Hydrodynamic Drivers of Dissolved Oxygen Variability Within a Highly Developed Tidal Creek in Myrtle Beach, South Carolina

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HYDRODYNAMIC DRIVERS OF DISSOLVED OXYGEN VARIABILITY
WITHIN A HIGHLY DEVELOPED TIDAL CREEK IN MYRTLE BEACH,
SOUTH CAROLINA

By
Douglas Matthew Pastore

Submitted in Partial Fulfillment of the Requirements for the Degree of Master of Science
in Coastal Marine and Wetlands Studies in the School of the Coastal Environment
Coastal Carolina University

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Dedication

This work is dedicated to my parents, Dan and Karen Pastore. Their support and love have made me who I am and given me the opportunity to achieve my goals.
Acknowledgements

I would like to begin by thanking my major advisors Dr. Erin Hackett and Dr. Rick Peterson. To Dr. Hackett, thank you for your support and belief in me. I will always be grateful for the hard work you have invested in my education and hope to continue to learn from you as it has been one of the best experiences of my life. To Dr. Peterson, your guidance along this path that has made this work possible has been unwavering and I will forever be grateful for that. I will always be thankful for the many conversations concerning science and any other facet of life our conversations stray to as they have kept me working and motivated at times when I felt it was impossible to accomplish my goal. I would also like to thank my committee members Dr. Diane Fribance and Dr. Rich Viso. Beginning my journey at Coastal Carolina is in many ways due to the help of Dr. Viso and I would not have had this opportunity without him. Finally, I would like thank Dr. Fribance for the endless field trips, meetings, and collaboration with respect to much of the field work for this project helping progress both this research and my education. This project would not have been possible without my funding through Horry County, Coastal Carolina University, and the School of the Coastal Environment.

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Abstract

Erosion and water quality degradation have been observed in Singleton Swash in Myrtle Beach, SC, and have been hypothesized to be related to migration of the beach-face channel. Dredging this channel temporarily fixes erosional threats to nearby infrastructure but the effects on water quality are not well understood. It is hypothesized that variations in dissolved oxygen (DO) concentration (used here as a proxy for water quality) within the water column are related to changes in vertical mixing and transport due to oceanic tidal forcing. This study utilizes current meters, pressure sensors, and optical DO probes to measure and study the relationships between flow characteristics, DO, and water level. Correlation analysis is used to examine the influence of each physical process on dissolved oxygen variations as well as to explore the relationships between hydrodynamic variables. Results show that larger tidal ranges are associated with higher mean levels of DO concentration in the swash. The larger tidal ranges are linked to greater magnitude currents, and due to tidal asymmetries in the swash, flood currents are stronger than ebb currents. Stronger currents are also linked to larger Reynolds shear stress indicating increased mixing on flood tides relative to ebb tides. It is concluded that the combined transport of oxygenated waters into the swash on large flood tides, and its subsequent mixing due to strong currents, result in increased DO concentrations in the swash. This result is also supported by the direct correlation found between DO concentration and tidal ranges. Health of the ecosystem depends on strong currents and higher magnitude Reynolds shear stress to maintain high DO concentrations within the system. It was found in Hoffmagle (2015) that lower beach-face elevations correlate with larger tidal ranges in the swash; thus, maintenance of the beach-face swash channel is imperative to support a large tidal range and therefore overall ecosystem health.
# Table of Contents

List of Figures ................................................................................................................... vii  
List of Symbols and/or Abbreviations .............................................................................. x  
1.0 Introduction ................................................................................................................... 1  
2.0 Background ................................................................................................................... 7  
   2.1 Overview of Southeastern Tidal Creeks ................................................................. 7  
   2.2 Tidal Creek Hydrodynamics .................................................................................... 8  
   2.3 Tidal Creek Water Quality .................................................................................... 11  
   2.4 Previous Research in Singleton Swash ................................................................. 13  
3.0 Research Objectives .................................................................................................... 16  
4.0 Field Observations ...................................................................................................... 18  
   4.1 Geomorphic Observations ....................................................................................... 18  
   4.2 Hydrodynamic Observations .................................................................................. 19  
   4.3 Water Quality Observations .................................................................................. 22  
5.0 Analytical Methods ..................................................................................................... 27  
   5.1 Quality Control ....................................................................................................... 27  
   5.2 Tidal Range, Currents and Discharge ..................................................................... 29  
   5.3 Reynolds Shear Stress .......................................................................................... 31  
   5.4 Pearson Correlation Analysis ................................................................................. 32  
6.0 Results and Discussion ............................................................................................... 37  
   6.1 Water Quality .......................................................................................................... 37  
   6.2 Hydrodynamics ........................................................................................................ 39  
      6.2.1 Water Velocities ............................................................................................ 39  
      6.2.2 Discharge ........................................................................................................ 41  
      6.2.3 Reynolds Shear Stress .................................................................................... 43  
      6.2.4 Tidal Range ..................................................................................................... 46  
   6.3 Geomorphology ....................................................................................................... 48  
   6.4 Implications ............................................................................................................ 48  
7.0 Summary and Conclusions ......................................................................................... 67  
8.0 Works Cited ................................................................................................................ 69  

List of Figures

**Figure 1:** Local sites showing tidal creeks traversing the beach-face (red arrows) defined as swashes: A) Singleton Swash, located 11 Km north of Myrtle Beach, SC; B) Withers Swash, located in Myrtle Beach, SC C) Floral Lake Swash, located in Surfside Beach, SC; D) Dogwood Lake Swash, located 1 mile north of Surfside Beach, SC (Google Earth ©, 2017)..........................5

**Figure 2:** Aerial photos of Singleton Swash tidal creek with surrounding watershed (A) as well as Singleton Swash beach site showing the most dynamic area of the swash in terms of geologic change (B). Movement of the swash interferes with surrounding infrastructure primarily the Dunes Golf Course (yellow circle) and Beach Club (blue circle) to the south (Google Earth ©, 2017)................................................................................................................................................ 6

**Figure 3:** Time series of Singleton Swash tidal range to ocean tidal range ratio (SSTR/OTR) and transition point elevation (considered a deposition “hotspot”). Correlation of these time series yields a strong inverse relationship (Correlation Coefficient ($R$) =-0.647) (Hoffnagle, 2015).....15

**Figure 4:** Singleton Swash beach-face where RTK-GPS elevation studies are performed. Transition point elevation (33°45'23.36"N 78°47'41.25"W) is determined for this area from the elevation surveys. The image date was 11/15/2017, one month after the last beach-face elevation survey of this study. ....................................................................................................................... 23

**Figure 5:** Singleton Swash including the transition point, Site A where the Argonaut, DO, and water level probes were deployed, Site B where a supplementary water level probe was deployed, and Site C where the ADV was deployed.................................................................24

**Figure 6:** Schematic of the instrument locations. The view is downstream (+x). Red circles denote locations of dissolved oxygen measurements, the black outlines are the sample volume locations for the Vector (ADV) and Argonaut (2-D ADCP), the blue circle is the location of water level measured and the blue dashed line is the water level. The vertical locations of the measurements are given.................................................................................................................25

**Figure 7:** Along channel (A) downstream (+x) and across channel (B) (-y) view of the Vector, seating, and frame at low tide on 28 March 2018. The Vector is centered in the channel approximately 6.9m from the bank in (B)........................................................................................................................................26

**Figure 8:** Sample time series of all three velocity components on 06 May 2018 beginning at 16:00 (A), where the along channel velocity $u$ is the blue, the across channel velocity $v$ is in red, and the vertical velocity $w$ is in yellow. Power spectral density for along channel velocity, $u$ (B), exhibiting a peak at a frequency within the approximate timescales for wind waves (~1 Hz).....35
Figure 9: Schematic of the cross-sectional measurement used to calculate discharge. $D$ is the total depth of the water column at a given position ($y$) and time ($t$) as calculated in Equation 2, where $z$ is the reference height at a given position ($y$) (measured on 28 March, 2018) and $h$ is the water level time series. 

Figure 10: Sample time series of percent (%) saturation of dissolved oxygen at Site C over a period of 15 days. Daylight hours (x) begin at 06:00 and end at 20:00 and non-daylight hours (x) range from 20:00 to 05:00. Regions of light blue denote times during spring tide while the grey denotes periods of neap tide.

Figure 11: (A) Time series of along-channel current velocity ($u$) measured by the ADV during the May, 2018 deployment. (B) Power spectral density (PSD) of current velocity ($u$) from the May deployment of the ADV. The first major spectral peaks (circle) occur at 12.5 hours with a harmonic occurring at a period of 6.25 hours (circle). Secondary spectral peaks (circle) correspond to periods of 25.6, 8.25, and 4.13 hours.

Figure 12: Average correlation coefficient between DO concentration and current velocity ($u$) versus lag time over 48 hours, 95% confidence intervals are shown in red; peak correlation is $R = -0.511$ at 1.75 hour-lag.

Figure 13: Time series of both the flood (diamond) and ebb (square) discharge ($Q$, Equation 3 and 4) per tidal cycle computed from the data collected during the May deployment of the ADV.

Figure 14: (A) Time series of Reynolds shear stress ($R_S$) calculated from velocity measurements recorded by the ADV over the May deployment. (B) Percent (%) water column above ADV (blue), in which times of high tide are denoted by x and times of low tide are denoted by x.

Figure 15: Average correlation between dissolved oxygen and flood (A) and ebb (B) discharge over ten-day periods (Equation 4 and 5). 95% confidence bounds are shown in red; flood peak $R = 0.608$ at 0 day-lag and peak ebb $R = -0.541$ at 0 day-lag.

Figure 16: Power spectral density ($x$) of Reynolds shear stress ($R_S$). Two dominant peaks correspond to 12 and 6-hour time scales.

Figure 17: Average correlation of Reynolds shear stress ($R_S$) and current velocity ($u$) magnitudes over 12 hours for the May deployment, 95% confidence bounds are shown in red; peak correlation is $R = 0.434$, 0 hour-lag.
**Figure 18:** Average correlation of dissolved oxygen concentration and wind speed over 14 days for the May deployment, 95% confidence bounds are shown in red; peak $R=0.274$ at 0.042 day-lag.

**Figure 19:** Average correlation of Reynolds shear stress magnitude ($RS$) and dissolved oxygen concentration over 12 hours for the May 2018 deployment, 95% confidence bounds are shown in red; peak correlation is $R=0.276$, 0 hour-lag.

**Figure 20:** Time series of tidal range (blue; left axis) in Singleton Swash at Site B and elevation of the transition point (orange; right axis) on the beach-face. The black line denotes the low pass filtered and resampled tidal range time series used in the correlation analysis between tidal range and transition point elevation.

**Figure 21:** Average correlation of tidal range between Sites A and B over 14 days using data from 1 September, 2017 to 1 January, 2018, 95% confidence bounds shown in red; peak correlation is $R=0.981$, 0 day-lag.

**Figure 22:** Average correlation of tidal range and maximum flood (A) and ebb (B) tide current velocity magnitude over 24 hours for the May deployment of the ADV, 95% confidence bounds shown in red; flood peak correlation is $R=0.927$, 0 hour-lag and ebb peak correlation is $R=0.842$, 0 hour-lag.

**Figure 23:** Average correlation of tidal range and flood (A) and ebb (B) tide discharge ($Q$) over the May deployment of the ADV, 95% confidence bounds shown in red; flood discharge correlation is $R=0.754$, 0 day-lag and ebb peak correlation is $R=0.977$, 0 day-lag.

**Figure 24:** Average correlation of tidal range and dissolved oxygen over 14 days using data from 1 September, 2017 – 1 June, 2018, 95% confidence bounds shown in red; Peak correlation is $R=0.277$, 1.5 day-lag.
### List of Symbols and/or Abbreviations

<table>
<thead>
<tr>
<th>Symbol</th>
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<td>( v )</td>
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<tr>
<td>( \rho )</td>
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<td>( \tau_w )</td>
<td>Bottom shear stress</td>
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<td>2-Dimensional</td>
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<tr>
<td>$R$</td>
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<td>Yellow Springs Instrument</td>
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<tr>
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<td>Bank inland (left) position</td>
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<tr>
<td>$y_{br}$</td>
<td>Bank oceanic (right) position</td>
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1.0 Introduction

Tidal creeks are a common and integral part of the coastline along the southeastern United States. Tidal creeks are defined by Healy (2005) as “… relatively long and narrow, are shallow, and exhibit tidal water level fluctuations and weak tidal currents.” Tidal creek systems function as habitats, fish breeding grounds, sediment and pollutant filters, as well as recreational areas (Leonard, 1997; Mitsch and Gosselink, 2000). Water level fluctuations in tidal creeks are primarily dictated by local tidal regimes (semidiurnal in the southeastern U.S.), which influence the environmental characteristics of the tidal creek systems.

Tidal creek systems are highly susceptible to change due to variations in tidal forcing. Sediment supply and nutrient concentration are examples of factors that vary with the tides. Along with variations in tidal forcing, the previously mentioned factors are also controlled by other natural processes such as local winds, non-tidal currents, and waves (Brinson, 1988). Anthropogenic modifications to these systems (e.g., dredging, construction, and runoff alteration) can also change the tidal creek system both intentionally and unintentionally. Runoff alteration to near-shore tidal creeks systems can change drainage characteristics and create what is known as a “swash”. A swash is a specific type of tidal creek that traverses the beach-face and connects directly with the ocean (Figure 1) (NERRA, 2018).
Natural changes to swash systems, specifically sediment deposition restricting tidal exchange, can lead to poor ecosystem health. Restricted tidal exchange decreases semi-diurnal flushing which can lead to impaired water quality conditions. Concentrations of bacteria (e.g., fecal coliform) and dissolved oxygen (DO) levels serve as useful indicators of impaired water quality conditions within similar systems (Mallin et al., 2001). Restricted water exchange can also lead to lower water velocities in the swash enabling salinity stratification that limits vertical mixing of DO within the water column of the creek. Reduced mixing as well as transport of DO and other nutrients can stress local organisms due to low supply of these life-sustaining compounds degrading overall ecosystem health.

Poor ecosystem health in areas that rely on tourism can adversely affect local economies. Many coastal communities include tidal marshland where residents and tourists enjoy fishing, water sports and other recreational activities. Poor environmental quality in these areas leading to beach closings due to high fecal coliform concentrations or low fish abundance (i.e., poor fishing conditions) caused by hypoxic (poorly oxygenated) waters, can deter tourists. Area managers along the South Carolina coast are concerned with these water quality issues due to its multibillion-dollar tourism industry. This study focuses on Singleton Swash (Figure 2), a tidal creek system located along the Grand Strand in Myrtle Beach South Carolina.

Singleton swash is a tidal creek located approximately 11 kilometers north of Myrtle Beach, SC. The local tides observed along the coast are microtidal with a roughly 1.5 m tidal range, differing from the 0.5 m tidal range observed within the swash. The swash creek extends approximately 2 km inland with an average depth and width of 0.75 m and 14 m, respectively. The creek terminates inland at a fresh water pond that drains
into the swash basin. Along with drainage from the pond, the local golf course and surrounding hotel comprise the 6.8 square kilometer watershed that empties into the creek (USACE, 2009). The path of the inland channel is highly stable due to dense vegetation, whereas the section of channel that traverses the beach-face is highly dynamic.

Littoral drift causes the creek channel to migrate southward threatening local infrastructure. Impending erosion frequently threatens the local golf course clubhouse as well as beachfront homes extending 200 m south of the golf course. Water quality of the inland salt marsh also becomes degraded during periods of increased creek channel migration. Horry County has attempted to mitigate erosion and water quality degradation by periodically (~ every 8 months) dredging the beach-face channel, effectively straightening and directly connecting the inland creek to the ocean.

A previous study of Singleton Swash performed by Hoffnagle (2015) suggests creek morphology is associated with DO variations within the creek channel. Stable salinity stratification within the swash was shown to vary indirectly with tidal range, which was used as a proxy for flushing (Hoffnagle, 2015). Hoffnagle (2015) also examined the relationship between bottom DO and stable salinity stratification but the results were inconclusive and so the author notes the need for more research to understand the mechanics of dissolved oxygen exchange and transport within the swash.

This study extends research in this area by linking specific physical phenomena observed in Singleton Swash to water quality variability within the tidal creek system. Observations of geomorphology, current velocities, water level, and mixing within the tidal creek provide insight into the relative role of these mechanisms in driving water quality variability. Correlation analysis is used to explore the relationships between the
aforementioned geomorphic, hydrodynamic, and water quality variables. The analysis establishes the type, significance, and temporal scale at which the relationships occur. The results of this study show that asymmetric tides within the swash control the physical mechanisms driving DO variability within the swash and that higher tidal ranges yield larger mean DO concentrations.

The proceeding sections are organized as follows: a background of southeastern tidal creeks along with a review of prior research in Singleton Swash is described in Section 2. Section 3 presents the research objectives and hypotheses. Section 4 details field observations, and Section 5 explains the analytical methods. Section 6 presents and discusses results from the study. Finally, Section 7 summarizes the study, presents conclusions and discusses the implications for the study site as well as future work.
**Figure 1:** Local sites showing tidal creeks traversing the beach-face (red arrows) defined as swashes: A) Singleton Swash, located 11 Km north of Myrtle Beach, SC; B) Withers Swash, located in Myrtle Beach, SC; C) Floral Lake Swash, located in Surfside Beach, SC; D) Dogwood Lake Swash, located 1 mile north of Surfside Beach, SC (Google Earth ©, 2017).
Figure 2: Aerial photos of Singleton Swash tidal creek with surrounding watershed (A) as well as Singleton Swash beach site showing the most dynamic area of the swash in terms of geologic change (B). Movement of the swash interferes with surrounding infrastructure primarily the Dunes Golf Course (yellow circle) and Beach Club (blue circle) to the south (Google Earth ©, 2017)
2.0 Background

2.1 Overview of Southeastern Tidal Creeks

Tidal creeks are located extensively along the southeastern coast of the United States and serve many functions for the surrounding ecosystem. Tidal creeks serve as a conduit to connect terrestrial ecosystems with coastal marine systems and often promote the growth and development of extensive salt marsh ecosystems along their flow paths. These systems are common due to the abundance of sediment along the coastline (both inland and in the coastal ocean) allowing for continued marsh surface accretion with rising sea level (Leonard, 1997). This accretion of sediments dictates the level of inundation of flora and fauna throughout daily tidal cycles within these systems. Species local to salt marsh systems are sensitive to the specific habitat dictated by tidal flooding and ebbing. A change in mean sea-level can impact species distribution due to changes in local habitat (Morris et al., 2002), which may in turn influence the overall function and health of tidal creek systems.

Small tidal systems offer prime habitats as well as quality breeding grounds for various organisms particularly at low trophic levels (i.e., primary producers and consumers). Such environments attract vertebrates and invertebrates due to the abundance of their food source. Salt marsh tidal creeks also relieve stressors from high energy environments (e.g., the surf zone) creating an environment ideal for reproduction (Herke, 1971; Peterson and Turner, 1994; Turner et al., 2000).
Small tidal ecosystems are highly vulnerable to anthropogenic alteration (Sanger et al, 2013). Modification of the coastal environment due to urbanization has intensified terrestrial runoff and nutrient/contaminant input into the surrounding environments, including tidal marsh systems (Smith, 2015). Yet, the rapid growth environment and frequent tidal flushing often prevent these inputs from degrading the tidal creek water quality. Sanger et al. (2013) observed that subtidal zones contain lower concentrations of nutrients and harmful bacteria (e.g., fecal coliform) than intertidal zones. This finding suggests that factors such as tidal flushing can efficiently counteract pollutant inputs within tidal creek systems.

2.2 Tidal Creek Hydrodynamics

Hydrodynamics (i.e., physical water flow characteristics) influence factors such as dissolved and particulate material transport, sedimentation, and morphology within tidal creek systems. Flow direction and magnitude through tidal creeks vary on event scales in response to stormwater runoff as well as on hourly, daily, and bi-weekly time scales due to tidal cycles. Wave and tidal energy can resuspend sediments, thereby changing the landscape of the tidal system (Bouma et al., 2005). Sediment deposition and transport are also directly related to the energetics of currents within the system and so ultimately influence the morphology of the area (Allen, 1990).

Tidal creek morphology, specifically channel geometry, is controlled by depth and flow. Semi-diurnal tidal cycles dictate the flow, and therefore energy, within tidal basins and so exert a controlling factor on sediment resuspension (Gardner and Bohn, 1980). Spring-neap cycles also influence sediment transport (both grain size and distance of transport); more energy from spring tides allows for larger particles to be resuspended and
transported farther (Smith and Mclean, 1984). As tides propagate inland through tidal basins, friction from the channel bed reduces current velocities and associated energy axially into the creek. This friction leads to asymmetric tides, resulting in spatial variability of sediment transport within creek systems (Dronkers, 1986).

Flow instabilities derived from shear (which is caused by friction at boundaries) result in a turbulent environment. A boundary layer within the channel due to bottom stress causes heterogeneity in the along-channel velocities with depth. Bottom stress varies with current speed and therefore varies in time and space causing spatiotemporal variations in turbulence within the channel. At the air-water boundary, wind stress on the surface plays a similar role in generating shear that can lead to flow instability and consequently turbulence. Variations in bathymetry (i.e., changes in water depth within the channel) also impact vertical water velocity distributions within tidal creek systems and consequently affect the magnitude of shear (O’Laughlin and van Proosdij, 2014). Variations in currents are related to both external (i.e., tides, waves, wind) and internal (i.e., vegetation, bathymetry, channel geometry, and water column stratification) factors, which can cause changes to turbulent environments.

Turbulent environments in open channels, such as tidal creeks, create an optimal setting for vertical mixing. The type of flow (turbulent or laminar) in a specific environment is best characterized by the Reynolds number \((Re)\). The Reynolds number is defined as \(UL/\nu\), where \(U\) is a characteristic along-channel velocity, \(L\) is a characteristic length (commonly depth for tidal creeks with large width:depth ratios) and \(\nu\) is the kinematic viscosity of the fluid (Cimbala and Cengel, 2008). Reynolds number is a dimensionless number representing the ratio of inertial to viscous forces, where large \(Re\)
indicates turbulent flow. The $Re$ in Singleton Swash calculated in this study is on the order of $10^6$ and similar turbulent systems have $Re \sim 10^5$ (Leonard and Luther, 1995). Based on this comparison, Singleton Swash is a turbulent environment.

Turbulent environments can enable more efficient mixing than laminar environments in which mixing is only achieved through molecular diffusion and advective dispersion. Dispersion is the spreading of solid particles due to momentum transfer (Fischer, 1973; Thorpe, 2005). Mixing is altered due to turbulent diffusion, which can enhance dispersion. Therefore, turbulent mixing influences the transport of sediments, nutrients and dissolved oxygen within the water column (Leonard and Luther, 1995), so it represents an important process in estuarine systems.

Shear is the primary driver of turbulent vertical mixing within a well-mixed boundary layer flow. Shear is defined as the change in $u$ (along-channel velocity) with respect to $z$ (vertical position) ($\partial u / \partial z$). The dominant turbulent shear stress that occurs in a boundary layer is the Reynolds shear stress ($-u'w'$), where $w$ is the vertical velocity, and the prime ('') denotes turbulent velocity. The shear or friction velocity is defined as $u* = \sqrt{\frac{\tau_w}{\rho}}$, where $\tau_w$ is the bottom shear stress and $\rho$ is fluid density (Cimbala and Cengel, 2008). In the logarithmic region of a boundary layer (i.e., where the law-of-the-wall is valid), the Reynolds stress is on the same order as $(u*)^2$; thus, Reynolds stress is often characterized by friction velocity ($u*$) (Thorpe, 2005). Fischer (1973) demonstrates that eddy diffusivity, which is the turbulent diffusion coefficient, is proportional to $u*$; thus, Reynolds stress can be considered an indicator of mixing via turbulent diffusion in boundary layers.
2.3 Tidal Creek Water Quality

The complicated flow characteristics of tidal creek systems impart influences on the distribution of materials that contribute to ecosystem health and water quality (e.g., nutrients, contaminants, larvae, and phytoplankton). Tidal creek systems in highly developed areas are of interest due to their potential influence on estuarine water quality (Mallin et al., 2001), because they experience enhanced terrestrial runoff and nutrient loading from engineered stormwater conveyance systems (Smith, 2015). An increase in nutrient-rich input via surface runoff often stimulates a heterotrophic response that can lead to poor water quality within tidal creeks (i.e., low levels of dissolved oxygen; hypoxia) (Hutchins et al., 2012).

Urban areas consisting of dense infrastructure often contribute nutrients to local watersheds due to decreased infiltration capacity as a result of impermeable surface and subsurface structures (Hopkinson and Vallino, 1995). In these urban environments, stormflow has been shown to transport higher concentrations of nutrients (e.g., ammonium, nitrate, nitrite, and phosphorus) and larger volumes of water compared to subsurface flow (Hopkins and Vallino, 1995; Hutchins et al., 2012). Watersheds are also adversely affected by pesticides transported through runoff that potentially kill fish, vegetation, and microorganisms, degrading the overall quality of the ecosystem (Fulton et al., 1999).

High nutrient concentrations from increased terrestrial runoff can lead to eutrophication within tidal creek systems. Eutrophication occurs when organic matter and nutrient concentrations are increased within aquatic ecosystems resulting in excessive algal growth” (Bricker et al., 1999; Tyler et al., 2009). This strong increase in primary
productivity subsequently increases the rate of aerobic decomposition, potentially
decreasing the concentration of DO within the system (Diaz et al., 2004).

The response of a tidal system to eutrophication depends on multiple parameters
(e.g., flushing and stratification). Residence times of water, controlled by flushing, within
a basin influence the overall nutrient concentrations (Paerl, 2006). Estuaries with shorter
residence times (i.e., faster flushing) are less susceptible to eutrophication as nutrient
contamination is flushed out relatively quickly. Conversely, longer residence times lead to
a stagnation of the system, in which biogeochemical processes have more time to utilize
nutrients and eventually degrade water quality conditions.

Stratification within the water column can also lead to more significant effects from
eutrophication. Water column mixing allows for the transport of dissolved oxygen from
the overlying atmosphere and surface waters to the bottom layer of the water column,
replenishing the oxygen depleted through respiration (Kemp et al., 2005). Reduction in
vertical mixing (due to stratification) therefore decreases the supply of oxygen to the
bottom layer of the water column. Stable stratification acts to inhibit vertical mixing as a
portion of the energy must be used to convert potential energy to kinetic energy. A
reduction in vertical mixing paired with no additional sources of DO can lead to hypoxic
(defined as dissolved oxygen concentrations below 2 mg/L) conditions from biological
respiration of oxygen (Dauer et al., 2000; Zhang et al., 2010). These hypoxic conditions
stress organisms resulting in relocation and in extreme cases mortality; ultimately, these
effects decrease the overall health of the ecosystem. Thus, making dissolved oxygen an
indicator of overall ecosystem health.
2.4 Previous Research in Singleton Swash

Singleton Swash, a tidal creek near Myrtle Beach, SC and the site of this study, has been previously studied because of extreme morphodynamics observed within the system. Previous work has explored water quality, morphodynamics, and hydrology of the area (USACE, 2009, Hoffnagle, 2015).

Hoffnagle (2015) examined linkages between water quality and beach morphodynamics in Singleton Swash. This study suggests that over time, the mouth of the swash migrates due to changing beach-face features. Sediment deposition “hotspots” along the beach-face directly correlate to decreases in Singleton Swash tidal range relative to ocean tidal range (Figure 3).

Dredging of the beach-face is the current solution for temporarily resolving this issue. Hoffnagle (2015) reports that as tidal flushing diminishes, which is inferred from a reduced tidal range, salinity gradients increase leading to stratification within the water column. Hoffnagle (2015) explored the relationship between dissolved oxygen saturation levels and water column stratification caused by salinity, but no clear relationship was found.

The United States Army Corps of Engineers (USACE) (2009) performed a study of Singleton Swash with the goal of modeling the swash system using CMS-Flow. CMS-Flow simulates coastal currents, water level, sediment transport, and morphology to explore changes due to engineering modification options such as jetty construction, shoal mining, channel filling and dredging (USACE, 2009). Acoustic Doppler current profiler (ADCP) data were collected to study currents within the swash. Water velocity
measurements indicate that tidal current magnitudes are greater during flood tide, indicative of a flood-dominated system.

The USACE (2009) study suggests future structural solutions should aim to maximize flow between the swash basin and ocean to decrease sediment accumulation and channel migration. The USACE recommends implementing a semi-permanent solution through the addition of rock structures on the southern bank to slow channel migration. As a permanent solution, the study recommends a vertical wall be installed on the northern bank of the swash. The wall must facilitate ocean-swash exchange and allow for continued recreation in the area while being constructed at a low enough elevation to ensure public safety.

These previous results form the basis of the experimental design of this study. This present study builds upon previous research in Singleton Swash by incorporating this historical data as well as adding information via direct measurement of additional hydrodynamic features of the swash. This extension enables more insight into the relationships between geomorphology, hydrodynamics, and water quality in Singleton Swash.
Figure 3: Time series of Singleton Swash tidal range to ocean tidal range ratio (SSTR/OTR) and transition point elevation (considered a deposition “hotspot”). Correlation of these time series yields a strong inverse relationship (Correlation Coefficient ($R$) = -0.647) (Hoffnagle, 2015).
3.0 Research Objectives

Previous research of this tidal creek system has demonstrated relationships between tidal range and creek channel elevation. Hoffnagle (2015) found an inverse relationship between maximum beach-face elevation and tidal range, i.e., larger elevations co-occurred (or preceded) reduced tidal ranges. Direct correlation between tidal range and tidal current velocities has been found in similar systems (Mao et al., 2004), and thus, similar relationships are expected for Singleton Swash. Strong tidal currents have been shown to result in greater Reynolds shear stresses in estuarine systems (Stacey et al., 1999). Similar trends are expected in this tidal creek system because greater magnitude currents cause greater shear in the bottom boundary layer of the channel resulting in increased Reynolds shear stresses (Stacey et al., 1999). It was also shown in Hoffnagle (2015) that a reduced tidal range co-occurred with low dissolved oxygen concentrations in bottom waters of the swash. Replenishment of oxygen in the tidal creek can originate from two primary abiotic sources: (i) transport by tidal currents directed into the swash from the ocean, and (ii) downward mixing of oxygen derived from the atmosphere.

This study aims to investigate the relationships between beach-face elevation, tidal range, and water velocities with respect to their combined influence on water quality in a highly dynamic tidal creek. The primary goal of this study is to delineate these physical mechanisms and their impact on driving water quality variability in the Singleton Swash
tidal creek system. The study uses a combination of geophysical, hydrodynamic, and geochemical measurements across a broad spectrum of temporal scales to explore these relationships in this setting.

Consistent with prior studies, an inverse relationship between geomorphologic indicators of beach-face variability (transition point elevation) and tidal range in Singleton Swash is expected. Furthermore, tidal range is hypothesized to have a direct relationship with current velocities measured in the thalweg of the tidal creek because it is assumed that the tides are the primary driver of currents within the swash. It is hypothesized that dissolved oxygen levels (a proxy for water quality in the system) have a direct relationship with the magnitude of current velocities due to their role in transport of DO into the swash as well as their role in mixing due to increased bottom stress. Because mixing is also enhanced with increased wind speed due to larger wind stress on the water surface, we also expect to observe direct relationships between dissolved oxygen concentrations and wind speed. Finally, due to the role of vertical mixing in influencing DO bottom concentrations, we expect a direct relationship between DO concentration and Reynolds stress. The relative magnitude of these observed relationships can shed light on the importance of the different abiotic factors driving water quality variability in Singleton Swash.
4.0 Field Observations

4.1 Geomorphic Observations

Beach-face real-time kinematic corrected global positioning system (RTK-GPS) surveys were conducted to collect geomorphic data on the transition point elevation. The surveys were made in State Plane NAD-83 meters along the WGS-84 geoid and collected on foot with a backpack-mounted receiver. Six surveys were conducted, each performed along a grid covering 37,390 m$^2$ of the dynamic beach-face including the channel. The surveys were performed at low tide to enable collection of the semidiurnally submerged transition point. Data acquired were corrected horizontally at 10 cm and vertically at 20 cm accuracy, sampling at a frequency of 10 Hz. These data were used to quantify change in elevation of the beach-face over time.

Transition point elevation of the creek channel is located at 33º45’23.36” N 78º47’41.25” W. Transition point, seen in Figure 4, is extracted from the beach-face data and was determined similarly to the methods used in Hoffhagle (2015). The author defined the transition point as “[The point] found at the landward side of the surveys where the stable part of the channel meets the unstable part of the dynamic creek channel”. The transition point elevation is indicative of the sediment deposition/erosion within the channel on the beach-face.
Water depth of the creek channel was measured in 25 cm increments across the width of the channel at Site C (Figure 5). This water depth was used as a reference for water heights measured at Site C (see Section 5.2). The channel cross-section was assumed to be constant for the duration of the study and was used to calculate discharge.

### 4.2 Hydrodynamic Observations

Hydrodynamic measurements consist of water level and water velocities. Water level data were collected at three separate locations (see Figure 5): (i) at Site A; (ii) at Site B which is approximately 350 meters upstream of Site A; and (iii) at Site C (160 m upstream of Site A). The pressure sensor at Sites A and B (Onset Hobo water level logger) measured absolute pressure on 15-minute intervals. The sensors were deployed in PVC housings approximately 0.3 and 0.1 m above the creek bed, at Sites A and B respectively. An internal pressure correction was applied using concurrent barometric pressure measurements made from Apache Pier, located approximately one kilometer from Site B. Following the barometric pressure correction, water levels were referenced to height above NAVD-88. Water pressure measured at Sites A and B on 1-second intervals were converted to water height above the sensor by standard sea-level atmospheric pressure. Water level at Site C was measured at approximately 0.83 m above the creek bed at this site.

The $u$ and $w$ velocities were measured by a 2-dimensional acoustic Doppler current profiler (2D ADCP), *Sontek Argonaut-SW*, at Site A. This site is located in the center of a relatively straight section of the tidal creek channel which is assumed to contain relatively low cross channel velocities because it is not near any bends in the channel. The *Argonaut* was mounted on a frame at the bottom of the channel and recorded data from the water column as an “upward looking” device (Figure 6). The frame consists of a polycarbonate
plate and was secured to the channel floor using sand screws, connected at each corner of the plate. The plate was elevated 3-4 cm above the channel bottom to decrease measurement interference from sedimentation. The *Argonaut* was deployed from 4 November, 2017 to 4, December, 2017. Measurements of water velocity, sampled at 1 Hz, were averaged across 5-minute bursts every 15 minutes. The *Argonaut* measured water velocity in cells 20 cm in height from 7 cm above the instrument (13 cm above seabed) to the top of the water column; thus, the number of cells was variable in time. The *Argonaut* calculates a single depth integrated velocity by averaging velocity measurements over the entire range of cells accurate within ±1% (±0.05m/s) of the measured velocity (Sontek, 2006).

Velocity measurements collected at Site C were performed in two separate autonomous deployments of a *Nortek Vector* acoustic Doppler velocimeter (ADV). The first deployment occurred 14-28 March, 2018, while the second deployment occurred 1-31 May, 2018. The *Vector* was mounted to a PVC frame drilled into the channel bed. The instrument was oriented “facing down” and the sample volume was located 15 cm below the probe head or approximately 55 cm above the creek bed (deployment 1: 58.5 cm; deployment 2: 56.5 cm) (Figure 7). The ADV measured three velocity components ($u$, $w$, and cross-channel velocity, $v$) within the sample volume. Measuring water velocities over this small sample volume (300 cubic millimeters) allows for more accurate velocity measurements (Nortek, 2005). The ADV collected data in an along channel ($x$), across channel ($y$) and vertical ($z$) coordinate system. During these deployments, a nominal velocity of ±1.00 m/s (vertical = ±0.6 m/s, horizontal = ±2.1 m/s) is used to minimize
Doppler noise in the measurements without exceeding this range. The Vector is accurate within ±0.5% (±1 mm/s, whichever is greater) of the measured velocity.

The first deployment used a sampling frequency of 64 Hz for 15-minutes (57,600 samples) at an interval length of 45-minutes (sample for 15 minutes; dormant for 30 minutes). During the second deployment, a sample frequency of 64 Hz was also utilized, but due to rapid change in tidal currents observed from the first deployment, the sample period was 5-minutes (19,200 samples) at an interval length of 15-minutes. The chosen sampling frequency is the highest frequency at which the instrument can measure and presumably resolves the majority of the range of turbulent scales. Prior studies have estimated turbulence characteristics in similar environments using an ADV with a sampling frequency of 10 Hz\(^1\) (Voulgaris and Meyers, 2004). Deviations from the burst-mean velocity are considered turbulent fluctuations. Data were screened for wave-induced motion using fast Fourier transforms; bursts showing wave-induced flow were not used to estimate turbulence fluctuations due to potential contamination of turbulence estimates due to the waves (Trowbridge, 1999; Feddersen and Williams, 2007).

To supplement the water velocity and turbulence measurements collected in the swash, wind speed measurements from a meteorological station (collected every 5-minutes) at Apache Pier (Long Bay Hypoxia Monitoring Consortium; sutronwin.com) are utilized in the study. Wind speed measurements are utilized as an indicator of wind stress magnitude at the water surface, which varies with the square of wind speed (Thorpe, 2005).

\[^1\] An analytical model was used to estimate the turbulent kinetic energy dissipation rate. (Voulgaris & Meyers, 2004)
4.3 Water Quality Observations

Dissolved oxygen concentrations within Singleton Swash was measured using an Onset dissolved oxygen data logger at Sites A and C (Figure 5). DO concentration was measured continuously at Site A from 1 September 2017 through 1 June 2018 and at Site C from 30 April through 1 June 2018. The data loggers optically measure dissolved oxygen (concentration in mg/L) and temperature (°C). The logger is accurate at ± 0.2 mg/L. Dissolved oxygen concentration and temperature were measured on 5-minute time intervals continuously throughout the study. At Site A, the DO probe was attached to a PVC pipe anchored to the seabed, so the sensor sat approximately 30 centimeters above the seabed. At Site C, the probe was also attached to a PVC pipe anchored to the creek bed but sat approximately at 56.5 cm above the sea bed (depth of Vector sample volume during deployment 2). To supplement the dissolved oxygen data, a YSI ProODO was used to measure dissolved oxygen measurements concentration to reference the start and end DO concentrations measured by the Onset data logger, with end values occurring on removal from housing for data collection.
**Figure 4:** Singleton Swash beach-face where RTK-GPS elevation studies are performed. Transition point elevation (33°45'23.36"N 78°47'41.25"W) is determined for this area from the elevation surveys. The image date was 11/15/2017, one month after the last beach-face elevation survey of this study.
**Figure 5:** Singleton Swash including the transition point, Site A where the *Argonaut*, DO, and water level probes were deployed, Site B where a supplementary water level probe was deployed, and Site C where the ADV was deployed.
Figure 6: Schematic of the instrument locations. The view is downstream (+x). Red circles denote locations of dissolved oxygen measurements, the black outlines are the sample volume locations for the Vector (ADV) and Argonaut (2-D ADCP), the blue circle is the location of water level measured and the blue dashed line is the water level. The vertical locations of the measurements are given.
Figure 7: Along channel (A) downstream (+x) and across channel (B) (-y) view of the Vector, seating, and frame at low tide on 28 March 2018. The Vector is centered in the channel approximately 6.9m from the bank in (B).
5.0 Analytical Methods

The approaches for quantifying relationships between tidal creek transition point elevation, tidal range, current velocities, and Reynolds shear stress as well as their influence on water quality are discussed in this section. In the first subsection, details of the efforts to ensure quality control of the data are discussed. The second subsection describes the methods used to estimate tidal range, current velocities, and discharge within the creek system. The third subsection defines the methods by which Reynolds shear stress is estimated. Finally, the fourth subsection details the correlation analysis methods used extensively throughout this study to explore relationships between hydrodynamic, water quality, and geomorphic variables.

5.1 Quality Control

Water velocities measured by both the Vector and Argonaut were processed for erroneous measurements before analysis. Each quality control process to identify erroneous values is performed over each burst interval. First, for the Vector, beam pulse series correlations (including all axes: x, y, z) estimated by the Vector are used to filter the data. Auto-correlation performed by the ADV accounts for variations in the Doppler shift estimated due to different types and concentrations of scattering particles near the sample volume (Elgar et al., 2005). The threshold for acceptable data is a correlation of 0.90 (Nortek, 2005); data falling below this threshold have been excluded from the data set. Along with signal correlation, data were screened for extremely low (< 8 dB) signal-to-
noise ratios (SNR) (Elgar et al., 2005). Values < 8 dB indicate low concentrations of acoustic reflecting materials in the sample volume or water levels falling below the transducer head. No extremely low SNR values were observed, and it is therefore assumed that the water level never fell below the sensor head. To compliment the internal quality control parameters, velocity measurements exceeding ±3.5 standard deviations from the burst-mean velocity have been removed from the data. The threshold of 3.5 standard deviations was determined by trial and error; this threshold removed any clear outliers without removing seemingly valid large/small velocities.

Standard error of velocity is internally calculated within the Argonaut corresponding to each velocity measurement. Velocities with standard errors greater than two are considered erroneous (Sontek, 2006) and have been removed from the data set. In conjunction with standard error, SNR was also evaluated to identify erroneous data. SNR fluctuations greater than 20 dB between consecutive measurements indicate burial of the instrument; data meeting this criterion have been removed.

Dissolved oxygen measurements are corrected for any drift in the dissolved oxygen monitoring record using HOBOware® and the calibration DO measurements made with the YSI. These corrections were performed for every interval of dissolved oxygen deployment (approximately 2 weeks); calibration DO measurements are used for both the starting and ending values. Consecutive DO measurements exceeding a difference of 20% saturation were considered erroneous and removed from the data set. This threshold of 20% saturation removed outliers within the time series without affecting accuracy of the measurements.
Water level data were also screened for erroneous measurements by removing values that varied more than 0.5 m between consecutive measurements. This procedure assumes that within a 15-minute measurement interval, water level will not vary by more than 33% of the normal tidal range in the region (mean tide range of 1.5 m) (Barnhardt, 2009). These values deemed erroneous were removed from the data set.

5.2 Tidal Range, Currents and Discharge

Tidal range of the swash is computed by subtracting consecutive (maximum) high and (minimum) low water levels. These high and low water levels are determined using the temporal derivative of the water level time series, $\frac{\partial h}{\partial t}$. The zero crossing points of $\frac{\partial h}{\partial t}$ represent the maximum and minimum water levels at high and low tides, respectively. A boxcar filter (eight data points corresponding to two hours) was applied to the measured water level time series to reduce the effects of noise in determining the timing of zero-crossings. The filter is applied in both directions along the time series to ensure no phase shift of the data. The difference in water level at the times between consecutive zero crossing points of $\frac{\partial h}{\partial t}$ (high and low tide) is calculated for the entire time series and serves as an estimated tidal range. Once calculated, the tidal range time series for Site B was low pass filtered with a moving average filter of 14 days (removing the spring neap cycle) then resampled using linear interpolation to a sample interval of 14 days. This interpolated tidal range time series is used in the correlation with transition point elevation (e.g., see Figure 20 in section 6).

This study utilizes both the Argonaut and Vector to measure axial current velocities within Singleton Swash. The depth integrated velocity calculated internally by the
Argonaut is used as the measured current velocity at Site A, representing mean depth integrated velocity over a 15-minute interval. Current velocity ($\bar{u}$) measured by the Vector at Site C, similarly, is the mean of the measured velocities over a burst-interval (i.e., 15-minutes for the March deployment and 5-minutes for the May deployment). Statistical convergence of mean velocities was confirmed over these time periods.

Calculation of discharge is computed from cross-channel area at Site C and the current velocity as measured by the Vector. The cross-sectional area is calculated as:

$$A(t) = \int_{y_{BR}}^{y_{BL}} D(y, t) dy$$

where $y$ is the cross-channel position, the subscripts ($BR$ and $BL$) indicate the position of the right (oceanic) and left (inland) banks, respectively, and $t$ is time. Water depth, $D$, is calculated by:

$$D(y, t) = [z(y) + h(t)]$$

where $h(t)$ is the water level time series measured by the Vector. The water level measurements performed by this instrument measure the water level above the sensor, which was located at 0.83 m above the creek bed (see Figure 6 and Figure 7B). $z(y)$ is the water depth at various cross channel locations from the reference measurement recorded 28 March, 2018 (Figure 9). When water level fell below the pressure sensor ($h(t) = 0$), then $D = 0.83$ m at the location of the sensor. This assumption is presumably valid because as mentioned in section 5.1, the water level did not fall below the Vector’s transducer head and the distance between the transducer head and pressure sensor is 0.15 m; thus, any difference in water depth from 0.83 m is considered negligible.
Cross-sectional area is used to determine the flood and ebb discharge ($Q_f$ and $Q_e$), respectively:

\[ Q_f = \int_{t_L}^{t_H} A(t) \bar{u}(t) dt \]  

\[ Q_e = \int_{t_H}^{t_L} A(t) \bar{u}(t) dt \]

$Q_f$ and $Q_e$ are calculated over the rising and falling tides, respectively, and $t_H$ is time of slack high tide while $t_L$ is the time of slack low tide. Times corresponding to zero crossing current velocities are slack tides, where high slack tides are distinguished by preceding negative velocities and vice versa.

### 5.3 Reynolds Shear Stress

To characterize turbulent quantities within the swash, velocities measured by the ADV were decomposed as follows:

\[ u(t) = \bar{u} + u'(t) + \bar{u}(t) \]  

\[ w(t) = \bar{w} + w'(t) + \bar{w}(t) \]

Equations 5 and 6 denote the decomposition for the along channel ($u$) and vertical velocities ($w$) at time $t$, and $u'$ and $w'$ are the turbulent fluctuations in the along channel and vertical velocities, respectively, and $\bar{u}$ and $\bar{w}$ are wave orbital along channel and vertical velocities, respectively. Wave-induced velocities are assumed to be negligible therefore, bursts identified that were not consistent with this assumption were excluded from the data set.
Spectral analysis using a fast Fourier transform of the along channel velocity is used to identify wave velocities. Spectral peaks that indicate variability in water velocity at frequencies corresponding to wave orbital velocities (approximately 0.1 - 1 Hz) are visually identified as wave contaminated bursts. Welch’s method is used to calculate the power spectral density – each burst was first detrended then sectioned into 1024 data points (16 seconds) with 50% overlap between adjacent sections. A Hamming window was used over the sectioned data to reduce spectral leakage. The power spectral density is estimated for each section and subsequently averaged over all sections to reduce noise. An example of a burst with significant wave-induced motion is shown in Figure 8.

Equations (5) and (6) are applied after removal of burst-intervals containing evidence of wave-induced velocities, and then Reynolds shear stress is calculated as:

\[ R_s = -\rho \overline{u'w'} \]  

(7)

where \( \rho \) is the density of sea water, 1029 kg/m\(^3\), and the overbar indicates temporal average. Reynolds stress was averaged over each (non-wave contaminated) burst-interval for both deployments of the Vector.

5.4 Pearson Correlation Analysis

Pearson correlation is a statistical method used throughout this study to explore relationships between variables. This method generates a coefficient representing the strength of a linear relationship between two time series of discrete measurements. The correlation coefficient \( (R) \) is computed as a function of discrete lag:

\[ R_{p,q}(m) = \frac{\sum_{i=1}^{n-|m|} (p_i + m - \bar{p})(q_i - \bar{q})}{\sqrt{\sum_{i=1}^{n} (p_i - \bar{p})^2 \sum_{i=1}^{n} (q_i - \bar{q})^2}} \quad m \geq 0 \]
\[ R_{p,q}(-m) = \sum_{i=1+|m|}^{n}(p_{i+m} - \bar{p})(q_i - \bar{q}) \left[ \sum_{i=1}^{n}(p - \bar{p})^2 \sum_{i=1}^{n}(q - \bar{q}) \right]^{-\frac{1}{2}} \quad m < 0 \] (8)

where \( m \) is discrete lag, \( n \) is the number of discrete samples, the overbar denotes ensemble average, and the two variables being cross-correlated are \( p \) and \( q \). This technique quantifies the type of relationship (direct, inverse, or none), the strength of the relationship, and any lag between variables \( p \) and \( q \). High correlation at a positive lag indicates that variability in \( q \) precedes the variability in \( p \), whereas negative lag indicates variability in \( q \) follows variability in \( p \). The variables corresponding to \( p \) and \( q \) for this study are given in Table 1.

To assess the significance of the correlation coefficient, the 95 % confidence interval is calculated for each correlation calculation; correlation coefficients with confidence intervals that do not encompass zero are considered to denote significant relationships.
Table 1: Time series used in equation (8) for both $p$ and $q$, sampling period, location of the measurement corresponding to each variable, and the instrument used to take the measurement.

<table>
<thead>
<tr>
<th>Correlation Variable ($p$)</th>
<th>Correlation Variable ($q$)</th>
<th>Sample period (days)</th>
<th>Location of measurement ($p/q$)</th>
<th>Instruments ($p/q$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tidal Range (m)</td>
<td>Transition-point elevation (m)</td>
<td>112</td>
<td>Site B / beach-face</td>
<td>HOBO water level logger/ RTK-GPS</td>
</tr>
<tr>
<td>Tidal Range (m)</td>
<td>Tidal Range (m)</td>
<td>131</td>
<td>Site A / Site B</td>
<td>HOBO water level logger/ HOBO water level logger</td>
</tr>
<tr>
<td>Tidal Range (m)</td>
<td>Max Flood (m/s)</td>
<td>31</td>
<td>Site A / Site C</td>
<td>HOBO water level logger/ Vector</td>
</tr>
<tr>
<td>Tidal Range (m)</td>
<td>Max Ebb (m/s)</td>
<td>31</td>
<td>Site A / Site C</td>
<td>HOBO water level logger/ Vector</td>
</tr>
<tr>
<td>Tidal Range (m)</td>
<td>Flood Discharge (m$^3$)</td>
<td>31</td>
<td>Site A / Site C</td>
<td>HOBO water level logger/ Vector</td>
</tr>
<tr>
<td>Tidal Range (m)</td>
<td>Ebb Discharge (m$^3$)</td>
<td>31</td>
<td>Site C / Site C</td>
<td>HOBO water level logger/ Vector</td>
</tr>
<tr>
<td>Reynolds shear stress (N$^2$/m)</td>
<td>$\overline{u}$ (current velocity; m/s)</td>
<td>31</td>
<td>Site C / Site C</td>
<td>Vector/ Vector</td>
</tr>
<tr>
<td>Dissolved Oxygen (% saturation)</td>
<td>Reynolds shear stress (N$^2$/m)</td>
<td>31</td>
<td>Site C / Site C</td>
<td>HOBO DO logger/ Vector</td>
</tr>
<tr>
<td>Dissolved Oxygen (% saturation)</td>
<td>Flood Discharge (m$^3$)</td>
<td>31</td>
<td>Site C / Site C</td>
<td>HOBO DO logger/ Vector</td>
</tr>
<tr>
<td>Dissolved Oxygen (% saturation)</td>
<td>Ebb Discharge (m$^3$)</td>
<td>31</td>
<td>Site C / Site C</td>
<td>HOBO DO logger/ Vector</td>
</tr>
<tr>
<td>Dissolved Oxygen (% saturation)</td>
<td>$\overline{u}$ (current velocity; m/s)</td>
<td>31</td>
<td>Site C / Site C</td>
<td>HOBO DO logger/ Vector</td>
</tr>
<tr>
<td>Dissolved Oxygen (% saturation)</td>
<td>Tidal Range (m)</td>
<td>271</td>
<td>Site A / Site A</td>
<td>HOBO DO logger/ HOBO water level logger</td>
</tr>
<tr>
<td>Dissolved Oxygen (% saturation)</td>
<td>Wind Speed (m/s)</td>
<td>31</td>
<td>Site C / Apache Pier</td>
<td>HOBO DO logger/ Meterological station</td>
</tr>
</tbody>
</table>
Figure 8: Sample time series of all three velocity components on 06 May 2018 beginning at 16:00 (A), where the along channel velocity $u$ is the blue, the across channel velocity $v$ is in red, and the vertical velocity $w$ is in yellow. Power spectral density for along channel velocity, $u$ (B), exhibiting a peak at a frequency within the approximate timescales for wind waves ($\sim 1$ Hz).
**Figure 9:** Schematic of the cross-sectional measurement used to calculate discharge. $D$ is the total depth of the water column at a given position ($y$) and time ($t$) as calculated in Equation 2, where $z$ is the reference height at a given position ($y$) (measured on 28 March, 2018) and $h$ is the water level time series.
6.0 Results and Discussion

6.1 Water Quality

Dissolved oxygen concentration varies over both solar and tidal cycles observed within the swash. Tidal and solar cycle fluctuations in dissolved oxygen concentrations have a period of approximately 12 and 24 hours respectively, and spring-neap tidal cycles vary over two weeks. The light/dark cycle influences DO concentration due to net autotrophy (photosynthesis) during daylight hours, which increases DO concentration within the water column; net heterotrophy (respiration) during dark hours decreases DO concentrations. Periods of greatest DO saturation within the swash were observed when flood tides occurred simultaneously with hours of maximum sunlight. These periods of high saturation were almost always proceeded by hypoxic conditions when DO concentrations were below 30% saturation.

Figure 10 shows a sample time series of DO concentration over 15 days in May 2018, illustrating variations in DO concentrations during periods of concurrent flood/daylight and ebb/night events. On 12 and 13 May, two periods of supersaturation (over 100%) were followed by very low concentrations (<~20% saturation). On these days, high tides occurred during mid-day hours (14:00 – 15:00) and low tides occurred during night hours (00:00 – 1:00). Over the course of the time series in Figure 10, the day/night and tidal cycles become asynchronous due to the slightly different periodicities of these cycles. The greatest effect of this change is observed on 19-20 May (Figure 10) when the
range in DO concentration is the smallest. When flood tide occurs during the night, removal of oxygen due to net heterotrophy appears to be offset by a supply of oxygen from flood tide resulting in a low range in daily DO concentration fluctuations. These observations of dissolved oxygen dynamics over the day/night cycle suggests that both biological and physical mechanisms drive DO concentration variability in Singleton Swash.

Eighty percent of the time over the duration of the study, spring tides show higher mean DO concentration relative to neap tide mean DO concentration. This result suggests that larger tidal amplitudes increase the overall inventory of dissolved oxygen within the swash. During neap tides (grey shaded areas in Figure 10), a large variability of DO concentration is observed, with lower maxima concentrations and a slightly decreasing mean DO concentration over tidal cycles. These differences in trends between spring and neap tides also reflect the effects of the relative timing between tidal cycles and daily solar cycles. For example, during the neap cycle in Figure 10 (19-27 May) higher DO concentration maxima occur when daylight hours and high tide are concurrent (yellow peaks); inversely, periods when high tide and daylight have become asynchronous show lower maximum DO concentrations (purple peaks).

Delineating between biological and physical mechanisms that cause variation in dissolved oxygen concentration is beyond the scope of this study; yet based on the trends observed in Figure 10, we speculate that both are important. Biological dynamics generally act to decrease DO concentrations during dark periods and increase DO concentrations during light periods. Significant variations in DO concentration minima and maxima occur during daylight hours. A minimum DO concentration during daylight hours implies removal of DO exceeding that of the biological production. This result suggests
hydrodynamic mechanisms may play a substantial role in driving DO variations. Based on the periodicity of the variations (approximately 25 and 12 hours; verified through Fourier analysis) shown in Figure 10, tidally-driven processes within the swash likely drive DO dynamics in the swash, thus, long term DO concentration trends in the swash are likely dependent upon the resupply of dissolved oxygen to the water column (via both biology and hydrodynamics) to maintain a healthy ecosystem.

6.2 Hydrodynamics

6.2.1 Water Velocities

Figure 11A illustrates the observed current direction and magnitude ($\bar{u}$) during the May 2018 deployment of the ADV. The average flood and ebb current velocities observed throughout the deployment were 0.23 and 0.16 m/s, respectively. Average current speeds measured by the other velocity sensors are similar to those in Figure 11A. During the deployment of the 2-D ADCP (4 November - 4 December, 2017), average current velocity during flood tide was 0.26 m/s while the average ebb tide current was 0.14 m/s. Similar results were found in the March deployment of the ADV as well (data not shown). The difference between average flood and ebb tidal current velocity suggests tidal asymmetry within the swash. Average cross-channel currents ($\bar{v}$) over the May deployment were on the order of $10^{-5}$ m/s confirming the assumption of low cross channel velocities.

Tidal currents within Singleton Swash are asymmetric with higher magnitude velocities over shorter periods during flood tide. This result has been observed in similar estuarine systems (Speer and Aubrey, 1984; O’Laughlin and van Proosdij, 2014) and is attributed to friction and time-variable channel cross-sectional area within a system.
Current velocities also demonstrate a mixed semidiurnal tidal cycle. Each day consists of two ebb and two flood tides with one of each being of noticeably greater magnitude than the other. This characteristic is shown in Figure 11A by the discrepancy in consecutive peaks of both the ebb and flood tides. This same slightly asymmetric semidiurnal signal is observed in the ocean tides in this region. Figure 11B shows a Fourier transform of $\bar{u}$ for the May time series, in which spectral peaks occur at a period of approximately 12 hours (with a harmonic at 6 hours) denoting the influence of ocean tidal cycles within the swash. These data indicate that water current velocities within the swash are predominantly tidally driven.

Secondary peaks shown in Figure 11B highlight variations between flood and ebb tides within the swash and indicate asymmetric tides. Flood dominated asymmetry, such as that seen here, occurs predominantly in systems with no river input and little to no variation in channel width as a function of water level (Dronkers, 1986; Blanton et al., 2002). These conditions are observed within Singleton Swash and cause peak flood tide velocities to exceed those during ebb tide, whereas ebb tide lasts almost twice as long as flood tide. Such tidal asymmetry may be associated with secondary peaks at periods of 4 and 8.5 hours (Figure 11B) because these time periods are similar to the observed time periods of flood and ebb tides, respectively.

Fluctuations in tidal velocity are concurrent with the periodicity in dissolved oxygen concentration. Average correlation (Equation 8) between DO and $\bar{u}$ over two days using data from the May deployment shows a significant inverse relationship ($R = -0.511$, 1.75-hour lag) as illustrated in Figure 12. As flood velocities are represented by negative values, an inverse relationship implies that flood currents are strong when DO saturation
is high, whereas strong ebb currents are correlated with low DO saturation. Supporting this result is the alignment of peak DO concentrations with peak flood currents, whereas the peak ebb currents lead DO minima and slack low tide occurs simultaneously with DO minima. These relationships suggest that current velocity is an important factor in controlling DO in the swash. A lag in the correlation of DO and $\bar{u}$ of approximately 1 hour and 45 minutes indicates that variation in DO concentration occurs after a change in current velocity. Transport of DO saturated water into the swash occurs during flood tide with higher magnitude currents increasing saturation levels; inversely, ebb tides transport oxygenated waters out of the swash, transporting lower DO water from the upstream salt marsh into the swash channel. The inverse relationship is supported by both ADV deployments as well as the 2-D ADCP deployment in 2017.

6.2.2 Discharge

Discharge was calculated using data measured over 114 tides (57 flood, 57 ebb) during the May deployment of the ADV. The swash is ebb dominated with respect to discharge: despite stronger current velocity during flood tide, the prolonged ebb tidal periods discharge more water volume than flood tide injects. Over the May deployment, the average ebb discharge per tidal cycle was $4.52 \times 10^5$ m$^3$ (total ebb discharge over May was $2.58 \times 10^7$ m$^3$) while the average flood discharge per tidal cycle was $9.97 \times 10^4$ m$^3$ (total flood discharge over May was $5.68 \times 10^6$ m$^3$) (Figure 13). For the first spring tide (approximately 1 – 7 May), flood and ebb discharge are closer in magnitude, but afterward, ebb discharge greatly exceeds flood discharge by at least a factor of 3. This result indicates that the swash is not storing water (at least during this time) as other sources entering the swash (e.g., rainfall, runoff, and groundwater) likely lead to the larger water export. These
results are consistent with those from the March deployment as well, which suggests that this result is not seasonally dependent.

A possible source of error contributing to the large asymmetry in ebb and flood discharge may result from a sampling artifact. Fixed sampling height of the ADV at 0.55 m above the creek bed allows for variability in which part of the water column current velocity is being measured as water level changes. In open channel flows, maximum velocities occur within the top 25% of the water column (Cimbala and Cengel, 2008). The sample volume is persistently within this region for prolonged periods during ebb tides (approximately 8.25 hours) while it is occasionally in this region during flood tides (approximately 4 hours) (Figure 14B). This discrepancy, coupled with the assumption of velocity homogeneity over the cross-section, could skew ebb discharges to higher values and likely explains some of the difference between ebb and flood discharges shown in Figure 13. Thus, the ebb discharge may be somewhat lower, making flood and ebb discharge more similar than that shown in Figure 13.

Correlations between current velocity and DO concentrations as well as the relationship between current velocity and discharge (Equations 3 and 4), imply that there would also be a relationship between DO concentration and discharge. Dissolved oxygen concentration variations are significantly related to both the flood \( R = 0.608, 0\text{-day lag} \) and ebb \( R = -0.541, 0\text{-day lag} \) tide discharge as seen in Figure 15. These relationships suggest transport of oxygenated water into the swash during flood tides; as incoming water volume increases, the concentration of DO also increases. The opposite is true for the ebb tide: as discharge volumes during ebb tide increases, DO concentrations diminish, possibly associated with farther downstream transport of marsh waters that contain lower DO.
concentration to Site C. This effect of DO occurs immediately within the same tidal cycle (i.e., lag less than 12 hours). Following ebb tide, DO concentrations remain nearly constant until either the next flood tide or enough daylight hours lead to an increase in oxygen as a product of photosynthesis. The cyclical distribution of the correlation with lag is likely associated with the periodicity of the mixed tide.

6.2.3 Reynolds Shear Stress

Reynolds shear stress ($R_s$) was calculated over the two ADV deployments. For the May deployment, $R_s$ was calculated for every 5-minute burst. Figure 14A shows the variation in $R_s$ over the May deployment in which the mean $R_s$ was -0.025 N/m$^2$. Negative $R_s$ is generally observed during flood currents and positive $R_s$ with ebb currents, so negative mean $R_s$ indicates larger $R_s$ on flood tides. In fact, the magnitude of the mean $R_s$ for flood tides is 0.186 N/m$^2$, whereas for ebb tides, it is only 0.049 N/m$^2$. These results indicate that $R_s$ magnitude is flood dominated.

Reynolds shear stress varies similarly over the tidal cycles as the other hydrodynamic variables. The power spectral density of $R_s$ (Figure 16) illustrates that $R_s$ varies on daily (25.6 hours), tidal (6.4 and 12.85 hours), as well as the asymmetric flood (4 hours) and ebb (8.25 hours) tidal time periods. Slack periods in the swash correspond to lower $R_s$ and vice versa, thus, $R_s$ appears to vary with current speed. Variation in $R_s$ can also be caused by depth of the sampling volume. Periods of flood tide, characterized by increased water height and velocity, change boundary layer conditions (Schlichting and Gersten, 2016). These variations imply a varying location of the sample volume within the boundary layer. Figure 14B shows percent of the water column above the instrument versus
time, where percent water column above the sensor varies from 0% (the sensor was at the water surface) to ~35% (i.e., the sensor was submerged under 35% of the water column). When the instrument is deeper in the water column, $R_S$ variations are presumably dominated by changes in bottom stress, while when it is near the surface, variations in $R_S$ are less affected by bottom stress.

Increases in $R_S$ can be attributed to increased current velocities within the swash. Correlation analysis of the magnitude of $R_S$ and magnitude of $\bar{u}$ reveal a significant relationship ($R = 0.434$, 0 hour-lag) (Figure 17). This result suggests that an increase in the magnitude of $\bar{u}$ will increase the magnitude of the Reynolds shear stress within the swash; as current speed increases so will $\frac{\partial u}{\partial z}$ at the seabed, increasing the bottom stress and consequently the Reynolds stress above the seabed (Marusic et al., 2010).

Interactions between surface and bottom boundary layers within the swash create a complex relationship between the hydrodynamic variables and $R_S$. An increase in wind speed can change the stress on the surface and cause increases in $R_S$ beyond the aforementioned relationship between current speed and bottom stress. The influence of either the surface or bottom boundary layer upon $R_S$ is greatly affected by the position of the sensor in the water column. To examine the influence of wind speed on $R_S$ (effects from surface boundary layer), correlation analysis was performed. Statistical measures indicate that no significant relationship exists, suggesting that changes in $R_S$ are primarily related to the variability of tidal currents within the swash also indicating that bottom stress is most likely the primary influence on $R_S$ for the measurement position in this study.
DO concentration and wind speed time series were correlated to further investigate the effects of wind within the swash. The results (Figure 18) suggest that wind speed impacts variations in dissolved oxygen concentration within the channel ($R = 0.277, 0.042$ day-lag). This relationship challenges the result previously discussed that changes in $R_S$ are not related to wind speeds. An increase in dissolved oxygen concentration within the water column concurrent with increased wind speeds suggests that oxygen is being injected from the atmosphere across the air-water boundary as a result of processes associated with wind shear. Thus, wind changes the surface boundary layer due to increased wind stress, promoting a response in the Reynolds shear stress within the water column. The correlation between DO concentration and wind speed despite no correlation between $R_S$ and wind speed indicates that surface layer mixing processes occurred closer to the water surface than the ADV measured on average and/or that the variations in $R_S$ due to wind stress are negligible compared to those associated with the current speed. This interpretation is also supported by the lag of 0.042 days (1-hour) observed in the correlation of DO concentration and wind speed. Shear in the water column driven by wind stress would cause a more immediate response in dissolved oxygen concentration if the concentration was being measured at the same location as the enhanced mixing.

The role of $R_S$ in influencing DO concentration variability within Singleton Swash is not negligible. The relationship between $R_S$ and DO concentration (Figure 19) is significant ($R = 0.276, 0$ hour-lag), suggesting that Reynolds stress does influence DO concentrations within the swash. Assuming that measurements by the ADV were performed within the bottom boundary layer, this relationship also suggests that boundary layer mixing redistributes DO throughout the water column (although the vertical extent of
mixing within the water column cannot be determined from the data). The lower correlation coefficient between DO concentration and $R_5$ compared to that between DO and discharge suggests that the transport of oxygen into the swash plays a slightly larger role in DO concentration variability than does the redistribution of DO throughout the water column.

6.2.4 Tidal Range

Water level also reveals spring-neap cyclicity as well as a mixed semidiurnal tidal regime. Tidal range was calculated from the water level time series as discussed in Section 5.2. Periods of large tidal range indicate spring tides while periods of low tidal range indicate neap tides at Site B (Figure 20). Tidal range fluctuates between 0.12 m and 0.881 m (Figure 20).

Tidal range for Singleton Swash was calculated at Sites A, B, and C with similar ranges at all sites, suggesting that tidal wave propagation through the swash is spatially uniform. Figure 21 illustrates the correlation of tidal range between Sites A and B using water level data collected from 1 September, 2017 to 1 January, 2018, which shows consistent variation in tidal ranges throughout the swash ($R = 0.987$, 0 day-lag).

Tidal range has been linked to morphological and salinity gradients within Singleton Swash (Hoffnagle, 2015). Results discussed in this study (i.e., current velocities, discharge, Reynolds shear stress) reveal distinct tidal signatures. Water level measurements have been conducted over a much longer time period than the two deployments of the ADV in this study; thus, if the characteristics investigated with the ADV can be linked to tidal range, the findings on mechanisms of transport and mixing previously discussed can be extrapolated to longer time periods of swash monitoring. To explore these connections,
tidal range was correlated to each of the aforementioned physical processes through correlation analysis (Table 2).

A significant relationship between tidal range and both maximum flood ($R = 0.927$, 0 day-lag) and ebb ($R = 0.842$, 0 day-lag) tidal velocities was found (Figure 22). This result suggests that as tidal range increases, maximum current velocities increase as well. This relationship also suggests that tidal range can be used as a relative indicator of current velocity magnitude within the swash. The relationship between current velocity and discharge (Equations 3 and 4) implies that tidal range will also vary with discharge on flood ($R = 0.754$, 0 hour-lag) and ebb ($R = 0.977$, 0 hour-lag) tides. This relationship, seen in Figure 23, suggests tidal range is a suitable proxy for tidal flushing of the swash. Correlation of tidal range and Reynolds shear stress revealed no statistically significant relationship, although it is plausible that $R_S$ increases during higher tidal range due to the relationships described above between tidal range and current velocity and the prior finding that increased current velocity correlates with increased $R_S$.

The relationship describing tidal range as a possible indication of higher velocities within the swash can also be used to describe DO concentration. The significant relationship between current velocity and dissolved oxygen links tidal range to DO concentration as well. Correlation analysis of DO concentration and tidal range over 14 days using data from September 2017 through June 2018 (Figure 23) reveals a significant relationship ($R = 0.277$, 1.5 day-lag). DO concentration was averaged over 12-hours, down-sampling the data to match the tidal range sampling interval. Average correlation of this DO concentration time series and tidal range is computed over 14 days and captures variations in dissolved oxygen and tidal range occurring over timescales greater than one
tidal cycle. The direct relationship suggests that as tidal range increases, dissolved oxygen concentration also increases. A lag of 1.5 days between tidal range and dissolved oxygen concentrations indicates that an increase in tidal range precedes an increase in DO concentration. This result suggests higher tidal ranges promote higher DO concentrations within the system over time.

6.3 Geomorphology

The shape of the beach-face at Singleton Swash varies over weekly to monthly timescales due to variations in sediment transport (Hoffnagle, 2015). Elevation of the transition point decreases June through August then increases until the conclusion of the surveys in October (Figure 20). Correlation analysis revealed no statistically significant relationship between transition point elevation and tidal range for this period of time. Contrary to these results, a strong inverse relationship between transition point elevation and tidal range ($R = -0.647$) has been observed in Singleton Swash by Hoffnagle (2015); however, this previous study measured transition point elevations at higher resolution over a longer period of time. The author also linked increases in beach-face elevation to sedimentation events in the swash (Hoffnagle, 2015) most likely driven by littoral drift, the dominant sediment transport process in Long Bay (Barnhardt, 2009).

6.4 Implications

Variabilities in current velocity, Reynolds shear stress, and discharge were observed over flood and ebb tidal stages in Singleton Swash. These variabilities suggest that ocean tidal forcing exerts a large influence on these physical processes within the Singleton Swash system. Variations in dissolved oxygen concentrations over similar cycles/timescales (Figure 10) also suggest significant variations due to tidal forcing.
Effects from the day/night cycle were also observed in relation to DO concentration variations suggesting biological processes also significantly contribute to DO concentration variability in the system. Variations in current velocity were found to occur 1.75 hours before dissolved oxygen variations, where the variations were found to be inversely related (recall flood currents are negative). This result suggests that flood tides increase DO concentrations whereas DO concentration decreases during ebb tides. The same relationships occur between flood and ebb discharge with dissolved oxygen, suggesting that highly oxygenated ocean water is transported into the swash during flood tides and transported out of the swash during ebb tides. The tides were found to be mixed and asymmetric, with larger magnitude currents during flood tide but for shorter duration of time. An observation of higher magnitudes of Reynolds shear stress over flood tides suggests that more vertical mixing within the water column occurs during periods of higher water velocities, which were observed to occur during flood tides within the swash. The relationship between DO concentration and Reynolds stress was found to be weaker than that between DO concentration and discharge suggesting that vertical mixing is of secondary importance.

The relationship between DO concentration and current velocity, as well as the relationship between current velocity and tidal range indicate that faster flood currents in the swash increase DO concentrations relative to mean conditions. A positive feedback mechanism impacts this behavior as well because Reynolds shear stress was also found to increase with stronger current velocities, indicating enhanced vertical mixing. In contrast, stronger ebb tide velocities result in larger ebb discharge, which was found to decrease DO concentrations. Furthermore, the system was found to be dominated by ebb discharge,
although part of this discrepancy may be attributable to differences in sensor positioning within the water column during ebb and flood currents. Combined, these results suggest that management of flow conditions would be beneficial to retain higher DO concentrations within the swash over a tidal cycle. It was also found here and in Hoffnagle (2015) that increases in the transition point elevation result in a decrease in tidal range in the swash, which results in lower current speeds. This result suggests that ensuring low transition point elevations would maintain high DO concentrations in the swash. The direct relationship between DO concentration and tidal range over the course of the study also indicates that the facilitation of increased tidal ranges can improve water quality within the system.
Figure 10: Sample time series of percent (%) saturation of dissolved oxygen at Site C over a period of 15 days. Daylight hours (x) begin at 06:00 and end at 20:00 and non-daylight hours (x) range from 20:00 to 05:00. Regions of light blue denote times during spring tide while the grey denotes periods of neap tide.
Figure 11: (A) Time series of along-channel current velocity ($\bar{u}$) measured by the ADV during the May, 2018 deployment. (B) Power spectral density (PSD) of current velocity ($\bar{u}$) from the May deployment of the ADV. The first major spectral peaks (circle) occur at 12.5 hours with a harmonic occurring at a period of 6.25 hours (circle). Secondary spectral peaks (circle) correspond to periods of 25.6, 8.25, and 4.13 hours.
**Figure 12:** Average correlation coefficient between DO concentration and current velocity ($\bar{u}$) versus lag time over 48 hours, 95% confidence intervals are shown in red; peak correlation is $R = -0.511$ at 1.75 hour-lag.
Figure 13: Time series of both the flood (diamond) and ebb (square) discharge ($Q$, Equation 3 and 4) per tidal cycle computed from the data collected during the May deployment of the ADV.
**Figure 14:** (A) Time series of Reynolds shear stress ($R_S$) calculated from velocity measurements recorded by the ADV over the May deployment. (B) Percent (%) water column above ADV (blue), in which times of high tide are denoted by $x$ and times of low tide are denoted by $x$. 
Figure 15: Average correlation between dissolved oxygen and flood (A) and ebb (B) discharge over ten-day periods (Equation 4 and 5). 95% confidence bounds are shown in red; flood peak $R = 0.608$ at 0 day-lag and peak ebb $R = -0.541$ at 0 day-lag.
Figure 16: Power spectral density (x) of Reynolds shear stress ($R_s$). Two dominant peaks correspond to 12 and 6-hour time scales.
Figure 17: Average correlation of Reynolds shear stress ($R_s$) and current velocity ($\bar{u}$) magnitudes over 12 hours for the May deployment, 95% confidence bounds are shown in red; peak correlation is $R = 0.434$, 0 hour-lag.
Figure 18: Average correlation of dissolved oxygen concentration and wind speed over 14 days for the May deployment, 95% confidence bounds are shown in red; peak $R=0.274$ at 0.042 day-lag.
Figure 19: Average correlation of Reynolds shear stress magnitude ($R_s$) and dissolved oxygen concentration over 12 hours for the May 2018 deployment, 95% confidence bounds are shown in red; peak correlation is $R=0.276$, 0 hour-lag.
Figure 20: Time series of tidal range (blue; left axis) in Singleton Swash at Site B and elevation of the transition point (orange; right axis) on the beach-face. The black line denotes the low pass filtered and resampled tidal range time series used in the correlation analysis between tidal range and transition point elevation.
Figure 21: Average correlation of tidal range between Sites A and B over 14 days using data from 1 September, 2017 to 1 January, 2018, 95 % confidence bounds shown in red; peak correlation is $R = 0.981$, 0 day-lag.
Figure 22: Average correlation of tidal range and maximum flood (A) and ebb (B) tide current velocity magnitude over 24 hours for the May deployment of the ADV, 95% confidence bounds shown in red; flood peak correlation is $R = 0.927$, 0 hour-lag and ebb peak correlation is $R = 0.842$, 0 hour-lag.
Figure 23: Average correlation of tidal range and flood (A) and ebb (B) tide discharge ($Q$) over the May deployment of the ADV, 95% confidence bounds shown in red; flood peak correlation is $R = 0.754$, 0 day-lag and ebb peak correlation is $R = 0.977$, 0 day-lag.
Figure 24: Average correlation of tidal range and dissolved oxygen over 14 days using data from 1 September, 2017 – 1 June, 2018, 95% confidence bounds shown in red; Peak correlation is $R = 0.277$, 1.5 day-lag.
Table 2: Time series used in equation (8) for both \( p \) and \( q \) with the corresponding maximum correlation coefficients, associated upper and lower 95% confidence intervals (CI), lag (\( m \)), correlation resolution (\( C_r \)), and correlation interval (\( C_i \)).

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<th>( p )</th>
<th>( q )</th>
<th>( R )</th>
<th>95% Lower CI</th>
<th>95% Upper CI</th>
<th>( m ) (hours)</th>
<th>( C_r ) (hours)</th>
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7.0 Summary and Conclusions

Singleton Swash is a tidal creek located in Myrtle Beach, SC that flows 2 km inland from the beach-face to a stormwater pond. The swash functions as a recreational area, a conduit for runoff from surrounding infrastructure, and an ecosystem for a diverse population of species. Migration of the beach-face channel as a result of littoral drift inhibits exchange between the ocean and inland creek, which may be associated with degraded water quality observed in the system. Investigation of the interactions between beach-face morphology, tidal creek hydrodynamics, and DO dynamics in Singleton Swash may help inform mitigation strategies for related ecological and economic problems.

The primary goal of this study is to delineate the physical mechanisms influencing water quality. Field observations of water velocity, wind speed, water level, and dissolved oxygen concentration occurred from May 2017 to June 2018. Water level and dissolved oxygen concentration were continuously monitored over the course of the study, while beach-face elevation was surveyed from May through October 2017, and water velocities were primarily measured during deployments in March (14 days) and May (31 days) 2018. The latter observations were used to estimate discharge and Reynolds shear stress, an indicator of vertical mixing.

Results show that larger tidal range yields larger mean DO concentrations within the swash. Flood tides within the swash occur over shorter duration with larger currents as opposed to ebb tides, which occur over longer periods with weaker currents. Higher tidal
currents during flood tide generate larger Reynolds shear stress indicating greater mixing within the system. Flood tides import and mix oxygenated waters from the ocean throughout the system. Conversely, ebb tides, which transport oxygenated waters out of the swash and bring water with lower DO concentrations from the stormwater pond closer to the channel mouth, are associated with periods of lower dissolved oxygen concentration. Larger tidal ranges cause stronger flood currents that transport higher volumes of oxygenated water into the swash and cause more mixing of DO throughout the system. This process overcomes the transport of water with lower DO concentrations from upstream on ebb tide resulting in a mean increase in DO over an entire tidal cycle. Consistent with these observations, smaller tidal ranges generate less transport into the swash coupled with less mixing, causing more retention of water in the swash (i.e., less exchange of waters). The lack of mixing and low transport of water from the ocean may also enable increased water temperatures in the swash. These effects facilitate biological factors to potentially become more significant and lower the mean DO concentration within the system, possibly due to promoting net heterotrophic conditions.

To facilitate larger tidal ranges within the swash, maintenance of low beach-face elevation is required. Both this study and Hoffnagle (2015) indicate that beach-face elevation exerts a primary control on the tidal range observed within Singleton Swash. This result confirms that maintaining lower beach-face elevation within the swash channel will lead to larger tidal ranges, which generate greater mean DO concentrations and therefore promote a healthier ecosystem. Future studies should examine the asymmetry of the tidal cycle and its relationship to morphology over a longer period than this study to aid in promoting a healthy ecosystem.
8.0 Works Cited


USACE (2009). Singleton Swash Planning Assistance to States (PAS) Study Horry Country, South Carolina(Rep.). Charleston, SC.
