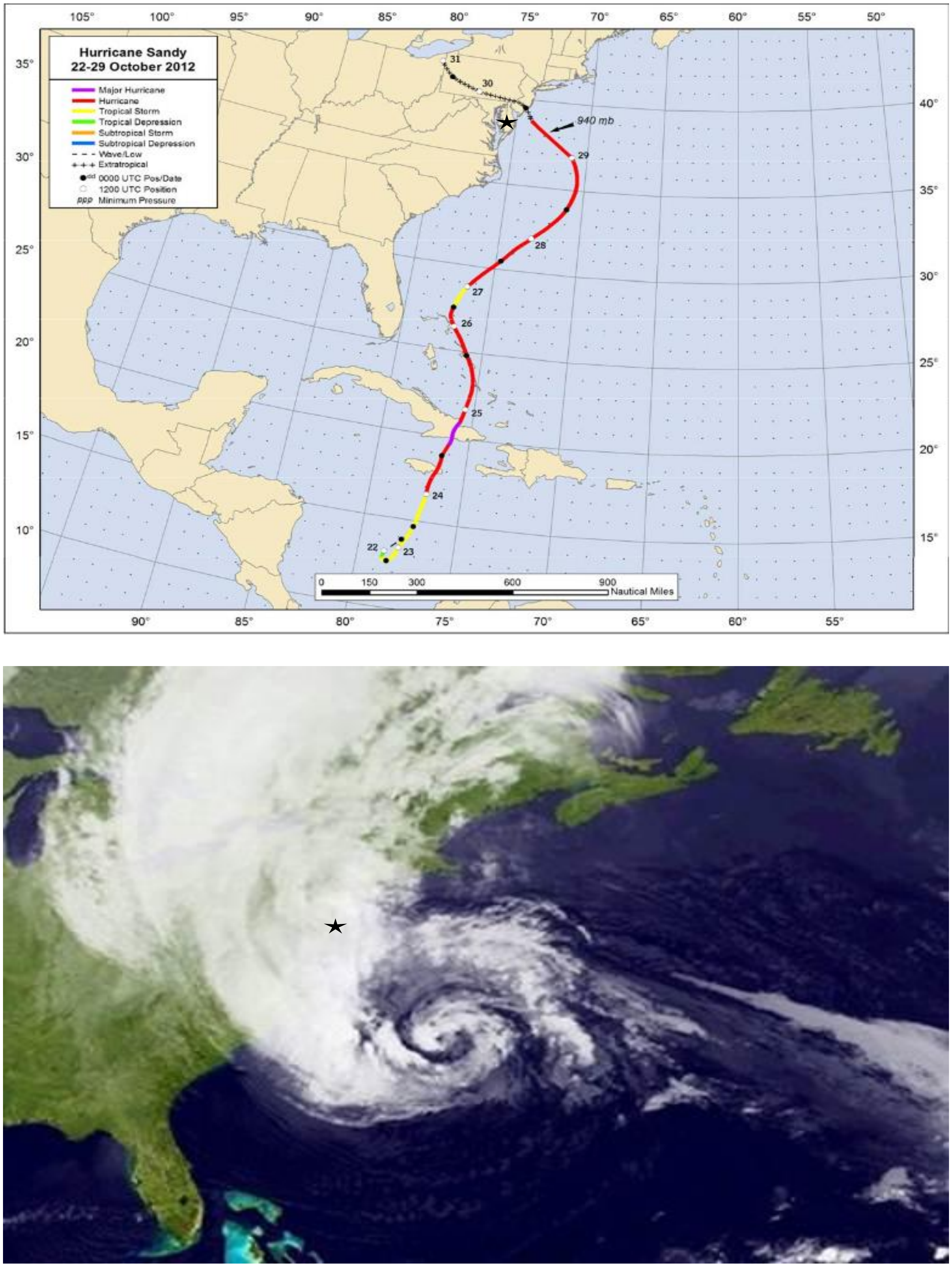


Figure 4.1: (A) Hurricane Sandy Track. Source: National Hurricane Center (NHC) Sandy Report (B) Satellite image of Hurricane Sandy Source: NOAA. Black star shows location of field sites.

Study sites

This chapter will focus on two pairs of research sites, one pair on the Atlantic Coast and the other on the Chesapeake Bay.

Atlantic Coast Sites-

The Atlantic study sites are a pair of tidal marshes located on Eastern Shore Maryland, Atlantic Coast, located in the EA Vaughn Wildlife Management Area (Worcester County) (Figure 1.9). Sites will be referred to as Atlantic marshes; Atlantic Unditched site is located at N 38° 04'40.99", W75° 22'55.44" and the Atlantic Ditched site at N38° 04'34.75" W75° 22'33.09". The nearby bay tides are semidiurnal with a mean range of 5 cm. The study sites have salinities ranging from mesohaline to polyhaline (approximately 0.1 to 30 ppt). The dominant plant species in these marshes are *Spartina alterniflora* (smooth cordgrass) and *Spartina patens* (marsh hay cordgrass). Ditched site have an average ditch spacing of 40 meters. The unditched marsh does not have ditches, but contain a small tidal meandering creek approximately 30 meters long, portions buried. All marsh sites are connected to the adjacent bay by a tidal channel. Soils were sampled in summer 2012 at various core depths up to 100 cm. The sites consisted of peat layers with depth averages of 89 cm for the ditched site and 84 cm for the unditched site. Both sites do not show spatial variation at these depths (see Chapter 3).

Chesapeake Bay Sites-

The Chesapeake Bay study sites are brackish marshes located on the western shore of the Delmarva Peninsula within the Deal Island Wildlife Management Area, Maryland (Somerset County) USA, near Monie Bay. A set of paired marshes consisting of a ditched and an unditched site were studied (Figure 1.9). These sites will be referred to as Chesapeake Bay Sites. Chesapeake Unditched site is located at N 38° 11'08.78" W75° 54'33.27" and the Chesapeake Ditched site at N38° 11'02.21" W75° 54'23.02". Monie Bay is characterized by semi-diurnal tides, with an average tidal range of 54 cm and a diurnal range of 63 cm. Depending on the season and the location from the tidal source, salinities range from mesohaline to polyhaline (approximately 6 to 21 ppt). Study sites are dominated by plant communities consisting of *Juncus roemerianus* (black needlerush), *Spartina patens* (marsh hay cordgrass), and *Spartina alterniflora* (smooth cordgrass). Additional plants species, although in small quantities, include *Schoenoplectus americanus* (Olney's threesquare) and the invasive *Phragmites australis* (common reed). The substrate is peat approximately to 100 cm below ground followed by loam. Study sites were selected based on availability of ditched and unditched pairings close by, connectivity to the same tidal source, and ditches with similar geomorphology.

Methods:

Ditched and unditched marsh sites were equipped with hydrological monitoring equipment prior to Hurricane Sandy. Hydrological monitoring equipment at the ditch sites consisted of 3 transects of wells arrayed perpendicular to the ditches and parallel to the tidal creek. For ditch-drained sites, the first transect is located closest to the tidal creek and contains 7 wells, the second transect has 3 wells, and the far transect has 5 wells (Figure 4.2). Wells were installed laterally from a dominant ditch. Each well was installed 2.2 m deep. Wells were located at the midpoints, quarter points, and within ditches. Well installation at unditched sites were also arrayed with 3 transects of wells parallel to the main tidal channel, the first transect having 3 wells; the second transect 1 well, and the far transect 3 wells. Wells were equipped with Odyssey Data Loggers that were calibrated in the lab and the water levels were periodically field-checked. Calibration equations were developed for each data logger if required. Water levels were recorded every 15 minutes.

On site weather gauges failed during the storm, therefore data from nearby weather stations were used for precipitation, wind speed, and direction. Tide gauge data were obtained from National Oceanic Atmospheric Administration (NOAA) at their Ocean City gage (Station ID: 8570283) and the Bishop Head gage (Station ID: 8571421). Data were also obtained from the United States Geological Survey (USGS) station at Chincoteague Island. This station was only deployed for Hurricane Sandy. A comparison of the Ocean City and Chincoteague gages shows similarities in duration, magnitude, and timing. The Ocean City gage has a dataset available for a period before and after the storm surge, therefore, this gage was used in the data analysis along with the marsh gages. (Figure 4.3).

Mini wells and soil plots were located perpendicular to the tidal creek. Stratigraphy was evaluated at each soil plot by core sampling to a depth of approximately 90-100 cm and qualitatively described. Cores were separated by horizons and bagged in the field and refrigerated until laboratory analysis occurred. Cores were sub-sampled, weighed, dried at 105ºC for 24 hours, and then reweighed for water content. See Chapter 2 and 3 for more detailed information.

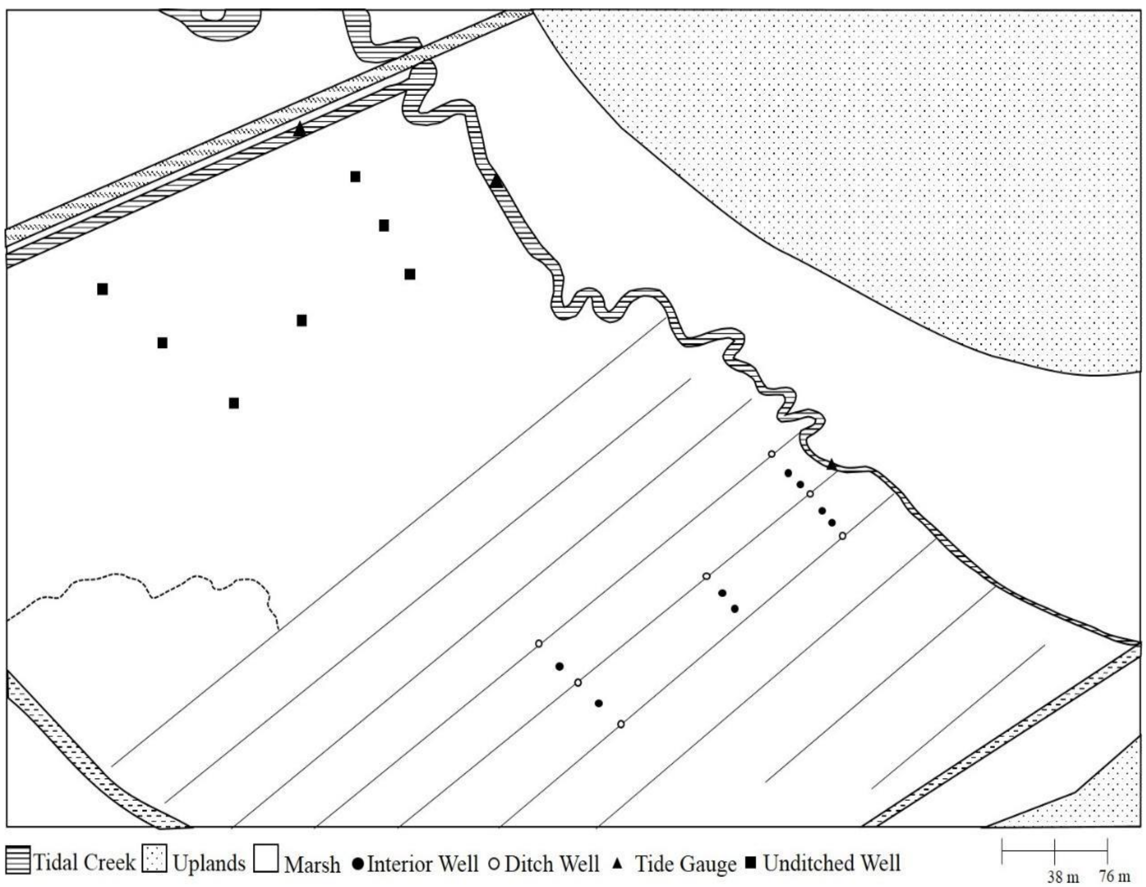


Figure 4.2: Site maps showing ditched and unditched sites at the Chesapeake Bay sites. Note well locations, ditch spacing, orientation, and tidal creek.

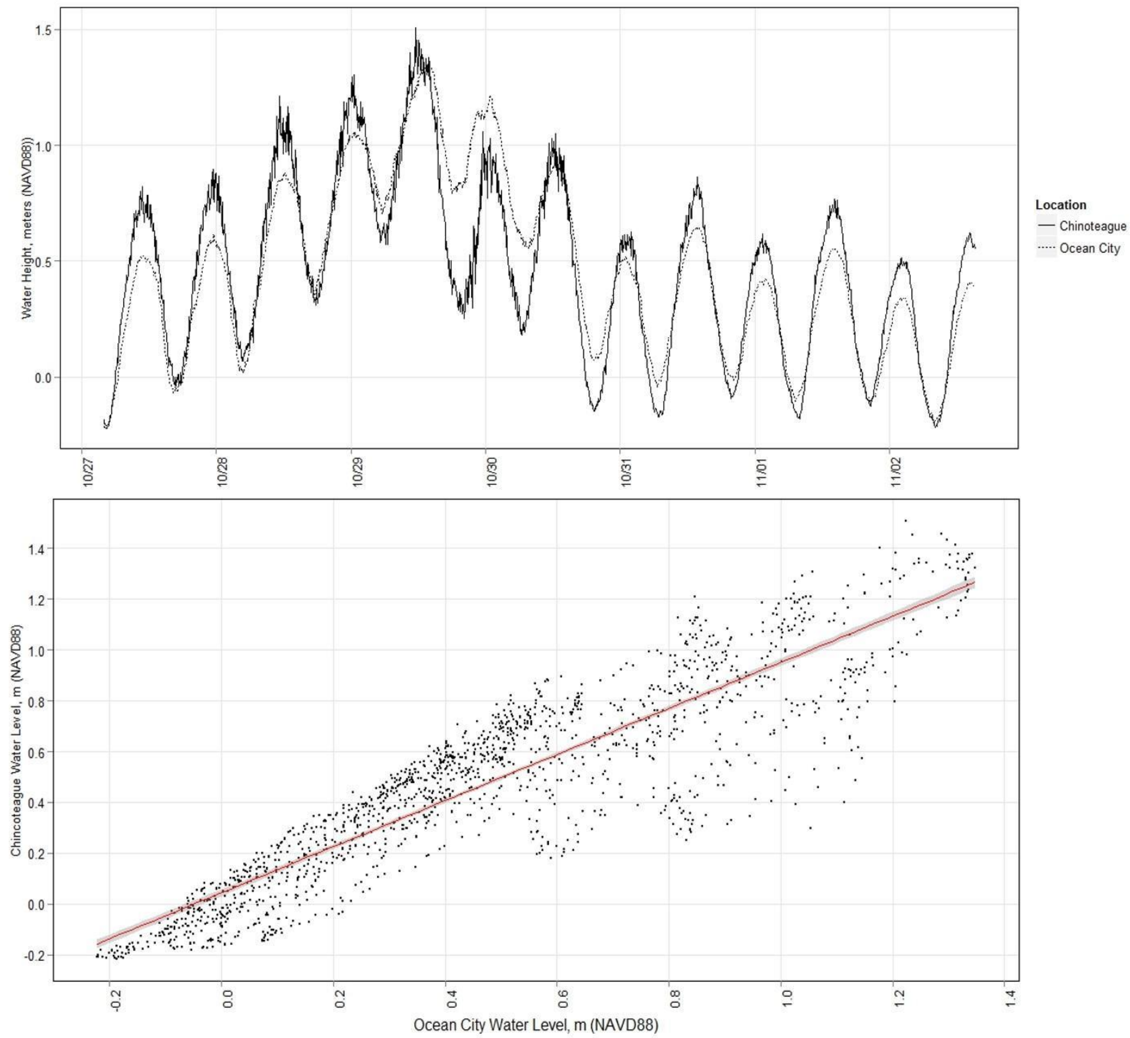


Figure 4.3: Comparison of NOAA tide gage at Ocean City Maryland versus USGS tide gage at Chincoteague. Atlantic research site lies between these gages on Chincoteague Bay (Figure 1.9).

Results

<u>Atlantic Coast Sites:</u>

Storm Track, rainfall, wind directions, and tidal surge during Hurricane Sandy

The cumulative rainfall during Hurricane Sandy was 22.5 cm at the research site. This is the highest rainfall event during the overall research project from 2011-

2015 (Table 4.1). In 2012, Hurricane Sandy rainfall contributed 20% of the yearly rainfall and increased rainfall for the month of October by 17 cm compared to the average. Hurricane Sandy also produced the highest recorded daily precipitation by day for 2012 on October 29th at 16.9 cm (Figure 4.4).

Hurricane Sandy was rated as a category 1 hurricane as it moved across the Delmarva coast, with a maximum wind speed of 65 km/hr on October 29, 2012 counter clockwise, with the direction from the North (Figure 4.4). High wind speeds coming from the North - Northwest generated a storm surge that reached 135 cm, 122 cm above the predicted tides for the area (Figure 4.4). Dominant wind direction then switched from the Northwest to the Northeast during the storm peak. Field sites are affected by winds from East, South, NE, and SE. The N and NE winds drove surge from Atlantic Ocean into the Chincoteague Bay then into Atlantic research sites.

Table 4.1: Annual Climatological Summary, National Centers for Environmental Information

Year	**Precipitation Yearly Total (cm)**	**October Precipitation (cm)**
2012	127.84	27.18
2013	129.24	12.19
2014	105.49	5.08
2015	132.89	13.28

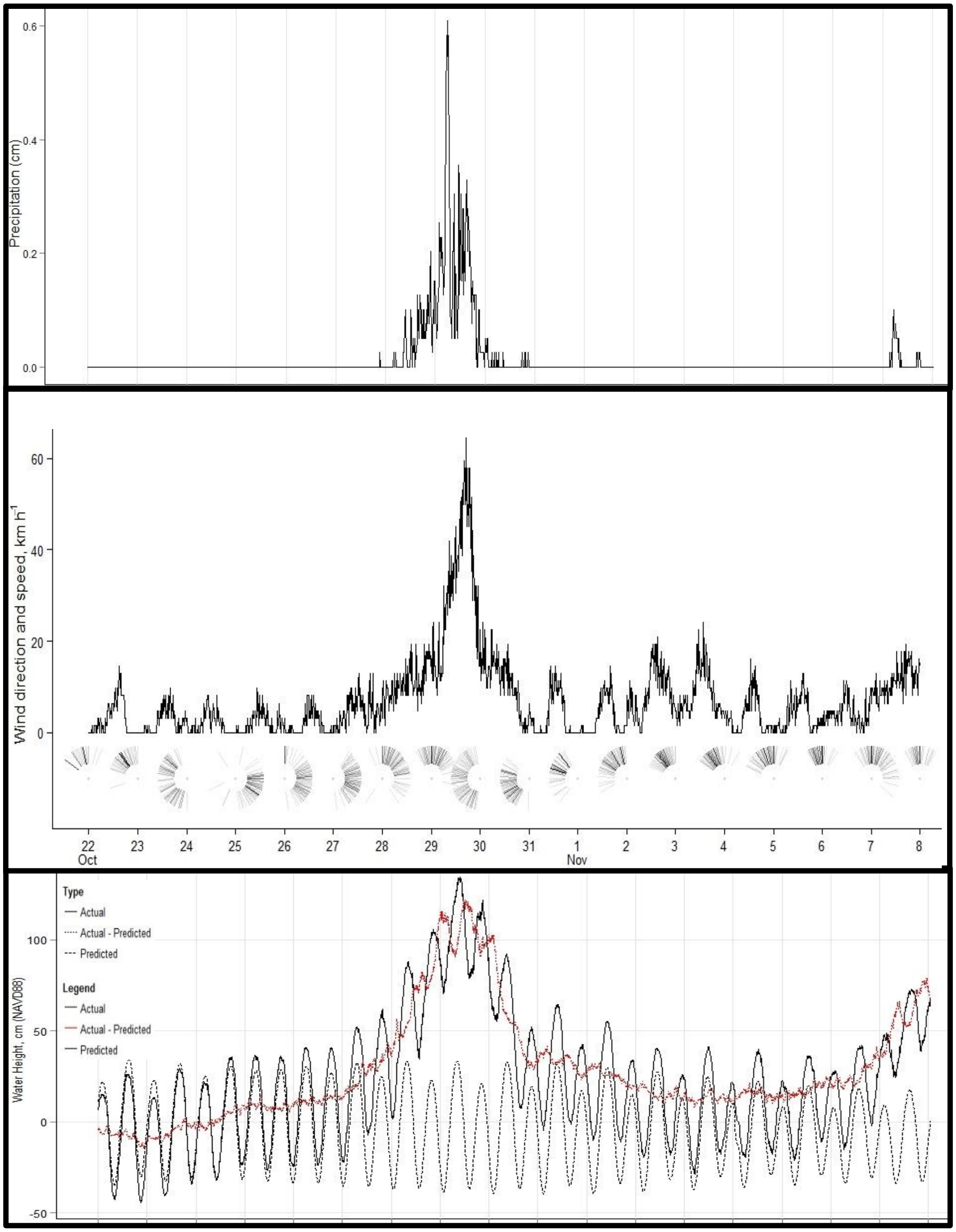


Figure 4.4: (a) Precipitation. Total accumulation over 3 days was 22.5 cm, (b) Maximum wind was 65 km/hr; wind direction shifted near the peak storm; (c) Tidal stage at Ocean City compared to predicted gauge height. Storm Surge shown in red (peak is 122 cm greater than the expected tidal stage)

Response of Marsh Water Levels during Sandy

The high water levels led to widespread failure of tide gauges and wells. Surviving water level loggers were used to define maximum water level, and overland flow depth (depth of the water above the ground). Water levels in wells provided information on the areas of inundation.

Ditched Sites: Wells located in the ditch, similar to tidal gages, showed higher water levels in all transects; however, transect 3, which lies in the at the back of the site, furthest from the tidal source (Figure 4.5) showed the highest in the entire site. Ditch wells, regardless of transect location were lost the visible tide cycle around October 25, three days before the storm, and the semidiurnal tidal influence did not return until November 4, roughly 3 days after the storm. Maximum water levels in the ditches furthest from the tidal source were -58.2 cm above Mean Low Low Water (MLLW), whereas closest to tidal creek only produced -45.8 cm above MLLW (negative values mean above ground).

Interior wells showed a similar pattern, water levels were highest in the middle of the sites at -46.23 cm, lowest levels at 29.4 cm closest to the tidal creek (negative values mean above ground). The back interior wells all failed during the storm. Interior wells also show lower water levels on average compared to in ditch wells. In addition, ditch wells maintained a constant water level during peak storm for longer period of time closest to the creek, 99.4 hours versus 83.4 hours (Table 4.2). Most wells peaked on October 30th 2012 between 2:35 and 3:35pm.

Unditched Sites: The well located closest to the tidal creek had a maximum storm surge of 53.8 cm above the individual MLLW. The well in the middle of the site had a maximum water level at 44.8 cm. The Unditched site showed significant difference from the Ditched site ditch wells and interior wells in the duration at maximum value, 7 hours versus 86.5 hours. Unlike the ditched site, unditched wells did not show any difference in water levels until the storm began and the water levels returned to normal almost immediately after the storm ended. Most wells peaked on October 30th 2012 around 12:45, almost 3 hours before the ditched site.

Water level cumulative distributions for the unditched site indicate that during Sandy, most wells showed water levels at or above ground 97% or more of the time for a 7-day period, with the exception of well 2 which only showed above ground measurements 50% of the storm. Water level probability curves indicate that if the Hurricane Sandy data are excluded from the 2012 dataset, water levels were at or below the marsh surface for 10% of the time, with the exception of well 2 for 85% of the time.

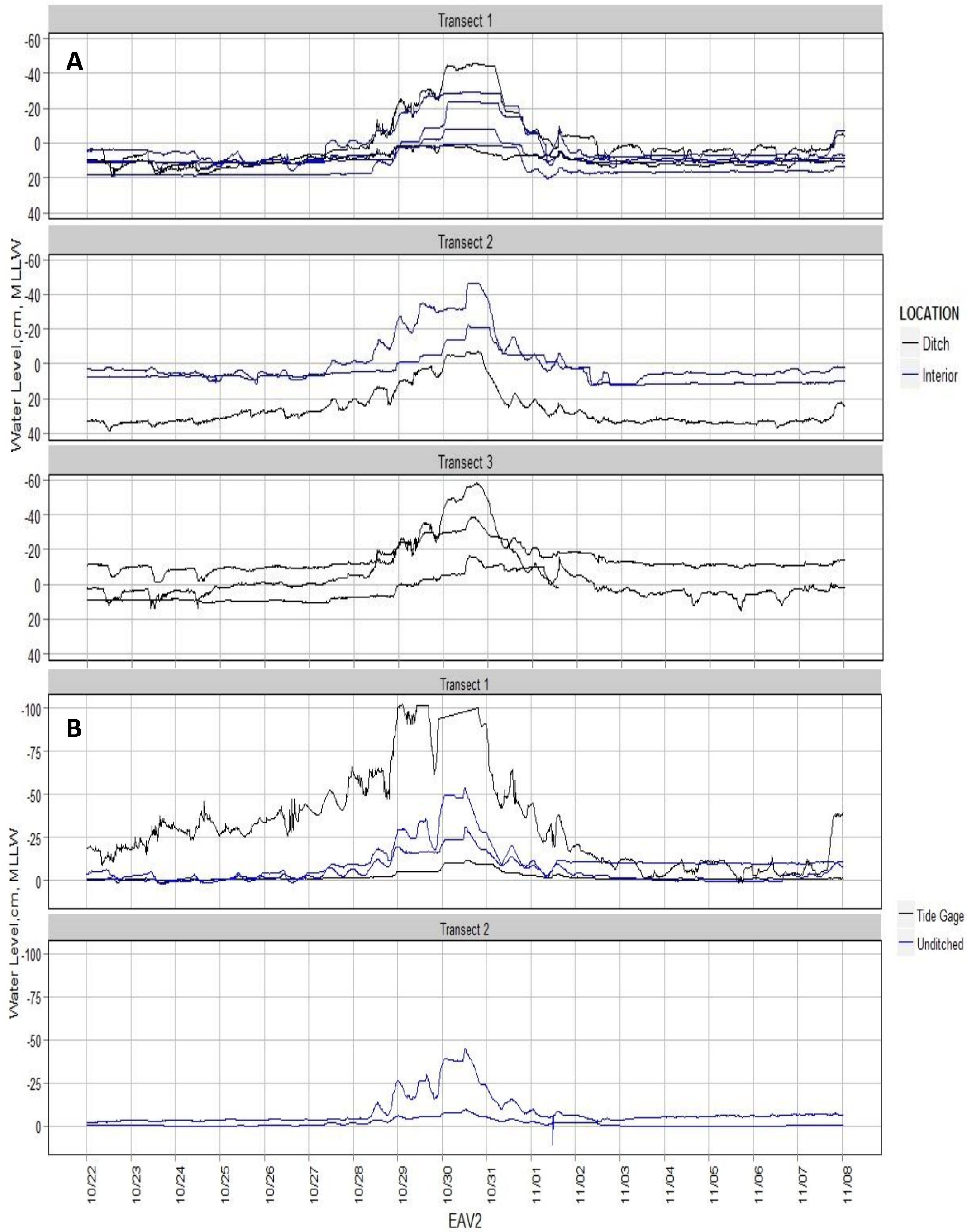


Figure 4.5: (A) Ditched Site- Time series showing water levels in wells located in the ditches and in the interior portions of the marsh. (B) Unditched Site- Time series showing water levels in wells located in the unditched wells and the tidal creeks. Note the duration at maximums and differences in the maximums levels.

Table 4.2: Storm Surge Duration at maximum (hours) NOTE: Transect 1 (T1) is closest to tidal source					
Ditched Wells	Duration	*Interior Wells*	Duration	*Unditched Wells*	Duration
3 (T1)	120	2 (T1)	67	1 (TC)	2
5 (T1)	78.8	4 (T1)	66.5	2	0.5
211 (T2)	77.3	14 (T1)	68.8	3	0.5
7 (T3)	116	15 (T1)	131.3	6	0.68
9 (T3)	88.9	16 (T2)	77.3	8 (TC)	40
11 (T3)	16.8	17 (T2)	129	9	0.75
AVERAGE (T1)	99.4	*AVERAGE* (T1)	83.4	*Interior*	0.61
AVERAGE (T2)	77.3	*AVERAGE* (T2)	103.2		
AVERAGE (T3)	73.9	*AVERAGE* (T3)	-		

Storm surges versus lunar spring tides time series and hysteresis:

Comparison of storm surge water levels during Sandy with monthly spring tides indicates Sandy storm surge water levels reached 40- 100 cm above local ground, whereas spring tides ranged from 18 cm above local ground surface at ditched sites, and 14 cm below ground for unditched sites. Both sites show little effect on the water levels with a full moon in the months of September and October 2012 (Figure 4.6/4.7/4.8). Comparison of marsh water levels to local Atlantic Coast storm surge water levels indicates height attenuation of 50%; approximately 4 cm/km. Atlantic Coast tide gage (Ocean City) shows maximum peak almost a day before the marsh sites (Figure 4.3). Marsh water levels indicate that maximum water levels lagged behind the coastal storm surge, but also did not drain as quickly. The marsh water levels built to higher levels over the course of the storm surge, indicating that water was prevented from exiting the marsh system, which continued to raise marsh water levels.

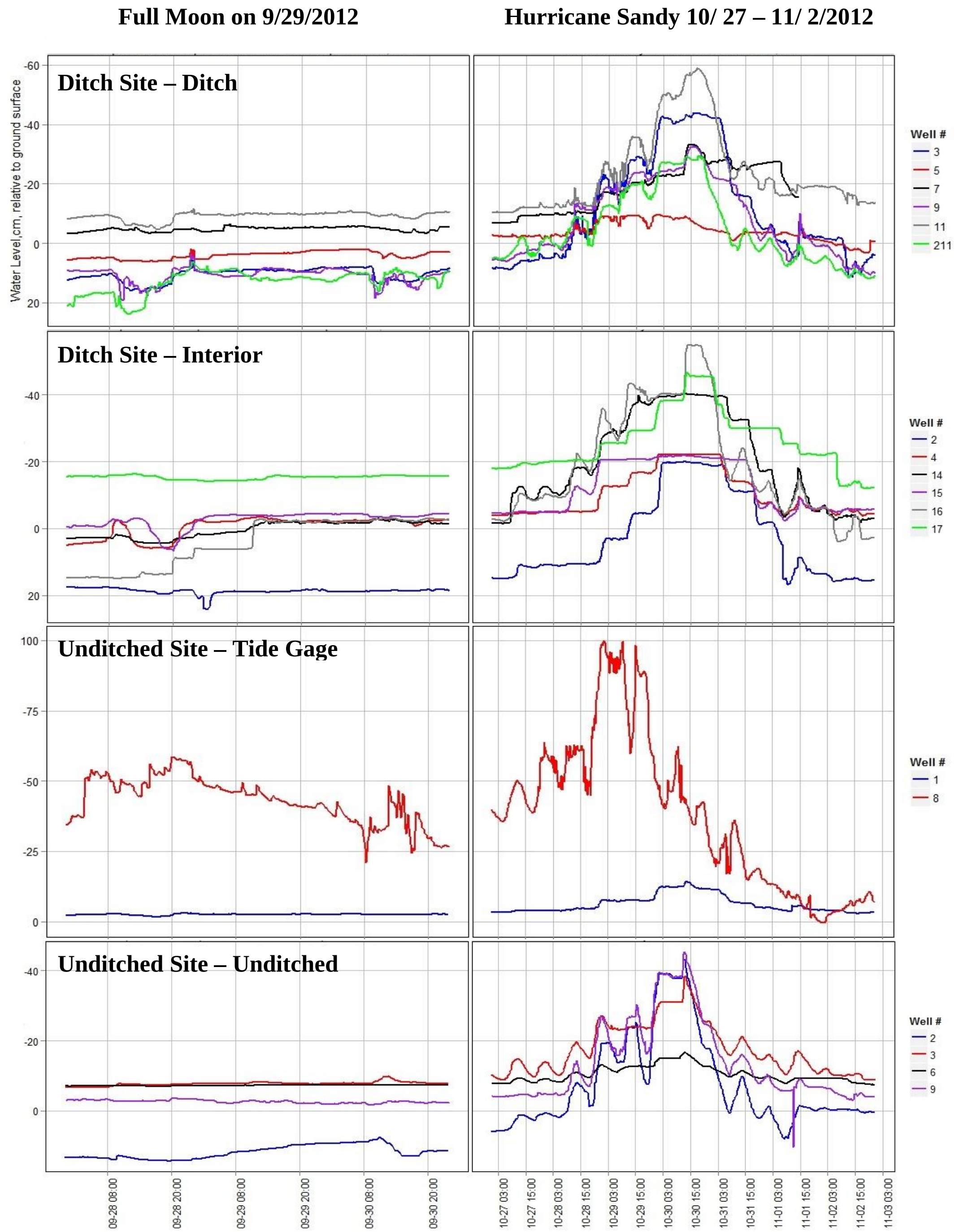


Figure 4.6: Time series showing water levels in wells located in the ditches and in the interior portions of the marsh for the Ditched Site, and time series showing water levels in wells located in the unditched wells and tidal creeks for a lunar high tide and Hurricane Sandy.

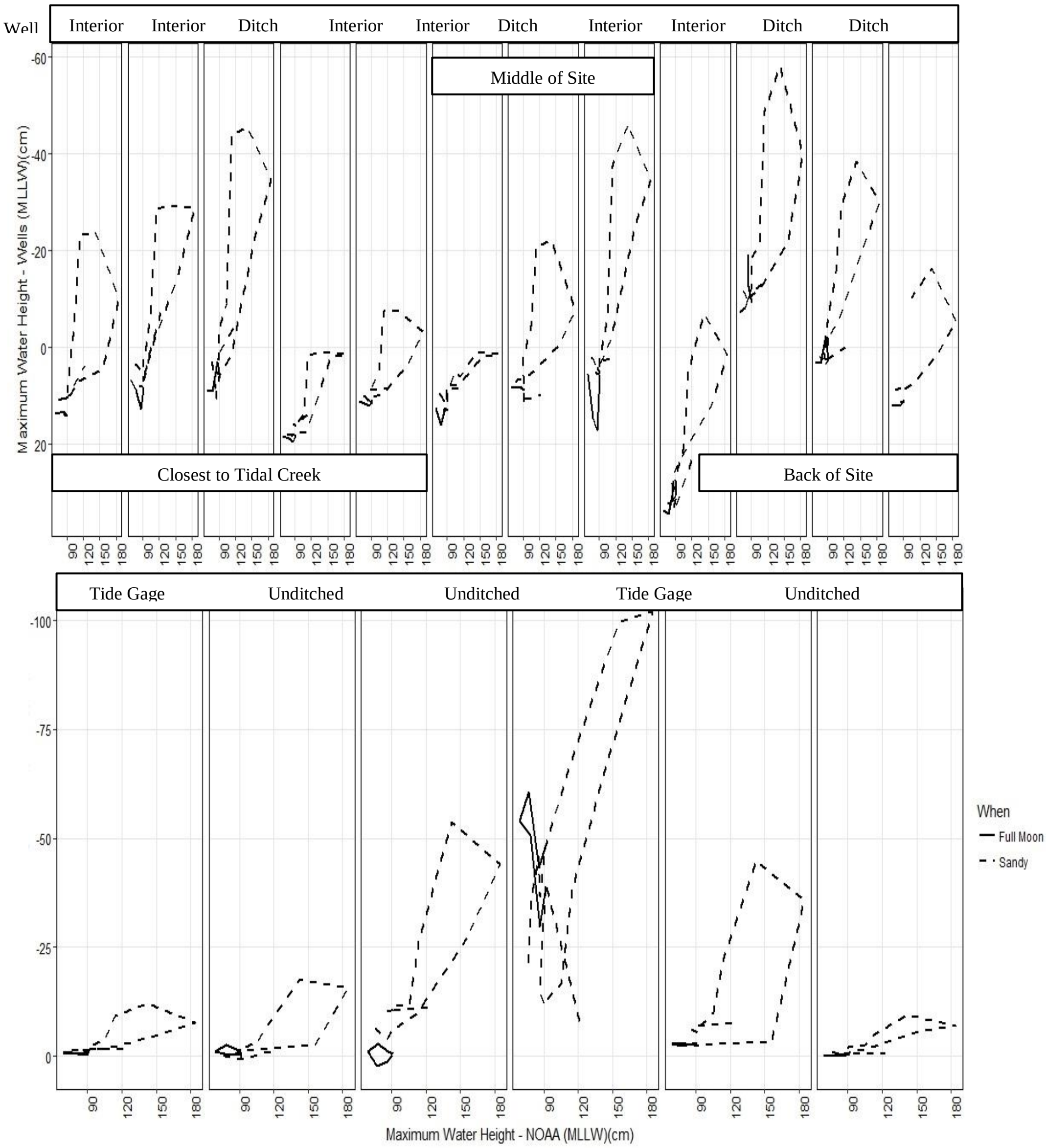


Figure 4.7: Hysteresis graphs for Ditched and Unditched Sites. See figure in appendix for comparison between Sandy, full moon, and second biggest storm.

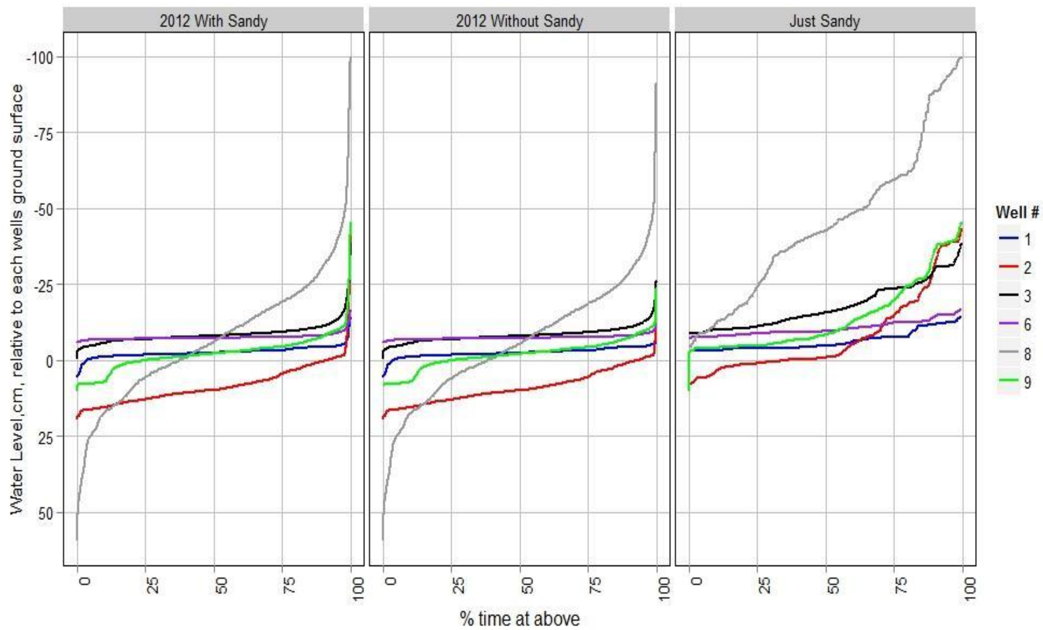


Figure 4.8: Probability curve water levels at or above the ground surface for unditched site for three time periods: Annual 2012 dataset with Sandy, Annual 2012 when Sandy subset data was removed, and the Sandy subset data.

Chesapeake Bay Sites:

Storm Track, rainfall, wind directions, and tidal surge during Hurricane Sandy

The cumulative rainfall during Hurricane Sandy was 25 cm at the research sites. This was the highest rainfall event during the overall research project from 2011-2015 (Table 4.3). If the measured yearly precipitation total is doubled to compensate for the missing 6 months of data (73.6 cm yearly), then in 2012, Hurricane Sandy rainfall contributed 34% of the yearly rainfall and increased rainfall for the month of October by 8.7 cm compared to the following October. Hurricane Sandy also produced the highest recorded precipitation by day for 2012 on October 29th at 17.2 cm (Figure 4.9).

Hurricane Sandy was rated as a category 1 hurricane as it moved across the Delmarva coast, with a maximum wind speed of 88.5 km/hr on October 29, 2012, with the direction coming from the North (Figure 4.9). High wind speeds and North and Northeast wind direction generated a storm surge that reached 93 cm, 65 cm above the predicted tides for the area (Figure 4.9). Dominant wind direction switched from Northeast to Southwest winds during the storm peak.

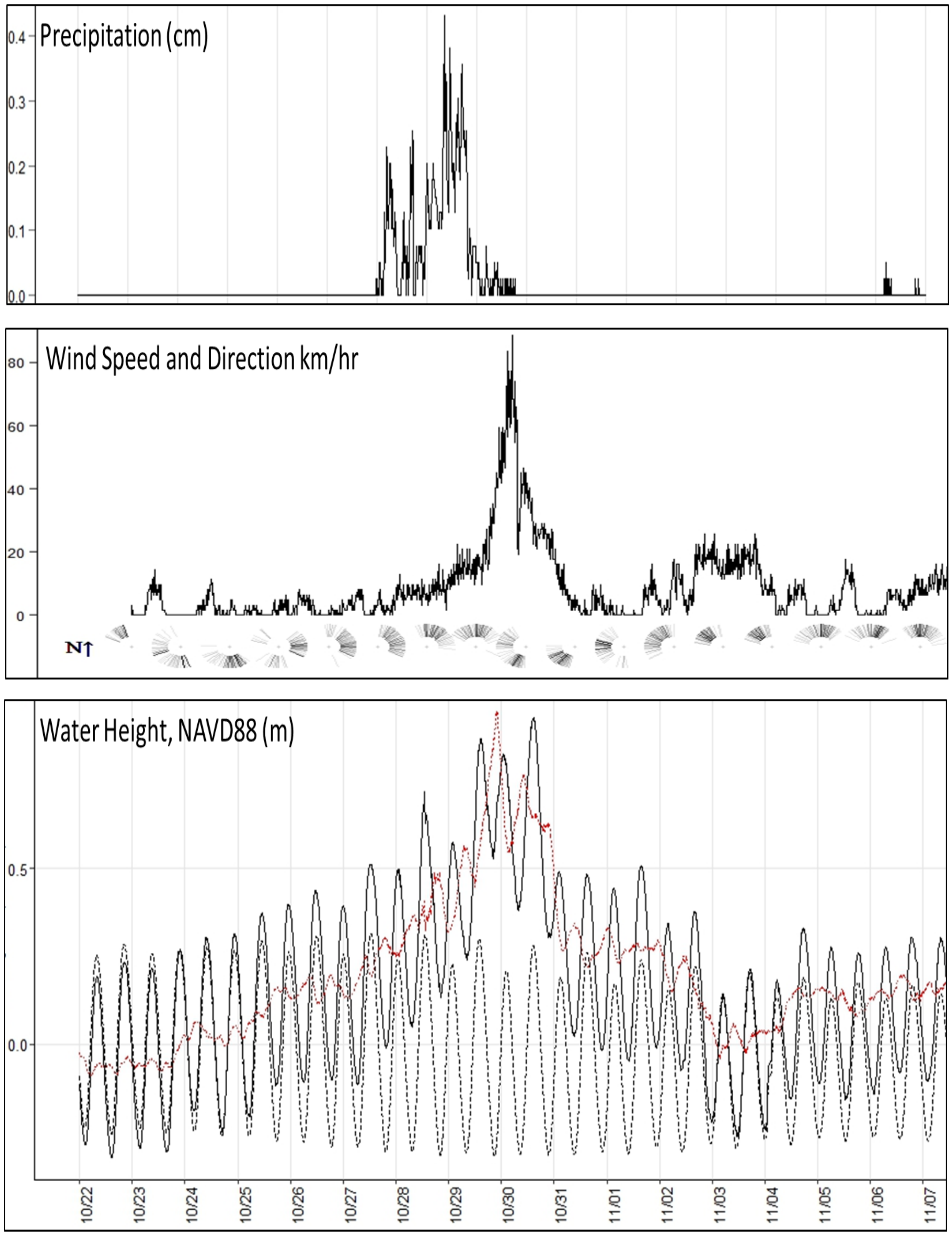


Figure 4.9: (a) Precipitation. Total accumulation over 3 days was 25 cm, (b) Maximum wind was 65 km/hr; wind direction shifted near the peak storm; (c) Tidal stage at Bishop Head, NOAA tide gage

Table 4.3: Annual Climatological Summary, National Centers for Environmental Information

Year	Precipitation Yearly Total	Precipitation Total - October
2012	36.8*	NA
2013	60.7*	16.3
2014	70.6**	5.5
2015	37.7*	NA

*missing 6 months of data **missing 4 months of data

Marsh Water Levels during Sandy

The high water levels led to widespread failure of tide gauges and wells. Surviving water level loggers were used to determine maximum water level, overland flow depth (depth of the water above the ground), and areas of inundation.

Ditched Sites: Wells located in the ditch, similar to tidal gages, showed lower water levels in all transects; however, transect 2, which lies in the middle of the site, (Figure 4.10; Table 4.4) showed the highest in the entire site. Ditch wells, regardless of transect location, were lost a visible tide cycle around October 27, one day before the storm, and the tidal influence did not return until November 1, roughly 1 day after the storm. Maximum water levels in the ditches furthest from the tidal source were -87 cm above Mean Low Low Water (MLLW), whereas closest to tidal creek only produced -140 and -50 above MLLW (negative means above MLLW).

Interior wells water levels were highest in the front of the sites at -67.4 cm, lowest levels (~ -55 cm) farthest to the tidal creek. Interior wells also showed slightly higher water levels on average compared to ditch wells. In addition, interior wells maintained a constant water level during peak storm for longer period of time closest to the creek, 13.25 hours versus 7.5 hours (Table 4.5). Most wells peaked on October 30th 2012 between 14:02 and 15:05.

Unditched Sites: The well located closest to the tidal creeks, had a maximum storm surge of 48.3 cm above the individual MLLW. The well in the middle of the site had a maximum water level at 45.8 cm. Unditched showed similar durations between ditched site and unditched site regardless of well type. Unlike the ditched site, unditched wells did not show any difference in water levels until the storm began and the water levels returned to normal almost immediately after the storm ended. Most wells peaked on October 29th 2012 around 23:23, approximately 15 hours before the ditched site.

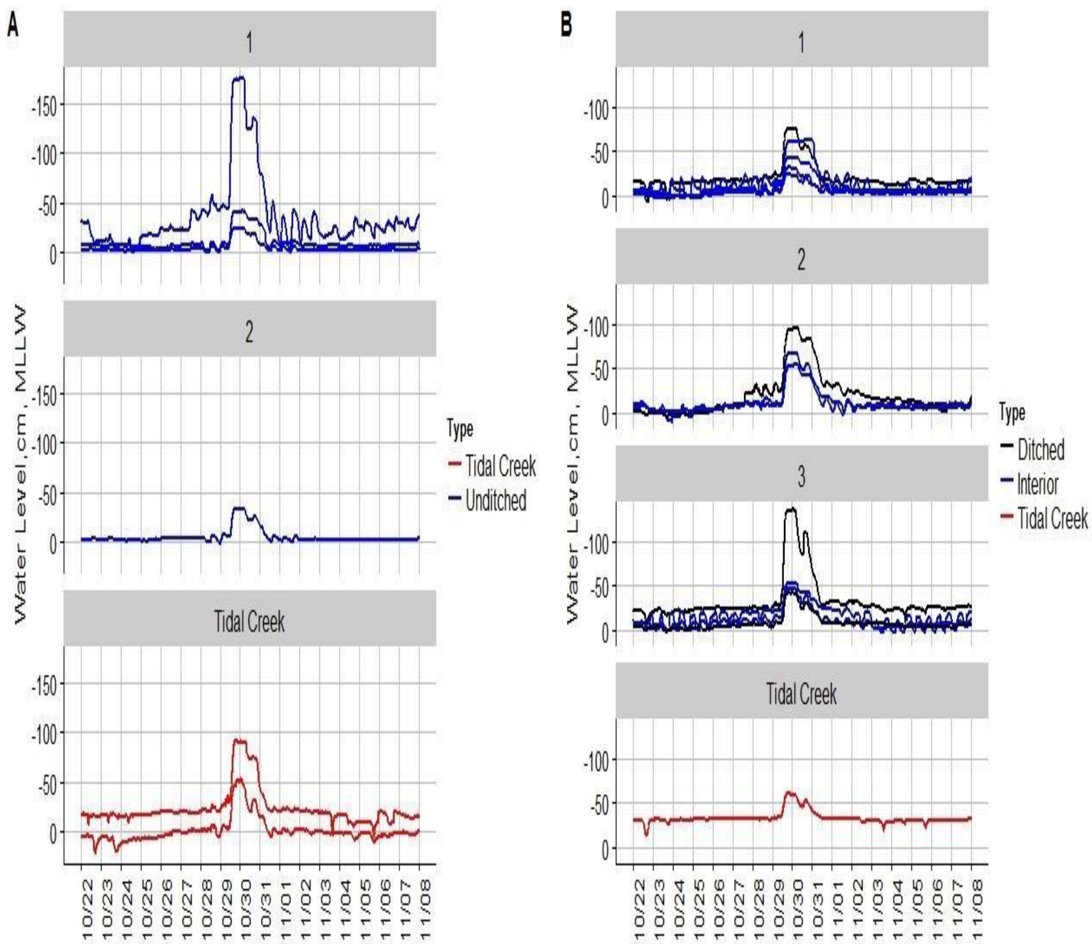


Figure 4.10: Time series of sites and transects. (A) Unditched Site- Time series showing water levels in wells located in the unditched wells and the tidal creeks. (B) Ditched Site- Time series showing water levels in wells located in the ditches and in the interior portions of the marsh.

Table 4.4: Duration at storm surge maximum (hours) NOTE: Transect 1 is closest to tidal source

Ditched Wells	Duration	*Interior Wells*	Duration	*Unditched Wells*	Duration
5 (T1)	7.5	2 (T1)	10.4	1	14
25 (T2)	13.5	212 (T1)	11.5	2	13.5
10 (T3)	8.3	26 (T1)	22	3	12.8
6 (T3)	8.25	4 (T1)	9.1	7 (TC)	12.5
11 (TC)	8.5	16 (T2)	12	8 (TC)	8.3
		210 (T2)	10.2	9	7.5
		9 (T3)	6.5		
		7 (T3)	8.5		
AVERAGE (T1)	*7.5*	*AVERAGE (T1)*	*13.25*	*Interior (T1)*	*13.4*
AVERAGE (T2)	*13.5*	*AVERAGE (T2)*	*11.1*	*Interior (T2)*	*7.5*
AVERAGE (T3)	*8.3*	*AVERAGE (T3)*	*7.5*	*Tidal Creek*	*10.4*
AVERAGE (TC)	*8.5*				

Storm surges versus Spring tide time series and tidal stage hysteresis:

Comparison of marsh water levels during a spring tide (full moon) indicates that high tide water levels are near the ground surface in almost all wells in both ditched and unditched sites. Both ditched and unditched sites show little water level response with a spring tide in September (Figure 4.11). Atlantic tide gage shows maximum peak the same day as the sites during Hurricane Sandy. Using hysteresis to describe storage shows ditched sites had higher water levels than the unditched site. Ditches in the ditched site are shown to reduce interior marsh flooding by creating extra storage space and acting as conduit flow (Figure 4.12).

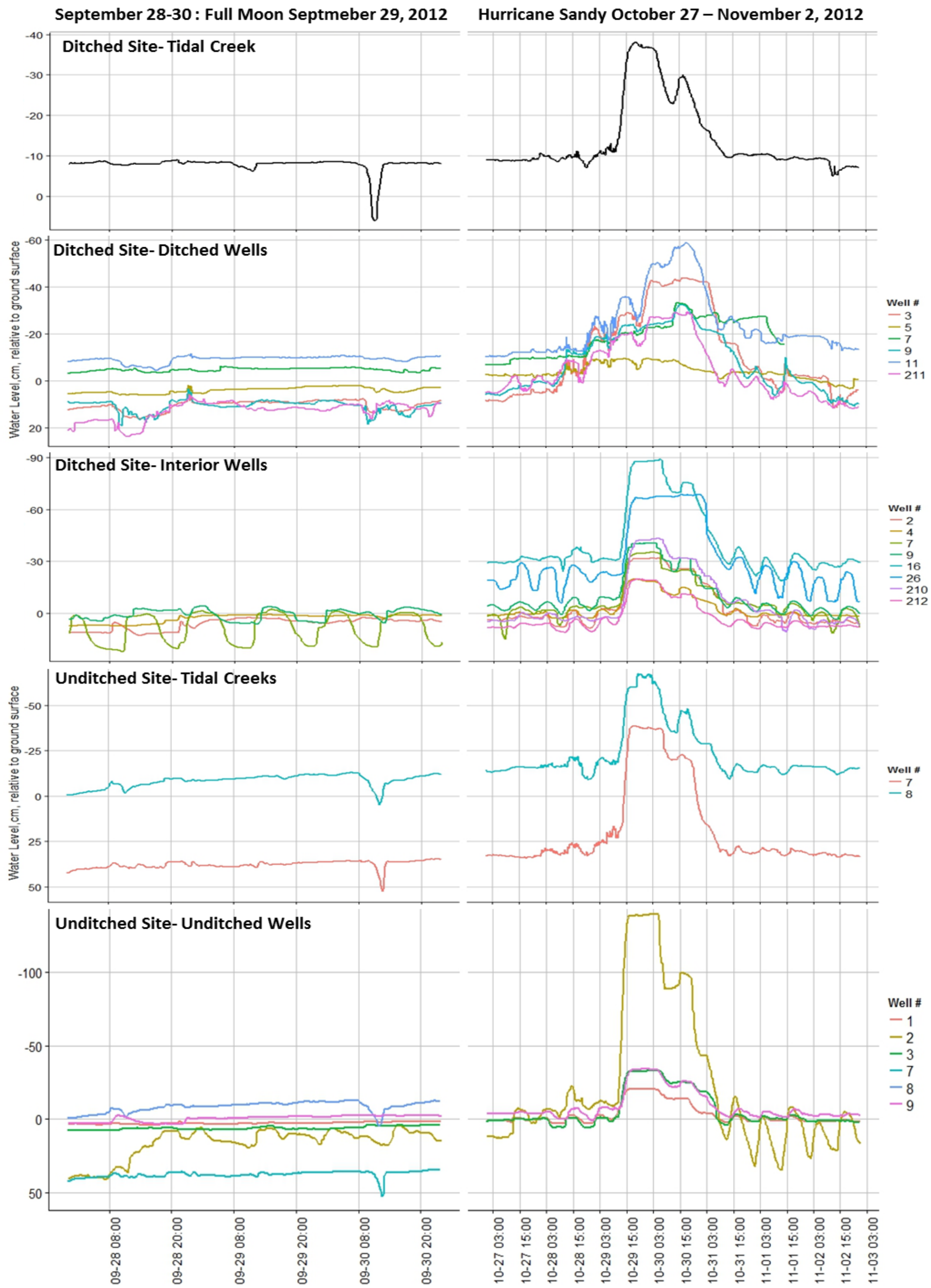


Figure 4.11: Maximum water levels on the NOAA tide gage and maximum marsh water levels for Ditched and Unditched Sites.

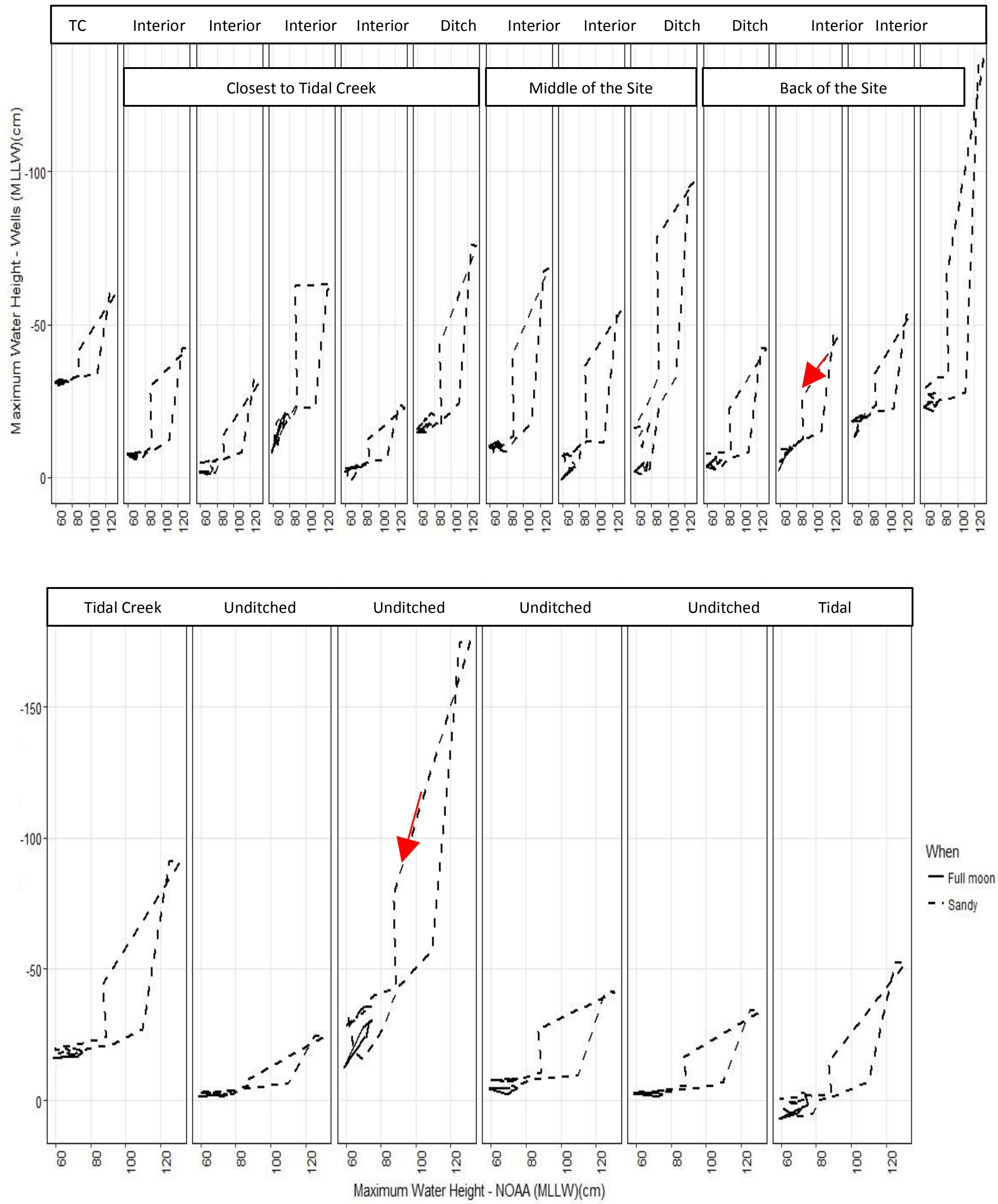


Figure 4.12: Hysteresis graphs for Ditched and Unditched Sites. Arrows indicate direction.

Discussion

Hurricane Sandy produced large storm surges causing flooding and wind damage in 24 states with an economic losses of tens of billions of dollars (FEMA 2013). Along both the Atlantic and Chesapeake Bay portion of Maryland, tidal bays overflowed into the nearby marshes and subtidal ditches. These associated coastal marshes experienced prolonged flooding and storm tide conditions similar to studies reported from New Jersey (Miselis et al. 2015). While major storms often have prominent and long-term effects on coastal marshes, these data suggest that Hurricane Sandy had localized and temporary impacts on these microtidal salt marshes.

Supporting modeling studies by Loder et al. (2009) who showed that the effects of ditches or channels on storm surge amplification decreased with channel size. This study suggests that small ditches had relatively little effect on storm surge attenuation in the marshes. Due to the position of the site within tidal embayments or behind barrier islands, the storm surge attenuation mostly occurred within Chincoteague Embayment, the Chesapeake Bay, and Monie Bay. The storm surge appeared to be damped by these embayments and by flow across the marsh before reaching the sites. This information is important for predicting and modeling future storm impacts and for local wetland managers to understand the role of storms in marsh dynamics.

The duration of high tides was significantly different between the Chesapeake and Atlantic sites. The Chesapeake Bay marshes had a much shorter time period of flooding compared to Atlantic Coast marshes, likely due to the position of the marsh and its tidal channels relative to the wind and storm tide directions. At the Atlantic Coast sites, the extremely large area of storm force winds north and west of the center of counterclockwise circulating around Hurricane Sandy generated large ocean and bay water storm surges, over multiple tidal cycles. Along the coast of Maryland, Assateague barrier islands were breached, Chincoteague Bay overflowed into the subtidal shallow ditches, and coastal marshes experienced prolonged flooding and storm tide conditions (Miselis et al. 2015).

This study documented the effects of Hurricanes Sandy on the hydrological responses on Atlantic Coast marshes in Maryland. The Sandy surge was dissipated by the coastal bays and marshes. The hysteresis of water levels between the NOAA tide gage and marsh gages indicated that storage of water in the marsh (in channels, ditches, and marsh surfaces) was likely responsible for the long duration of high water levels in the Atlantic marsh. This long-duration of high water levels in the marsh may be a water storage mechanism that protects inland areas, including coastal communities. It is possible, however, that long durations of high water levels may lead to disturbance in plant or animal communities. From the perspective of the role of ditches, however, results of this study suggest that ditches did not have a significant effect on either storm surge height or duration. Therefore, the restoration of ditches in these types of interior marshes in microtidal settings for the purposes of improving storm surge attenuation would not an effective use of restoration funds.

Chapter 5: Evaluation of hydrological consequences of ditch plugging restoration projects

Introduction

Coastal marshes have a long history of anthropogenic alterations (ditching, impoundments, dikes, open water management systems, etc.). Drainage ditches are common geomorphological feature of East Coast marshes; most were dug since the 1930's in an attempt to control the breeding of salt marsh mosquitoes. It is estimated that 90% of marshes from Maine to Virginia have been ditched (Bourn and Cottam 1950). Ditches are on average spaced approximately 40 m apart and tend to lower marsh water tables (Stearns 1940; Bourn and Cottam 1950; Turner 1997; Adamowicz 2005; Gedan 2009); although not all studies indicate declines in water levels relative to the ground surface (Vincent 2013). Some studies indicate little change in water table several years after ditches were installed. The study by Stearns et al. (1940) indicated water table lowering of about 4 cm two years after ditch installation. It has also been shown that ditches drain and decrease the prevalence of natural pools on marsh surfaces, which are important to fish and waterfowl (Adamowicz 2005; Lathrop, Cole, and Showalter 2000). Ditches in some marshes replace tidal creek functions, leading to changes in the size and discharge in the natural tidal channels (Adamowicz and Roman 2005). These changes in marsh hydrology caused by ditches are associated with changes in natural vegetation patterns (Bourn and Cottam 1950; Daiber 1986; Miller and Egler 1950; Niering and Warren 1980; Clark et al. 1984; Raposa and Roman 2001) and decreased bird habitat.

Almost all of Maryland's coastal marshes have been impacted by the excavation of mosquito drainage ditches. These straight, narrow ditches were designed to drain the upper reaches of salt marshes under the assumption that this would remove shallow water breeding habitat for mosquitoes. Coastal managers now know that this drainage approach was poorly designed; draining the high marshes eliminated vast amounts of habitat for marsh fishes that prey upon mosquito larvae. Furthermore, ditches that partially fill with sediment can become stagnant shallow ponds that support mosquito larvae.

Efforts are underway in Maryland and elsewhere to restore the hydrologic and ecological functions of ditched salt marshes by filling or plugging ditches. The goal of these projects is to restore natural tidal hydrodynamics to the marshes as an initial step in marsh habitat restoration.

In this chapter, data on the changes in marsh hydrology that occurred as a result of a ditch restoration project conducted by Maryland Department of Natural Resources will be presented and evaluated. The goals of the ditch plugging restoration project were to modify marsh hydrology and raise marsh water levels, support permanent and semi-permanent water bodies, and increase populations of aquatic invertebrates, fish and submerged aquatic vegetation (SAV) common to marshes. These aquatic organisms provide food and habitat for wetland birds. Another goal of the ditch plugging restoration project was to enhance flow in the tidal channels and to enhance connectivity between the main tidal channels and restored marshes.

Ditch Plugging as a restoration practice:

Ditch plugging is a recent method used to enhance salt marsh habitat and to provide mosquito control. Successful restoration projects will require careful choice of ditches to plug and post-restoration monitoring to determine whether long-term effects from this management practice are successful. Ditch plugs are currently being used by state wildlife management groups in Maryland to create ponds which can enhance fish and wildlife habitat. Plugging ditches should inhibit drainage, which should create wet areas and pools in the region near the plugged ditches. The interactions among marsh surface elevation, soil characteristics, and hydrologic regimes regulate the marsh self-maintenance processes, but these interactions can be expected to vary by site location and with hydrologic modification.

Ditch plugs used in the restoration projects studied in this project are small dams inserted in the ditch close to the tidal source. Plugs are designed to block the conduit (ditch) water flow from entering and leaving the marsh, and intended to raise ditch water levels and thereby reduce drainage into the ditches from the interior portions of the marsh. Their design is variable because they have to fit the site topography, subgrade soils and required foundation treatments, location and suitability of backfill materials, embankment fill heights and slopes, settlement allowances, and stabilization requirements.

The goal of this research project is to determine whether these hydrological objectives of the restoration project have been attained. This research compares the marsh hydrology before restoration (Chapter 3) with marsh hydrology after restoration. Water levels in different locations in the marsh before and after restoration were examined. The response of marsh water table levels to storms and tides were also reviewed.

Methods and Study Sites

The hydrology of the ditched marshes was described in Chapter 3. This chapter reports the hydrological measurements that were used to characterize post-restoration conditions at the paired (restored-ditched and unditched) study sites at both the Chesapeake and Atlantic marsh sites. At these marshes, the monitoring networks described in chapter 2 were continued. The plugs were installed on 3/25/2014 at the Atlantic sites and on 4/15/2014 at the Chesapeake sites. Monitoring of precipitation, tidal stage, marsh water levels and salinity continued through 7/31/2014. Field methods and the general monitoring network are described in Chapters 2 and 3 and reviewed briefly below.

Hydrological Monitoring

Each marsh monitoring site contained a series of wells laid out as described in Chapter 2 and shown in Figure 1.9. Sites were instrumented with wells in 3 transects arrayed perpendicular to the ditches and parallel to the tidal creek. Wells were installed before restoration laterally from a dominant ditch that was to be plugged by the restoration. Wells at unditched sites also arrayed with 3 transects of wells parallel to the main tidal channel. Each well was installed 2.2 m deep. Wells were equipped with Odyssey Data Loggers that were calibrated in the lab and the water levels were periodically field-checked and recorded in field data every 15 minutes. Nested mini wells were used for manually read salinity measurements using an YSI probe. Locations of ditch plugs are also found on Figure 5.1.

Ditch plugging restoration was conducted in the ditched marshes on both the Atlantic and Chesapeake Bay marshes described in Chapters 2 and 3. The approach in this chapter is to: a) Compare the pre- and post-restoration marsh water levels at each site, and b) compare restored ditched marshes to adjacent unditched marshes during the same time period. These data were used to determine whether marsh restoration through ditch plugging achieved restoration objectives.

The hydrological monitoring network outlined in detail in Chapters 2 and 3 was continued for the post-restoration time period at the paired study marshes on both the Chesapeake and Atlantic marsh sites: Monitoring of precipitation, wind speed, tidal creeks, marsh ground water levels and salinity for the post-restoration analysis extended for the time period of 4/15/2014 to 7/31/2014 for the Chesapeake site, and 3/25/2014 to 7/31/2014 for the Atlantic Site. Salinity was synoptically sampled in tidal creeks and groundwater wells on several dates (6/18/2014 and 7/1/2014) in the post-restoration time period in both restored ditched and unditched marshes.

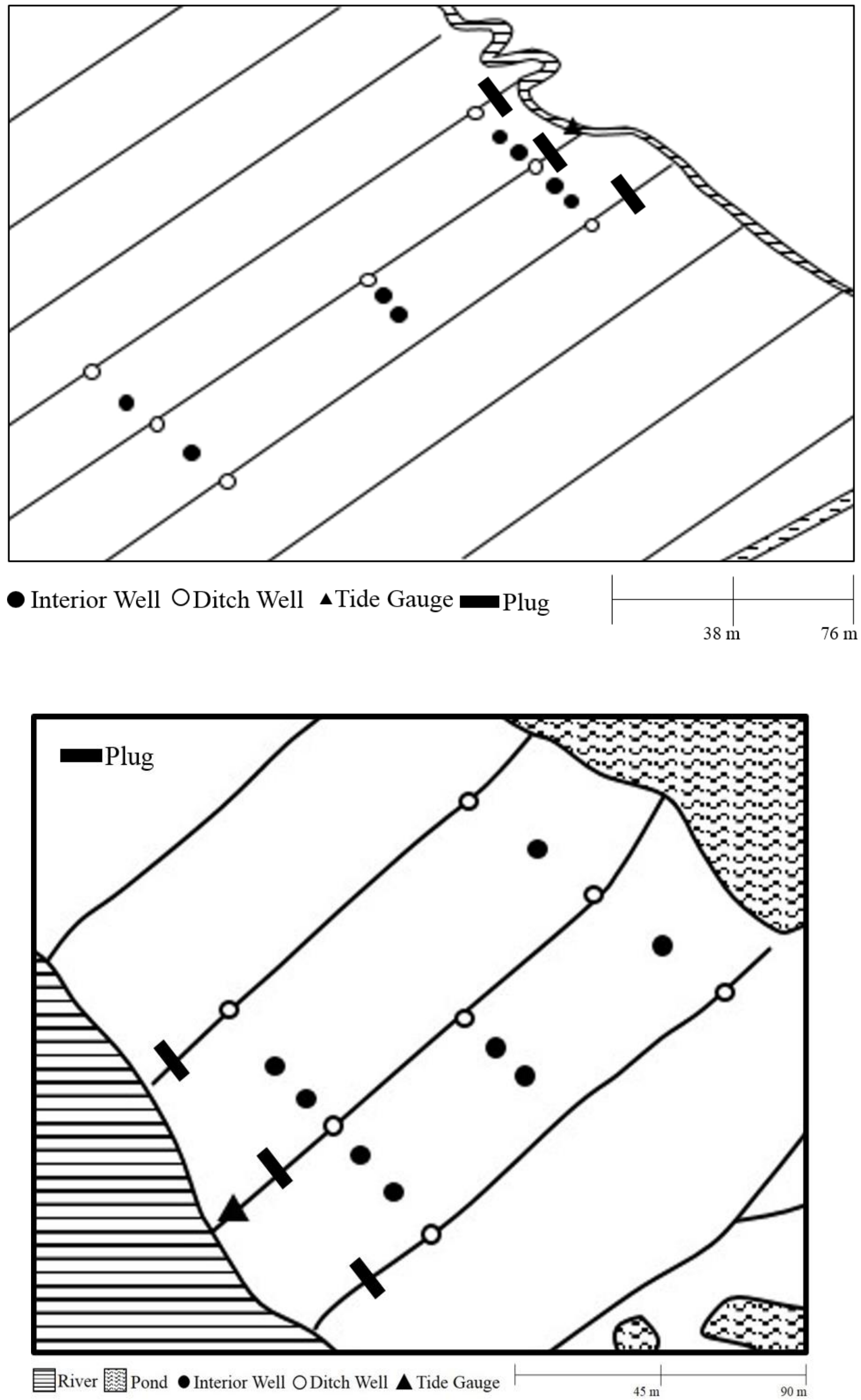


Figure 5.1: (A) Chesapeake Site (B) Atlantic Site with locations of wells and ditches

Figure 5.2: Photo of a ditch plug at the Atlantic site facing the tidal creek

Results:

Chesapeake Bay Marshes

Tidal Range:

The post-restoration tidal range was analyzed from data obtained from tide gage wells located within the restored-ditched and unditched marsh sites (Figure 5.3). Tide gage wells at the ditched site indicate an average tidal range of 9.02 cm for the post-restoration time period. The adjacent unditched marsh tide gage had an average tidal range of 6.78 cm for the same time period. Comparing these values to the few months prior to restoration indicates a significant reduction (25.5 to 9.02 cm) in tidal range from the pre-restoration to the post-restoration time period, however when compared to the overall pre-restoration tidal range from Chapter 2, the difference is not as significant (11.4 pre restoration to 9.02 post restoration) (Table 5.1). This can be explained by natural variations in each time period used and the short duration of the monitored post-restoration time period. The tidal variations in restored Chesapeake ditched marsh sites were compared to the unditched wells. These data indicate that the restored sites have lower tidal ranges than the unditched sites during the post restoration period (9.02 Ditched vs 6.78 Unditched). Chesapeake Unditched shows very little change, while the restored ditched site shows an initial decline during restoration followed by a gradual rise in tidal range. The restored ditched site also shows more events with high ranges in both the pre-and post-restoration time periods.

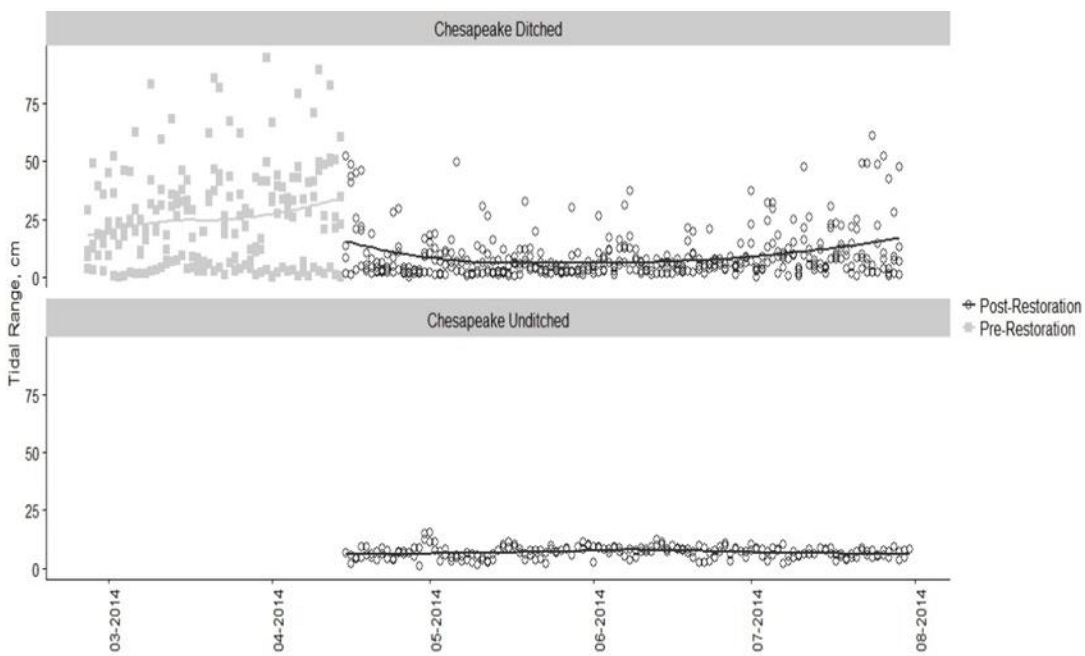


Figure 5.3: Tidal range during Post-restoration period for Chesapeake Unditched site and Chesapeake Ditched site

Table 5.1: Pre (2013, 2014) and –Post (2014) restoration tidal range

	Pre-rest period (2013; 1 year) Tidal range, cm	Immediate pre-rest 2014 tidal range (1.5 months), cm	Post-restoration 2014 tidal range, cm
Unditched	9.8	25.5	6.78 (2.4)
Ditched	11.4	-	9.02 (10.1)

( ) = standard deviation

Salinity characteristics

Synoptic salinity sampling was conducted in the ditched-restored and the unditched marsh sites on (6/18/2014 and 7/1/2014). The Chesapeake unditched sites show higher salinity values (29.4 - 20.2 ppt) than the restored ditched sites, which ranged from 26.9 - 17 ppt). Box plots of salinity are shown in Figure 5.4. Both unditched and restored-ditched sites show significantly higher salinities than during the pre-restoration time period, suggesting secular changes that are not related to the restoration project. Ditched-restored and unditched sites had similar average salinity values (22.3 vs 24 ppt).

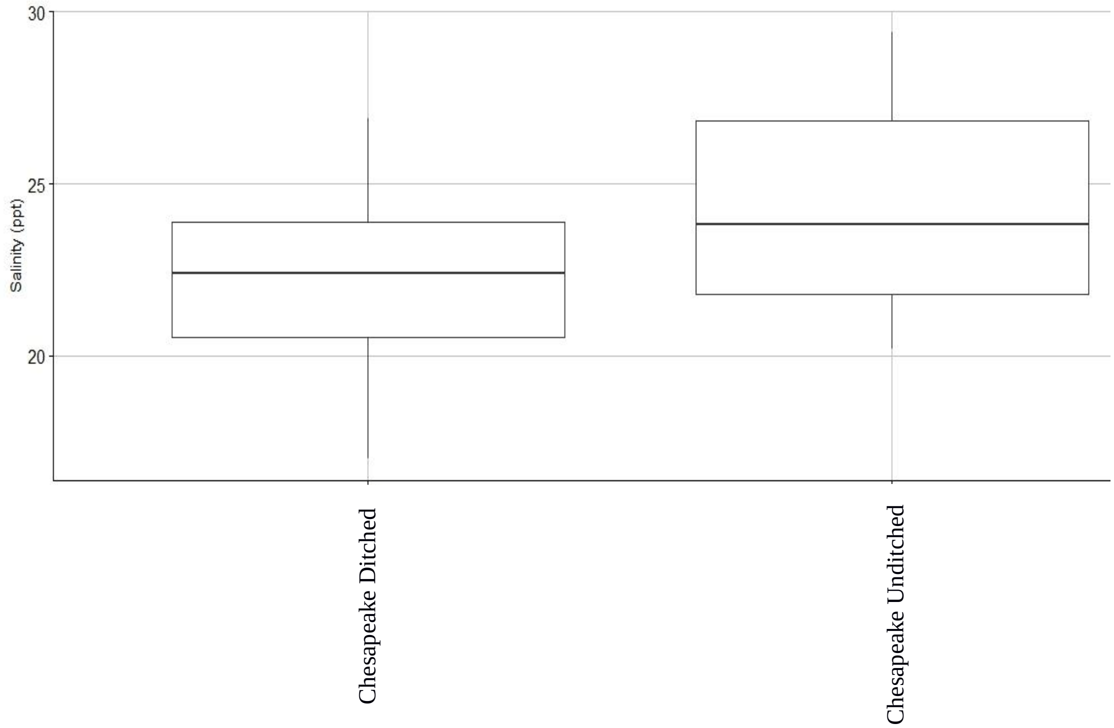


Figure 5.4: Salinities during post-restoration period indicate similar average values for the restored ditched and unditched marshes.

Hydrologic Behavior of restored ditched and unditched marshes

Time series data of marsh water levels

Water levels from marsh groundwater wells were obtained for the post-restoration time period at the restored ditched site and at the unditched sites. The well data during the post-restoration period indicated an increase in the overall water level compared to the pre-restoration period. This increase in water level was not a dramatic/step change but a gradual one (Figure 5.5). The plugs at the restored sites also appeared to have an influence on interior well water levels, which also increased as time passed. The restored-ditch wells and the unditched wells did not have similar characteristics. One main difference is that the restored ditched wells exhibited a reduction in water level minimums. This means the water levels are fluctuating less. This may create marsh water levels and pools with more constant levels in the post–restoration time period. Unditched marsh water levels remained relatively constant. Approximately 5 months after the ditch plugging restoration, the restored ditched wells began to show water table minimums that were lower than the unditched marsh and were below ground. These data are also apparent on boxplots of the wells for each site. Post-restoration data indicated that water levels were higher than pre-restoration data (Figure 5.6).

The cumulative probability distribution curves displayed some differences among the sites and between pre-and post-restoration (Figure 5.7). Cumulative probability curves show a larger range within Ditched Sites both before and after restoration. The months before restoration, most wells were below ground 90% of the time (10% above ground) then after restoration most wells were seen to be below ground on average 75% of time. This increase in percent time above ground supports the time series data in which water levels were shown to increase post restoration.

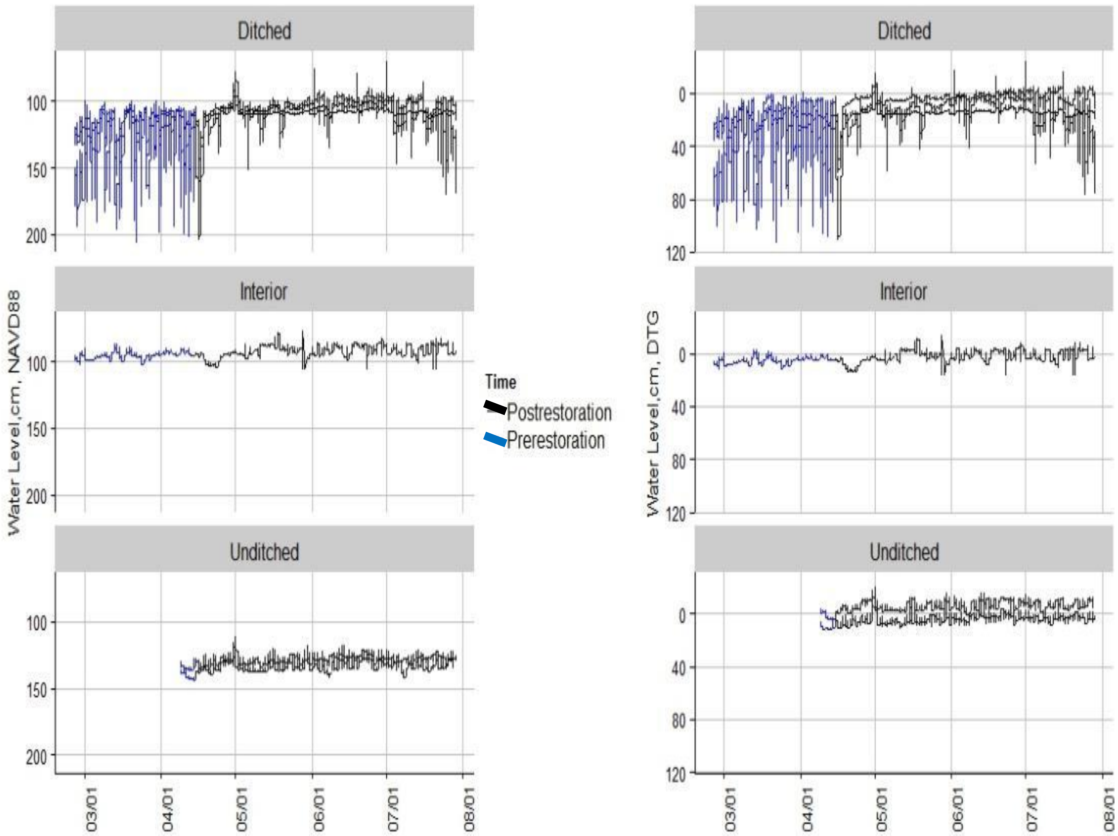


Figure 5.5: Time series of Post-restoration water level data for ditched site and unditched Chesapeake Bay sites

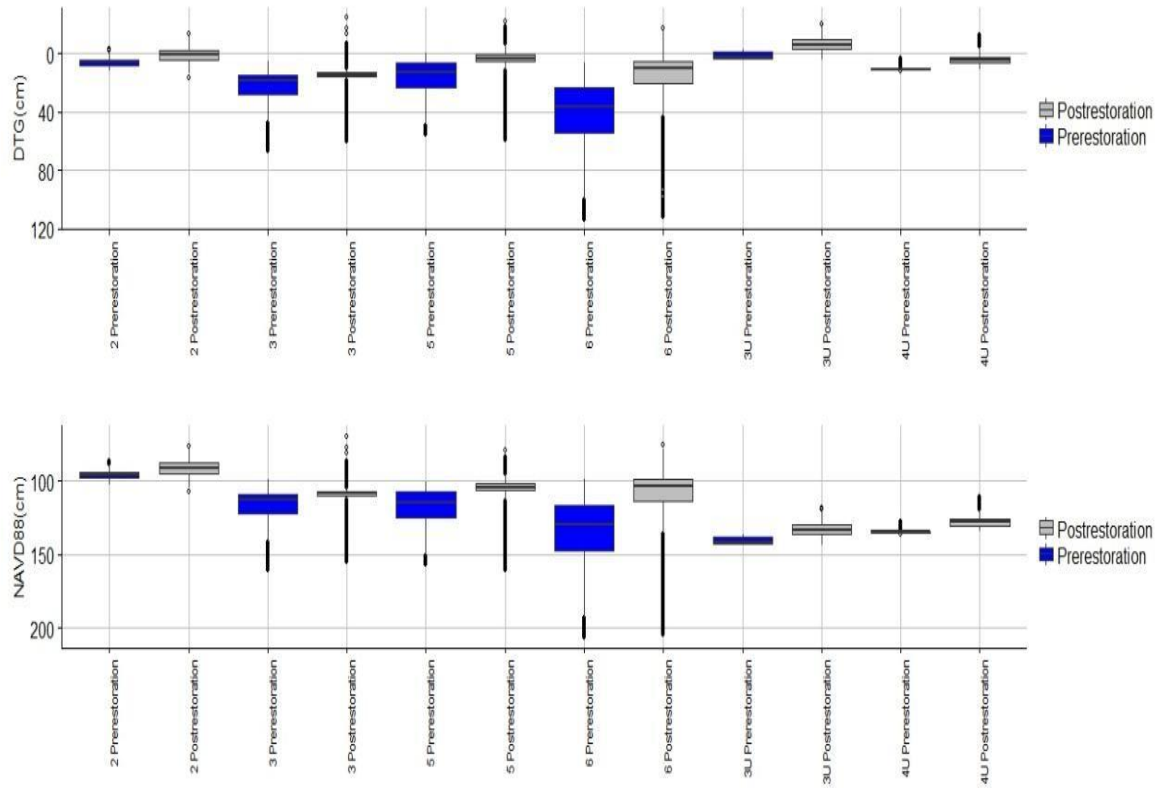


Figure 5.6: Boxplots showing pre- and –post restoration differences between wells and sites in ditched and unditched sites. Unditched sites are wells labeled with a “U” on x axis. Box plot symbolism: center line = median; box boundaries = 1st and 3rd quartile; circles = outliers.

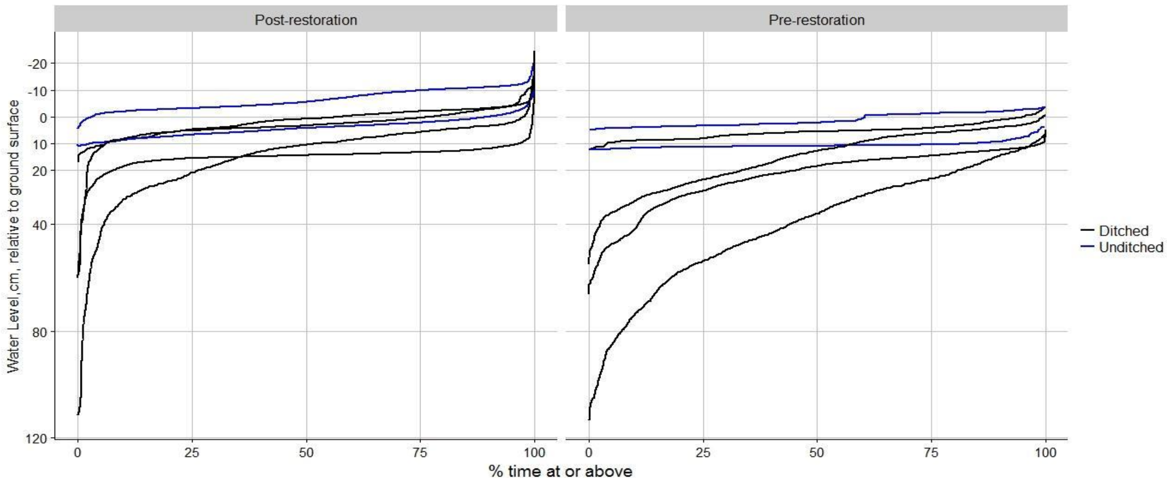


Figure 5.7: Water level cumulative probability graphs showing post-restoration (left) and pre- restoration (right) cumulative probability distributions of ditched and unditched wells.

Atlantic Sites

Tidal Range:

Post-Restoration tidal range was found using the tide gage wells located within the Atlantic sites (Figure 5.8). Ditched site average a tidal range of 2.83 cm for the post restoration project time period and unditched averaged 0.91 cm on the NAVD88 datum. Comparing these values to the few months immediately prior to restoration indicates that there is a not a reduction but instead a slight increase (2.68 to 2.83 cm) in tidal range post restoration. When compared to the overall pre-restoration tidal range from Chapter 2, however, the post restoration period shows that the tidal range was reduced by half (5.68 pre restoration to 2.83 post restoration). One explanation could be natural variations in tidal range and difficulties with having short duration time series data to evaluate consequences of ditch plugging and restoration projects in general. Another possible cause is the ditch restoration caused disconnect in tidal connectivity.

Comparing Atlantic Ditched site to Atlantic Unditched, the unditched wells have much lower tidal ranges during the post restoration period (2.83 Ditched vs 0.91 Unditched). The reduction in ditched post restoration could be reducing the tidal range to be more similar to the unditched sites than the pre-restoration ranges. Atlantic unditched shows very little change, while the ditched site shows a gradual rise post restoration. Ditched site has higher high tidal events post restoration compared to pre-restoration period.

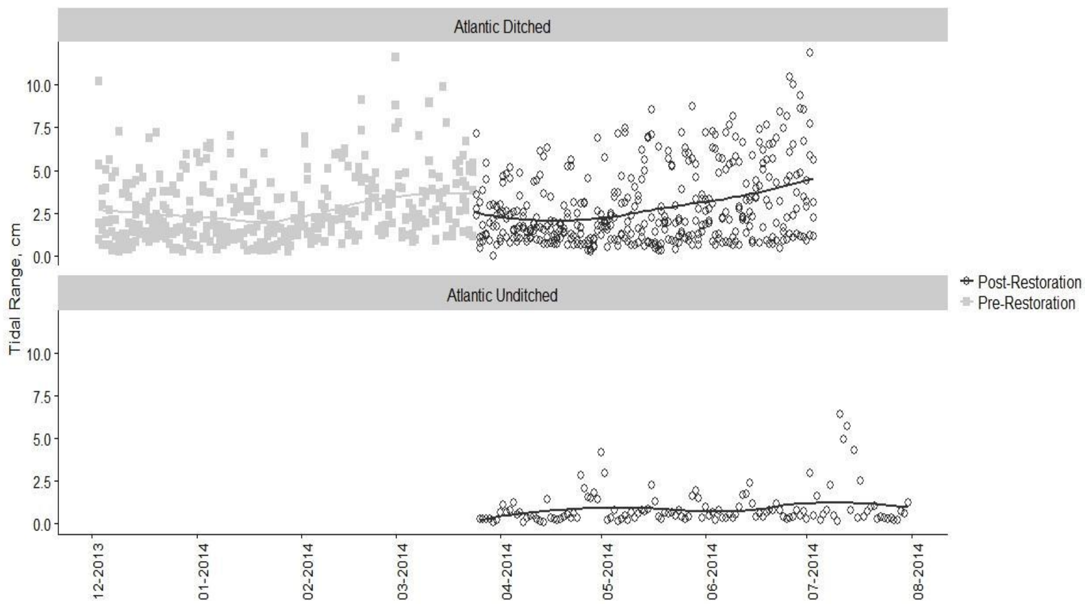


Figure 5.8: Tidal range during Post-restoration period for Atlantic Unditched site and Atlantic restored Ditched site

Table 5.2: Post restoration tidal range Atlantic sites

	Pre-rest period Tidal range, cm	Immediate pre-rest 2014a tidal range, cm	Post-restoration 2014b tidal range, cm
Unditched	4.5	0.91	0.91 (1.0)
Ditched	5.7	-	2.82 (2.2)

**Standard Deviation represents ( )

Salinity characteristics

The Atlantic Ditched sites showed higher average salinities than their unditched counterparts. The unditched sites showed lower salinity ranges (34.6 - 3.26 ppt) while the ditched site was higher around (48.8 - 7.7 ppt). Both sites exhibit higher salinities in the post-restoration time period than were observed during the pre-restoration time period that was reported in chapter 2 (Figure 5.9).

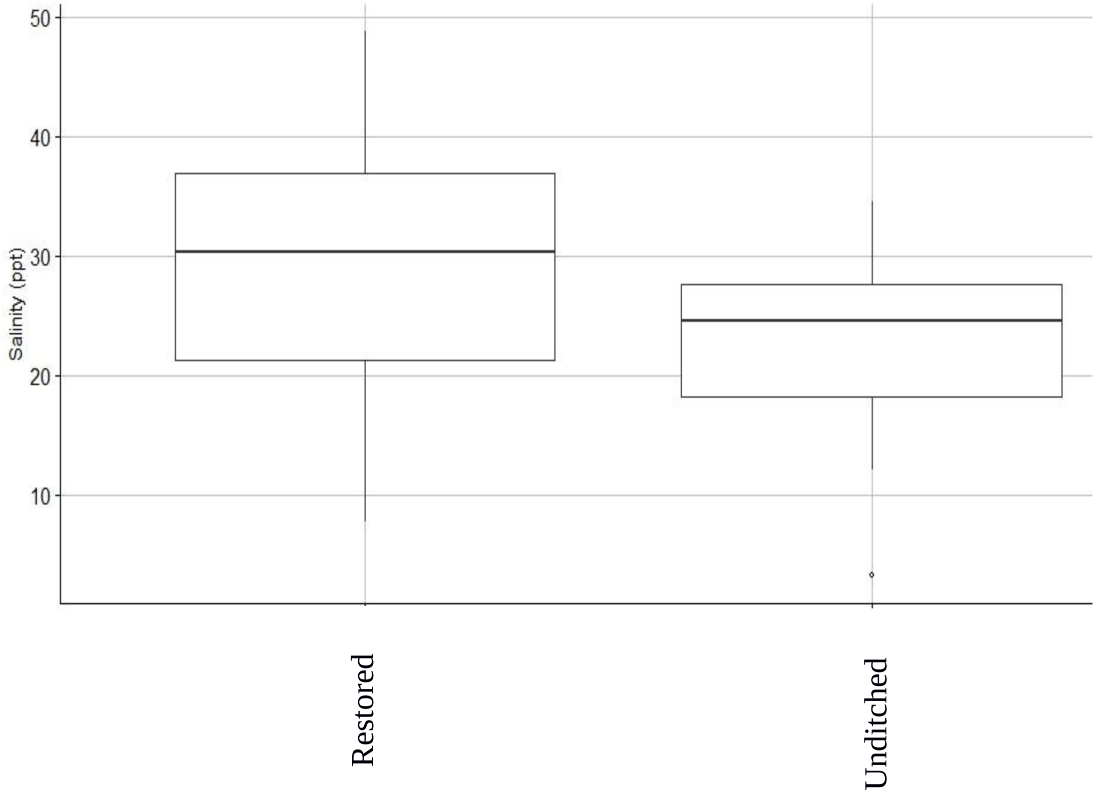


Figure 5.9: Salinity during post-restoration period for the restored ditched site and the unditched reference site. Salinity collected on 6/18/2014 and 7/1/2014 combined for each site.

Hydrologic Behavior of restored ditched and unditched marshes

<u>Time series data of marsh water levels</u>

Water levels in the ditched marsh wells in the post-restoration time period exhibited an increase in the overall water level compared to the pre-restoration period;

developed a new average water level (Figure 5.10). The response of the interior wells located up-marsh from the plug also indicated an increase in water levels as time passed. One main difference was in the ditched wells where there is a reduction in water level lower minimums but the interior wells had approximately the same amount. Unditched site marsh water levels remained fairly constant. Approximately 3 months post restoration the ditched wells did start to show lower minimums. This was confirmed through boxplots of the wells for each site. Post restoration data indicated that water levels were higher than pre-restoration data (Figure 5.11). There is a larger difference closer to the tidal creek, although there is still an increase in the back of sites.

Cumulative curves show a larger range within Ditched Sites both before and after restoration (Figure 5.12). The months before restoration, most wells were above ground 90% of the time (10% below ground) then after restoration most wells were seen to be below ground on average 0% of time. These data are shown to increase post restoration and creating pooling on the surface.

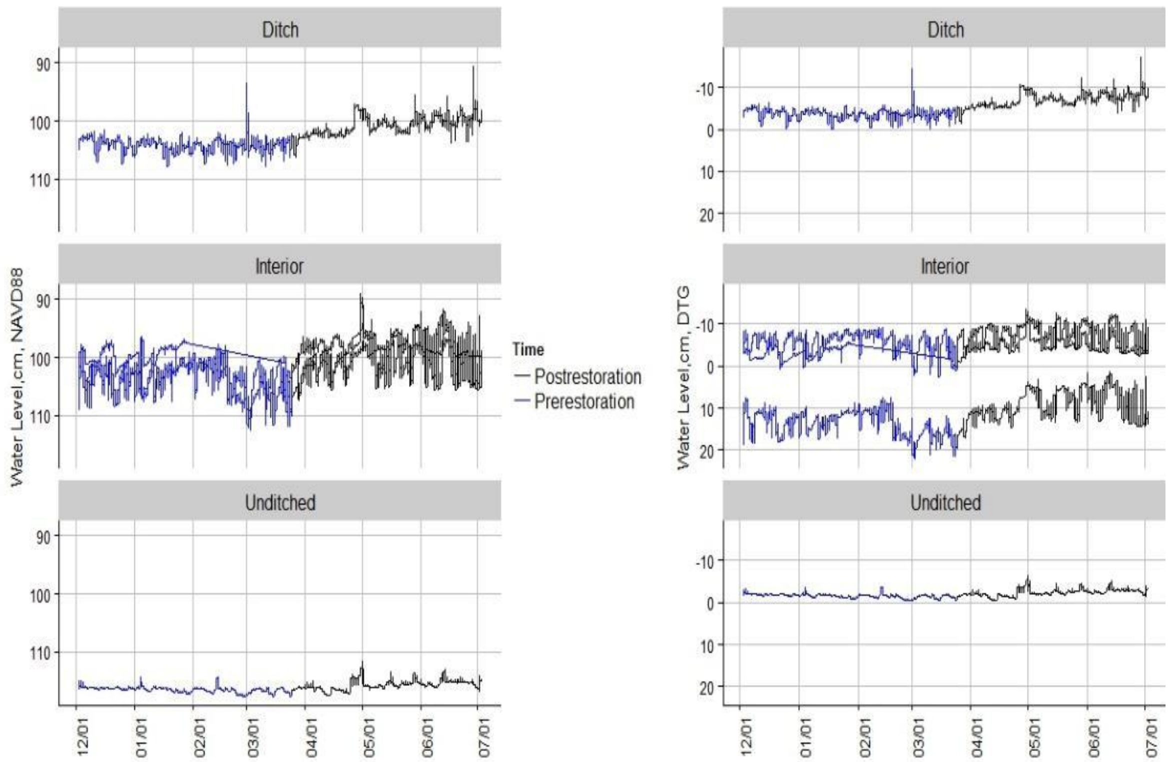


Figure 5.10: Time series of Pre- and –Post restoration water level data for ditched site and unditched Atlantic sites.

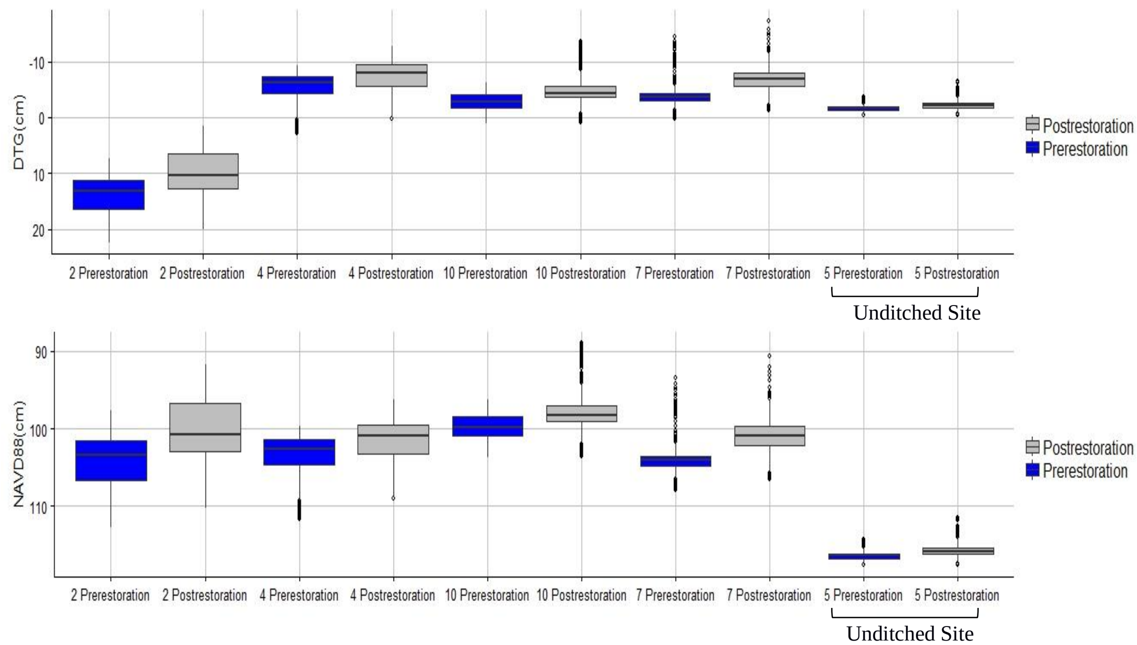


Figure 5.11: Boxplots showing pre- and –post restoration water levels wells and sites in ditched and unditched sites. Unditched sites are wells labeled with a "U" on x axis. Box plot symbolism: center line = median; box boundaries = 1st and 3rd quartile; circles = outliers.

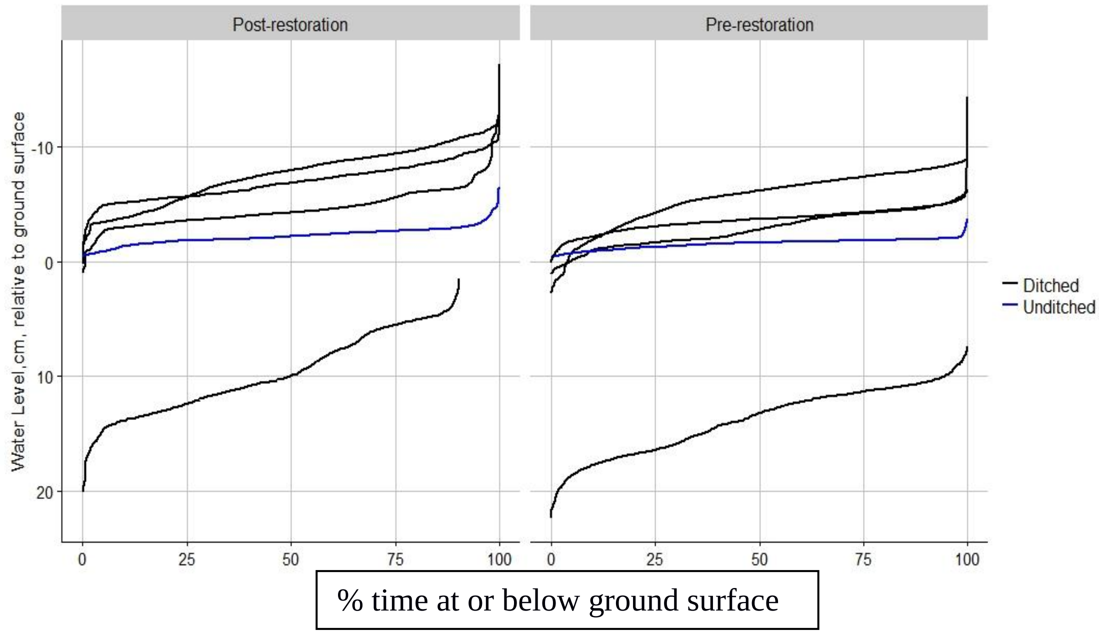


Figure 5.12: Cumulative curves showing pre- and –post restoration differences between wells and sites in ditched and unditched sites.

<u>*Summary:*</u>

Comparison of Atlantic and Chesapeake Marshes

At both the Atlantic and Chesapeake Marshes, tidal range was higher in the ditched marshes compared to their natural, unditched, counterparts. This is despite their geomorphic position (ditched site closer to bay in Atlantic site vs unditched site closer to bay for Chesapeake site), which suggests that ditches amplify the tidal range, a result also discussed in Chapter 3. All sites showed a post-restoration decrease in tidal range in the sites when using the longer pre-restoration time period (1 year) and not the immediately before pre-restoration period (1.5 months).

The salinity differences at the two locations showed different patterns. Salinity was higher in the ditched sites at the Atlantic marshes, but higher in the unditched sites in the Chesapeake Bay marshes. These differences could be due to the geomorphic position of the sites relative to the contributing tidal bays. At the Atlantic sites, the ditched site is closer to the bay, and the unditched site at the Chesapeake location is closer to the bay.

During the post-restoration period, both the Chesapeake Bay and the Atlantic Coast marshes showed higher water level minima immediately after the ditch plugs were installed. This effect was less prominent in the Atlantic coast ditch wells, which appeared to regain drainage conditions, although water levels remained the same in the interior wells. The Chesapeake site interior wells showed an overall increase in water level after ditch plugging and water levels were more frequently at the ground surface. All the wells in post restoration period showed higher water levels than the pre-restoration periods at both locations. At the restored sites, Chesapeake marshes were at or above the surface more often and for longer durations compared to the Atlantic marshes (Table 5.3).

Table 5.3: Site and parameter	Chesapeake 2013	Chesapeake 2014 (PR)	Atlantic 2013	Atlantic 2014 (PR)
Unditched Tidal range (cm)	9.8	6.8	4.5	0.91
Ditched Tidal Range (cm)	11.4	9.0	5.7	2.82
Salinity Unditched (ppt)	15.5	24.2	15.2	23.4
Salinity Ditched (ppt)	12.8	22.3	20.9	29.0
Marsh GW probability above ground unditched	50%	50%	50%	50%
Marsh GW probability above ground Ditched	10%	25%	90%	100%

(PR) = Post Restoration

Discussion:

The majority of previous studies of the effects of ditches indicate that they substantially changed salt marsh hydrological and ecological environments (Daiber 1986; Roman et al. 2000). An assessment of human impacts to salt marshes on a global scale found that ditching was damaging to salt marshes, but less so than many other human alterations of marsh environments (Gedan et al. 2009). Marsh ditches are responsible for impacting estuarine hydroperiods as shown here and in previous studies. As such, ditching may reduce the processes by which natural unaltered salt marshes handle nutrients and organic matter, vegetation composition, and formation of the marsh platform.

This study showed that hydrologic regimes in ditched and restored ditched marshes differed from unditched marshes. Higher tidal ranges were observed in the ditched marshes, but these extended tidal ranges are associated with lower tidal minima not higher tidal maxima. Increases in tidal range have been associated with salt marsh building processes and the alleviation of stressful soil conditions (Chmura et al. 2001) and are directly related to maintenance of marsh surface elevation (Vincent 2013). Morris et al. (2002) determined that flooding had positive effects on primary production and marsh accretion, but flooding past optimal levels would negatively impact plants and marsh accretion processes. Marsh accretion ceases when peat building processes (i.e. deposition and organic production) are interrupted such as when flow into the marsh is restricted (Morris et al. 2002).

The results of this project indicate that marsh water levels were higher (Chesapeake) and lower (Atlantic) in ditched marshes than their natural unditched counterparts when comparing the 2-year time series. These results are similar to previous studies that suggested ditches decreased soil pore-water levels (Bourn and Cottam 1950; Lesser 1982; Adamowicz and Roman 2005), and supported results of a recent study in which ditching did not result in lower water levels relative to the surface (Vincent 2013). This may not have always been the case during the last 80 years; the marsh soils likely became better drained when ditches were originally dug (Anisfeld 2012). Over time with decomposition and subsidence, elevations could have decreased, and waterlogging could have increased, explaining why the ditched sites show elevated water levels compared to their natural counterparts. Ditched marshes indicated a reestablishment of a new equilibrium following the initial disturbance.

Ditch plugging was shown to increase water levels post restoration, which was one of the goals of the project. There were some indications that drainage conditions were re-established subsequent to plugging at some of the sites.

Conclusion:

1. Chapter 2 presented data that indicate higher tidal amplitudes in ditches and ditched marsh interiors than in unditched marshes. The soil bulk density and hydraulic conductivity data were stratified by depth, with higher K values and lower bulk density values in the upper, peat rich layers and lower K values and higher bulk density values at depth. There were no statistically significant differences in either K or bulk density between ditched and unditched marshes, which suggests that ditching does not cause significant compaction in these microtidal settings.
2. In Chapter 3, marsh water level data in ditched and unditched marshes were compared and their relationship to plant species composition was evaluated. It was determined that water levels were at or above the ground surface a greater percentage of the time in the unditched marshes. Evaluation of the hydraulic head data indicated that ditched marshes had consistent horizontal gradients towards the tidal creeks. Although variations in average species composition appeared to be different in ditched and unditched marshes, the greater heterogeneity among plots in the unditched marshes caused significant overlap in their distributions. Thus, plant species coverage was not statistically different in the ditched and unditched marshes
3. In Chapter 4, the response of water levels to the Hurricane Sandy storm surge was evaluated. These data indicated that the marshes attenuated the Hurricane Sandy storm surge and stored water within the marsh and tidal creeks during

the storm surge. These effects were observed in the reduction in height of peak storm surge stage and the hysteresis in marsh water levels compared to the NOAA tide gages at both sites. The duration of high water levels was significantly longer at the Atlantic sites. The effects of ditches on marsh attenuation could not be directly assessed because larger geomorphic features (main tidal creeks) were more important in directing the storm surge than the ditches. Differences in storm surge maximum between ditched and unditched portions of marshes were not significant.

4. In Chapter 5, the hydrological consequences of ditch plugging restoration projects were examined. These data indicate that the marsh restoration project modified the marsh hydrology and salinity. Ditched marshes were still hydrologically different than the unditched marshes, but restoration made the sites more similar to the natural unaltered counterparts. The marsh water level exceedance curves indicated a post restoration increase in the percent of time that marsh water levels were at or above ground (by 15%). These data indicate that post-restoration water levels are closer to the surface for longer periods of time, and are more similar to, but still distinct from, the unditched sites.

Implications:

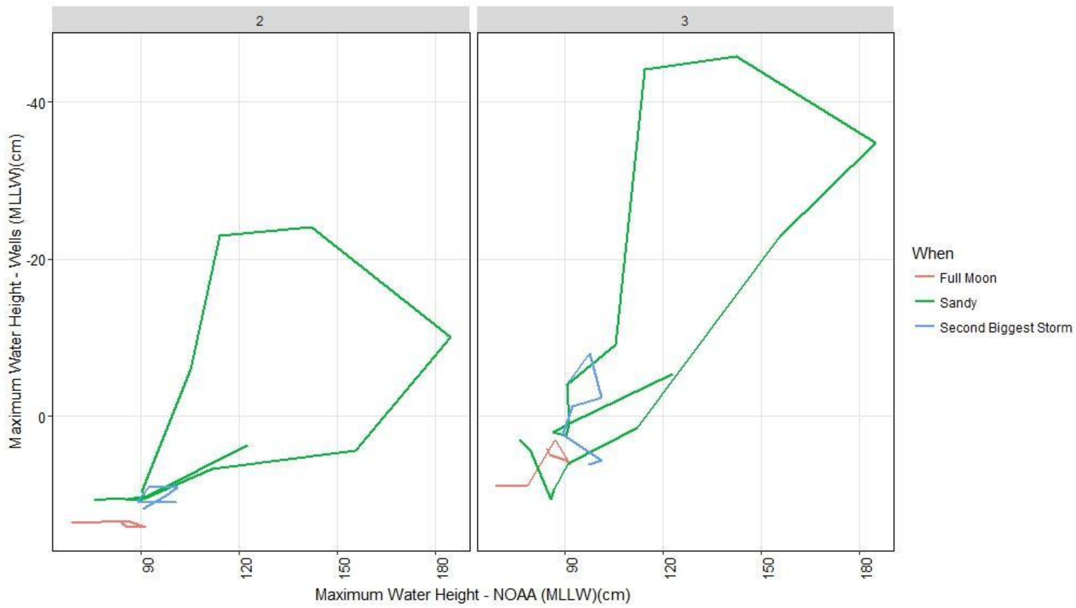
Aside from human disturbances, the most significant threat to coastal marshes is sea-level rise and the increase in frequency and magnitude of storms. Small ditches oriented perpendicular to tidal creeks do not appear to affect the attenuation of storm surges by friction generated by flow over marshes. It is also possible that these ditches were able to store more storm surge. This may have increased the storage and perhaps the duration of high water levels in ditched marshes in comparison with unditched marshes. Restoration projects with the goal of improving storm surge attenuation may not be appropriate for these microtidal, inland, ditched marshes. On the other hand, there were significant differences in the position of the water table in ditched and unditched marshes. Ditch plugging restoration projects raised water levels in marsh interiors, but did not fully recreate the water levels found in unditched marshes. This suggests that ditch restoration may be an effective procedure in these marshes if a rise in marsh water levels would improve marsh habitat or other ecological functions.

Ditching can cause a disconnect in tidal flow and associated sediment transport between the tidal source and the interior portions of the marsh. It creates many challenges for local land management and adaptation strategies. This project showed that restoration increased water levels with more above ground events and raised water levels closer to the surface for extended periods of time. One of the goals of the project was to recreate pools on the surface for wildlife, and restoration was shown to be successful in doing so within the 4-5 months that post restoration data were available. Restoring tidal flow similar to unditched marshes could ensure that the marshes are utilized to its full potential hydrologically and ecologically. Reducing the ditches and ditch impacts could increase connectivity and the permeability of the estuary to enhance different forms of movement and interactions instead of conduit flow in and out of the ditches.

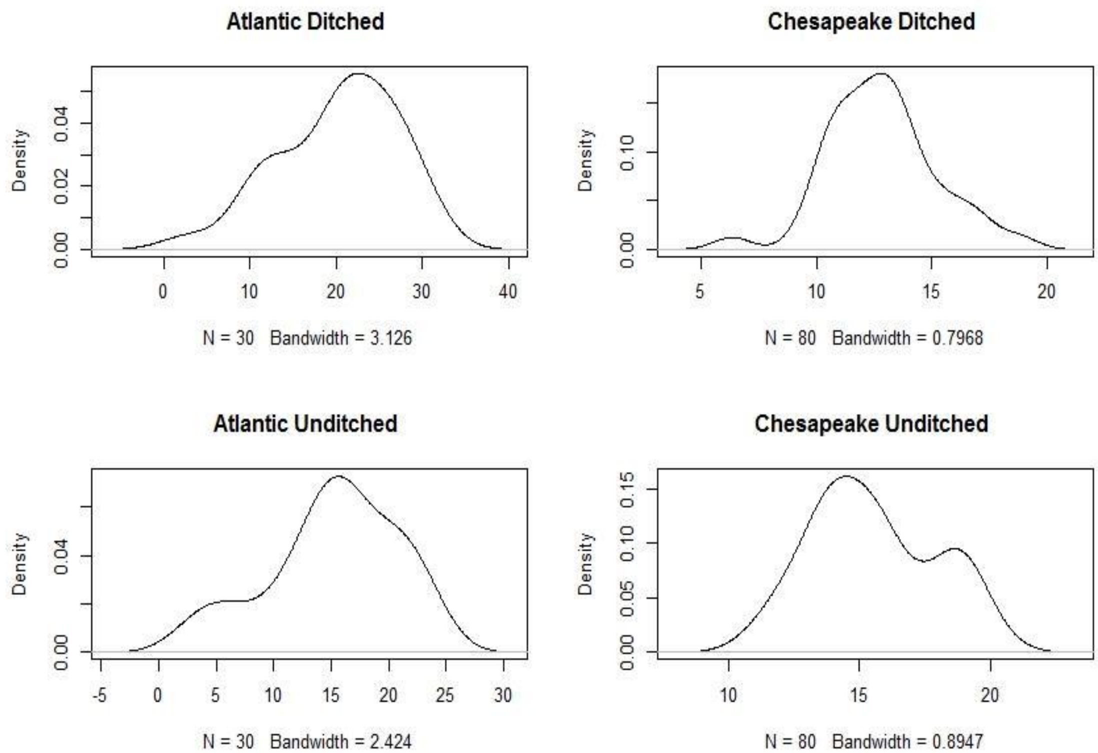
After completing the research for this dissertation, I recommend long-term planning to address redesigning protected marshes to plan for vegetation and habitat shifts, transitional habitats upland of the marshes, and restoring more areas to tidal flows to support wildlife in the short-term. This would also allow the marshes the opportunity to start accreting again, as sediments and nutrients will no longer be trapped in the ditches and unable to reach interior marsh between ditches. Resource managers and scientists should not rush to judgment concerning the success of restoration shortly after restoration. As a result of restoration, the marsh water levels increased and there was visible ponding on the surface almost immediately, however vegetation and biota shifts may go through a longer transition leading to reestablishment of vegetation composition more suited for the increase in tidal flow and water levels.

Full recovery relative to natural unditched marshes, if it is ever achieved, may require more than 1-2 decades similar to the restoration of diked marshes. Full structural and functional equivalence similar to unditched marshes may not be a realistic goal, but recovery of some marsh attributes and functions should be regarded as a successful outcome of restoration efforts.

Appendices

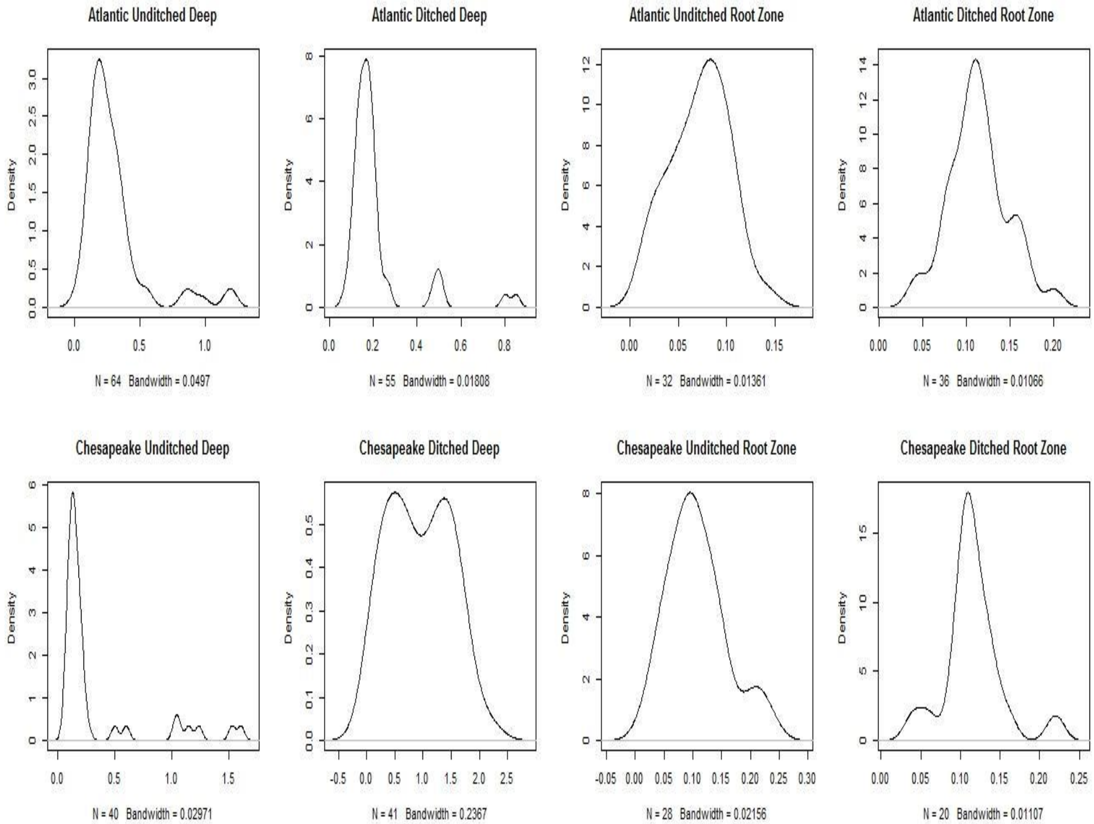


Hysteresis showing Hurricane Sandy, full moon, and the second biggest storm that occurred during the pre-restoration period.



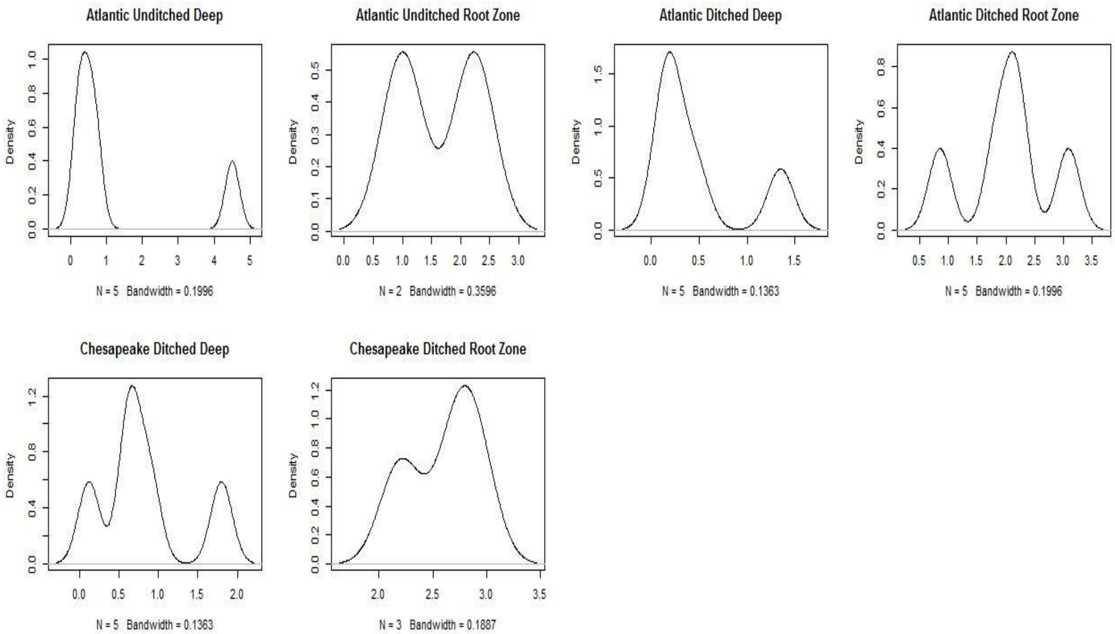
Statistics Figure 1: Salinity

- Atlantic Ditched and Unditched sites; p-value = 0.0014 Statistically Different
- Chesapeake Ditched and Unditched sites; p-value = 1 Statistically Similar
- Atlantic Ditched vs Chesapeake Ditched; p-value = 5.034e-14 Statistically Different
- Atlantic Unditched vs Chesapeake Unditched; p-value = 0.658 Statistically Similar



Statistics Figure 2: Bulk Density

- Chesapeake Ditched site comparing deep soil cores vs root zone cores; p-value = 6.3 e-09 Statistically Different
- Chesapeake Unditched site comparing deep soil cores vs root zone cores; p-value = 1.6 e-07 Statistically Different
- Atlantic Unditched comparing deep and root zone cores; p-value = 3.347e-07 Statistically Different
- Atlantic Ditched comparing deep and root zone cores; p-value = 0.00014 Statistically Different
- Atlantic Unditched Deep vs Chesapeake Unditched Deep p-value = 0.6924 Statistically Similar
- Atlantic Ditched Deep vs Chesapeake Ditched Deep p-value = 1 Statistically Similar
- Atlantic Unditched Root Zone vs Chesapeake Unditched Root Zone p-value = 0.999 Statistically Similar
- Atlantic Ditched Root Zone vs Chesapeake Ditched Root Zone p-value = 0.5997 Statistically Similar



Statistics Figure 3: Hydraulic Conductivity

- Chesapeake Ditched site comparing deep soil cores vs root zone cores; p-value = 0.998 Statistically Similar
- Atlantic Ditched comparing deep and root zone cores; p-value = 0.9967 Statistically Similar
- Atlantic Unditched comparing deep and root zone cores; p-value = 0.5961 Statistically Similar
- Atlantic Ditched Root Zone vs Chesapeake Ditched Root Zone; p-value = 0.8529 Statistically Similar
- Atlantic Ditched Deep vs Chesapeake Ditched Deep; p-value = 0.8216 Statistically Similar

Well Construction:

Each well was equipped with a water level data logger and measured the depth to water below the probe’s O-ring seat at a 15-minute intervals. Wells were constructed out of 3.05 meter PVC 40 pipe. The bottom 2.15 meter of the pipe was machined screen slotted 0.010 with 0.50 cm spacing between each slot. A PVC point was fixed to the bottom of the well and a cotton sock covered the screened portion of the well to prevent sediments from entering. Wells were installed using hand augers 2.2 m deep along 3 transects. Mini wells were also installed to determined vertical stratification of the hydraulic head. Mini wells were also constructed with a bottom point and socked PVC 1.5 meter 40 pipe screen machined slotted 0.010 with 0.50 cm spacing between each slot. Mini wells were slotted at depths of 5-20, 20-50, and 50-80 cm. The mini wells were used to hand measure snapshot data events.

Well Calibration:

Odyssey by Dataflow Systems Pty Ltd capacitance water level data loggers were installed at an average depth of 2.2 meters in closed wells with ventilation and constructed to remain above ground 1 meter. Prior to deployment, each data logger was calibrated in the lab using the bucket method. Loggers received two marked points on the sensor cable, one at 200mm up from the top of the sensor and then another mark 2 meters down for a 2 meter logger and 3 meters for the 3 meter loggers. Caution was taken to not allow the sensing cord to make contact with the bucket sides. The logger was then lowered to the two marks and calibrated with the Odyssey computer program. The point of zero is at the top of the logger where the cap meets the logger body. The manufacturer specifies that properly maintained probes have a resolution of 0.8 mm and can achieve ±5 mm accuracy (Dataflow Systems, 2008).

References

Allan, Philip F. 1950. Ecological bases for land use planning in Gulf Coast marshlands. Journal of Soil and Water Conservation. 5: 57-62, 85.

Adamowicz, S. C. and C. T. Roman, 2005. New England salt marsh pools: a quantitative analysis of geomorphic and geographic features. Wetlands 25:279 – 288

Allen, J. R. L., and K. Pye 1994. Saltmarshes: Morphodynamics, Conservation, and Engineering Significance, 196 pp., Cambridge Univ. Press, U. K.

Anisfeld, S. C., and G. Benoit. 1997. Impacts of flow restrictions on salt marshes: an instance of acidification. Environmental Science and Technology 31: 1650–1657.

Anisfeld, S. C., M. J. Tobin, and G. Benoit. 1999. Sedimentation rates in flow-restricted and restored salt marshes in Long Island Sound. Estuaries 22:231–244.

Bertness M.D., Ellison A.M. 1987. Determinants of pattern in a New England salt marsh community. Ecological Monograph 57:129-147

Bertness, M. D. 1991. Zonation of Spartina patens and Spartina alterniflora in a New England Salt Marsh. Ecology 72:138–148.

Blake, E., Kimberlain, T.,Berg, J.,Cangialosi, J., Beven, J., 2012. Tropical Cyclone Report Hurricane Sandy

Blake GR, Hartge KH. Bulk density. In: Klute A, editor. Methods of soil analysis. Part 1, physical and mineralogical methods. 2nd ed., Agronomy Monograph No. 9. Madison, WI: Soil Science Society of America; 1986. p. 363–375.
Bourn, W. S. and Cottam, C., 1950. Some biological effects of ditching tidewater marshes. U. S. Fish and Wildlife Service, Res. Report 19. 17 p.

Brewer JS, Levine JM, Bertness MD (1997) Effects of biomass removal and elevation on species richness in a New England salt marsh. Oikos 80:333-341

Buchsbaum R.N., Catena J, Hutchins E., James-Pirri M.J. 2006. Changes in salt marsh vegetation, Phragmites australis, and nekton in response to increased tidal flushing in a New England salt marsh. Wetlands 26:544–557

Burdick, D. M., M. Dionne, R. M. Boumans, and F. T. Short. 1997. Ecological responses to tidal restorations of two northern New England salt marshes. Wetlands Ecology and Management 4:129–144.

Cahoon DR. 2006. A review of major storm impacts on coastal wetland elevations. Estuaries Coasts 29:889–98

Chmura G.L., Hung G.A. 2004. Controls on salt marsh accretion: a test in salt marshes of Eastern Canada. Estuaries 27:70–81

Daily G.C, Alexander S, Ehrlich P.R, et al. 1997. Ecosystem services: benefits supplied to human societies by natural ecosystems. Issues Ecol 2:1–16.

Day , J. W., R. R. Christian, D. M. Boesch, A. Yáñez, Arancibia, J. Morris, R. R. Twilley, L. Naylor, L. Schaffner & C. Stevenson, 2008. Consequences of climate change on the ecogeomorphology of coastal wetlands. Estuaries and Coasts 31: 477-491

Daiber, F C. 1986. Conservation of tidal marshes. VanNostrand Reinhold Co., New York.

de Groot R.S, Wilson M.A, and Boumans R.J.M. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol Econ 41:393–408.

Dionne, M., F. T. Short, and D. M. Burdick. 1999. Fish utilization of restored, created, and reference salt-marsh habitat in the Gulf of Maine. Pages 384–404 in L. Benaka, editor. Fish habitat: essential fish habitat and rehabilitation. American Fisheries Society, Symposium 22, Bethesda, Maryland.

Duncan, Wilbur H.; Duncan, Marion B. 1987. The Smithsonian guide to seaside plants of the Gulf and Atlantic Coasts from Louisiana to Massachusetts, exclusive of lower peninsular Florida. Washington, DC: Smithsonian Institution Press. 409 p.
FEMA. 2013. Hurricane Sandy FEMA After-Action Report

Ferris, J.G., and Knowles, D.B., 1963, The slug-injection test for estimating the coefficient of transmissibility of an aquifer, in Bentall, Ray, comp., Methods of determining permeability, transmissibility, and drawdown: U.S. Geological Survey Water-Supply Paper 1536–I, p. 99–304.

Fetter, C. W. (2001). Applied Hydrogeology (4th ed.). Prentice Hall. Uper Saddle River, New Jersey.

Foti R, del Jesus M, Rinaldo A, Rodriguez-Iturbe A (2012) Hydroperiod regime controls the organization of plant species in wetlands. Proc Natl Acad Sci U_S_A 109: 19596-19600

Gedan K.B., Silliman B.R., Bertness M.D., 2009, Centuries of human-driven change in salt marsh ecosystems: Annual Review of Materials Science. 1:117–141

Gillham, M. E. 1957. Coastal vegetation of Mull and lona in relation to salinity and soil reaction. J. Ecol. 45: 757-778.

Kearney MS, Grace RE, Stevenson JC (1988) Marsh loss in the Nanticoke Estuary, Chesapeake Bay. Geogr Rev 78: 205–220

Kennish, M.J. 2002. Environmental threats and environmental future of estuaries. Environmental Conservation 29, 78–107.

Konisky R.A., Burdick D.M., Dionne M, Neckles H.A. 2006. A regional assessment of salt marsh restoration and monitoring in the Gulf of Maine. Restoration Ecol 14:516–525

Krauss, K.W., Doyle, T.W., Doyle, T.J. et al. Wetlands (2009) 29: 142. doi:10.1672/07-232.1

Lathrop, R.G.; Cole, M.B., and Showalter, R.D., 2000. Quantifying thehabitat structure and spatial pattern of New Jersey (U.S.A.) salt marshes under different management regimes. Wetlands Ecologyand Management, 8(2/3), 163–172.

Loder, N. M., Irish, J. L., Cialone, M. A. & Wamsley, T. V. Sensitivity of hurricane surge to morphological parameters of coastal wetlands. Estuar. Coast. Shelf Sci. 84, 625–636 (2009).

Lovelace, J. K., & McPherson, B. F. (1998). Restoration, creation, and recovery: Effects of Hurricane Andrew (1992) on Wetlands in Southern Florida and Louisiana. National water summary on wetland resources. U. S. Geological Survey Water Supply Paper, 2425

McKenzie NJ, Jacquier DJ, Isbell RF, Brown KL (2004) Australian Soils and Landscapes An Illustrated Compendium. CSIRO Publishing: Collingwood, Victoria.

Michener, W. K., Blood, E.R.., Bildstein, K.L., Brinson, M.M., and Gardner, L.R.. 1997 Climate change, hurricanes and tropical storms, and rising sea level in coastal wetlands. Eco-logical Applications 7:770-801.

Miller, W.R. and Egler, F.E., 1950. Vegetation of the Wequetequock-Pawcatuck tidal marshes, Connecticut. Ecological Monographs, 20(2), 145–170

Mitsch,W.J., Gosselink, J.G., 2000.Wetlands.VanNostrand Reinhold, New York.

Morris, J.T., Sundareshwar, P.V., Nietch, C.T., Kjerfve, B., Cahoon, D.R., 2002. Responses of coastal wetlands to rising sea level. Ecology 83: 2869-2877.

Morton, R. A., and J. A. Barras. 2011. Hurricane impacts on coastal wetlands: A half - century record of storm - generated features from southern Louisiana. Journal of Coastal Research 27:27 - 43.

Niering, W. A., and R. S. Warren. 1980. Vegetation patterns and processes in New England salt marshes. BioScience 30:301-307.

Park RA, Lee J.K, Mausel P.W, and Howe R.C. 1991. Using remote sensing for modeling the impacts of sea level rise. World Resour Rev 3: 184–220.

Peck, M. A., P. E. Fell, E. A. Allen, J. A. Gieg, C. R. Guthke, and M. D. Newkirk. 1994. Evaluation of tidal marsh restoration: comparison of selected macroinvertebrate populations on a restored impounded valley marsh and an unimpounded valley marsh within the same salt marsh system in Connecticut, USA. Environmental Management 18:283–293.

Peet, R. K., T. R. Wentworth, and P. S. White. 1998. A flexible, multipurpose method for recording vegetation composition and structure. Castanea 63:262–274.

Pfeffer, W.T., J.T. Harper, and S. O'Neel. 2008. Kinematic constraints on glacier contributions to 21st-century sea-level rise. Science 321:1340-1343.

Pincus, S., 1938. Mosquito control in greater New York. In: Proceedings of the 25th Annual Meeting of the New Jersey MosquitoControl Association

Portnoy, J. W. 1991. Summer oxygen depletion in a diked New England estuary. Estuaries 14:122–129.

Portnoy, J. W. 1999. Salt marsh diking and restoration: biogeochemical implications of altered wetland hydrology. Environmental Management 24:111–120.

Portnoy, J.W. Anda . E. Giblin. 1997. Effects of historic tidal restrictions on salt marsh sediment chemistry. Biogeochemistry 36:275-303.

Raposa K.B. 2008. Early ecological responses to hydrologic restoration of a tidal pond and salt marsh complex in Narragansett Bay, Rhode Island. J Coast Res 55:180–192

Raposa KB, Roman C.T. 2003. Using gradients in tidal restriction to evaluate nekton community responses to salt marsh restoration. Estuaries 26:98–105

Raposa, K. B. 2002. Early responses of nekton to restoration of a tidally restricted New England salt marsh. Restoration Ecology. 10:665-676.

Raposa, K.B., and C.T. Roman. 2001 Seasonal habitat-use patterns of nekton in a tidal-restricted and unrestricted New England salt marsh. Wetlands 21: 451-461

Reed, D.J. & Cahoon, D.R. 1992. The relationship between marsh surface topography, hydroperiod, and growth of Spartina alterniflora in a deteriorating Louisiana salt marsh. J. Coastal Res. 8: 77-87.

Redfield, A.C., Rubin, M., 1962. The age of salt marsh peat and its relation to recent changes in sea level at Barnstable, Massachusetts. Proceedings of the National Academy of Sciences USA 48: 1728-1735.

Rochlin, I.; James-Pirri, M.-J.; Adamowicz, S.C.; Wolfe, R.J.; Capotosto, P.; Dempsey, M.E.; Iwanejko, T.; Ninivaggi, D.V., 2012. Integrated Marsh Management (IMM): A new perspective on mosquito control and best management practices for salt marsh restoration. Wetland. Ecol. Manag. 20: 219–232.

Roman, C.T., R. W. Garvine, and J. W. Portnoy. 1995. Hydrologic modeling as a predictive basis for ecological restoration of salt marshes. Environmental Management 19: 559–566.

Roman, C.T., W. A. Niering, and R. S. Warren. 1984. Salt marsh vegetation change in response to tidal restriction. Environmental Management 8: 141–150.

Rotzoll, K. and Fletcher, C.H., 2013. Assessment of groundwater inundation as a consequence of sea-level rise. Nature Climate Change, 3(5), 477–481.

Shisler, J. K. 1990. Creation and restoration of coastal wetlands of the northeastern United States. p. 143–170. In J. A. Kusler and M. E. Kentula (eds.) Wetland Creation and Restoration: The Status of the Science. Island Press, Washington, DC, USA.

Sinicrope, T. L., P. G. Hine, R. S. Warren, and W. A. Niering. 1990. Restoration of an impounded salt marsh in New England. Estuaries 13:25–30.

Singh, P.B. and Nathan, K., 1965. Hydraulic studies of drainageditches under tidal influence. Transactions of the American Societyof Agricultural and Biological Engineers, 8, 460–463, 469.

Smith, J.B. 1904. Report of the NJ State Agricultural Experiment Station upon the mosquitoes occuring within the state, their habits, life history, etc. MacCrellish & Quigley, Trenton, N.J.

Stearns, L. A., D. MacCreary, and Daigh, E.C. 1940. Effect of ditching for mosquito control on the muskrat population of a Delaware tidewater marsh. University of Delaware Agriculture Experiment Station Bulletin 255:1-55.

Stevenson, J.C., Kearney, M.S., Pendleton, E.C., 1986. Vertical accretion rates in marshes with varying rates of sea level rise. In: Wolf, D. (Ed.), Estuarine Variability. Academic Press, New York, pp. 241-160.

Stoddart, D.R., Reed, D.J., French, J.R., 1989. Understanding salt-marsh accretion, Scolt Head Island, Norfolk, England. Estuaries 12: 228-236.

Sheikh, P. A. 2005. “The impact of Hurricane Katrina on biological resources.” Congressional Research Service Rep. to Congress, Order Code RL33117, Washington, D.C.

Smith, T. M., & Smith, R. L. (2009). Coastal and Wetland Ecosystems. In B. Wilbur (Ed.), Elements of Ecology (6th ed., pp. 547). San Francisco, CA: Pearson Benjamin Cummings

Stedman, S. and T.E. Dahl. 2008. Status and trends of wetlands in the coastal watersheds of the Eastern United States 1998 to 2004. National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Department of the Interior, Fish and Wildlife Service.

Stumpf, R.P. 1983. The process of sedimentation on the surface of a salt marsh. Estuarine, Coastal and Shelf Science 17:495-508.

Swenson, E.M. and Turner, R.E., 1987. Spoil banks: Effects on a coastal marsh water-level regime. Estuarine, Coastal and Shelf Science, 24: 599-609.

Temmerman, S., T. J. Bouma, G. Govers, Z. B. Wang, M. B. De Vries, and P. M. J. Herman (2005b), Impact of vegetation on flow routing and sedimentation patterns: Three-dimensional modeling for a tidal marsh, J. Geophys. Res., 110, F04019, doi: 10.1029/2005JF000301

Thien. S.J. 1979. *A flow diagram for teaching texture by feel analysis.* Journal of Agronomic Education. 8:54-55.

Turner, R.E. & Lewis, R.R. (1997) Hydrologic restoration of coastal wetlands. Wetlands Ecology and Management 4: 65–72.

Vincent, Robert E., David M. Burdick, and Michele Dionne. 2013. "Ditching and Ditch-Plugging in New England Salt Marshes: Effects on Hydrology, Elevation, and Soil Characteristics." *Estuaries and Coasts* 1-16.

Warren, R. S., P. E. Fell, R. Rozsa, A. H. Brawley, A. C. Orsted, E. T. Olson, V. Swamy, and W. A. Niering. 2002. Salt marsh restoration in Connecticut: 20 years of science and management. Restoration Ecology 10:497–513.

Wamsley, T. V., M. A. Cialone, J. M. Smith, J. H. Atkinson, and J. D. Rosati (2010), The potential of wetlands in reducing storm surge, Ocean Eng. ,37 ,59 – 68, doi:10.1016/j.oceaneng.2009.07.01

Weis JS, Butler C (2009) Salt marshes: a natural and unnatural history. Rutgers University Press, New Brunswick, NJ

Williamson, C.T., 1951. Review of mosquito control in Suffolk County,N.Y. In: Proceedings of the 38th Annual Meeting of the New Jersey Mosquito Control Association

Wilson, W.H., T. Brush, R. Lent, and B. Harrington. 1987. The effects grid - ditching and open marsh water management on avian utilization of Massachusetts salt marshes. Pp. 333-349 in W.R. Whitman and W.H. Meredith, eds. Waterfowl and wetlands symposium: proceedings of a symposium on waterfowl and wetlands management in the coastal zone of the Atlantic Flyway. Delaware Department of Natural Resources and Environmental Control, Dover