ABSTRACT

KURKI-FOX, JALMAR JACKSON. Characterization and Evaluation of the Condition and Long-Term Trajectory of Forested Wetlands in North Carolina: Implications for Management and Restoration. (Under the direction of Dr. Michael R. Burchell II).

There is a general recognition of the valuable ecosystem services wetlands provide including water quality improvement, flood mitigation, and nutrient cycling. However, there is a lack of information available regarding the structural and functional characteristics of natural wetlands and trends in the condition of wetlands in North Carolina. This project was an effort to narrow the knowledge gaps and improve wetland management, protection, and restoration. Specifically, regarding characterizing the hydrology of several different wetland types, defining background conditions for natural wetland water quality, assessing the level of metals in wetland soils, and modeling the impacts of climate change on wetlands in North Carolina. For this project, 16 natural wetland sites across the Piedmont and Coastal Plain of North Carolina were monitored for hydrology, soils and water quality for three years. The sites consisted of basin, headwater, and riverine wetlands. These data were combined with long-term data collected at these same sites for various periods from 2005 to 2013.

The wetland hydropattern is the most important factor impacting wetland structure and function, but it is rarely adequately addressed in wetland restoration projects. The wetland hydropatterns were analyzed, and general patterns were identified that characterized the different wetland types. The observations were compared to the standards for wetland mitigation projects. The observed durations generally far exceeded the criteria thresholds, and long-term simulations showed a wide range of saturation durations.

Water quality standards for wetlands were proposed by US EPA close to three decades ago; however, little progress has been made in implementing water quality criteria. Wetland water quality results were compared to background levels for streams, and draft nutrient

thresholds for reference conditions were developed. These thresholds were calculated using a method which combined a US EPA percentile-based approach (90th percentile) and bootstrap confidence intervals. The thresholds were calculated as 0.07 mg/L for NO_3^--N, 0.15 mg/L for NH_4^+-N, 2.9 mg/L for ON, 3.1 mg/L for TN, and 0.3 mg/L for TP. In general, background levels for streams were lower for TN, NH_4^+-N, and TP but higher for NO_3^--N compared to wetlands.

Soils samples were collected to determine the level of soil organic matter (SOM) and assess the potential risk to the wetland biota from Zn and Cu in the soils. SOM levels in the soil were variable, ranging from less than 5% in some Piedmont sites to over 50% in some Coastal Plain sites. Mean plant available metal concentrations were similar to previously reported levels for wetlands but were much lower than average levels for upland soils. Mean concentrations were around 1 mg/kg for Cu and 5 mg/kg for Zn. Based on these results, mean background levels for identification of disturbance could be established at 2 mg/kg for Cu and 7 mg/kg for Zn. The impacts of land use were evident on the concentration of metals; absent a direct anthropogenic source, the levels of Zn and Cu in wetland soils likely pose little danger to the biota of most North Carolina wetlands.

Calibrated and validated DRAINMOD models and downscaled climate projections were used to assess the long-term impacts of climate change on the hydrology of Coastal Plain wetlands. Nine models were selected from an ensemble of climate models to assess the range of possible impacts. Mean water levels were predicted to decline from 0-64 cm by the middle of this century and 25-84 cm by the end of this century, depending on the changes in precipitation and temperature. Water table declines were generally largest during the growing season. Overall, this study showed the impacts of climate change on the hydrologic regimes of NC wetlands could be severe.

Characterization and Evaluation of the Condition and Long-Term Trajectory of Forested Wetlands in North Carolina: Implications for Management and Restoration

by
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DEDICATION

To Bertha M. Kurki

A woman that had a profound impact on my life and always selflessly exemplified the following quote:

"I expect to pass through this world but once. If, therefore, there be any kindness I can show, or any good thing I can do any fellow human being let me do it now. Let me not defer or neglect it, for I will not pass this way again." ~ attributed to several Quaker historical figures

BIOGRAPHY

Jalmar "Jack" Kurki-Fox was born near Ashland, Wisconsin in December of 1988. He spent his formative years in Wisconsin, before moving to Gainesville, FL in the late 1990s. He eventually attended the University of Florida where he graduated Summa Cum Laude with a B.S. in Civil Engineering in 2012. He received a M.E. in Civil Engineering with a concentration in Water Resources from UF a year later. During his time at UF, he worked on a research project modeling the impacts of sea level rise on coastal aquifers. After receiving his Masters, he worked for a water resources engineering consulting firm in Tampa, FL for about two years. Jack enrolled at NC State in January of 2015. After graduation, he hopes to eventually pursue a career in water resources and ecosystem research and management in the state or federal government.

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Introduction

Forested wetlands have been disappearing at a faster rate than any other wetland type (Dahl, 2011). This is concerning because forested wetlands provide many valuable ecosystem services to society, including water quality improvement, flood mitigation, wildlife habitat, timber production, carbon sequestration, and recreation (Mitsch and Gosselink, 2007; Walbridge, 1993), and some less recognized benefits such as cultural, aesthetic, and educational values (Mitsch et al., 2015). In North Carolina, approximately 70 percent of the rare and endangered plant and animal species are dependent on wetlands (Fretwell et al., 1996). Forested wetlands are particularly important for water quality improvement for watersheds in the southeastern United States (Walbridge, 1993); while southeastern states only account for one quarter of the land area in the conterminous United States, they contain over 65% of the total forested wetland area (Ainslie, 2002). The latest *Status and Trends of Wetlands in the Conterminous United States* report from the U.S. Fish and Wildlife Service (FWS) indicated forested wetlands lost more area than any other wetland type from 2004 to 2009 (Dahl and Stedman, 2013). This translates into a loss of ecosystem services and the associated economic benefits (e.g. De Groot et al 2013) for the Southeast. While increased restoration has the potential to buffer some of the losses to forested wetlands, the ability of restored wetlands to achieve functional equivalence with natural ecosystems has not been demonstrated; thus, even with increased restoration efforts, the long-term impact continues to be a loss of wetland function and biodiversity (Dahl, 2011). Therefore, efforts are needed to better protect high quality natural wetlands, expand the understanding of the current condition and functions of wetlands to inform wetland management, improve the

outcomes of wetland restoration projects, and assess the possible long-term trajectory of the remaining wetlands in North Carolina.

Large-Scale Wetland Monitoring and Research in North Carolina

The North Carolina Department of Environmental Quality Division of Water Resources (NC DEQ DWR) has a history of monitoring wetlands in North Carolina dating back to 2004. Their activity was primary funded through multiple U.S. Environmental Protection Agency (U.S. EPA) Wetland Program Development Grants. These projects included monitoring of hydrology, soils, water quality, macroinvertebrates, and amphibians. The primary objectives of these projects were to assess the condition and trends of natural wetlands in North Carolina. These efforts continued until 2013 when the wetland monitoring program was eliminated. When the program ended, the NC State University Department of Biological and Agricultural Engineering stepped in to continue the important work of monitoring wetlands in North Carolina. The objectives of the new project were to add three years of data to the existing long-term dataset that was developed by NC DEQ and use the combined dataset to assess trends in wetland conditions, identify possible stressors, and develop guidelines and metrics to evaluate wetland restoration projects.

Wetland Trends

Forested Wetlands in the Southeast

Over half of the wetland area in the conterminous United States has disappeared since pre-European settlement (Dahl, 1990). Losses have been most severe in the southeastern United States (Hefner, 1994) and coastal watersheds in particular (Dahl and Stedman, 2013). The southeastern United States contain just under half of all wetlands and two thirds of forested wetlands (Ainslie, 2002). There are about 5.7 million acres of wetlands remaining in North

Carolina (Fretwell et al., 1996), 68% of which are forested wetlands (Ainslie, 2002). The Southeast has also suffered some of the greatest losses in wetland area, as 85 percent of wetland losses from 1950 to 1970 (Hefner and Brown, 1984) and 90 percent of the losses from the mid-1970s to the mid-1980s (Hefner, 1994) occurred in the 10 southeastern states. Wetland losses in North Carolina have largely reflected the nation overall as about 44 percent of historical wetlands in the state have been lost (Dahl, 1990). Wetland losses in North Carolina were particularly severe from the mid-1970s to the mid-1980s, a decade in which 1.2 million acres were lost (Hefner, 1994). Since that time, losses have slowed substantially; however, Coastal Plain wetlands, accounting for 95 percent of North Carolina wetlands, still lost about two percent of their area between 1994 and 2001 (Carle, 2011).

Nationwide, wetland losses have also slowed considerably as a result of federal initiatives, and there was even a modest gain of net wetland area between 1998 and 2004 (Dahl, 2005). However, the most recent *Status and Trends of Wetlands in the Conterminous United States* report from 2004 to 2009 again showed losses outpacing gains with forested wetlands suffering the largest losses in terms of acreage (Dahl, 2011). Furthermore, coastal watersheds continue to lose wetland area at a rate far exceeding the nation as a whole. From 2004 to 2009 coastal watersheds lost over 360,000 acres of wetlands, which is more than six times the loss rate for the conterminous United States as a whole. The losses were concentrated in the Atlantic and Gulf Coast states (Dahl and Stedman, 2013). This is of particular concern given coastal watersheds contain about 38% of wetlands while only accounting for 13% of the land area in the conterminous United States (Dahl and Stedman, 2013). A significant percentage of this area lies on the Coastal Plain of the southeastern United States.

Causes of Wetland Loss

Causes of wetland loss in the Southeast vary, but losses have primary been driven by intensive drainage for agricultural use as clearing and drainage were largely encouraged and incentivized by federal government policy in the 19th and most of the 20th centuries (Mitsch and Gosselink, 2007). Some of the largest losses occurred in the period after World War II until the passage of Swampbuster in 1985 (Ainslie, 2002). More recently losses can be attributed to silviculture in forested wetlands, urbanization, and other development (Dahl, 2011). However, the relatively modest estimates of the change in net area over the last two decades (e.g. about 60,000 acres lost from 2004-2009) have masked what appears to be more serious losses in wetland function. The Army Corps of Engineers (USACE) defines areas that are "inundated either permanently or periodically at mean water depths ≤ 6.6 ft" to be wetlands (USACE, 1987). As a result, the creation of ponds is tallied as a gain in wetland area by the FWS (Dahl, 2011). This results in concentrated losses of some wetland types and replacement with a set of homogeneous wetlands that do not support functions at the same level as the lost wetland ecosystems. Additionally, hundreds of thousands of acres of wetland have been reclassified as other wetland types (e.g. forested wetland to shrub wetland) because of disturbance and alteration (Dahl, 2005, 2011). While this conversion to other wetland types does not result in a net loss of overall wetland area, it can still result in decreased ecosystem function (Whigham, 1999).

Overall, wetland losses have slowed considerably, but wetland function is still declining, and will remain an issue as population continues to grow and development competes with wetlands for land area. Population growth will stress water resources and poses the risk of wetland water sources being diverted to other needs (Burkett and Kusler, 2000), such as

agriculture and consumptive use. In already stressed coastal watersheds, where population growth has nearly tripled the national rate over the last four decades (NOAA, 2013) the impacts could be the most severe.

Wetland Regulations

Currently, there is no single federal law that governs overall wetland protection (Mitsch and Gosselink, 2007). Instead, a number of different laws are applied by several agencies including the US EPA, USACE, and the U.S. Department of Agriculture (USDA). This often contributes to legal conflicts over the extent of wetland protections (Copeland, 2013). Until the 1970s, the official policy of the U.S. federal government was largely to encourage wetland drainage, and most of the wetland loss that occurred in the southeastern U.S. was federally subsidized (Mitsch and Gosselink, 2007). With the passage of the Clean Water Act (CWA) and subsequent amendments, the protection of wetlands was codified into law at the federal level. In 1985, Congress passed Swampbuster (a provision of the Food Securities Act of 1985), which reduced the loss of wetlands on agricultural lands by making farmers ineligible for USDA subsidies if they drained, filled, or otherwise altered wetlands for agricultural use. However, this has not halted loss of wetlands on agricultural lands completely as a majority of farmers do not receive subsidies (Copeland, 2013).

Further recognizing the important ecosystem services wetlands provide to society, the US EPA, in conjunction with other federal agencies, established a policy of "No-Net-Loss" of the nation's wetland functions in 1989. This was prompted by a report from the Conservation Foundation's National Wetland Policy Forum, which advocated for achieving a no-net-loss of wetland area and function, as well as, long-term gains in both (Mitsch and Gosselink, 2007). The policy was instituted by a memorandum of agreement between the US EPA and USACE (US

EPA, 1990). The goal of achieving "No Net Loss" is primarily facilitated by the USACE through the enforcement of Section 404 of the CWA. Section 404 requires that permits be obtained for dredging or filling "Waters of the United States" (Mitsch and Gosselink, 2007). "Waters of the United States" currently refers to waters that are used for interstate commerce or adjacent to these waters and their tributaries (US EPA, 2018). This definition includes many wetlands and was expanded to include isolated wetlands and small streams in 2015; however, this change was never implemented and is now under revision (US EPA, 2018). Despite these protections, wetlands can still be impacted as some activities such as silviculture are exempt from Section 404 permits (Copeland, 2013).

The USACE permits usually require compensatory mitigation for wetlands that are impacted or lost due to permitted projects. Compensatory mitigation comprises the creation, restoration, enhancement, or preservation of wetlands required by developers to offset the loss of wetland functions due to development projects (National Research Council, 2001). In 2008, the US EPA and USACE released Mitigation for Losses of Aquatic Resources; Final Rule, which sets minimum standards for mitigation projects planning and monitoring and included specific requirements for mitigation plans (Copeland, 2013). In addition, the 2008 rule promotes the use of ecologically based performance standards for wetland mitigation projects (Federal Register, 2008).

In North Carolina, NC DEQ Division of Mitigation Services (DMS) is charged with handling all compensatory mitigation permitting in conjunction with the USACE. The North Carolina Interagency Review Team (NCIRT) develops and publishes mitigation performance standards through the USACE regional office in Wilmington, NC. The NCIRT is made up of representatives from the USACE, the U.S. Fish and Wildlife Service, US EPA, National Marine

Fisheries Service, NC DEQ DWR, NC Wildlife Resources Commission, NC Division of Coastal Management, and the NC State Historic Preservation Office. The latest version of the NCIRT standards for streams and wetland mitigation projects was issued in 2016. These guidelines were intended to raise mitigation monitoring standards to those set in the 2008 Compensatory Mitigation for Loss of Aquatic Resources; Final Rule (USACE, 2016).

Wetland Mitigation Outcomes

Although the original objective of "No-Net-Loss" was to maintain and expand wetland area and function, measurement of failure and success has for the most part, followed an area only accounting approach. According to the last two *Status and Trends of Wetlands in the Conterminous United States* report estimates, there was a net gain of wetland area from 1998 to 2009 (Dahl, 2005, 2011). While wetland losses have undoubtedly slowed, this finding must be examined in the context of the overall trends of wetland loss. A gain of tens of thousands of acres is trivial considering the millions of acres lost over the last century (Mitsch and Gosselink, 2007). Also, if the creation of urban, industrial, and agricultural ponds is discounted from the analysis, the result is a net loss of several hundred thousand acres of wetland area over the same period (Dahl, 2005, 2011). Additionally, out-of-kind replacement of wetlands appears to be a common practice; easier to create or restore wetland types are favored for mitigation in place of the more complex wetlands that were destroyed (Dahl, 2011). However, the ability of marshes or ponds to perform the same functions as other wetland types at the same level has not been established (Dahl, 2011). Whigham (1999) contends that with current practices "No-Net-Loss" will not be achieved and continued loss of wetlands will lead to a decrease in biodiversity. Many others have come to the same conclusion wetland mitigation is not replacing lost functions or recreating the lost ecosystems (e.g. Moreno-Mateos et al., 2012). The National Research

Council's (NRC) Committee on Mitigating Wetland Losses concluded in 2001 that the goals of "No-Net-Loss" were not being achieved (National Research Council, 2001). However, there has been little progress on improving restoration practices or developing better design guidelines (Brooks and Gebo, 2013). As a result, natural wetlands are continually destroyed for development based on the long-standing, misguided assumption that wetland functions can be easily recreated in restoration projects (Whigham, 1999). However, this has been proven unlikely in the short term.

Mitigation Outcomes in North Carolina

Aside from comparing soil properties between restored and natural wetlands (Bruland et al., 2006; Bruland and Richardson, 2006), published studies comparing natural wetlands to mitigation wetlands in North Carolina are scarce. One of the few published studies on mitigation projects recently evaluated the regulatory success of mitigation projects from 2007 to 2009. The results showed 75% of compensatory mitigation wetland projects in North Carolina were considered regulatory successes (Hill et al., 2013). However, the success rate was closer to two thirds when preservation and enhancement were removed from the analysis. The reported success rate was much higher than a previous study regarding mitigation compliance in North Carolina that reported a failure rate above 50% (Pfeifer and Kaiser, 1995), and better than many studies from across the nation (e.g. National Research Council, 2001). The authors attributed observed failures to inadequate soil properties and topography that prevented the establishment of the target hydrologic regime for the desired ecosystem type (Hill et al., 2013). However, this study did not give an indication of the functional performance or comparison to natural wetlands. It is important to recognize regulatory success is not synonymous with functional equivalence, as

regulatory targets are often not based on restoring wetland function (Matthews and Endress, 2008).

Hydrology and Wetland Restoration

Hydrology is the primary factor influencing the development, evolution, structure, function, and persistence of wetland ecosystems (Fretwell et al., 1996). Hydrology governs the composition and structure of the vegetation, species and persistence of fauna, development of hydric soils, accumulation of organic matter, sediment transport, and nutrient cycling in wetland environments (Mitsch and Gosselink, 2007).

Wetland hydrology is the combination of the hydroperiod or hydropattern, hydraulics, and dominant water source for the wetland (Reddy and DeLaune, 2008). The wetland hydropattern is the pattern of the water level fluctuations though time (US EPA, 2008a) or the hydrologic signature of the wetland (Mitsch and Gosselink, 2007). The terms hydropattern and hydroperiod are often used interchangeably by different authors. In this study, hydropattern will be used to refer to the seasonal pattern of water level fluctuations. Hydropatterns can vary considerably by wetland type. For example, hydropatterns can range from riverine wetlands where the primary water source is overbank flooding and inundation patterns that are dependent on extreme events, to Carolina Bay wetlands where the soil can be inundated for more than 300 days, and extreme fluctuations are rare (Caldwell et al., 2011). The hydropattern is directly tied to many of the habitat services the wetlands provide, such as amphibian habitat (Babbitt et al., 2003) and macroinvertebrate habitat (Brooks, 2000). As a result, hydrology is often considered one of the most important factor in the design of wetland restoration projects (Kentula and Kusler, 1989), and one of primary reasons some restoration projects fail to meet their objectives (National Research Council, 2001).

Many studies point to an inability to recreate natural hydrology as one of the primary factors in mitigation wetlands differing from their natural counterparts (e.g. National Research Council, 2001; Brooks et al., 2005; Hoeltje and Cole, 2007; Brooks and Gebo, 2013). In some cases, mitigation wetlands were wetter than natural wetlands and exhibited static, higher water levels that did not display the characteristic fluctuations of natural systems (e.g. Shaffer et al., 1999; Cole and Brooks, 2000a; National Research Council, 2001; Cole et al., 2006). This is likely because most designs are geared towards exceeding jurisdictional hydrology requirements and thus satisfying permit compliance (Bledsoe and Shear, 2000; Cole et al., 2006; Pfeifer and Kaiser, 1995; Ruiz-Jaen and Aide, 2005).

For the past 30 years, wetland mitigation performance standards for hydrology have been based on the USACE definition of wetland hydrology. However, the USACE requirements do not differentiate between different wetland types (i.e. saturation periods are not wetland specific) (Bledsoe and Shear, 2000; Johnson et al., 2012); the standards only indicate if a *jurisdictional* wetland exists (Johnson et al., 2014). Some designs based on exceeding jurisdictional hydrology fail to allow for wetlands to dry out (Cole and Brooks, 2000a; National Research Council, 2001), and result in large areas of open water that are not representative of natural wetlands of the same class (Cole, 2017; Shaffer et al., 1999).

The USACE definition of wetland hydrology requires the water table be within 30 cm or above the soil surface for a continuous period of 14 days during the growing season in most years (5 of 10 years) (USACE, 2012). Historically, targets for this period of continuous saturation have ranged from 5-12.5% of the growing season for mitigation projects in North Carolina (Hill et al., 2013). The monitoring period was increased from 5 to 7 years in 2013 (NCIRT, 2013) after being set at 5 years since the late 1990s (Hill et al., 2013). In 2016, the

NCIRT released new performance criteria for wetland mitigation projects. The new criteria set periods of continuous saturation that must be achieved based on the soil series underlying the restoration site. The NCIRT monitoring requirements also allow for mitigation project designers to use data from reference sites or hydrologic models if they believe the given hydrology standard is not appropriate for their restoration site. However, the reference data cannot be used as a replacement for a hydrology standard, only as supporting evidence (NCIRT, 2016).

These new guidelines from NCIRT may be considered an improvement in that a given soil series generally occurs in similar landscape positions (Richardson and Vepraskas, 2001), and there are different requirements for different ecoregions (i.e. Coastal Plain, Mountains, Piedmont) (NCIRT, 2016). However, while the NC DEQ DMS restoration program is serving its function, there is a desire to move more towards the recommendations in the Federal Compensatory Mitigation for Losses of Aquatic Resources; Final Rule, which advocates for the use of performance standards that may create higher functioning mitigation projects (Federal Register, 2008).

Characterizing Wetland Hydrology

Forested wetlands are particularly difficult to restore (Beissel and Shear, 1997; Mitsch and Gosselink, 2007) because of their complex hydrology and time required for trees to establish and mature. In addition, there is limited information for recreating hydrologic regimes for wetland practitioners (Brooks and Gebo, 2013). Dating back to the mid-1990s, several studies of wetland hydrology have sought to establish characteristic hydrologic metrics that can be inferred from the hydrogeomorphic (HGM) classification alone (e.g. Cole et al., 1997). HGM is a wetland classification system developed by the USACE and is based on the landscape position of the wetland, its primary source of water, and its hydrodynamics (Brinson, 1993a). In

Pennsylvania, Cole et al. (1997) and Cole and Brooks (2000) compared hydrologic regimes across four HGM wetland subclasses. They found median water table depth was a good differentiating parameter between wetland types. However, the same could not be concluded for minimum and maximum water levels. They concluded basic inferences about hydrologic regime can be drawn from the HGM classification for a given geographic region; however, inferences may breakdown for disturbed or degraded sites (Cole and Brooks, 2000b).

Similarly, Shaffer et al. (1999) compared hydrology by HGM subclass for wetlands in Oregon. Hydropatterns were characterized by reporting intervals for the mean water level, minimum and maximum water levels, 90-day minimum and maximum water levels, and ranges for water levels, and extent and duration of inundation. They showed hydrologic conditions were comparable among wetlands in the same HGM subclasses, and that natural and altered wetland hydrology showed differences (Shaffer et al., 1999). More recently, Cole (2017) used cluster analysis to group wetland by type using summary statistics of the wetland hydroperiod. This analysis separated the created sites from the natural sites but was not able to precisely differentiate between the different HGM subclasses of the natural wetlands (Cole, 2017). These studies demonstrated that wetlands of a specific class might be identified by certain hydrologic measures, and these general finding support the prospect of developing design and performance standards based on specific wetland classes. However, a single measure (e.g. median) is not adequate to use as the basis for designs.

Hychka et al. (2013) used empirical cumulative distribution functions (ECDFs) to differentiate between different wetland types in a study of long-term hydrology in the Mid-Atlantic region of the U.S. They were able to identify different patterns and differences between

reference and disturbed sites, but the limited sample size prevented anything more than general inferences.

Menberu et al. (2016) compared peatland restoration sites to reference wetland sites in Finland. They used ECDFs to compare pre and post restoration periods. Generally, sites were wetter and more closely resembled the reference sites after restoration. They also examined storage gain in the peat soils post rainfall and the water table recession curves. Water table fluctuations were also compared between the pre and post restoration periods. They concluded that restoration success can be determined based on the measures they evaluated; however, this approach requires reference sites with the same monitoring period of record as the restoration site. One significant drawback of these methods is that they do not provide a quantifiable measure of restoration (e.g. within x% of the reference), but rather are tools to allow for qualification of restoration based on a case-by-case basis that relies on best professional judgement. Overall, ECDFs are a valuable tool to visualize differences between hydrologic time series (Hychka et al., 2013), but they may not be applicable for an objective design or monitoring criteria.

The pattern of hydrologic pulsing control species compositions and productivity and can be both a pressure and a subsidy to wetlands (Odum et al., 1995). Pulsing refers to the frequent fluctuations in water level causes by storm events or seasonal flooding. Wetland systems that exhibit pulsing hydropatterns have been shown to exhibit higher productivity (Mitsch and Gosselink, 2007). Studies in North Carolina have examined the importance of the hydroperiod in forested wetlands but have largely focused on the relationship between hydroperiod and plant community composition. Beissel and Shear (1997) found flood frequency is a good predictor of species composition and suggest restoration designs could be based on flood frequencies for

forested riverine wetlands. Similarly, Townsend (2001) reported extremely wet years (90th percentile) have the largest impact on species distribution. Bledsoe and Shear (2000) monitored hydrology, soils, and vegetation in alluvial swamps, and then used stream gauge data to analyze the long-term hydrologic impacts on plant species composition. They reported slight elevation changes can have marked impacts on flooding regimes and species composition. Such relationships create a great deal of uncertainty in prediction and design for restoration. These studies did not provide specific guidelines for restoration; although, Bledsoe and Shear (2000) advocate the establishment of a quasi-natural hydrologic regime and soils based on standards developed from reference systems. However, these studies do emphasize the importance of adequately capturing and predicting extreme events when characterizing hydropatterns for forested riverine wetlands.

In two related papers, Johnson et al. (2012) and Johnson et al. (2014) advanced the ideas put forward by Beissel and Shear (1997) and Bledsoe and Shear (2000) by comparing finer measures of forested wetland hydropatterns with plant community composition. They assessed a network of 13 reference wetlands by identifying parameters of the hydrologic regimes that were correlated with plant community composition and used these parameters to distinguish between forested wetland types. Specifically, they identified correlation between plant community composition and the Indicators of Hydraulic Alteration (IHA) using multivariate ordination techniques and multiple regression. IHA parameters were originally developed to assess alterations to stream systems (Richter et al., 1996), but can be adjusted to assess water level fluctuations (Johnson et al., 2012). They found several IHA parameters were highly correlated with plant community structure and could be used to distinguish the minor difference between hydropatterns of different forested wetland communities. They contend the measurement of

wetland restoration success should be based on hydrologic parameters that are highly correlated with plant community composition and developed from regional reference wetlands networks (Johnson et al., 2014). They also showed there is no correlation between plant composition and jurisdictional hydrology (Johnson et al., 2012). Johnson et al. (2014) contended non-growing season fluctuations do have an impact on wetland structure. This is reinforced by the fact that many biochemical processes are controlled by hydrology regardless of the time of year (e.g. Kadlec and Wallace, 2009). Thus, non-growing season hydrology should be considered in the design process.

Caldwell et al. (2011) used DRAINMOD to model the long-term hydrology at reference sites and established recommendations for restoration designs for four wetland community types in the Coastal Plain of North Carolina. They concluded restoration should be based on the thickness of organic soil and the median hydroperiod (total days inundated per year). Their ability to show that long-term modeling can characterize differences in hydrologic regimes among different wetland types is important. They demonstrated that each wetland type exhibited distinguishable hydrology based on periods of inundation. They recommended potential restoration sites be monitored for a year and then modeled to guide restoration design (Caldwell et al., 2011). While this approach would probably be successful, it is unlikely to be adapted for many mitigation projects under the current regulatory requirements.

Aside from these studies, there is limited published information on creating hydrologic design guidelines or even baseline characterizations of what the natural hydrologic regimes look like for many wetland communities. Still, the general consensus among researchers indicates recreating natural hydrologic regimes is the key to creating high functioning wetlands that approach the level of natural wetlands (e.g. Beissel and Shear, 1997; Bledsoe and Shear, 2000;

Zedler, 2000; National Research Council, 2001; Brooks et al., 2005; Cole et al., 2006; Mitsch and Gosselink, 2007; Brooks and Gebo, 2013; Johnson et al., 2014).

One of the challenges of characterizing wetland hydrology and developing design guidelines is the long-term nature of the data required. In order to evaluate normal conditions, 7 to 10 years of data is desirable; however, there is limited published information on long-term hydropatterns of natural wetlands. One of roadblocks preventing these datasets from expanding is the resource intensive process of data collection (He et al., 2002), and the nature of research projects, which mostly last two to three years (Cole and Kentula, 2011). While running long-term simulations of wetland hydrology using hydrologic models (e.g. Caldwell et al., 2011) provides a substitute for the long-term data, there is some inherent error involved (average absolute error of 10-20 cm between observed and simulated water table depths is considered "good" for DRAINMOD simulations (Skaggs et al., 2012)). However, for some wetland subclasses, a few centimeters may be greater than what is required to differentiate between wetland types (e.g. Johnson et al., 2014).

Reference Wetlands

The concept of reference wetlands, or a group of wetlands of a specific class that mitigation projects can be compared against was described by Brinson and Rheinhardt (1996). They defined references wetlands as "sites within a specified geographic region that are chosen to encompass the known variation of a group or class of wetlands." They argued reference wetlands should form the basis of standards for evaluating and comparing impacts and designing restoration projects. Designs based on existing high functioning ecosystems can remove potential bias and allow for more objective design and performance criteria (Brinson and Rheinhardt, 1996). In the subsequent decades, the reference approach to design and performance criteria has

been widely endorsed as having the potential to improve restoration and management outcomes (e.g. Whigham, 1999; Zedler, 2000; National Research Council, 2001; Brooks et al., 2005; Clewell, 2009; Cole and Kentula, 2011; Craft and Hopple, 2011; Johnson et al., 2014). However, little progress has been made in creating reference wetland networks or organizing and distributing collected wetlands data.

Some studies have used a single reference site for comparison with restoration projects (e.g. Jarzemsky et al., 2013). While this may be appropriate for gauging restorations that are part of a larger, interconnected wetland system, the use of a single reference does not capture the natural variation among the reference population (Craft and Hopple, 2011). Using a network of reference wetlands allows for more complete inferences on wetland condition (e.g. Gebo and Brooks, 2012).

Wetlands within reference networks should be located in the same geographic region (e.g. Gebo and Brooks, 2012; Johnson et al., 2014) to eliminate regional climate deviations or other physical factors that could impact comparisons (Craft and Hopple, 2011). Design guidelines should be based on the highest functioning wetlands from the reference population or reference standards. Thus, reference wetlands for design should exhibit minimal levels of anthropogenic disturbance (Kentula et al., 1992). However, given the history of anthropogenic disturbance in most areas, reference wetlands may need to be defined as the "best attainable" condition in a given region (Smith et al., 1995).

Reference wetland networks have been developed successfully. For example, the Penn State Cooperative Wetland Center maintains a database of 222 reference wetlands that includes vegetation, hydrology and soils data (Gebo and Brooks, 2012). The Montana National Heritage Program assessed 68 wetlands across the state to develop a reference network (Newlon and

Vance, 2011). Vegetation reference networks are more common compared to wetland reference networks (Lane and Texler, 2009).

There are some drawbacks to a reference-based approach to restoration. Currently, there are very few reference wetland systems (Cole and Kentula, 2011). The primary reason reference wetland networks have not been developed is that they are costly and time consuming to establish (Cole and Kentula, 2011; Johnson et al., 2014). Additionally, once wetland reference networks are developed they require ongoing commitments to collect, maintain, and manage the data. Further resources are required to protect the reference sites from disturbance and degradation. Current federal environmental regulation in the United States does not require the use of reference wetlands for the design or monitoring of mitigation projects (Ruiz-Jaen and Aide, 2005), and thus reference comparisons are seldom employed.

Overall, the current restoration policies and methods have resulted in a loss of wetland functions and a trend towards a set of restored wetlands that lack the heterogeneity found in natural wetland populations (Brooks et al., 2005; Dahl, 2011). In order to reverse this trend and improve wetland construction and restoration outcomes, practitioners need design and performance criteria, derived from reference wetlands, specific to the target wetland type. This approach could improve the construction and monitoring of systems that could improve both long and short-term success (Brooks and Gebo, 2013). Although, reference networks can be resource intensive to establish, their creation should be considered one of the most urgent priorities for wetland managers (Cole and Kentula, 2011) because of the potential to advance scientific understanding of wetland functions and improve restoration outcomes. Brooks et al. (2016) advocated the development of a national reference wetland registry to facilitate data sharing, help guide policy, and inform decision-making at the regional scale.

Wetland Water Quality

Wetland water quality is a major area of wetland science that is understudied. As previously noted, wetlands provide many unique ecosystem services to society such as non-point source pollution mitigation, wildlife habitat for many threatened species, flood mitigation, and other important services (Mitsch and Gosselink, 2007), yet water quality standards for protecting aquatic life do not explicitly apply to wetlands in most states, and numeric nutrient criteria have not been developed for wetlands. However, states do hold the legal authority under the CWA to develop and apply water quality standards to wetlands (Kusler and Christie, 2012).

Nearly three decades ago, in an effort to expand the regulatory protection of wetland ecosystems, the US EPA issued *National Guidance Water Quality Standards for Wetlands* in 1991 (US EPA, 1994). This document provided guidance for the states on developing state water quality standards specific to wetlands. Extending water quality standards to wetlands would strengthen the regulatory framework to better ensure the overall goal of conserving wetland resources and protecting the important natural functions of wetlands (US EPA, 1994). According to the US EPA (1994), the purpose of water quality standards specific to wetlands are to:

- Establish a baseline that supports the further development of programs for regulating wetlands.
- Create a point of reference against which impacts can be measured.
- Set goals for regulatory programs.

The *National Guidance Water Quality Standards for Wetlands* document encouraged several steps to ensure the protection of wetlands through the application of water quality standards: (1) expand the definition of state waters to include wetlands, (2) designate specific

uses for wetland waters, (3) adopt narrative and applicable numeric criteria, (4) adopt biological criteria, and (5) apply the state's anti-degradation policy to wetlands (US EPA, 1994). While most of these steps are not theoretically complicated, implementation of numeric criteria specific to wetlands is technically demanding and time-consuming. At least 15 states have adopted narrative criteria, specific use designations, and anti-degradation policies (Kusler and Christie, 2012); however, few states have adopted numeric criteria specific to wetlands. Nebraska is one exception, as they published numeric criterion specific to wetlands in 2014. However, the standards are not particularly stringent. For example, the maximum allowable concentration for nitrate plus nitrite was 100 mg N/L (Nebraska DEQ, 2014), which is several orders of magnitude higher than typical values for surface waters (US EPA, 2012a). Efforts at developing water quality standards have not been entirely unsuccessful. For example, the Florida Department of Environmental Protection developed and implemented nutrient criteria for phosphorus in the Florida Everglades, which has helped improve ecosystem health (US EPA, 2008c), but numeric standards have not been applied across the state. Other efforts are underway at the state level across the country, but the results have not been widely disseminated or implemented. For example, Minnesota collected water quality samples from wetlands across the state to establish baseline conditions (Genet, 2012). However, to date, recommendations for background conditions or baselines have not been published. In North Carolina, several monitoring efforts were undertaken (e.g. Baker et al., 2008; Savage et al., 2015), but the results have not been implemented in any way.

In North Carolina, there are currently no numeric water quality standards for wetlands; however, narrative criteria are covered in the North Carolina Administrative Code- 15A NCAC 2B .0231 Wetland Standards. The North Carolina regulations designate uses for wetland waters,

and the narrative criteria language focuses on avoiding "adverse impacts" to the wetland uses. Specifically, avoiding "concentrations… of substances which are harmful to human, animal or plant life… which individually or cumulatively may cause adverse impacts on existing wetland uses" and maintaining the hydrologic conditions necessary to avoid adverse impacts to the chemical, nutrient, DO, and pH conditions in the wetland (NCAC, 2015). However, without any numeric values or background ranges that would allow for the identification of changes indicative of "adverse impacts," these narrative standards would be very difficult to enforce with respect to wetland water quality.

The lack of progress in implementation by the states is likely the result of the highly dynamic nature of wetland water quality that is dependent on many outside factors and temporal variability (US EPA, 2008c), and the wide range of hydrologic conditions that often result in an absence of surface water for long periods (Kusler and Christie, 2012). Another major impediment to developing wetlands water quality standards is a lack of data and the costs associated with large-scale data collection efforts. In addition, many of the environmental factors present in wetlands (US EPA, 2008c) contribute to make normal conditions for wetlands fall outside the range of typical surface water concentrations (Kusler, 2011). Furthermore, while studies on stream water quality have been published for decades (e.g. Schröpfer 1942), there are very few published, long-term studies on surface water quality in natural wetlands.

Developing Nutrient Criteria

In an effort to better protect wetlands from nutrient enrichment, the EPA issued *Nutrient Criteria Technical Guidance Manual for Wetlands* in 2008. This document recommended the establishment of nutrient criteria specific to wetlands and provided guidance for developing the criteria (US EPA, 2008c). Methods for developing nutrient criteria for wetlands include the use

of reference conditions to characterize criteria based on minimally disturbed natural systems, frequency distribution-based approaches, ecological thresholds, applying predictive relationships between nutrient concentration and wetland function, and developing standards based on peer reviewed literature (US EPA, 2008c).

According to the US EPA (2008c), water quality criteria should be established for each wetland type or class within a given ecoregion as wetlands of the same class typically function similarly and have similar hydrologic regimes. For example, riverine wetlands typically have pulses of influent water that continually import nutrients and sediment (Mitsch and Gosselink, 2007), while basin wetlands (e.g. Carolina bays) may only receive inputs from rainfall (Sharitz, 2003). Hydrogeomorphic (HGM) class may be a good candidate for grouping wetlands (US EPA, 2008c) because HGM classes differentiate wetland types by landscape position, hydraulics, and water sources (Brinson, 1993b).

The US EPA's nutrient criteria document provides guidance for two reference-based methodologies for developing proposed criteria for wetlands. Both approaches require water quality data for a group of wetlands over an extended period, but do not require in-depth information about specific wetland conditions.

The least complicated approach to define reference conditions as a starting point for nutrient criteria put forward by US EPA (and used in their reference conditions for streams and lakes) is to define reference conditions based on a percentile of the distribution of observations. This defined percentile is assumed to approximate relatively undisturbed conditions. The percentile can vary based on the population of wetlands monitored. For a large population, representing a range of conditions, US EPA recommends a percentile ranging from 5th to 25th

(typically the 25th) for an approximation of reference conditions. If the population is made up of more degraded sites this percentile may be adjusted closer to the 5th percentile (US EPA, 2008c).

When the population is large enough to identify a subset of minimally disturbed reference wetlands, US EPA recommends using the 75-95th percentile of the reference subset for the development of criteria. This percentile can be extended closer to the 95th percentile when there is high confidence the population is made up of high-quality reference sites. These approaches have been confirmed by several studies conducted in New York and Tennessee. These studies showed the 75th percentile of the reference population is approximately equivalent to the 25th percentile of all the sites (US EPA, 2000a). Suplee et al (2007) found that the 86th percentile of the reference was a good match for the 25th percentile of the entire population for streams in Montana, but also that these percentiles vary considerably between ecoregions and among target constituents.

Another approach summarized by US EPA (2008c), relies on identifying reference wetlands in a region that exhibit the least disturbed conditions, typically based on best professional judgment, and then defining the water quality criteria based on monitoring these reference sites (US EPA, 2008c). For this method and the distribution-based approach, by setting the criteria as some approximation of relatively undisturbed sites, the assumption is made that maintaining water quality levels within the range of undisturbed reference sites will maintain the biological integrity of wetlands (US EPA, 2008c).

Reference Conditions for Streams and Lakes

Other studies have used the percentile-based approach for defining background conditions. For example, USGS used the 75th percentile approach of a large dataset of streams in undisturbed areas to define background conditions (USGS, 2010). The US EPA has published

recommended water quality reference conditions for lakes and reservoirs, and rivers and streams for most Level III Ecoregions in the United States (e.g. US EPA, 2000). The published parameters include TP, TN, chlorophyll-a, and Secchi depth. The reference levels are based entirely on the 25th percentile of the distribution of the population in each ecoregion. In this approach, the median is calculated for each site in each season. Then the 25th percentile of all the sites is calculated for each season. The final reference condition is the median of the 25th percentiles for the four seasons (US EPA, 2000a). A lack of sufficient data has prevented the same approach from being implemented for wetland reference conditions. However, reference conditions for streams and lakes may serve starting point for characterizing the water quality of wetlands. Table 1.1 presents the different reference and background conditions applicable to streams and lakes in North Carolina.

Table 1.1: Proposed nutrient criteria and background conditions for NC.

Water Body	TP (mg/L)	TN (mg/L)	NO_3^--N (mg/L)	Method	Source
Lakes and Reservoirs-Ecoregions IX, XIV-63	0.02	0.40 – 0.46	0.02- 0.10	25th percentile	(US EPA, 2000b, 2001)
Rivers and Streams-Ecoregions IX, XIV-63	0.04-0.05	0.42 - 0.55	0.04- 0.13	25th percentile	(US EPA, 2000c, 2000a)
National Background for Streams	0.034	0.58	0.24	75th percentile	(USGS, 2010)

A more complicated approach for defining nutrient criteria is to identify breakpoints or thresholds in the plotted relationship between constituent concentration and some biotic response (i.e. algal blooms, macroinvertebrates, etc.) (Evans-White et al., 2013; US EPA, 2008c). However, this requires significant data collection efforts, and can be difficult when the response curve is broad and there are no evident defined breakpoints (Robertson et al., 2008; US EPA,

2008c, 2010). This approach also assumed that one predictor is responsible for the threshold when the response may be dependent on multiple stressors.

Other approaches for defining water quality background conditions for streams include multiple linear-regression or other modeling techniques to relate watershed attributes that have been impacted by anthropogenic factors to water quality parameters. Reference conditions are then defined based on setting the coefficients for the anthropogenic parameters equal to zero in the model (Dodds and Oakes, 2004; Robertson et al., 2008). Other model-based approaches for determining nutrient criteria include spatial regression tree models (Robertson et al., 2006) and Random Forest models (Olson and Hawkins, 2013). However, with highly variable hydrology this method may not be applicable to wetlands.

Some states have developed recommendations for reference conditions using a modified version of the distribution-based approach. The Montana Department of Environmental Quality (MTDEQ) developed ambient background values for streams in different western river basins using a percentile-based approach. MTDEQ based their proposed criteria on a subset of identified reference streams. MTDEQ chose the 80th, 85th and 90th percentiles of the concentrations measured in the reference sites as starting points for draft nutrient criteria (MTDEQ, 2008)

While EPA recommends the use of ecoregion specific nutrient criteria for wetlands and streams, research suggests that for streams the use of ecoregion grouping does not always appropriately account for the natural variation that exists among nutrient concentrations (Herlihy and Sifneos, 2008; Robertson et al., 2006). The grouping of sites within ecoregions for nutrient criteria will inevitably lead to the overprotection of some sites and the under protection of others (Olson and Hawkins, 2013). As a result, sub-ecoregion or ecosystem specific criteria may be

more appropriate. While the development of stream criteria is a complex and multivariate problem, wetlands water quality characterization can be even more complicated due to the wide range in hydrologic regimes. For wetlands, an ecoregion specific and wetland type specific criteria should be considered as a starting point (US EPA, 2008c).

Background Water Quality for Streams, Lakes, and Manmade Systems

While criteria development is an intensive process that would take years of data collection and significant financial and time investments (Kusler, 2011), defining the ambient or background conditions in wetlands would be a significant step towards better understanding wetland ecosystems and recognizing and addressing degradation. To begin this process, it is important to understand how wetlands compare relative to other water bodies. Table 1.1 includes selected studies of water quality from various sources that may be similar to wetland in some respects or influence wetlands (drain into or from wetlands). These values give a general idea of the ranges encountered in surface waters and natural treatment systems.

Table 1.2: Selected studies of stormwater, stream, lake, and agricultural drainage nutrient and metal concentrations.

Water Body	Number of sites	pH	Spec. Cond. (us/cm)	TP (mg/L)	TN (mg/L)	NO3 (mg/L)	NH4 (mg/L)	ON (mg/L)	Cu (ug/L)	Pb (ug/L)	Zn (ug/L)	Source
Rivers in North Carolina	4 rivers	6.4 (5.3-9.3)	104 (40-450)	0.31 (0.02-1.5)	1.49 (0.36-3.6)	0.69 (0.02-2.7)	0.07 (0.01-4.9)	0.67 (0.33-1.8)	11.6 (0-100)	14.9 (0-200)	27.4 (0-120)	Crawford, (1985)
Coastal Plain Ag Streams-	background	6.0 (4.2-6.8)	98* (49-264)	0.10* (0.02-1.14)	1.0* (0.42-2.3)	0.05* (<PQL-1.51)	0.05* (<PQL-0.93)	0.76* (0.23-1.7)	-	-	-	Harden (2015)
	Ag-influenced	6.2* (4.7-6.8)	132* (48-328)	0.12* (0.01-0.98)	1.5* (0.36-17.0)	0.17* (<PQL-15.9)	0.1* (<PQL-3.42)	0.82* (0.27-2.0)				
Jordan Lake 2016	42	7.8 (6.3-9.5)		0.07 (0.04-0.24)	1.03 (0.88-1.77)							NC DEQ (2016)
National Stormwater Quality Database		7.5* (5.3-9.9)	121*	0.30* (0.01-7.9)	-	0.60* (0.02-7.3)	0.31* (0-7.80)	-	14.0* (0.71-1360)	16.0* (0.13-1200)	110* (1.1-8100)	Maestre and Pitt (2005)
Ag Drainage Mineral soil in NC	1	5.86		0.14	3.73	1.00	0.91	-	13	-	7	Skaggs et al. (1980)
Ag Drainage Organic soil in NC	1	5.32		1.21	2.65	0.29	0.28	-	22	-	5	Skaggs et al. (1980)
Restored Wetland	1			0.17	1.29	0.007						Bruland et al. (2003)
Ag Drainage in NC	1			0.12	1.59	0.21						Bruland et al. (2003)
Stormwater Wetland				0.23	1.11	0.17	0.08					Lenhart and Hunt (2011)
Stormwater Wetlands								0.8*				Moore et al. (2011)
"background" concentrations for FWS constructed wetlands				<0.2	(1-3)	<0.1	<0.1					US EPA, (2000d)

All values are means unless otherwise noted -- brackets indicate range -- *median PQL: Practical quantitation limit

Water Quality in Natural Wetlands

While wetlands are often lauded for their ability to serve as nutrient, sediment, and pollutant sinks, and are often referred to as "nature's kidneys" (Mitsch and Gosselink, 2007), there are very few long-term studies on the ambient water quality of natural, undisturbed wetlands. Many of the academic studies on natural wetland water quality that have occurred have been limited by short sampling duration of only a few months or less (e.g. Yu et al., 2015; Justus et al., 2016) or limited number of sites (e.g. Skaggs et al., 1980). However, wetland size, surrounding land use, and hydrologic connectivity to nearby surface water all have an influence on wetland water quality (e.g. Azous and Horner, 2000; Houlahan and Findlay, 2004; Yu et al., 2015). For example, riverine wetland water quality would likely be influenced by flooding frequency and watershed land use because of the horizontal hydraulics (i.e. connectivity to the river) (Brinson, 1993b). For example, Weilhoefer et al. (2008) monitored water quality in a floodplain wetland over a two-year period. During flood events in which the wetland and river were hydrologically connected, TP, TN and conductivity dropped in the wetland, due to dilution caused by the large volume of floodwater. Overall TN, TP and conductivity were higher in the wetland than the river. Mitsch and Gosselink (2015) also report that riverine swamp water quality can vary significantly from the water quality in their adjacent rivers. Riverine swamps are typically nutrient and mineral rich while the opposite is true for basin systems.

Isolated wetlands are more sensitive to changes in surrounding land use, and thus water quality impacts are more noticeable (Yu et al., 2015). Not surprisingly, catchment land use is one of the primary factors influencing wetland nutrient concentration (Craft et al., 2007; Reiss, 2006; Yu et al., 2015); the percentage of developed land has a direct impact on wetland water quality (Azous and Horner, 2000). A wetland's underlying geology has also been shown to influence

water quality (Babb et al., 1997; Cole et al., 1997), as does soil type (e.g. Skaggs et al., 1980). Soil type (organic or mineral) differ in their capacity to immobilize/ mobilize dissolved constituents and regulate pH (Mitsch and Gosselink, 2007).

pH varies broadly among wetland types (see Table 1.3) and can even vary by several units daily in wetlands were algal productivity is high (Reddy and DeLaune, 2008). pH is typically acidic to near neutral in wetlands and can range from less than 4.0 in wetlands on thick organic soils (Johannesson et al., 2004; Walbridge and Richardson, 1991) to near neutral and above in many riverine systems (Justus et al., 2016). pH influences nutrient cycling and the precipitation dissolution reactions of many metals, as well as biological processes such as nitrification, denitrification, and mineralization (Reddy and DeLaune, 2008). Mitsch and Gosselink (2015) note that riverine freshwater swamps in the Southeastern United States typically have pH in the range of 6-7, while the pH for precipitation driven basins is usually in the range of 3.5-5.0.

Many of the important functions of wetland are the result of anaerobic conditions (e.g. carbon sequestration, denitrification) (Mitsch and Gosselink, 2015); however, dissolved oxygen (DO) levels actually vary widely in the surface water of wetlands. Seasonal changes are typical as DO is temperature dependent, and like in other aquatic ecosystems, diurnal DO fluctuations can occur due to photosynthesis by autotrophic organisms in the water column. Justus et al. (2016) observed DO in the water column ranging from 0.05 to 7.0 in riverine depressions during the summer months.

Conductivity is the measure of a water's capacity to conduct an electrical current and is largely a function of the level of dissolved inorganic ions (US EPA, 2012b). Conductivity is temperature dependent, increasing with higher temperatures. In undisturbed wetlands,

conductivity is typically low (Reiss, 2006; Walbridge and Richardson, 1991); however, across all wetlands there is a large range (Justus et al., 2016). The specific conductivity (normalized to 25 °C) of rivers generally ranges from 50 to 1500 µS/cm. However, these values can be higher for coastal systems where salinity is a factor. High quality streams for fish habitat generally exhibit specific conductivity in the range of 150-500 µS/cm (US EPA, 2012b). Specific conductivity has been shown to fluctuate over short time in wetlands. For example, Bosserman (1984) found that specific conductivity fluctuated diurnally in the Okenfenokee Swamp in Georgia; sometimes varying 20-40 µS/cm. Chow et al. (2016) reported specific conductivity varied by up to 20 us/cm over three to four-day periods in small ephemeral wetlands in South Carolina.

Inorganic nutrient concentration in surface waters of undisturbed wetlands are generally low (e.g. Skaggs et al., 1980; Reiss, 2006; Lane et al., 2015). For example, Skaggs et al. (1980) monitored drainage water from natural areas on poorly drained soils in the Coastal Plain of NC. Average concentrations were 0.15 mg/L for NH_4^+-N, 0.04 mg/L for NO_3^--N, and 0.05 mg/L for TP over a three-year period. Reiss (2006) monitored depressional-forested wetlands in FL, mean nutrient concentrations across the reference wetlands were 0.15 mg/L for NH_4^+-N, 0.09 mg/L for NO_3^--N, and 0.08 mg/L for TP. While the inorganic nitrogen values are low for undisturbed wetlands, especially compared to stormwater (e.g. Maestre and Pitt, 2005) and agricultural drainage (Skaggs et al., 1980), concentrations in disturbed wetlands are more comparable to these developed land uses. Wetlands in or adjacent to urbanized area receive stormwater influent (Craft et al., 2007), and thus the ambient concentration in these systems may be more similar to typical stormwater concentration, with elevated metal levels (e.g. Azous and Horner, 2000). Walbridge and Richardson (1991) reviewed the available water quality data on pocosins in eastern NC and found NH_4^+-N concentrations ranges from 0-1.62 mg/L and nitrate-N varied 0-

1.34 mg/L across natural and disturbed (agricultural sites). However, inorganic nutrient levels in undisturbed pocosins were much lower. For example, surface water mean ammonium-N was 0.04 to 0.11 mg/l and nitrate-N was 0.03 to 0.12 mg/L (Daniel, 1981). In agricultural watersheds in the Midwest, Craft et al. (2007) reported nutrient concentrations increased in wetlands located in areas most impacted by development and agriculture, and lowest in undeveloped areas.

Natural wetlands can serve as a source of organic nitrogen to downstream aquatic ecosystems, with ON increasing as percentage of wetland area in the watershed increases (e.g. Pellerin et al., 2004). This is the result of organic material accumulation due to the decreased decomposition rate under anaerobic conditions (Reddy and DeLaune, 2008). Organic nitrogen levels appear to be dependent on disturbance (e.g. Reiss 2006), but also the underlying soil type (e.g. Skaggs et al 1980) and retention time (i.e. flow through vs depressional wetlands). The bioavailability of organic nitrogen is generally assumed to be limited, however, research suggests that organic nitrogen may be an important source of nitrogen for organisms in coastal systems and thus could be important in contributing to algal blooms (Stepanauskas et al., 1999). Very few studies of natural wetlands report ON concentrations.

Table 1.3: Natural wetland water quality studies.

Description	Period	Number of sites	pH	DO (mg/L)	Spec Cond (µS/cm)	NH_4^+ (mg/L)	NO_3^- (mg/L)	ON (mg/L)	TN (mg/L)	TP (mg/L)	Cu (µg/L)	Zn (µg/L)	Pb (µg/L)	Source
Drainage water from undeveloped natural wetland in the Tidewater Region of NC	4 years – weekly composite samples	1 mineral soils	4.9			0.15	0.11	0.67	0.93	0.05	8	2		Skaggs et al. (1980)
		2 organic soils	3.6		(75-140)	0.14	0.04	0.95	1.13	0.05	12	15		
Wetlands Puget Sound Basin, WA	5 years ~9 samples per year	Natural	6.4		72.5 (na-230)	0.059 (na-1.37)	0.37 (na-3.2)			0.05 (na-0.85)	3 (na-15)	8 (na-21)	3 (na-21)	Azous and Horner (2000)
		Moderate impact	6.5		142 (na-275)	0.13 (na-2.27)	0.60 (na-7.2)			0.092 (na-0.78)	4 (na-7)	10 (na-33)	3 (na-13)	
		Highly impacted	6.7		151 (na-271)	0.07 (na-0.52)	0.40 (na-1.1)			0.11 (na-1.94)	4 (na-12)	20 (na-73)	5 (na-22)	
Review of water quality in NC Pocosins- displaying a range of disturbance			3.0-4.0		51-108	0-1.62	0-1.34							Walbridge and Richardson (1991)
Drainage water from non-riverine swamp forest and Pocosin in NC	2 years of monitoring	1					0.008		1.73	0.05				Bruland et al. (2003)
Riverine Depressions in Arkansas	3 events May-July 2012	24	6.7* (5.6-7.4)	1.7* (0.05-7.65)	147* (71-834)	0.31* (0.08-13.0)	0.04* (0.01-0.30)			0.24* (0.10-3.93)				Justus et al. (2016)
Wetlands in Indiana	Seasonally for 1 year	30				0.14	0.80							Craft et al. (2007)
Van Swamp, NC	3 years-monthly samples	1	2.5-5.7		91 (50-142)		0.2		1.2	0.02				Daniel (1981)
49 Carolina Bays	1 sampling event at each site in Jan. 88	49	4.6* (3.4-6.7)		76* (29-177)									Newman and Schalles (1990)
Geographically isolated wetlands in NC and FL	1 sample at each site	14	4.3 (3.6-5.9)			0.038 (0.017 - 0.135)	0.006 (<PQL-0.016		1.24 (0.77-2.02)	0.025(0.006-0.057)				Lane et al. (2015)
Depressional Forested Wetlands in FL	1 sample collected during summer	25 reference	5.2		81	0.15	0.09		2.02	0.08				Reiss (2006)
		25 AG	6.2		136	0.33	0.01		3.17	0.81				
		25 Urban	6.4		231	0.19	0.02		1.84	0.23				
Okenfenokee Swamp			3.8-4.9	<1-9.5	13-87									Bosserman (1984)
South Carolina Piedmont Ephemeral Wetlands	6 months	3	~6.5		50		0.01		0.33					Chow et al. (2016)

Metals in Surface Waters

Metals can be present in the water column of wetlands in both dissolved form as well as bound to suspended particles. Toxic metals can also be present in the soil profile bound to iron and manganese oxides and clay minerals (Gambrell, 1994). The formation of metal-organic matter complexes is an important factor governing the availability of metals in the dissolved phase (Sparks, 2003). The stability of metals in the wetland aquatic environment is dependent on pH and redox state. For example, zinc, copper and lead are all more bioavailable at low pH because adsorption is decreased (Gambrell, 1994).

Metal contamination in wetlands is typically associated with urbanization and industrial runoff. Sources of zinc, copper, and lead in the environment include buildings and infrastructure, cars, industrial sources, mining and impervious surfaces (ATSDR, 2005). Sources of metals can also include runoff from livestock operations (Nicholson et al., 1999; Zhang et al., 2012).

Zinc and copper are essential micronutrients for animals and humans, but exposure to high levels can cause serious health effects (ATSDR, 2005). Exposure to elevated levels in humans can cause dermal reactions, respiratory problems, neurological affects, and organ failure (ATSDR, 2005).

Elevated levels of heavy metals pose risks to many aquatic macroinvertebrates (Malaj et al., 2012) and amphibians (Vitt et al., 1990) and can cause shifts in species composition to more tolerant species as well as physiological reactions (Malaj et al., 2012). The elevated levels interfere with species growth and inhibit the life cycle of macroinvertebrates. Bioaccumulation through the food chain can also be a potential concern for contaminated sites (Goodyear and McNeill, 1999). In addition, high levels of toxic metals can inhibit microbial and enzymatic

processes. This can affect or inhibit many of the significant processes in wetlands, such as mineralization, nitrification, and denitrification (Reddy and DeLaune, 2008).

Mean zinc levels in rivers across the United States were below 50 µg/L and below 5.0 µg/L in some cases (ATSDR, 2005). However, contaminated waters can have levels above 30 mg/L. Zinc levels measured in the Great Lakes in the early 1990's were well below 1 µg/L (Nriagu et al., 1996). The median stormwater concentration from the NSWQ Database was 116 µg/L (Maestre and Pitt, 2005). A summary of selected studies of metals levels in surface water is presented in Table 1.4.

Copper is also detected in most surface water bodies in the U.S. A study of over 50,000 samples reported a mean concentration of 4.2 µg/L for total copper (Eckel and Jacob, 1988). Levels observed in natural, undisturbed wetlands are typically low (e.g. Skaggs et al 1980; Azous and Horner 2000), while waters impacted by development are higher (Azous and Horner 2000). The median stormwater concentration from the NSWQ Database was 14 µg/L (Maestre and Pitt, 2005). The background level for reference waters is about 3.0 µg/L (Reddy and DeLaune, 2008).

Lead is present in most surface water in the United States, concentrations range from below the detection limit to several hundred micrograms per liter in contaminated areas. Undisturbed areas typically have low concentrations. The median stormwater concentration from the NSWQ Database was 16 µg/L (Maestre and Pitt, 2005). Eckel and Jacob (1988) reported mean lead levels below 4.0 µg/L based on a study of 50,000 surface water sampling stations across the United States.

Table 1.4: Metal concentrations in surface waters for selected studies.

Description	Cu (µg/L)	Zn (µg/L)	Pb (µg/L)	Source
NSWQ Database	14*	116*	16*	Maestre and Pitt (2005)
50,000 surface water sampling stations	4.2*	-	<4.0	Eckel and Jacob (1988)
Background level for reference waters	3	-	-	Reddy and DeLaune (2008)
Great Lakes	0.8**	0.2**	<0.01**	Nriagu et al. (1996)
NC Rivers	12**	27**	15**	Crawford, (1985)
Background in surface waters	-	<50	-	ATSDR (2005)
Typical values unless designated as: * Median or **Mean				

From 2007 to 2014, the North Carolina DEQ DWR monitored 120 stream stations in their Random Ambient Monitoring System (RAMS) network. Over the 7-year study, twelve sites exceeded the chronic standard for copper, two sites exceeded the chronic standard for lead, and five stations were above the allowable limit for chronic zinc (NCDENR, 2014). Land use or other potential factors contributing to the elevated concentrations at the noncompliant sites were not reported.

Freshwater Water Aquatic Life Criteria for Metals

Existing aquatic life water quality criteria for metals and other parameters may be applicable to wetlands for some constituents or can serve as a starting point for development of wetland appropriate standards (Kusler, 2011). The US EPA and NC DEQ have set limits for several water quality constituents for the protection of freshwater aquatic life shown in Table 1.5.

Table 1.5: Selected water quality criteria (ug/l unless otherwise noted).

Parameter	Standard	Acute	Chronic	Source
NH_3-N**	-	17	1.9	Freshwater Aquatic Life, EPA
pH	6.0-9.0	-	-	Freshwater Aquatic Life, NCAC
DO	≥ 5.0	-	-	Freshwater Aquatic Life, NCAC
Cu***	-	10	7.0	Freshwater Aquatic Life, NCAC
Pb***	-	47	1.8	Freshwater Aquatic Life, NCAC
Zn***	-	92	93	Freshwater Aquatic Life, NCAC

**(mg/L) at pH 7 and 20 C

*** Hardness dependent, values shown are for hardness = 75 mg/L

Water quality standards for metals have two thresholds: a chronic and an acute level. The chronic level refers to the average of samples collected in one hour and the acute level refers to the concentration measured over four-day period (NC DEQ, 2017). Current aquatic life water quality standards are based on dissolved metal concentration (NC DEQ, 2017), because dissolved metals are more bioavailable and toxic to aquatic life (US EPA, 1996). Dissolved metals are defined as the portion passing through a 0.45-micron filter. These standards for dissolved metals (Cu, Pb, Zn) are hardness-dependent; meaning the threshold concentration increases for increasing hardness levels. The metals are less toxic to aquatic organisms at higher hardness levels because aquatic organisms more readily uptake magnesium and calcium (primary contributors to hardness) than the toxic metals (NCDENR, 2015). The hardness dependent thresholds are calculated using equations developed by US EPA (2004a).

Because samples for metals are often analyzed for total metals (dissolved plus particulate bound), US EPA (1996) provided guidance for developing site-specific conversion factors (EPA, 1996). The high DOC concentration in some wetlands (e.g. Wright and Reddy, 2012) may pose problems for applying the aquatic life standards for some constituents. For example, US EPA (2004) notes that copper may be substantially less toxic when DOC concentrations are elevated because a majority of the copper might be bound to the DOC and not bioavailable. This is because DOC humic and fulvic acid functional groups form strong complexes with dissolved copper (studies have shown that the toxicity of total copper is inversely related to the DOC concentration in streams) (Kramer et al., 2004). Thus, the aquatic life standards for copper may not be applicable to wetlands unless site-specific conversion factors are used. For example, a concentration of 6 mg/L copper might be measured in the sample, which might violate the standards, however, because of the high DOC in wetlands, the concentration may not actually

pose a risk to aquatic organisms. A large portion of this copper may not be available to aquatic organisms because it is strongly bound to DOC.

Wetland Soils

Overview

Wetland soils serve as the substrate for the important biogeochemical functions performed in wetland ecosystems and serve as a major sink for global soil carbon. Wetlands are estimated to store up to 30 percent of the global soil carbon. This is significant because wetlands account for less than 8 percent of the global land area (Mitsch and Gosselink, 2007).

All wetland soils are subject to extended periods of inundation and/or saturation that lead to the formation of anaerobic conditions (Mitsch and Gosselink, 2007). These conditions lead to biological, physical and chemical conditions that are unique to wetland environments (Reddy et al., 2000). Wetland soils are typically associated with organic matter accumulation (Bernal and Mitsch, 2012), resulting from high productivity relative to decomposition (because of anaerobic conditions) (Reddy and DeLaune, 2008). However, wetland soils occur across a wide range of textural classes (e.g. Richardson and Vepraskas, 2001) and organic matter content can vary from only a few percent to more than 80 percent in peat soils (Mitsch and Gosselink, 2007).

Because wetlands often occur at low areas in the landscape, topography naturally directs surface and subsurface flows through wetland areas (Richardson and Vepraskas, 2001), and as a result suspended or dissolved substances from the upland can enter wetland areas. These constituents can then be retained or removed in wetlands due to several factors including burial in sediment, microbial immobilization, sorption, plant uptake, precipitation reactions, and other chemical processes (Reddy and DeLaune, 2008). Studies have shown wetland soils have the

ability to more readily retain toxic and trace metals than upland soils environments (Gambrell, 1994).

The degree of alteration of the landscape surrounding wetland and the wetland type impacts the concentration of constituents entering wetlands (Richardson and Vepraskas, 2001). Development can alter the physical and chemical makeup of wetland soils in many environments. For example, agricultural drainage can lower the water table in adjacent wetlands and lead to the oxidation of organic matter and subsidence (Richardson and Vepraskas, 2001). Wetlands downstream from agricultural development receive increased nutrient and sediment loads (e.g. Gilliam, 1994). Excessive sediment loading can change the wetland morphology and thus alter the hydrologic regime and vegetative structure (Mitsch and Gosselink, 2007). Urbanization and development in upstream watersheds increase runoff volume and sediment loading (Richardson and Vepraskas, 2001) and stormwater pollutants can enter wetlands. This can lead to elevated metal levels in urban wetlands. For example, Azous and Horner (2000) found that forested wetland soils in highly urbanized areas had higher levels of heavy metals than wetlands in rural areas. Overall, urban wetlands serve as sinks for nutrients, sediment, and metals (Faulkner, 2004). Changes to wetland soils chemical content and physical structure can influence the plant species composition, and nutrient enrichment of soils can lead to a decline in species diversity (Bedford et al., 1999).

Soil Properties

Wetlands soil generally exhibit a high spatial variability in soil chemical composition (US EPA, 2008b). Organic soils are formed from the decomposition of organic materials, while mineral soils form from weathering processes and sediment transport and deposition (Richardson and Vepraskas, 2001). Organic soils are characterized as soil with organic content greater than

20 to 35 percent, depending on the clay content. The pH of organic soils is acid whereas mineral soils will be closer to neutral. Organic soils have much lower bulk density (less than 0.3 g/cm^3 vs 1.0 -2.0 g/cm^3) and much higher porosity (~80% vs. ~50%) than mineral soils. However, organic soils do not necessarily have higher hydraulic conductivity than mineral soils (Mitsch and Gosselink, 2007). Organic soils generally have higher cation exchange capacity (CEC) and nutrient content compared to mineral soils. However, mineral soils typically have higher nutrient availability than organic soils (Mitsch and Gosselink, 2007). The reactivity of soils is largely dependent on surface area and ionic charge. Most wetland soils have a negative charge and thus providing bonding sites for many metal cations (Faulkner and Richardson, 1989).

In mineral soils, the exchange sites are dominated by metal cations. The CEC of organic soils is dominated by hydrogen ions (Faulkner and Richardson, 1989). However, soil organic matter can play an important role in the immobilization of metals. The formation of metal ion-soil organic matter complexes is very important in governing the availability of toxic metals in the wetland environment. This can include the formation of complexes with both SOC and DOC (Sparks, 2003). In North Carolina, the majority of organic soil wetlands are found in the Coastal Plain.

Soil chemical and physical properties in riverine wetland systems can be highly variable, both vertically and spatially. This is the result of variable sediment deposition patterns resulting from several factors including distance from the stream, and the topographic variability related to the different floodplain features (Richardson and Vepraskas, 2001). Soils in basin wetlands typically include low permeability, clay layers that restrict water movement and root growth. The subsoils have very high bulk density and organic matter accumulation is limited to the upper surface layer (Richardson and Vepraskas, 2001).

Soil Organic Matter

It has been suggested that SOM might be a useful indicator of wetland condition (e.g. Shaffer and Ernst 1999) and is frequently used in the evaluation of the progress of wetland restoration projects (e.g. Shaffer and Ernst, 1999; Campbell et al., 2002; Bruland and Richardson, 2006; Bantilan-Smith et al., 2009; Hossler et al., 2011). This is because SOM is typically correlated with many important wetland functions and increased SOM typically results in decreased bulk density and higher water holding capacity. For example, Brettar and Höfle (2002) reported a direct link between denitrification and SOM content in forested wetlands; Stolt et al. (2000) and others reached similar conclusions. Soil carbon provides electrons to power the major reducing reactions in wetlands (denitrification, iron reduction, sulfate reduction, methanogenesis) (Mitsch and Gosselink, 2007). In addition, SOM provides nutrients for plant uptake, a source of energy for soil microorganisms, forms complexes with metals, strongly adsorbs toxic organic compounds, and stores nutrients long-term (Reddy and DeLaune, 2008). SOM can be extremely variable between sites in the same region and even within the same site (e.g. Bruland and Richardson 2006). Table 1.6 shows reported values for selected natural wetlands in the Southeast U.S.

Table 1.6: Selected SOM values from regional wetlands.

Site	Soil Material	Depth	pH	BD** (g/cm^3)	Organic C (%)	SOM (%)	Source
Forested Depressional Wetland in GA	Mineral	0-30 cm	-	0.47	10.0	17*	Craft and Casey, (2000)
Forested Floodplain in GA	Mineral	0-30 cm	-	1.02	5.2	8.8*	
VA Forested palustrine	Mineral	5-15 cm	4.4-5.9	-	0.7-15.4	1.2-26.5*	Stolt et al. (2000)
NC Peatlands	Organic	0-30 cm	-	0.02-0.36	-	50-97	Bridgham and Richardson (1993)
NC Headwater	Both	0-20 cm	-	-	-	10	Bruland and Richardson (2006)
NC Riverine	Both	0-20 cm	-	-	-	20	
NC Non-riverine	Mineral	0-20 cm	-	-	-	18	
NC Non-riverine	Organic	0-20 cm	-	-	-	65	
NC Pocosins	Both	0-10 cm	3.9-4.6	-	15.5-47.0	47-80*	Ducey et al. (2015)
NC Swamp	Organic	0-20 cm	4.1	-	-	17	Faulkner and Richardson (1989)
NC Pocosin	Organic	0-20 cm	3.9	-	-	77	
Forested Swamp	Organic	0-20 cm	4.5	-	-	59	

*Estimated using a conversion factor of SOM=1.72*OC from Mitsch and Gosselink (2007).
**BD: bulk density

Metals in Wetland Soils

Trace metals are naturally present at very low background concentrations in all soils. Elevated concentrations are typically only encountered due to anthropogenic disturbance (Gambrell, 1994). The NC Department of Agriculture and Consumer Services (NCDA&CS) reported average soil heavy metal concentrations from their testing of soils across North Carolina from 2005-2007. They reported mean *Mehlich 3 extractable* concentrations of 9.2 mg/kg for Cu and 27.2 mg/kg for Zn based on 3286 soil samples (Hardy et al., 2008). US EPA (US EPA, 2007b, 2007a) reported median background concentrations of 20 mg/kg dry weight and 50 mg/kg dry weight for total Cu and total Zn, respectively, for the eastern United States, although there was a wide range of variability for both parameters. US EPA (2007c) reported background

concentrations of 56 mg/kg and 34 mg/kg for total Zn and total Cu, respectively for North Carolina soils. When evaluating soil metal levels attention needs to be given to the laboratory analytical methods used because the results can vary considerably based on the analysis method employed. For example, for metals analysis, the National Wetland Condition Assessment used EPA method 3050B, which is a strong acid extraction that is a reasonable approximation of the total recoverable metals (EPA, 1996). Many state labs in the Southeast use the Mehlich 3 extraction to estimate plant available metals, which is the primary concern for agricultural production (Mehlich, 1984).

The NCDA&CS has publiched recommended contamination thresholds for Cu and Zn for agricultural crop production. For example, the critical toxicity levels for plants are 120 mg/kg and 60 mg/kg for Zn and Cu, respectively (Hardy et al., 2003). Cu levels are typically only measured above 20 mg/kg due to anthropogenic disturbances (Tucker, 1999). Others suggest that 50 mg/kg and 20 mg/kg for Zn and Cu, respectively, should be considered warning levels for agricultural applications (Rutgers Soil Testing Laboratory, 2017).

The US EPA has defined Ecological Soil Screening Levels (Eco-SSLs) or concentration of metals in soils that may be indicative of potential risks to the ecological receptors of plants, soil invertebrates, birds, and mammals (US EPA, 2007a). These values are intended for risk assessments and can help identify areas that may need further study, but should not be used as an indicator for the need of remediation (US EPA, 2007a). While, Eco-SSLs where developed and intended for application in upland soils, US EPA (2005) indicates that they may be useful for initial screening of wetland soils, particularly for soil invertebrate and plant Eco-SSLs. The concentrations for the ECO-SSLs are meant to be compared to total metal concentrations, not extractable results. Table 1.7 provides the Eco-SSL values for Cu and Zn.

Table 1.7: Eco-SSL levels for total zinc and copper (mg/kg dry weight of soil).

Constituent	Plants	Soil Invertebrates	Avian Wildlife	Mammalian Wildlife
Copper	70	80	28	49
Zinc	160	120	46	79

While some studies have used conversion factors to convert from extractable to total metals, the ratio of extractable to total varies widely across different soil type and conditions. Mcbride et al. (2009) found that the extractable fraction of Zn and Cu both about 0.30, but there is a wide range of published ratios. For example, Sims et al. (1991) reported that the correlation coefficient between Mehlich 3 extracted Cu and Zn and the EPA Method 350 (total metals) were only 0.56 and 0.43, respectively for agricultural soils in Delaware.

Climate Change

Impacts of Climate Change on Wetlands

The impacts of climate change on wetland area and function will vary greatly across different regions (Brinson, 2006), and while there are uncertainties to the extent, important wetland functions will be impacted (Burkett and Kusler, 2000). The primary impacts will result from the alteration of precipitation and temperature patterns and an increase in the frequency of extreme events (Erwin, 2009). As temperature and precipitation patterns change, wetlands are especially at risk because of their intermediate landscape position, where small changes in precipitation and evapotranspiration could have disproportionate impacts on the areas of some wetland types (Burkett and Kusler, 2000). In addition, plant species composition will likely shift as temperatures rise, and extinctions will increase along with increasing prominence of invasive plant species (Brinson, 2006). Extreme events (both depth and duration) tend to influence species composition (e.g. Beissel and Shear, 1997; Foti et al., 2012). Thus, wetland plant composition could trend towards a more homogeneous state, facilitated by the elimination of non-tolerant,

native species. Wiens (2016) found that climate induced local extinctions are already occurring among many plant and animal communities.

Climate change creates the risk of irreversible changes (likely unavoidable in some regions) to wetland ecosystems (Pachauri et al., 2014). For example, prairie pothole wetlands and vernal pools may disappear completely in some areas (Burkett and Kusler, 2000; Erwin, 2009). However, restoration has the potential to buffer some of the impacts of climate change. For example, large-scale restorations could increase carbon sequestration, coastal marsh restoration can help buffer shorelines against rising sea levels (Erwin, 2009; Jimenez Cisneros et al., 2014).

In North Carolina, wetlands are threatened by climate change due to higher temperature, shifting precipitation patterns, and sea-level rise. Increasing salinity and higher water levels in coastal wetlands can lead to ghost forests in coastal riverine bottomlands (Doyle et al., 2007), and sea-level rise may overwhelm some coastal wetlands (US EPA, 2016). In peatlands in eastern NC, changes to hydrology due to climate change could lower water levels and lead to rapid oxidation and loss of the organic soils (Erwin, 2009). The functions of basin wetlands are likely at particular risk given their similarity to vernal pools, which may be severely impacted in some areas (Erwin, 2009). Riverine wetlands will be impacted by the possible changes to streamflow patterns and flooding caused by climate change (Erwin, 2009). In addition, the hydrology of riverine wetlands in the interim between flood events could also be impacted by increased evapotranspiration due to higher temperatures and changing precipitation patterns.

While the southeastern United States has experienced less warming than other regions of the United States during the 20th century, mean annual temperature has increased steadily since 1980, and the first decade of the 21st century was the warmest on record for the Southeast

(Kunkel et al., 2013). The National Oceanic and Atmospheric Administration (NOAA) projects the mean annual temperature to increase between 2.2 to 4.2 degrees C (25th and 75th percentiles) in the southeastern United States during the 21st century based on an ensemble of climate models (Carter et al., 2014). In addition to higher temperatures, the southeast United States will experience more extreme temperature regimes. For example, in North Carolina the number of days per year with maximum temperature exceeding 35 degrees C is projected to increase by 100-300% this century (US EPA, 2016). The ensemble of models indicated variable projections for changes in precipitation, however a majority of models show modest increases in precipitation in North Carolina and Virginia (Kunkel et al., 2013), along with increases in extreme precipitation (Carter et al., 2014). Higher precipitation and evapotranspiration (because of higher temperatures) may be offsetting in terms of hydrologic impacts on wetlands, but this has not been evaluated with long-term simulations.

Long-term hydrologic models that incorporate future climate scenarios have been demonstrated as a tool to make hydrologic predictions for a wetland's response to climate change (e.g. Lee et al., 2015). Comparable to any long-term predictions, there is a great deal of uncertainty associated with the projected impacts of climate change on wetlands (Brinson, 2006; Erwin, 2009); however, this approach provides the best available method of making reasonable inferences on the future trends of wetlands for a specific type and geographic region. In addition, with observed GHG emissions following the predictions of the most extreme climate scenarios (USBR, 2016), model output from these high-end scenarios could provide a reasonable scenario for planning purposes.

Climate Models

Global climate models (GCMs) are multi-component, numerical models that use known physical processes to simulate climate patterns across the globe, and typically include coupled atmosphere, land-surface, and ocean components (USGS, 2014). Many models now include components to represent atmospheric aerosols, biogeochemical cycles and land use changes (NOAA, 2014).

The latest generation of climate models from the World Climate Research Programme's Coupled Model Intercomparison Project (CMIP5) have made significant advances in better simulating 20th century climate and more accurately representing cycles of climate variability including El Nino, Atlantic Multi-decadal Variability and Pacific Decadal Variability (NOAA, 2014). CMIP5 models provide projections for four unique emissions scenarios widely used to make predictions on the impacts of climate change, including in the IPCC's Fifth Assessment Report (AR5). The four concentration scenarios or Representative Concentration Pathways (RCP) represented in CMIP5 models are: RCP2.6, RCP4.5, RCP6.0, and RCP8.5. The number designation following "RCP" represents a value for radiative forcing reached in the year 2100. For example, RCP6.0 refers to a scenario in which radiative forcing reaches 6.0 watts/m^2 in 2100. RCP2.6 represents a mitigation scenario in which CO_2 emissions peak by 2020 and begin to decrease, with CO_2 concentration stabilizing around 400 pm after 2100. RCP4.5 corresponds to CO_2 emissions peaking around 2040. RCP 6.0 corresponds to CO_2 peaking in about 2060. Emissions for RCP8.5 continue to increase through the end of the 21st century. For RCP4.5, 6.0 and 8.5, CO_2 concentrations do not stabilize until after 2100 (van Vuuren et al., 2011). Temperature projections for these emissions scenarios do not vary greatly prior to 2050, but begin to diverge in the latter half of the 21th century (Mote et al., 2011).

While the complexity and performance of CMIP5 models has improved over the previous generation of models (NOAA, 2014), the spatial resolution of most GCMs (typically 1-2 degrees) is too coarse to make predictions about the local impacts of climate change due the variability in weather exhibited at this scale (USBR, 2013). In order to make regional and local assessments regarding climate change, GCM are downscaled to the local level (USGS, 2014). There are two methods of downscaling GCMs: statistical downscaling and dynamic downscaling. Dynamic downscaling uses regional climate models that are based on the same processes simulated in the GCM, with the GCM acting as boundary conditions (Mearns et al., 2003; USBR, 2013; USGS, 2014). However, because dynamic downscaling uses the GCM as boundary conditions, biases present in the GCM will translate to the local scale (NOAA, 2014; USBR, 2016; USGS, 2014). Dynamic downscaling is a very computationally intensive process (NOAA, 2014; USGS, 2014; USBR, 2016) and as a result, dynamically downscaled datasets are not as widely available as statistically downscaled projections (NOAA, 2014). Statistical downscaling is based on developing statistical relationships between atmospheric processes in the GCMs and local observed climate data. The statistical relationships are then used to downscale the GCM projections to the local scale (Kunkel et al., 2013; USBR, 2013). Statistical downscaling allows for bias correction of the GCM and does not have the computational restrictions associated with dynamic downscaling (USBR, 2016). Statistical downscaling is dependent on the quality of the observed climate data. Because the downscaling is based in statistical relationships between local observed climate and the GCMs, the methods rely on the assumption that the current statistical relationships will hold in the future (NOAA, 2014; USBR, 2013; USGS, 2014). Because of the capability to downscale with high spatial resolution and relatively minimal computational demands, statistically downscaling methods can be more easily

applied to local studies (Mearns et al., 2003). There are numerous statistical methods developed for bias correction and statistical downscaling (Mearns et al., 2003; Pierce et al., 2014).

Selecting Climate Models for Impact Assessments

While it is not encouraged to disregard any emissions scenarios when conducting climate change assessments, since 2000, emissions have followed the RCP8.5 scenario closely (USBR, 2016). Additionally, without major emissions reductions in the immediate future, RCP2.6 appears unlikely to be achieved. Downscaled and bias-corrected climate change projections are available from a variety of sources (e.g. NASA, USBR). Projections corresponding to RCP4.5 and RCP8.5 emission scenarios are the most widely available for downscaled datasets.

A consortium of universities and government agencies headed by the U.S. Bureau of Reclamation (USBR) provide access to statistically downscaled datasets on 4 to 12 km grids across the United States (USBR, 2017). These projections provide outputs for minimum and maximum daily temperature and precipitation. Daily datasets are produced using the Localized Constructed Analogs (LOCA) and Bias Correction with Constructed Analogs (BCCA) downscaling methods (USBR, 2017). LOCA downscaled projections tend to have better replication of extreme events and reduces the issue of too many light precipitation days that tend to be problems with BCCA downscaled datasets (Pierce et al., 2014).

With the large (and expanding) number of climate model simulations available for climate change assessment, it is not practical to include all the models in climate change assessments (Lutz et al., 2016). Instead, several models should be selected that provide a good representation of the range of outcomes; however, there is no universal guidance on selecting representative scenarios. The most common methods for selecting downscaled projections for use in climate assessments or modeling studies can be categorized as uncertainty-based (also

referred to as envelope-based) methods and performance-based methods (USBR, 2016). Performance-based methods refer to selecting projections that best simulate observed historical climate. Uncertainty-based methods refers to selecting an ensemble of models that represent the range of possibilities in future climate projections (USBR, 2016). There is a tendency to favor models that simulate past climate well (i.e. performance-based selection) (Mote et al., 2011; USGS, 2014); however, there does not appear to be correlation between a model's ability to simulate historical climate and its accuracy in projecting future climate change (Knutti et al., 2010; Mote et al., 2011). In addition, this approach alone may lead to the elimination of possible climate futures (Lutz et al., 2016). While, comparing models to historical observation should not be overlooked, it should probably not be used as the only measure for model selection.

In some cases, the performance in comparison to historical data is dependent on the selection of the climate variables and no models perform the best for all variables (Flato et al., 2013). For example, ranking may shift slightly depending on the variables and metrics used for comparison (USGS, 2014). Other methods include using the mean of an ensemble of models (Knutti et al., 2010). While this approach provides a single model to use for assessments, it does not cover the range of possible scenarios that best represents the range of impacts (Lutz et al., 2016).

More advanced methods that combine past performance and uncertainty (e.g. Lutz et al 2016) may lead to more appropriate selections for downscaled climate projections, which represent the range of possible outcomes and also represent past climate reasonably well. Other more complicated methods for model selection include the incorporation of multivariate statistical methods much as SVD and hierarchical clustering (Barring and Wilcke, 2016), principal component analysis, and cluster analysis (Mendlik and Gobiet, 2016).

It is important to recognize that climate models are not intended to recreate the daily, monthly, or even yearly temperatures or precipitation observations (Flato et al., 2013). The objective of climate models is to recreate the central tendency of climate over a long time period due to changing input conditions (Mote et al., 2011). Therefore, climate model performance should not be based on comparing short-term metrics to observations. Instead, the climate models should be evaluated by how well the models recreate long-term climate measures and patterns, or whether they are able to simulate the frequency of extreme events. This is also the reason that long periods of time are compared (e.g. 1970 to 2000 vs. 2017-2100) because the models are simulating long-term trends or the changes in the statistics of future climate (Flato et al., 2013). 30 years is typically used because it is the interval used to define average conditions by the World Meteorological Organization (WMO, 2017).

DRAINMOD for Natural Wetland Hydrology Simulations

DRAINMOD is a hydrologic model that was developed to simulate subsurface drainage and water table fluctuations on low gradient, poorly drained agricultural soils with evenly spaced parallel drains (Skaggs, 1980). The model has since been applied to successfully simulate the daily water table fluctuations in natural, undrained non-riverine wetlands (e.g. He et al 2002; Caldwell et al 2007; Chescheir et al 2008; Caldwell et al 2011), as well as wetland restoration projects (Petru et al., 2014). One of the major advantages of DRAINMOD is its ability to simulate the water level fluctuations over long periods (i.e. 20-30 years) with only daily temperature minima and maxima and daily rainfall as inputs.

DRAINMOD simulations are based on a water balance approach for a unit area of soil at the midpoint between parallel drains. Two separate water balance calculations are employed: one

for the soil surface and the other in the soil profile. Water balance calculations in the profile use the following equation for each time-step.

$$\Delta V = D + ET + DLS - F \qquad \text{Equation 1.1}$$

Where ΔV is the change in drainable pore space (cm), D is drainage (cm), ET is evapotranspiration (cm), DLS is deep lateral seepage (cm) and F is infiltration (cm). Infiltration is calculated in the model using the Green and Ampt equation and drainage rate is calculated using the Hooghoudt and Kirkham equations, depending on the surface storage depth and water table height above ground surface, and ET is calculated from PET, depending on the water table depth in the profile and the upward flux. PET is calculated using the Thornthwaite method or observed values can be input as well (Skaggs, 1980). The water balance is calculated at the surface using the following equation for each time-step:

$$P = F + \Delta S + RO \qquad \text{Equation 1.2}$$

Where P is precipitation (cm), F is infiltration into the soil profile (cm), ΔS is the change in surface depressional storage (cm), and RO is runoff (cm), which occurs when ΔS exceeds the maximum surface storage.

Model inputs that are typically measured in the field include the saturated lateral hydraulic conductivity, drain depth and spacing, and depth to the impermeable layer. Common methods for measuring lateral saturated hydraulic conductivity include the Auger Hole Method (Van Beers, 1970) and the Compact Constant Head Permeameter (Topp et al., 1992). Vertical saturated hydraulic conductivity is measured using the constant head method (Klute et al., 1986). The soil water characteristic curve, which is used to calculate Green Ampt parameters and water table depth drained pore space relationship is developed using the standard pressure plate method (Klute 1986) with undisturbed soil cores.

Weather inputs for DRAINMOD include the hourly rainfall and daily minimum and maximum temperature (Skaggs, 1980). Because hourly rainfall is rarely available for long periods, DRAINMOD includes a utility package to disaggregate daily rainfall to hourly inputs. Daily rainfall is typically spread over a four-hour period that is split over the hours of 6 am or 6 pm (Skaggs et al., 2012). Long-term daily rainfall and temperature records are available from the North Carolina State Climate Office for many areas in North Carolina (State Climate Office of North Carolina, 2017). DRAINMOD uses the daily temperature extremes and the site latitude in the Thornthwaite Method to calculate potential evapotranspiration (PET). However, the Thornthwaite method tends to overestimate PET during the summer and underestimate PET during the fall, winter, and spring in North Carolina (Skaggs et al., 2012). To account for this inaccuracy, PET correction factors are used to adjust the PET values calculated in the model. The monthly PET correction factors are calculated as the ratio of the monthly PET calculated using the more accurate Penmen-Monteith method to the monthly PET calculated by the Thornthwaite method over a several year period (Skaggs et al., 2012).

Several different approaches have been used for calculating PET correction factors. Petru et al (2014) calculated PET correction factors using open water estimated of Penman-Monteith PET obtained from the North Carolina State Climate Office database. Others (e.g. Caldwell et al 2007) have used nearby weather data to calculate the Penman-Monteith PET. He et al (2002) used previously published PET correction factors and adjusted them during the calibration process; however, adjusting PET correction factors during calibration should be done with caution and adjustment should not exceed 15% (Skaggs et al., 2012).

DRAINMOD can be manipulated to simulate hydrology in wetlands. For simulations of natural wetlands, DRAINMOD is calibrated to minimize the difference between the observed

and simulated water table depths by adjusting the theoretical drain depth and spacing. Because natural wetlands do not have these features their adjustment in the model is treated as an additional calibration parameter (Caldwell et al., 2011). Other calibration parameters including hydraulic conductivity, surface storage, and some other parameters can be adjusted to obtain a better calibration as indicated in Skaggs et al (2012).

DRAINMOD has been successfully applied to simulate the long-term hydrology of undrained wetland systems including Carolina bays (Caldwell et al., 2007), mitigation projects (Petru et al., 2014), and other wetland types (Skaggs et al., 1994). These models have been used for various comparisons including setting goals for wetland restoration of specific wetland communities (Caldwell et al., 2011), planning restoration projects (Messer, 2015), correlating hydric soil indicators to long-term hydrology (He et al., 2002), and evaluating wetland performance criteria (Skaggs, 2012; Skaggs et al., 1994). In addition, DRAINMOD has been demonstrated as a tool to make hydrologic predictions of climate change for agriculture and forestry (Abdelbaki, 2015; Sun et al., 2000).

Calibration periods are typically more than one year (e.g. Caldwell et al 2007; Petru et al 2014). However, shorter periods are also feasible (He et al., 2002; Skaggs et al., 2012). Water table calibration and validation is evaluated by calculating goodness of fit measures between the observed and simulated water table depths. These measures often used include the Mean Absolute Error (MAE), the Nash-Sutcliffe Modeling Efficiency (EF) (Skaggs et al., 2012), Coefficient of Determination (R^2), and the Root Mean Squared Error (RMSE).

Research Objectives

The overall goals of this research were to quantify the condition of wetlands of North Carolina, to predict the possible future trends, to better quantify wetland functions, and to develop

guidelines and criteria and could improve wetland restoration, management, and regulation. This research could fill some of the knowledge gaps regarding wetlands in North Carolina and improve our understanding of wetland functions and conditions, which may lead to better restoration outcomes. Some of the specific objectives of the research include:

1. Characterize the hydropatterns of natural wetlands in North Carolina and compare the observed conditions to the current hydrologic criteria for wetlands mitigation
2. Develop proposed nutrient thresholds for reference conditions in North Carolina wetlands and compare wetland water quality to other aquatic systems
3. Compare the concentrations of Cu and Zn in North Carolina wetland soils to contamination thresholds
4. Assess the long-term impacts of climate change on two wetlands in the Coastal Plain of North Carolina

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Chapter 2: Hydrology of natural wetlands in North Carolina

Abstract

The wetland hydropattern is the most important factor governing wetland structure and function, but it is largely ignored in the design of wetland restoration projects. Hydrologic success criteria for wetland restoration is based a single measure of continuous saturation duration during the growing season; however, research has shown that wetland hydropatterns for different wetland types are not differentiable based on a single metric. Improper hydrologic regime has been cited as a leading reason restoration projects fail to achieve the structure and function of natural wetlands. What is needed is guidance for restoration design based on long-term data from natural, reference wetlands. The general objectives of this project were: (1) characterize the hydrology of the different wetland types and identify generalized hydrologic features that are representative of each wetland type, (2) determine if the hydropatterns were similar among the wetland types based on summary statistics and Empirical Cumulative Distribution Functions (ECDFs), (3) compare the measured saturation durations to the standards for wetland restoration projects and (4) compare long-term simulated water levels to these hydrologic criteria. Water levels were monitored for two and a half years in 16 natural wetlands across the Coastal Plain and Piedmont of North Carolina beginning in the summer of 2015. The sites included headwater, basin and riverine wetlands. Overall, all wetlands types generally had characteristics of water level above or near the surface during the winter and spring, while typically drawing down below the surface during the growing season, although there was considerable year-to-year variability. The observed variability clearly illustrated the need for long-term data to adequately characterize the hydrology of non-tidal, non-lacustrine wetlands. Summary statistics and ECDFs did not differentiate between the wetland types well, although maximum water levels due to overbank

flooding did tend to separate the riverine sites. Some general characteristics that were typical of each wetland type were identified. The observed saturation percentages generally far exceeded the criteria thresholds for restoration projects; however, the short monitoring period, variability in wetland type, and levels of anthropogenic disturbance limit the inferences on these standards. The long-term simulations indicated a wide range of saturation duration percentages. This study, along with previous research demonstrate the need for long-term data to improve hydrologic performance criteria for wetland restoration projects.

Introduction

Wetland hydrology is the most important factor governing wetland establishment, structure, function, evolution, and persistence (Fretwell et al., 1996). Hydrology acts as a control on the wetland physiochemical environment and biota, which in turn respond in a series of complex feedback mechanisms to influence wetland structure (Mitsch and Gosselink, 2007). The overall hydrologic character of a wetland is described by the wetland hydropattern. The hydropattern, as defined by US EPA (2008), is the pattern of water level fluctuations through time. This seasonal pattern of water levels is often referred to as the hydrologic signature of the wetland (Mitsch and Gosselink, 2007). Because of the hydrologic controls on wetland structure and function, hydrology is often considered to be the most important factor in the design of wetland restoration projects (Kentula and Kusler, 1989), and one of primary reasons some restoration projects fail to meet their objectives is the engineering of inappropriate hydrology for the target wetland type (National Research Council, 2001). For several decades scientists have sought to develop a better understanding of wetland hydrology that could be used in the design of restoration projects (e.g. Cole and Brooks, 2000; National Research Council, 2001). However, today there is still a shortage of information on the hydrology of natural wetlands and a lack of design guidance has negatively affected restoration outcomes (Brooks and Gebo, 2013). Currently the only metric used to determine whether wetland hydrology is considered successful in restoration projects is the U.S. Army Corp of Engineers (USACE) definition of wetland hydrology or some minor variation. This definition relies on a single measure of continuous saturation period during the growing season and does not take into account the entire hydropattern, which has been shown to be critical for supporting many important wetland functions.

Several previous studies have sought to improve how hydrologic success is measured in wetland restoration projects. Johnson et al. (2012, 2014) suggested using measures of wetland hydrology that were highly correlated with plant community composition. However, their use of ordination and multiple regression has little practical applicability to designing or evaluating restoration projects on a large scale. Several studies have grouped sites by wetland type and tried to assess if some summary measures of hydrology differentiate the wetland types (e.g. Cole and Brooks, 2000; Shaffer et al., 1999). Site medians were reported to be a good differentiating factor; however, upon closer examination, this conclusion was not well supported given a wide range in median values within each wetland type and relatively small differences between some wetland types. In addition, the median water level provides no information about the overall hydropattern and would not provide a useful goal in the design process. Caldwell et al. (2011) developed long-term hydrologic models to predict the average hydroperiod for specific wetland community types and recommended this approach be pursued further. While this approach was promising, the initial process of developing models is a resource intensive process and this approach may be difficult for some wetland types (e.g. flood prone riverine sites). Expanding upon the USACE's standards, the North Carolina Interagency Review Team (NCIRT) released new saturation duration targets for wetland restoration in 2016 that are soil type and ecoregion specific (NCIRT, 2016).

This project was originally started with the intention of building upon a database of high quality reference wetland water level data collected by the North Carolina Division of Environmental Quality Division of Water Resources (NC DEQ DWR). The objectives were to use 8-10 years of the NC DEQ DWR data and add 3 years to that dataset. This dataset would have been used to develop metrics to evaluate restoration projects that would better assess the

entire hydropattern as opposed to just the longest period of saturation that is currently used for mitigation projects. By improving guidance for restoration projects, some of the goals of the U.S. Environmental Protection Agency (US EPA) and the U.S. Army Corp of Engineers' (USACE) 2008 Compensatory Mitigation for Losses of Aquatic Resources; Final Rule (USACE, 2008) could have been advanced; mainly through the improvement of restoration performance standards based on reference data. However, analysis of the legacy data indicated problems with data gaps, sensor failure, and inconsistent field calibrations. Therefore, analysis was only completed on 3 years of data (2015-2018), which limited trend analysis and more in-depth characterization of wetland hydropatterns. As a result, the goals for this portion of the project had to be substantially scaled back. The revised primary objectives of this chapter were:

1. Provide a general overview and characterization of the hydrology of the different wetland types
2. Compare the wetland hydropatterns using summary statistics and empirical cumulative distribution functions
3. Compare the observed saturation percentages to the newly developed NCIRT standards for wetland mitigation projects in North Carolina
4. Compare the long-term simulated hydropatterns to the NCIRT standards and characterize average hydropatterns that could be used for the development of future hydrologic criteria

Materials and Methods

Study Sites

Sixteen natural wetland sites located in the Coastal Plain and Piedmont physiographic regions of North Carolina were monitored for this project (Figure 2.1). These sites were selected

by the NC DEQ DWR as part of a larger assessment of the condition of natural wetlands in North Carolina dating back to 2005. The sites were made up of three different wetland types: riverine, headwater, and basin. The sites were located in a wide range of surrounding land uses and displayed varying degrees of anthropogenic disturbance, including logging, road construction, ditching, agriculture, and upstream urbanization. There was a wide range of underlying soil series and soils textures range from mineral to mucky (Table 2.1).

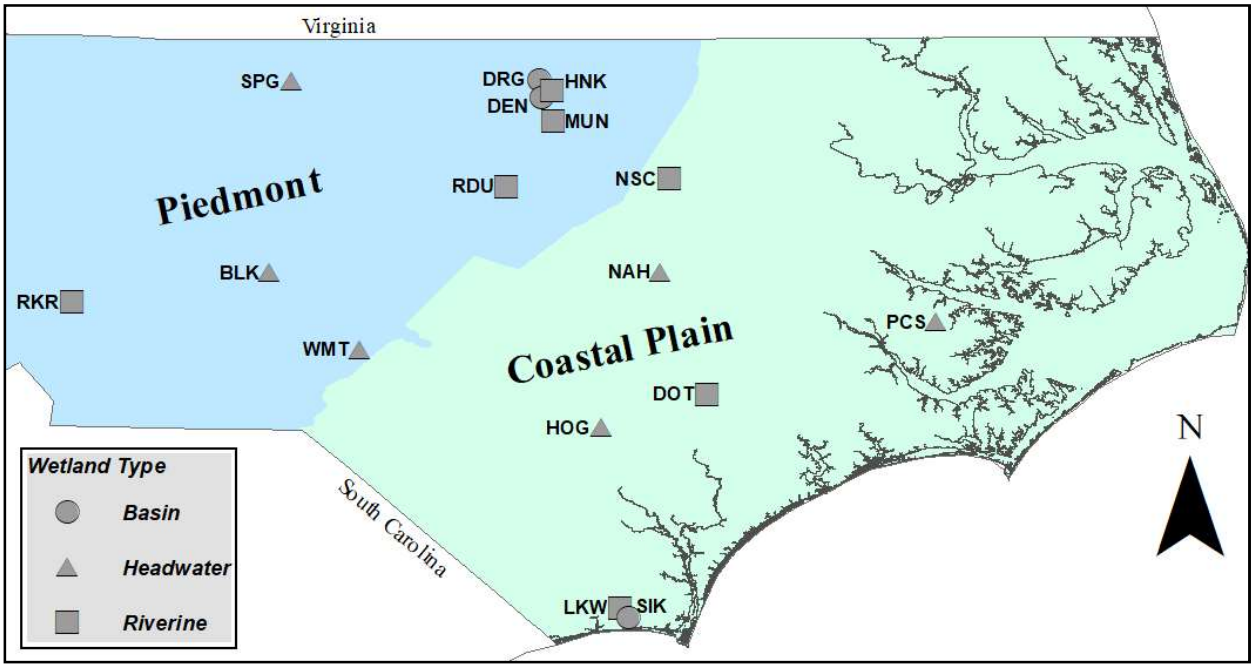

Figure 2.1: Site map with wetland type and site identifier code.

Table 2.1: Wetland site characteristics.

Site Code	County	Region	Wetland Type	Soil Series	Subgroup	Anthropogenic Disturbance
BLK	Montgomery	Piedmont	Headwater	Herndon	Typic Kanhapludults	Pasture; highway
SPG	Rockingham	Piedmont	Headwater	Codorus	Fluvaquentic Dystrudepts	Minimal
WMT	Moore	Piedmont	Headwater	Vauclose	Fragic Kanhapludults	Urban
RDU	Wake	Piedmont	Riverine	Chewacla	Fluvaquentic Dystrudepts	Upstream Development
RKR	Cabarrus	Piedmont	Riverine	Wehadkee	Fluvaquentic Endoaquepts	Upstream Development
HNK	Granville	Piedmont	Riverine	Chewacla	Fluvaquentic Dystrudepts	Upstream Development; highway
MUN	Granville	Piedmont	Riverine	Chewacla	Fluvaquentic Dystrudepts	Pasture; row crops
DEN	Granville	Piedmont	Depressional	Wehadkee	Fluvaquentic Endoaquepts	Row crops
DRG	Granville	Piedmont	Depressional	Wehadkee	Fluvaquentic Endoaquepts	Minimal
PCS	Beaufort	Coastal Plain	Headwater	Dorovan	Typic Haplosaprists	Minimal
HOG	Sampson	Coastal Plain	Headwater	Paxville	Typic Umbraquults	Pasture; spray field
NAH	Wayne	Coastal Plain	Headwater	Torhunta	Typic Humaquepts	Row crops; highway
DOT	Duplin	Coastal Plain	Riverine	Bibb	Typic Fluvaquents	Agriculture, drainage, development
NSC	Nash	Coastal Plain	Riverine	Wehadkee	Fluvaquentic Endoaquepts	Upstream agriculture + development
LKW	Brunswick	Coastal Plain	Tidal Riverine	Dorovan	Typic Haplosaprists	Upstream Development; highway
SIK	Brunswick	Coastal Plain	Depressional	Murville	Umbric Endoaquods	Minimal

All soil series descriptions and maps from Soil Survey Staff (2017)

Sampling Design and Methods for Water Level Monitoring

Two water table monitoring wells were installed to a depth of approximately 150 cm at each site following the US Army Corp of Engineers (USACE) monitoring well installation guidelines (USACE, 2005). The wells were installed in an upslope, downslope orientation in an effort to capture the range of hydrologic regimes at each site (i.e. one well was installed in the area likely to be inundated the longest and the other well was installed near the wetland edge) based on best professional judgement using topography and vegetation gradients as guides. The wells were equipped with HOBO water level data loggers (Onset Corp., Bourne, MA). The data loggers recorded measurements every hour. One logger was installed near the top of a well at each site to measure atmospheric pressure. The raw pressure files were processed into water level readings using HOBOware (Onset Corp., Bourne, MA). The data was downloaded seasonally, and manual water level measurements were recorded to adjust the water level data relative to the ground surface. The procedure depicted in Figure 2.2 was used to adjust for the raw water level data to ground surface. This method averages out any error or drift over time. Zero was set as ground surface, negative values were indicated below ground surface, while positive values indicated ponding on the surface.

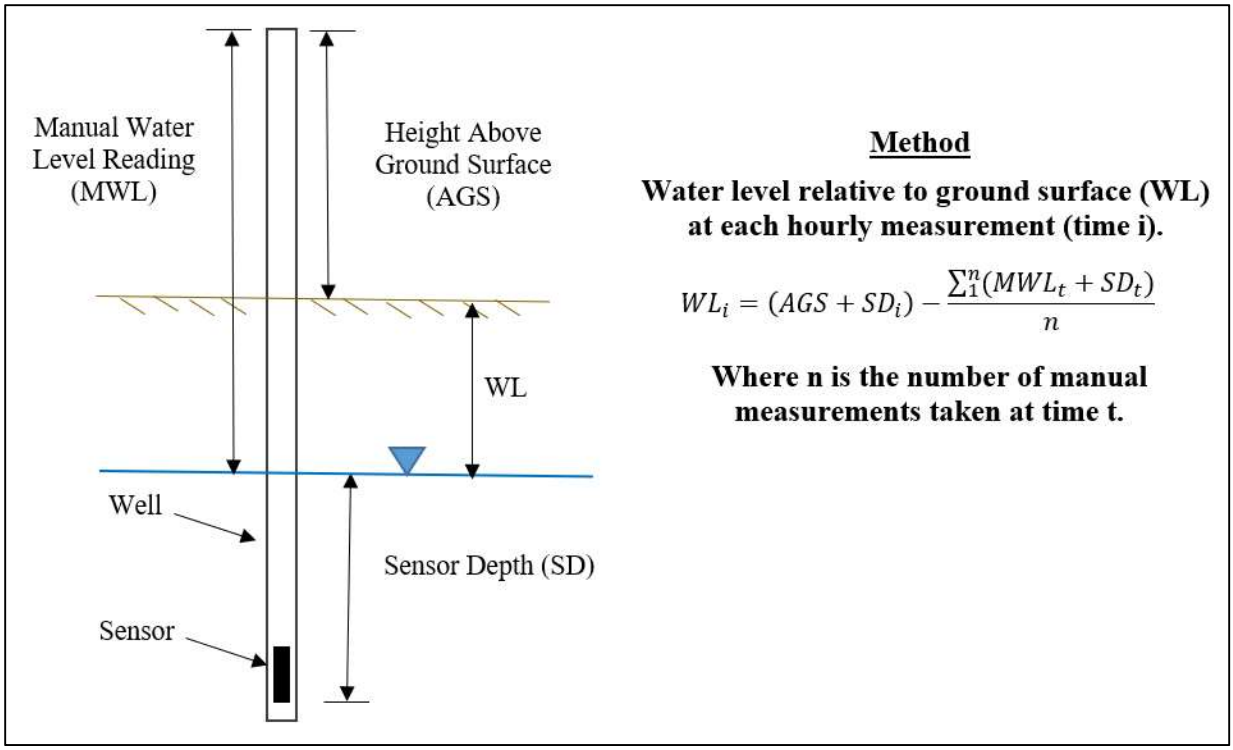

Figure 2.2: Procedure for adjusting sensor measurements to ground surface.

Hydropattern Analysis

Previous studies have compared summary measures of the hydropattern in an effort to assess whether wetlands of the same type have similar summary statistics, and whether these summary statistics are differentiable by wetland type (e.g. Cole and Brooks, 2000; Shaffer et al., 1999). For this analysis minimum, median, 25th percentile, 75th percentile, and maximum water levels during the growing season, along with percentage of the growing season when water level above the surface, above 30 cm depth, and below 30 and 60 cm depth were calculated and compared among wetland types.

Empirical cumulative distribution functions (ECDFs) are a valuable tool to visualize differences between hydrologic time series (Hychka et al., 2013), and can be used to easily visualize site-by-site differences and yearly differences, and where in the water level distribution these differences occur. ECDFs have been used to compare restoration sites to reference sites

(Menberu et al., 2016). For this project, ECDFs were compared for each site and growing season to assess if there were any clear differences in wetland type. Analysis was primarily completed on the downslope well or wetter area of the wetland (i.e. wet-end hydropattern) as this was assumed to show more differences than the marginal wetland edges. However, data for both wells was included in the Appendices.

Hydropatterns were plotted and the least disturbed sites were selected to draw some general inferences about wetland type specific generalized hydroperiods based on observations from this study and the literature. This was done in the form of a general description and a plot showing the typical pattern of water level fluctuations

Jurisdictional Hydrology Comparisons

The USACE definition of wetland hydrology is met if: "the site is inundated (flooded or ponded) or the water table is ≤30 centimeters below the soil surface for ≥14 consecutive days during the growing season at a minimum frequency of 5 years in 10 (≥50% probability)." (USACE, 2005). 14 days equates to about 5% of growing season in North Carolina. For mitigation projects, the saturation period (i.e. 14 days) is allowed to vary based on regional conditions (USACE, 2012). Historically, targets for this period have ranged from 5 to 12.5% of the growing season for mitigation projects in North Carolina (Hill et al., 2013). In 2016, the North Carolina Interagency Review Team (NCIRT) released new performance criteria for wetland mitigation projects. The new criteria set periods of continuous saturation based on the soil series underlying the restoration site. These new guidelines from NCIRT are an improvement in that a given soil series generally occurs in similar landscape positions (Richardson and Vepraskas, 2001), and there are different requirements for different ecoregions (i.e. Coastal Plain, Mountains, Piedmont) (NCIRT, 2016). More specifically, it recognizes that

soil types form in different landscape positions in response (in part) to hydrology. In order to make an adequate assessment if these standards are representative of the conditions in natural wetlands, several years of data from natural wetlands should be compared against the criteria. However, this is very rarely the case for academic studies, as most only last a few years (Cole and Kentula, 2011). For this study, a comparison of the two growing seasons of data were compared to the saturation criteria to get a general idea of the appropriateness of the new standards in comparison to natural wetlands. The growing season length was determined from WETS Tables. The growing season dates for each site and rainfall stations used can be found in Appendix D. The saturation percentages and all other calculations were completed in R 3.4 (R Core Team, 2017). The code is available in Appendix F.

Characterizing Long-Term Hydropatterns

Simulated long-term daily water level output from two validated and calibrated hydrologic models from Chapter 5 was used for this analysis. The two modeled sites were located in the Lower Coastal Plain and were not part of the 16 sites mentioned previously. The GDS site was a minimally disturbed area of the Great Dismal Swamp State Park in Camden County, NC. This site was a forested organic soil flat on Belhaven soils (Loamy, mixed, dysic, thermic Terric Haplosaprists) (NRCS, 1995). The site was primarily vegetated with cypress (*Taxodium distichum*), black gum (*Nyssa sylvatica*), and red maple (*Acer rubrum*). The NRF site in Carteret County, NC was a bottomland hardwood wetland on Deloss soils (Fine-loamy, mixed, semiactive, thermic Typic Umbraquults) (NRCS, 1987). The primary vegetation was black gum (*Nyssa sylvatica*), red maple (*Acer rubrum*) and redbay (*Persea borbonia*). Both sites were relatively undisturbed and had been used as reference sites for wetland restoration projects. The hydrology of the sites was modeled using DRAINMOD, a hydrologic model designed to

simulate agricultural drainage and water level in flat landscapes with parallel drains (Skaggs et al., 2012). DRAINMOD has been successfully adapted to simulate the daily water level in natural wetlands (e.g. Caldwell et al., 2007). More information on the site description and modeling process can be found in Chapter 5.

To analyze the simulated long-term water levels, the longest period of continuous saturation was calculated and compared to the NCIRT standards and the range of saturation durations was evaluated. Next, the hydropatterns and ECDFs for each year were plotted and percentiles (30th and 70th) of the hydropatterns were calculated to represent the bounds of average conditions. Finally, the year-to-year variability was analyzed and the implications for restoration were discussed.

Results and Discussion

Hydropatterns by Wetland Type

Piedmont Basin Wetlands

Basin wetlands occur in topographic depressions. The dominant water source is typically precipitation and the primary output is evapotranspiration. Surface inflow and outflow are generally limited and only occurs during periods of very high water level (Figure 2.3). In the Piedmont ecoregion, these small depressions generally occur where water is ponded in low areas due to the presence of an impermeable substrate. These systems are generally seasonally to semi-permanently flooded, which is characterized by prolonged inundation in the winter and spring and drawdowns during the summer months (Schafale and Weakley, 1990). The hydrologic regimes of these depressions allow for specific, unique functions. For example, basin wetlands are particularly important for amphibian habitat because they have standing water during the

amphibian breeding season and then dry out during the summer, which prevents the establishment of predatory fish (Batzer and Boix, 2016).

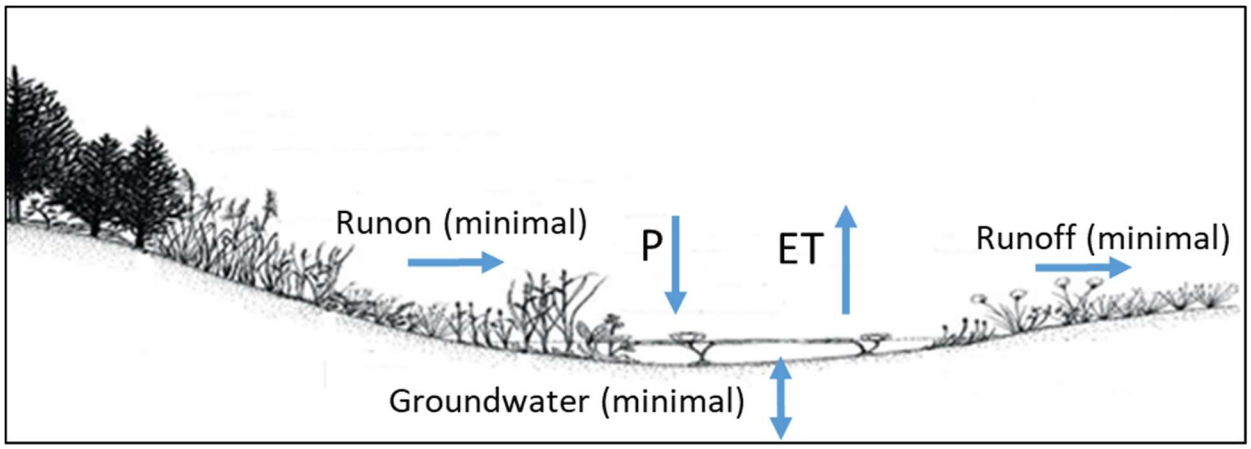

Figure 2.3: Cross sectional schematic and water budget components for basin wetlands (adapted from Gracz (2011)).

The DRG site was an isolated basin wetland located in the middle of a shrub/forested area in Granville County, NC. The basin was about a hectare in size and about 50 cm deep near the center (Figure 2.4). The area surrounding the wetland was logged in 2013. The areas most often inundated were vegetated with some aquatic macrophytes and submerged aquatic vegetation. The hummocks and shallow edge areas supported some red maple (*Acer rubrum*), oak (*Quercus spp*.) and small shrubs such as blueberry (*Vaccinium corymbosum*).

Figure 2.4: DRG site during June site visits.

The DEN site also located in Granville County, NC, was similar to the DRG site, but the basin was shallower with maximum depth of about 20-25 cm. The wetland was slightly lower topographically than the surrounding landscape and here was an elevated outflow so surface runon and runoff were more of a factor in the water budget at DEN. This likely contributed to generally larger increases in water level due to rain events (Figure 2.5). The woody vegetation species were similar to the species found at the DRG site. However, the DEN site and surrounding area were logged in 2015 and some of the basin, and the runon patterns to the basin, were severely altered by heavy equipment. The area monitored, which was less than half a hectare) remained undisturbed.

Both sites are characterized by periods of inundation in the winter and spring followed by a drawdown and variable dry period in the summer and/or fall. These two sites likely approximate the upper and lower range of the hydropatterns for Piedmont basin wetlands (Figure 2.5). These hydropatterns illustrate how hydrology controls specific wetland functions. For these isolated basin wetlands in the Piedmont, it is critical that there is standing water in the late winter and early spring for amphibian breeding habitat. In addition, these systems dry out for in the summer and fall, which prevent the establishment of fish species that would pose a risk to eggs and larval amphibians.

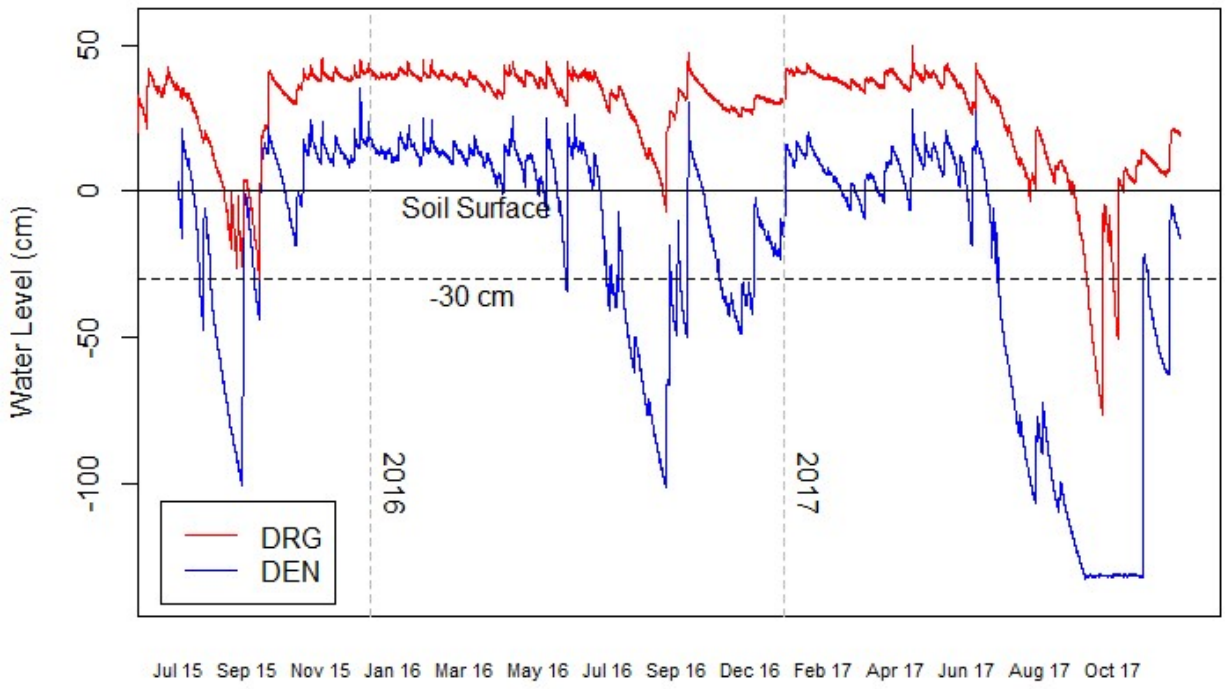

Figure 2.5: Wet-end hydropatterns for DEN and DRG Piedmont basin wetland sites.

This hydrologic control on amphibian habitat function of these basin wetlands was better illustrated at the DRG site, which had a slightly longer period of record than the other sites (5 years). This hydropattern data (Figure 2.6) indicated a range of annual variability in the timing of drawdowns and an interannual range of water table fluctuations. However, for each year there was inundated conditions in the winter and early spring and a drawdown and dry period in the summer/fall period. As previously mentioned, both of these characteristics are critical for amphibian habitat function.

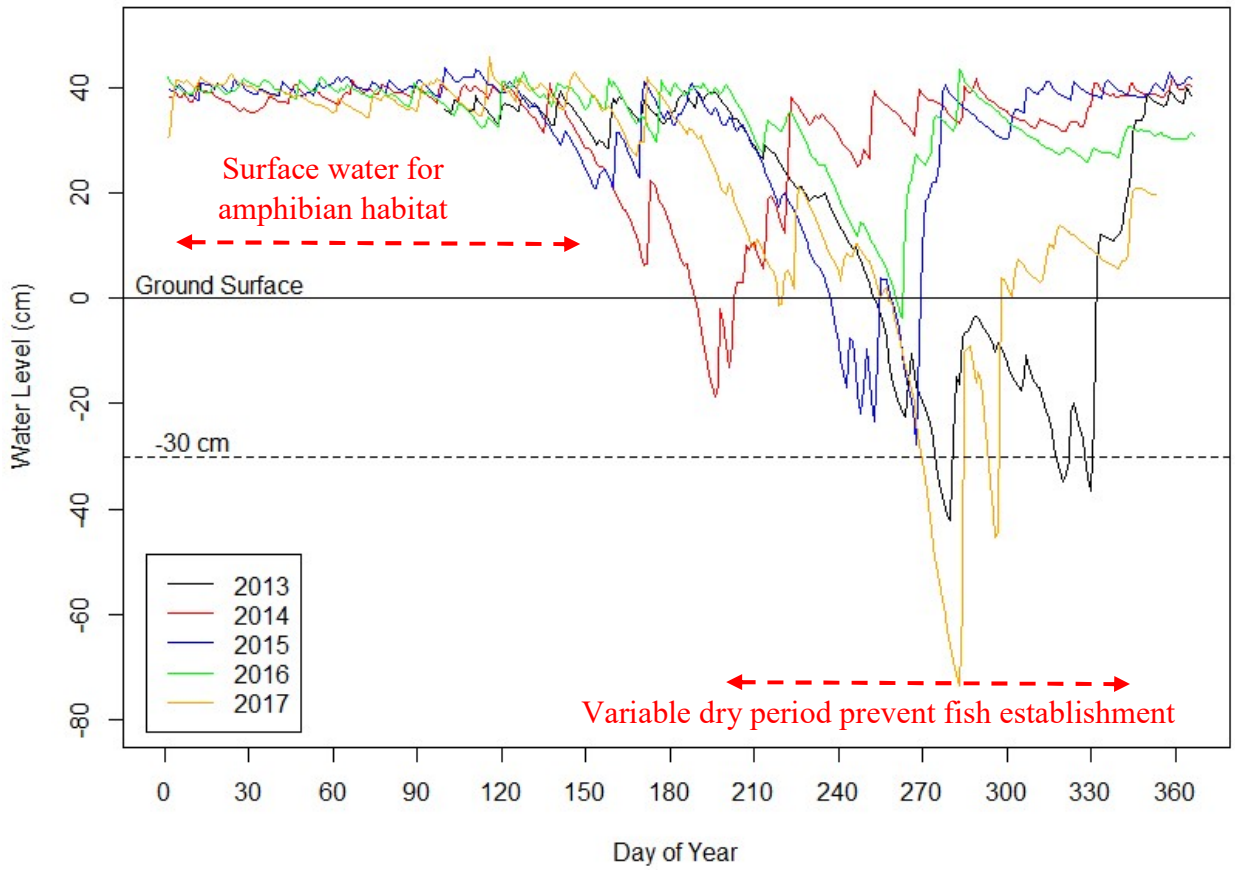

Figure 2.6: Yearly hydropattern at the DRG site.

In terms of restoring or creating a wetland with the functions of a basin wetland in a restoration project, it would be critical that the hydrology was similar to the natural system with continuous inundation of depth greater than approximately 15 cm and then a drawdown below the surface for some period in the summer and fall. This type of restoration could be used as a small component of larger non-riverine swamp forest or riverine floodplain restoration projects.

Carolina Bay Wetlands

Carolina Bays are oval shaped depressions that are oriented in a northwest – southeast direction along the long axis of the basin and typically have a sandy rim along the edge of the basin. They range from Georgia to Maryland along the Atlantic Coastal Plain. The basins range in size from less than a hectare to hundreds of hectares. Carolina Bay wetlands encompass a wide range of ecosystem types with hydrologic regimes ranging from nearly permanently

inundated to conditions where the water table is frequently below the surface. A wide range of vegetative communities occur in Carolina Bay ecosystems (Sharitz, 2003). The water budgets are typically similar to those shown for the basin wetlands (Figure 2.3), but with more groundwater interaction in some cases. Carolina Bays are typically precipitation driven systems although groundwater interaction can be a significant process at some sites (Caldwell et al., 2007; Sharitz, 2003).

The Carolina bay in Brunswick County, NC monitored was about 30 hectares and there was a wide range of hydrologic conditions observed across the basin. There were some very deep depressional areas (>1 m) although average maximum depth was probably closer to 20-25 cm. There was evidence that this site had been drained in the past. The site was logged within the last 10 years and the tree cover was primarily composed of 3-5 m pond pine (*Pinus serotine*), youpon holly (*Ilex vomitoria*), bald cypress (*Taxodium distichum*), which were 3-5 m in height. This bay appears to receive runoff, which was routed through ditches and culverts from a nearby partially completed housing development. This likely impacted the wetland hydropattern. For example, the rapid rises in water level were likely the result of runoff entering the site during storm events (e.g. Sept. 15, Sept 16, and May 17).

The hydropattern of the Carolina Bay was similar to the Piedmont basin wetlands with inundated conditions in the winter and spring and then a drawdown over the late growing season (Figure 2.7). Caldwell et al. (2007) observed a similar range of fluctuations over two-years of monitoring of several Carolina Bays in North Carolina. The overall hydropattern and timing of drawdowns were also similar, although lower water levels were observed in some sites, which would support the variability in Carolina Bay ecosystems indicated by Sharitz (2003).

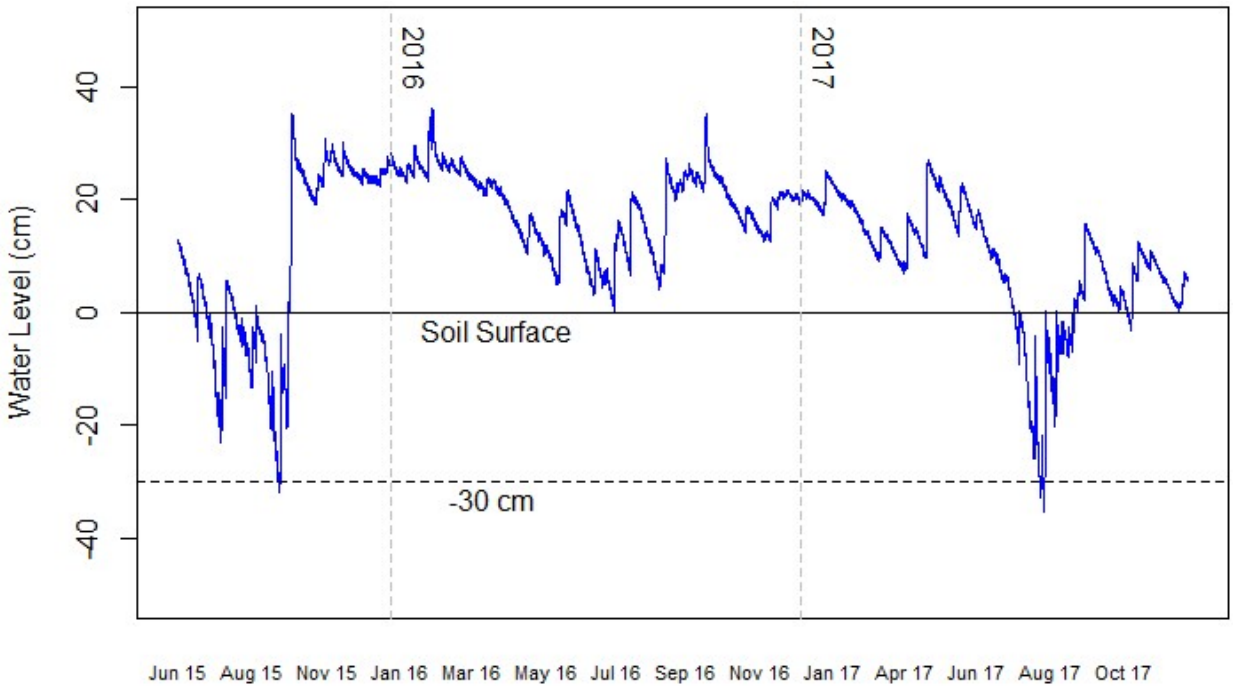

Figure 2.7: Hydropattern of the Carolina Bay wetland.

The basin wetlands in both the Coastal Plain and Piedmont had very similar patterns of inundation and drawdown. However, the depth of ponding observed varied, which was the result of basin morphology and even well placement. The water table elevations during the drawdown portion of the growing season were fairly shallow in the Carolina Bay, likely due to the low gradient landscape and high regional groundwater table in the Coastal Plain. Although the hydropattern of the Carolina Bay indicated it might provide suitable amphibian habitat in some areas, this was not the case due to very low surface water pH (<4.0). All three of the depressional wetland sites were logged within the last 10 years, which likely contributed to wetter conditions because of decreased transpiration (e.g. Dubé et al., 1995). This likely equated to slower drawdowns and shorter periods of dry conditions.

Headwater Wetlands

Headwater wetlands are generally shallow bowl-shaped areas in the upper reaches of watersheds where groundwater discharges and eventually forms first order streams. Both surface

water runoff and groundwater are important inputs to headwater wetlands (Figure 2.8). The relative contribution of each was largely related to the landscape gradient and local geology. Groundwater discharge can be a significant process in many headwater wetlands (Roulet, 1990), and thus surface water levels cannot always be directly explained by recent precipitation patterns. Groundwater discharge was observed in all the headwater sites where seeps formed small stream channels in the Piedmont and more braided channel systems or diffuse surface flow in the lower gradient Coastal Plain.

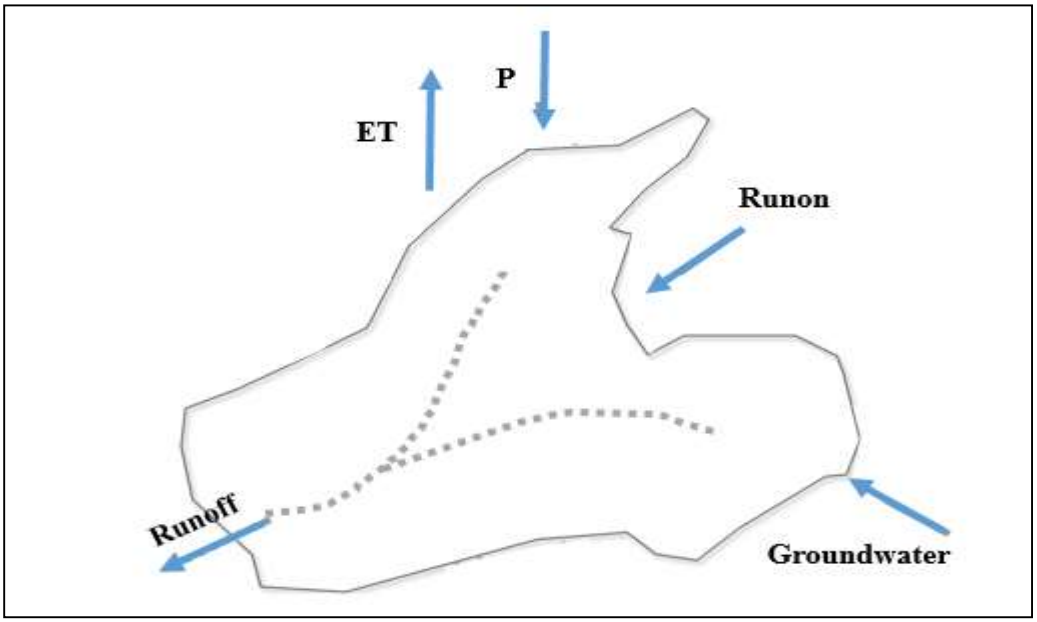

Figure 2.8: Plan view schematic and water budget components of a headwater wetland (adapted from MTU (2016)).

Coastal Plain headwater wetlands were the focus of this section because of the high level of disturbance in the Piedmont wetlands (the data can be found in Appendix A). The PCS site in Beaufort County, NC was a relatively flat, small (<2 hectares) headwater area located in the lower Coastal Plain. The wetland was underlain by deep organic soils and the dominant vegetation was redbay *(Persea borbonia*), red maple, (*Acer rubrum*) sweet bay (*Magnolia virginiana*), American holly (*Ilex opaca*), and swamp tupelo (*Nyssa biflora*). The upland

surrounding the area was largely forested, however there was some residential development within 50 m of the site. The site is relatively flat with some hummock and hollow topography and subsidence appears to have been a factor in the past (i.e., there were exposed tree roots in many areas). A road constructed downslope of the wetland concentrated outflow through a culvert under the roadway.

The NAH site in Wayne County, NC was very similar to the PCS site in that road construction had concentrated outflow to a culvert and it was relatively small (<1 hectare). The site received runoff from agricultural fields directly upslope of the wetland. Tree species were dominated by sweet gum (*Liquidambar styraciflua*), tulip poplar (*Liriodendron tulipifera*), red maple, (*Acer rubrum*), and tag alder (*Alnus serrulata*). Much of the under and mid-story at the site had been invaded by invasive Chinese privet (*Ligustrum sinense*). There was a histic epipedon overlaying sandy soils and there was distinct microtopography with many hummocks and hollows across the wetland.

The topography of the HOG site in Sampson County, NC was much steeper than the PCS or NAH sites. This combined with the sandy soils resulted in relatively low water levels and more rapid drainage. Dominant mature species include tulip poplar (*Liriodendron tulipifera*), red maple (*Acer rubrum*), and loblolly bay (*Gordonia lasianthus*). However, the understory and mid-story of the wetland and surrounding area were nearly monocultures of Chinese privet (*Ligustrum sinense*). The wetland was directly downslope from a hog waste spray field so some of the water table fluctuations may not have been the result of precipitation or groundwater inflow.

Figure 2.9: Coastal Plain headwater wetlands. NAH site (left) and PCS site (right).

The headwater wetlands also exhibited distinct seasonal hydropatterns with water tables highest in the winter and spring and then drawdowns during the growing season. Water levels were generally within 30 cm of the surface from October through July and then generally below 30 cm in August through October (Figure 2.10). Water tables below 30 cm were not observed in the lower Coastal Plain organic soil wetland (PCS) over this two-year study. This was likely due to the higher regional water table and higher storage capacity of organic soils (Mitsch and Gosselink, 2007). The single very high water level observed at the PCS site was because of localized flooding due to Hurricane Matthew. Lower water levels were observed in the steeper gradient sites (HOG site) and Upper Coastal Plain site (NAH site).

The PCS and NAH sites may have been wetter due to roads constructed downslope from the wetlands that impounded water in the lower areas of the wetland. The comparatively dry conditions at the HOG site (water table was never above the surface) was likely the result of a steeper gradient and the high permeability of the sandy soils. These two Coastal Plain headwater sites (PCS and NAH) that were impacted by road construction exhibited wet-end hydropatterns that were very similar to dryer basin wetlands, with shallow inundation during the winter and early spring before drawing down during the summer (Figure 2.10).

For the NAH site, the lower minimum water level observed in 2016 and 2017 (Figure 2.10), when compared to 2015, was likely the result of the outlet cleaning in March of 2016. All of these sites had some level of anthropogenic disturbance, which likely altered the hydropatterns to some extent. However, a site like PCS may represent one of the best available headwater sites given the extensive agricultural and silvicultural development in similar Coastal Plain landscapes.

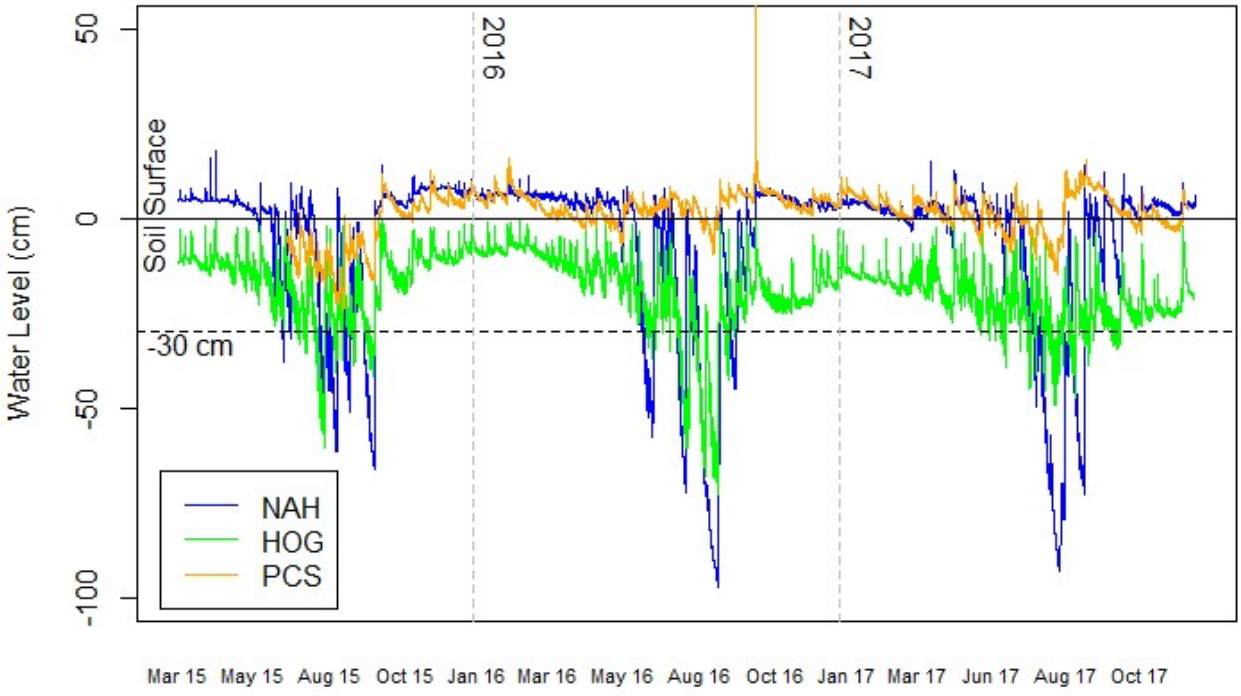

Figure 2.10: Wet-end hydropatterns for headwater wetlands in the Coastal Plain ecoregion.

Riverine Wetlands

Riverine wetlands are located adjacent to and on the floodplains of higher order streams and rivers. These systems receive and store floodwater during overbank events but also receive runon from surrounding uplands and tributaries and can function to recharge and discharge groundwater (Figure 2.11). The water budget is typically dominated by overbank flow events in riverine wetlands (Mitsch and Gosselink, 2007). One characteristic that separated riverine wetlands from the other wetland types in this study was the higher maximum water levels that

resulted from overbank flooding events. While these events are typically brief, they can have a major impact on the composition of wetland vegetation over the long term (Bledsoe and Shear, 2000). Depending on the size of the floodplain and landscape gradient, riverine wetlands can be made up of many different characteristics and the hydrologic regime can vary substantially across the floodplain. For example, large floodplains typically include levees, backswamps, various depressional areas from relic channels, terraces, and small streams (Richardson and Vepraskas, 2001).

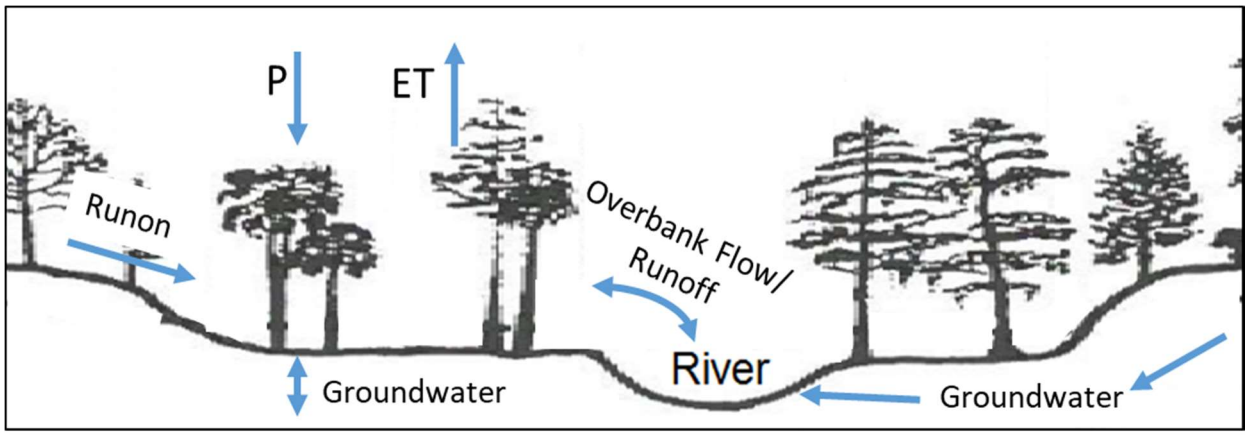

Figure 2.11: Cross sectional schematic and general water budget components for riverine wetlands. Adapted from Richardson and Vepraskas (2001).

There was substantial variability in the location, size of the wetland watersheds, width of the floodplains (Table 2.2) and wetland sizes among the riverine wetland sites, which likely limited the comparisons among these wetlands. The results should be interpreted in this context.

Table 2.2: Riverine wetland watershed area and floodplain width.

Site	Ecoregion	Watershed Area (Sq. km)	Floodplain Width (km)
DOT	Coastal Plain	984	1.1
NSC	Coastal Plain	34	0.25
HNK	Piedmont	49	0.1
MUN	Piedmont	21	0.1
RDU	Piedmont	13	0.15
RKR	Piedmont	210	0.25

Both the Coastal Plain riverine wetlands, NSC in Nash County, NC (Upper Coastal plain) and DOT in Duplin County, NC (Lower Coastal Plain) were backswamp areas of the floodplain complex. An additional site in Brunswick County, NC on the Lockwood Folly River (LWK) was not included in this discussion because it was tidally influenced (see Appendix A). The DOT site was part of the floodplain of the Northeast Cape Fear River and covered hundreds of hectares. Drainage water from adjacent agricultural fields was routed into portions of the site through a system of ditches and canals. The site had distinct microtopography with hummocks and hollows throughout. Some of the prominent tree species inhabiting the hummocks were red oak *(Quercus laurifolia)*, sweetgum (*Liquidambar spp.)*, water tupelo *(Nyssa aquatica)*, red maple (*Acer rubrum)*, and bald cypress (*Taxodium distichum*).

The upper coastal plain site (NSC) was substantially smaller than the DOT site; the floodplain was about 1/5 as wide and drainage area was about 1/30 the size of the DOT site. In addition, there was a much steeper elevation gradient from the wetland to the upland compared to other low gradient landscape of the mid and Lower Coastal Plain. During the winter months, flow through the site was observed regularly, which indicated the stream channel had relatively frequent connection with the floodplain (i.e. frequent overbank flooding). The dominant tree

species were red maple (*Acer rubrum*), sweet gum (*Liquidambar spp.)*, oak *(Quercus spp.)*, and ash (*Fraxinus spp.*),

Figure 2.12: Coastal Plain riverine wetlands. NSC site (left) and DOT site (right) during July site visits.

For the Piedmont riverine sites, the narrower floodplains and steeper gradients likely allowed for faster drainage, lower overall water table, and deeper drawdowns. The RDU and RKR sites were similar in that they both had mature hardwood canopies, similar sized floodplains, and severely incised stream channels that were likely the result of intense upstream development. Sediment deposition was a significant process at some of these sites where several inches of deposition was observed from single storm events. The tree species at these sites were dominated by bottomland hardwood species such as elder, ash, gum, maple, and oak.

The other two sites, HNK and MUN (both in Granville County, NC) were unique in that portions of the site resembled basin wetlands. The MUN site was a small (<1 hectare) floodplain depression in the backswamp that remained inundated for most of the year. The depression appeared to stay wet due to groundwater seepage from the base of the adjacent slope and runoff from fields in the upland. The HNK site was impacted by road construction and sewer line installation. This led to ponded water in the lower area of the wetland for long periods. The canopy at the HNK site was dominated by ash trees, although many of these trees were dead.

Invasive Japanese stiltgrass (*Microstegium vimineum*) was prominent in the understory at all of the Piedmont riverine sites monitored for this project.

Figure 2.13: Piedmont riverine wetlands. MUN site (left) and RDU site (right).

Overall, the riverine sites displayed similar patterns to the other wetland types with water level above or near the surface for the winter and spring and then a summer and fall drawdown period. The Lower Coastal Plain site (DOT) had water levels above the surface for a majority of the year, while water levels during the driest periods were much closer to the surface than for the upper Coastal Plain site (NSC) (Figure 2.14). However, this was dependent on rainfall. Conditions between the sites were very similar in 2017, which had a wetter spring and early summer that likely limited drawdown depth.

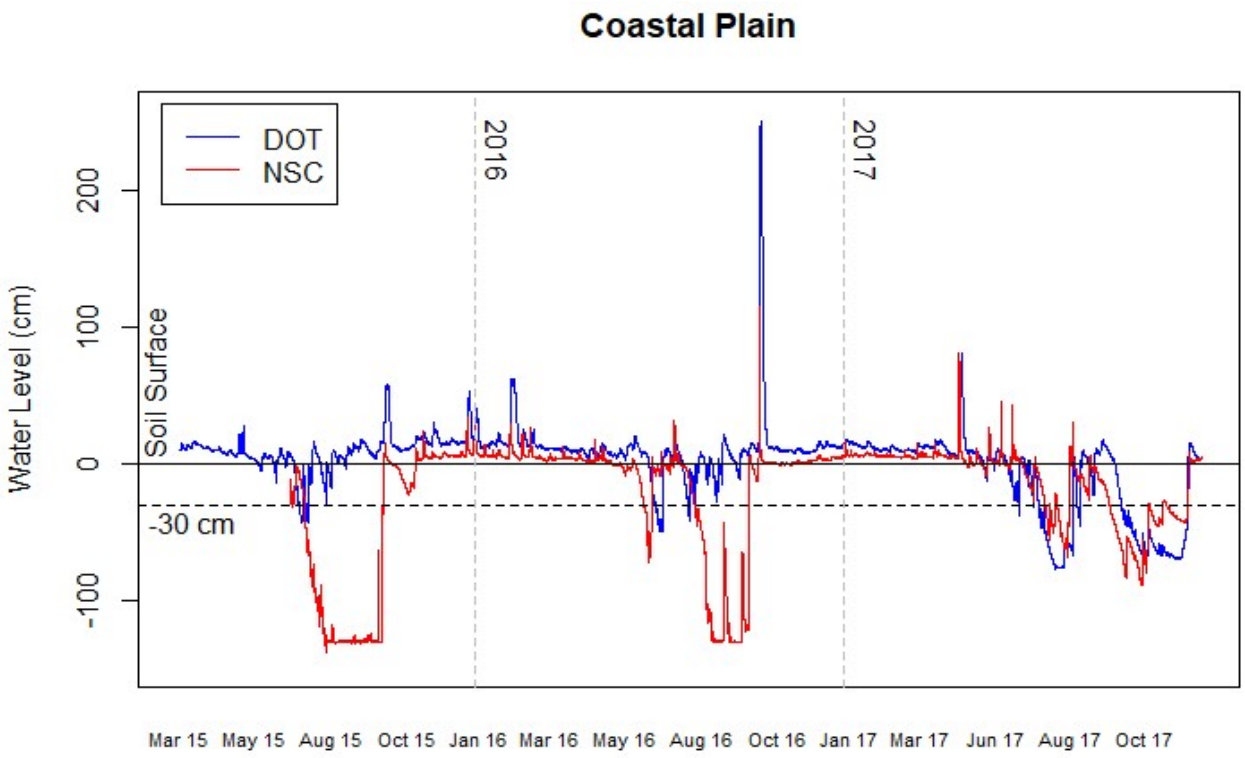

Figure 2.14: Hydropatterns for two riverine wetlands in the Coastal Plain ecoregion.

For the Piedmont riverine wetlands, that did not have floodplain depressions (RKR site in Cabarrus County and the RDU site in Wake County), the water level fluctuation were greater than the coastal plain sites. Many of these sites appeared to be dryer than they historically might have been due to deep channel incision. This was likely the result of increased peak flows due to development upstream. The increased peak flow may have led to faster drawdown following flood events due to channels in the floodplain that allowed for rapid drainage. The deep channel incision at these sites likely caused more rapid groundwater table recession after rain events because of the larger head difference between the floodplain and stream. Because of development and agricultural practices upstream, sediment loading was observed as a significant process that, over time, will likely modify the hydrology as the topography of the wetland was altered (Mitsch and Gosselink, 2007). For example, the highly developed Rocky River watershed (RKR site) had seven overbank events in 2017 and five in 2016, much greater than the typical

1.5-year recurrence interval that is generally assumed for bankfull events (Mccandless and Everett, 2002). These extreme events are important for influencing wetland plant species composition and can alter the wetland morphology through sediment deposition (Mitsch and Gosselink, 2007).

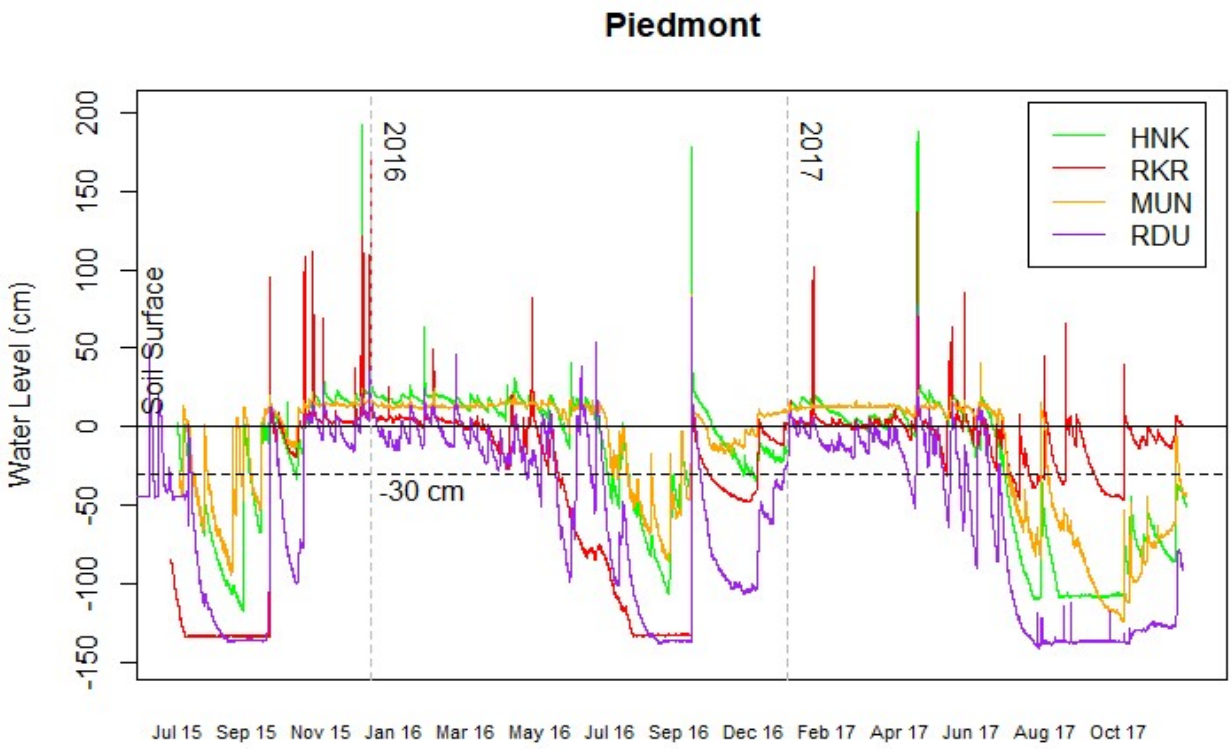

Figure 2.15: Hydropatterns for riverine wetlands in the Piedmont ecoregion.

Comparison of Wetland Types

There was no distinct grouping among the site medians or quartiles that separated the sites by region or type over the two-year study. The median growing season water levels for the wet areas of these wetlands ranged from -109 to +9 cm for the riverine sites, -27 to +3 for the headwater sites and -57 to +35 cm for the basin wetlands. However, for the most part, median water levels were between -30 and +10 cm (

Table **2.3**). There were also substantial differences in year-to-year median water level for

Type	Site	Region	Minimum (cm)		25th Percentile (cm)		Median (cm)		75th Percentile (cm)		Maximum (cm)	
			2016	2017	2016	2017	2016	2017	2016	2017	2016	2017
Basin	DRG	Piedmont	-7	-76	31	4	35	21	39	37	48	50
	DEN	Piedmont	-101	-133	-35	-119	2	-57	12	9	31	28
	SIK	Coastal Plain	0	-35	12	2	17	9	22	14	35	27
Headwater	BLK	Piedmont	-96	-97	-46	-88	-27	-23	-23	-21	-3	-6
	SPG	Piedmont	-5	-7	-1	-3	0	-2	1	-1	3	2
	WMT	Piedmont	-28	-28	-23	-23	-22	-22	-21	-21	-8	-11
	HOG	Coastal Plain	-73	-49	-25	-26	-19	-23	-13	-18	0	-2
	NAH	Coastal Plain	-97	-93	-17	-11	3	1	5	4	62	15
	PCS	Coastal Plain	-9	-15	1	-2	3	1	5	4	72	16
Riverine	DOT	Coastal Plain	-50	-77	4	-46	9	-2	11	8	251	81
	NSC	Coastal Plain	-131	-89	-32	-38	-1	-7	2	4	115	81
	HNK	Piedmont	-107	-110	-31	-107	7	-53	14	7	178	187
	RDU	Piedmont	-138	-142	-97	-137	-47	-109	-14	-20	82	78
	RKR	Piedmont	-134	-46	-117	-28	-39	-6	-10	2	82	137
	MUN	Piedmont	-86	-124	-22	-79	9	-23	14	12	84	54
	LKW	Coastal Plain	-7	-7	1	1	2	3	7	7	131	87

some of the sites, but 2017 was generally dryer with a few exceptions. The plots of hydropatterns for all the individual sites can be found in Appendix A and the comparison of plotted hydropatterns by type can be found in Appendix B.

There was no apparent grouping observed among the plotted ECDFs (Figure 2.16). The ECDFs clearly illustrated the considerable year-to-year varaibility in some of the sites. ECDFs for each site can be found in Appendix C. This observed overlap of the summary statistics and general overlap of the wetland hydropatterns could be due to the small sample size, the wide variability in location and size among the sites, as well as the varying levels of anthropogenic disturbance. Cole and Brooks (2000) indicated that separating wetland types using summary statistics tended to be the most challenging for disturbed sites.

Table 2.3: Growing season summary statistics for the wet-end hydropatterns at each site.

Type	Site	Region	Minimum (cm)		25th Percentile (cm)		Median (cm)		75th Percentile (cm)		Maximum (cm)	
			2016	2017	2016	2017	2016	2017	2016	2017	2016	2017
Basin	DRG	Piedmont	-7	-76	31	4	35	21	39	37	48	50
	DEN	Piedmont	-101	-133	-35	-119	2	-57	12	9	31	28
	SIK	Coastal Plain	0	-35	12	2	17	9	22	14	35	27
Headwater	BLK	Piedmont	-96	-97	-46	-88	-27	-23	-23	-21	-3	-6
	SPG	Piedmont	-5	-7	-1	-3	0	-2	1	-1	3	2
	WMT	Piedmont	-28	-28	-23	-23	-22	-22	-21	-21	-8	-11
	HOG	Coastal Plain	-73	-49	-25	-26	-19	-23	-13	-18	0	-2
	NAH	Coastal Plain	-97	-93	-17	-11	3	1	5	4	62	15
	PCS	Coastal Plain	-9	-15	1	-2	3	1	5	4	72	16
Riverine	DOT	Coastal Plain	-50	-77	4	-46	9	-2	11	8	251	81
	NSC	Coastal Plain	-131	-89	-32	-38	-1	-7	2	4	115	81
	HNK	Piedmont	-107	-110	-31	-107	7	-53	14	7	178	187
	RDU	Piedmont	-138	-142	-97	-137	-47	-109	-14	-20	82	78
	RKR	Piedmont	-134	-46	-117	-28	-39	-6	-10	2	82	137
	MUN	Piedmont	-86	-124	-22	-79	9	-23	14	12	84	54
	LKW	Coastal Plain	-7	-7	1	1	2	3	7	7	131	87

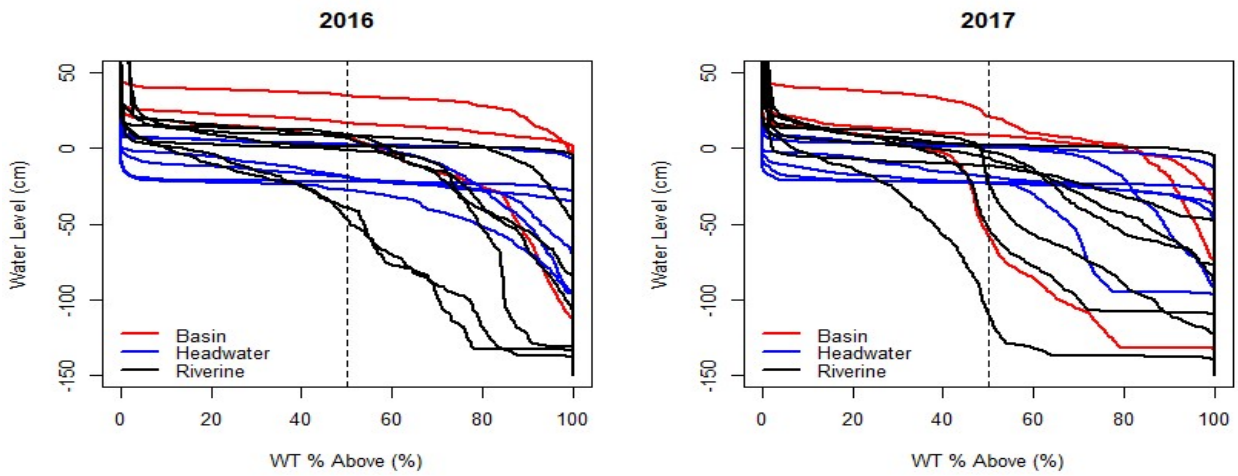

Figure 2.16: Comparison of growing season ECDFs for the wet-end wells of the different wetland types.

Some of the failure to differentiate the wetlands based on summary measures of hydrology was likely the large geographic distance between the sites and the relative uniqueness of some of the sites. Some of the sites were very small (<2 hectares) and likely atypical for the given wetland type. For example, the riverine MUN site was very dissimilar from the other riverine wetlands monitored. This wetland was actually a backswamp depression that received groundwater discharge and some runoff similar to headwater wetlands, yet still received water from occasional overbank flooding, and has long periods of relatively constant inundation much like the basin wetlands. Hydropatterns of selected sites from the three different wetland types were plotted together to illustrate this phenomenon in which the other wetland types resemble the attributes of a basin system (Figure 2.17). All of the sites have the same general pattern, but with varying depths of inundation. The headwater site here (NAH site) was impacted by road construction, which may contribute to the long-periods of standing water.

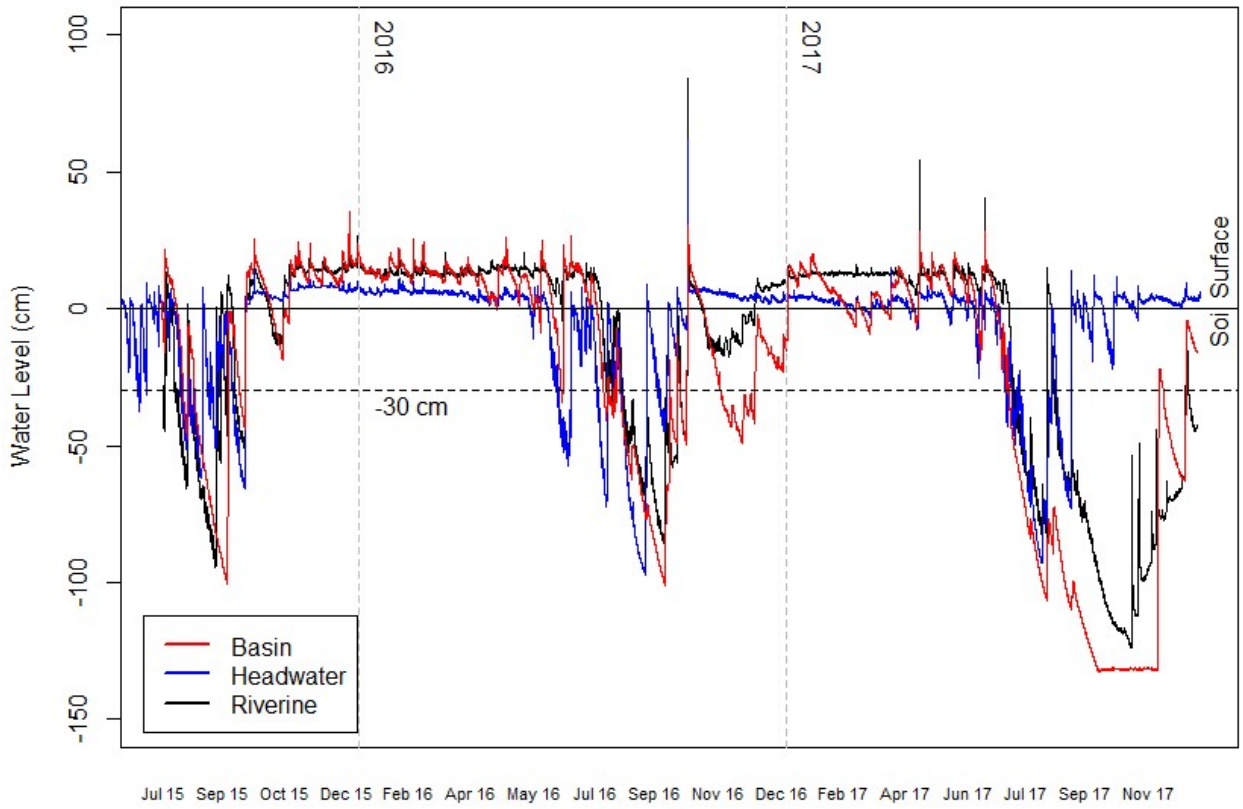

Figure 2.17: Comparison of selected hydropatterns showing how different wetland types can actually appear similar.

While the wetland sites in this study did not appear to be distinguishable by wetland type based on summary statistics or ECDFs and there was considerable overlap of the hydropattern plots, among the less disturbed sites there were distinct characteristics and patterns that were representative of certain wetland types. Figure 2.18 shows some of the less disturbed wetlands that provides a good representation of the range of hydropatterns observed in these sites. The riverine wetlands were characterized by the deep drawdowns during the growing season and higher maximum water levels due to overbank flood events. Otherwise, water levels were generally near or above the surface during the winter and spring. Basin wetlands had the highest water levels observed, and generally had inundated conditions for a majority of the year aside from the dry period in the later summer or fall. Headwater sites had the smallest range of water level fluctuations of the wetland types monitored. Water levels were generally within 30 cm of

the soil surface or slightly above in lower gradient Coastal Plain sites. Drawdowns were generally less than about 60 cm below the soil surface.

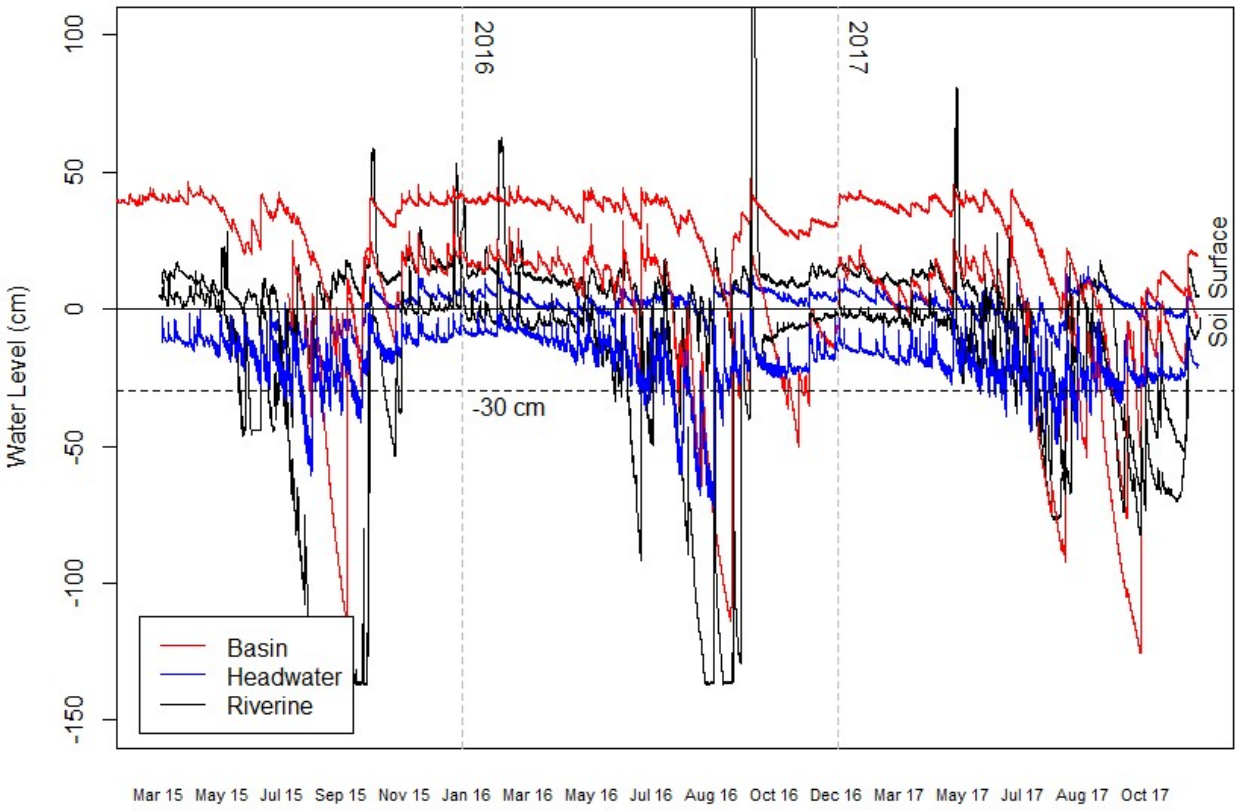

Figure 2.18: Hydropatterns for selected best available wetlands to illustrate the general range of hydropatterns for different wetland types.

Summary of the General Characteristics of Wetland Hydropatterns

Observations from this study and previous research show generally similar characteristics of hydropatterns of non-tidal, non-lacustrine forested wetlands. These general characteristics included water levels above or near the surface in the winter and early spring before a variable drawdown occured due to increased evapotranspiration during the summer. Water levels typically recovered during the late fall and winter; however, there can be considerable year-to-year varaibility (Mitsch and Gosselink, 2007). Gilliam and Skaggs (1981) observed water table depths near the surface from November through March and then drawdowns of one meter or more during the growing season in undisturbed pocosins in eastern North Carolina. Johnson et al.

(2014) showed similar patterns in non-riverine swamp foresting in the Coastal Plain. Comparable patterns were observed by Hunter et al. (2008) and Jarzemsky et al. (2013) in natural bottomland hardwood forests. While many of the wetland hydropatterns had the same general characteristics, there were unique features of each type. Table 2.4 summarizes the typical feature of each wetland type based on observations from this study and other sources (e.g. Mitsch and Gosselink, 2015; NC DEQ, 2016) and Figure 2.19, Figure 2.20, and Figure 2.21 provide a general graphical representation of typical hydroperiods for these wetland types. The figures were developed based on observations of the patterns exhibited in the least disturbed sites.

These observations are not intended to be a blanket description for all wetlands of a given type and there are certainly some subtypes and disturbed wetland sites that will not fall within the general conditions described here. Nevertheless, these observations generally describe what natural wetland hydroperiods might look like in these types of wetlands in North Carolina. However, substantial year-to-year variability was observed in this study due to climate and antecedent conditions (e.g. Mitsch and Gosselink, 2007). This inherent variability because of climate makes it difficult to draw substantial inferences with a high level of confidence from these short-term datasets. This is one of the many reasons why long-term data is so important for characterizing average, wet, and dry conditions in wetland ecosystems.

Table 2.4: General characteristics of each wetland type.

Wetland Type	General Hydrologic Characteristics
Basin	Semipermanently to seasonally flooded. Water depth can range from 15 to 60 cm. Primarily processes are precipitation and evapotranspiration. These systems typically persist due to high regional groundwater table or an impermeable substrate. Some groundwater influences have been documented in Carolina Bays. Piedmont basins typically dry out completely in the late summer.
Headwater	Small, shallow bowl-shaped areas near watershed divides. Headwater wetlands are typically lower gradients and wider in the Coastal Plain. Groundwater discharges to eventually form first order streams. The water table is generally within 30 cm of the surface during the non-growing season for Piedmont sites. Shallow ponding typically occurs in low gradient Coastal Plain sites. Main sources of water are groundwater discharge and surface runon. Range of water level fluctuations relatively smaller due to groundwater inputs.
Riverine	Wide range of hydrologic regimes from intermittently to seasonally flooded. High peak water levels due to overbank flooding. Water level can drawdown meters below surface during the growing season in some sites, particularly along incised channels. Water level near or above the surface is typical for winter and early growing season periods. Water levels are generally higher for Coastal Plain headwater wetlands.

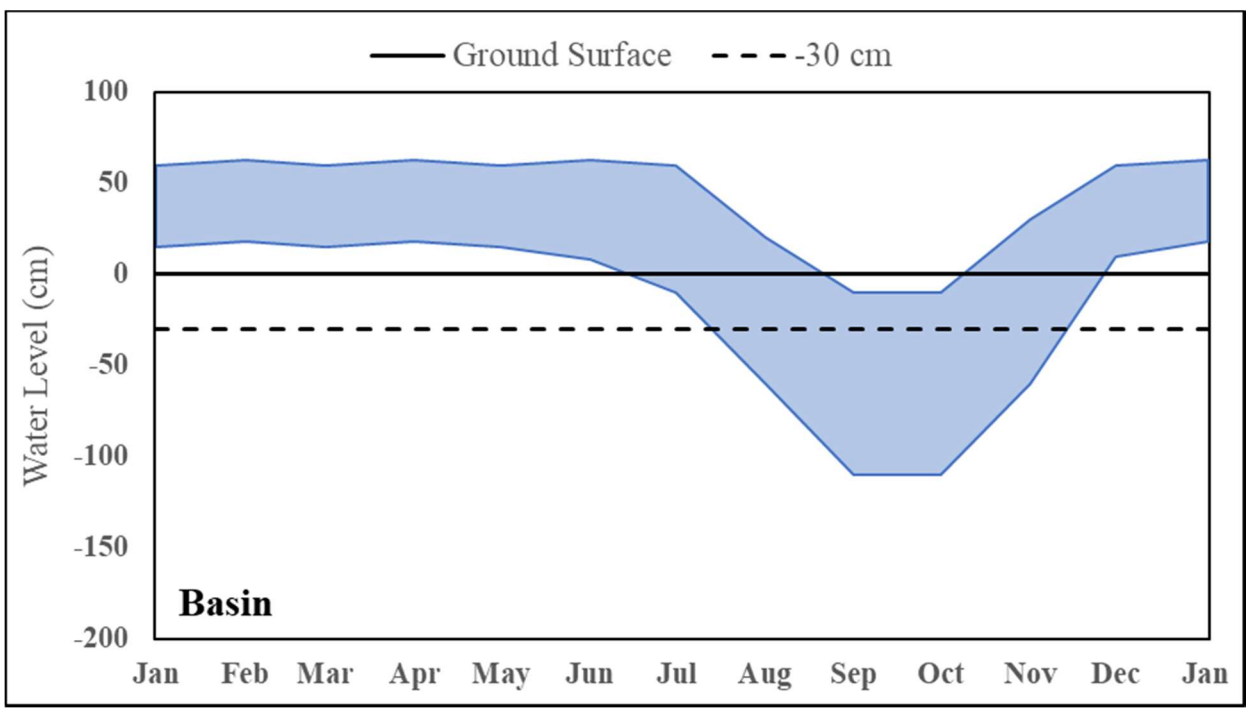

Figure 2.19: General range of hydropatterns for basin wetlands.

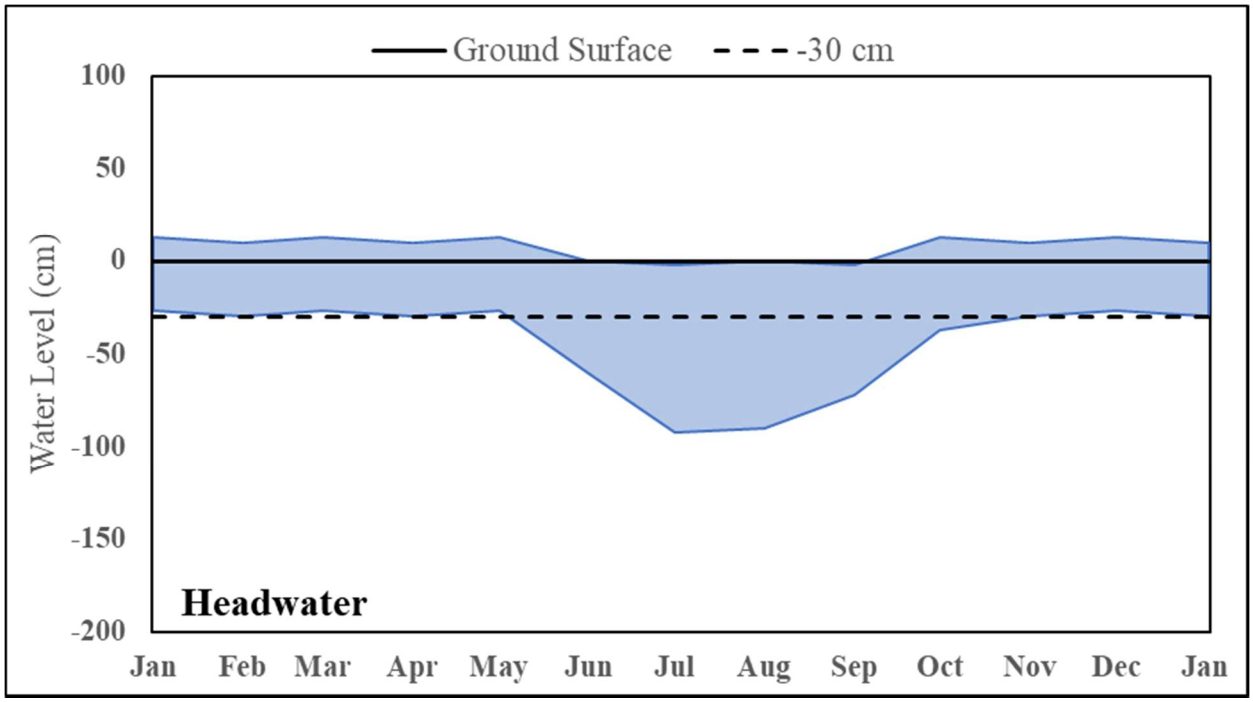

Figure 2.20: General range of hydropatterns for headwater wetlands.

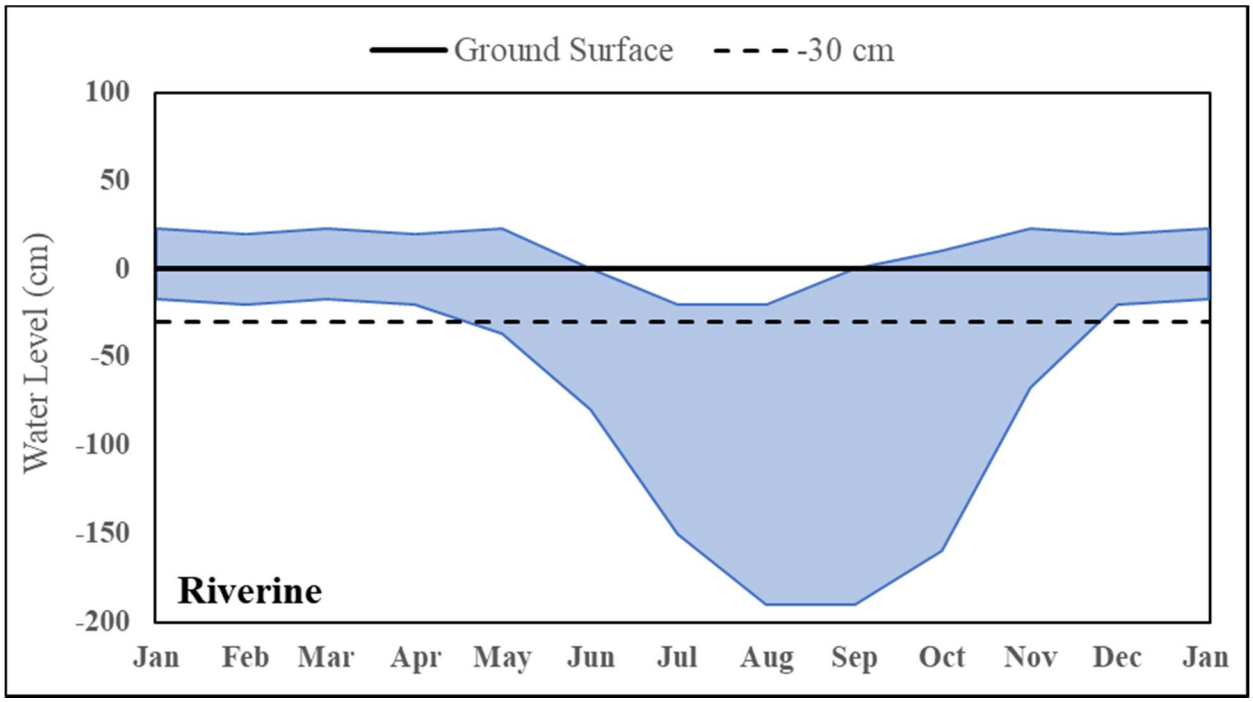

Figure 2.21: General range of hydropatterns for riverine wetlands. Note: peak water levels due to overbank flood events are not shown here. 1 to 8 overbank flood events per year were observed in these sites. The flood events did not follow a seasonal pattern.

Comparison to Wetland Hydrology Standards

With a few exceptions, the periods of saturation at the wetland sites exceeded the 5% of the growing season threshold for the definition of a wetland, and the periods of saturation were generally much longer than the NCIRT revised saturation duration recommendations. During this period, rainfall generally fell in the average range (See Appendix A). A large range of saturation percentages were observed in the same soil series, however well placement could affect this outcome. Because of the general characteristics of the hydropattern discussed earlier (water level above or near the surface early in the year and then variable drawdown over the growing season), the longest period of saturation typically occurred at the beginning of the growing season.

Table 2.5: Comparison of observed continuous saturation periods during the growing season to the NCIRT standards. Bolded values indicate the saturation durations fell short of the standards.

Site	Region	Type	Soil Series	Subgroup	NCIRT %	2016		2017	
						Wet (%)	Dry (%)	Wet (%)	Dry (%)
DRG	P	Basin	Wehadkee	Fluvaquentic Endoaquepts	12-16%	100	39	80**	34**
DEN	P	Basin	Wehadkee	Fluvaquentic Endoaquepts	12-16%	55	39	46**	48**
SIK	CP	Basin	Murville	Umbric Endoaquods	12-16%	100	100	56	48
BLK	P	Headwater	Herndon	Typic Kanhapludults	NA	44**	**3****	43**	**0****
HOG	CP	Headwater	Paxville	Typic Umbraquults	10-12%	40*	30*	26**	**5****
NAH	CP	Headwater	Torhunta	Typic Humaquepts	12-16%	40*	32*	47	19
PCS	CP	Headwater	Dorovan	Typic Haplosaprists	12-16%	100*	100*	100	55
SPG	P	Headwater	Codorus	Fluvaquentic Dystrudepts	7-9%	100	36	100	35
WMT	P	Headwater	Vauclose	Fragic Kanhapludults	NA	100*	21*	100**	9**
DOT	CP	Riverine	Bibb	Typic Fluvaquents	12-16%	42	40	40	27
HNK	P	Riverine	Chewacla	Fluvaquentic Dystrudepts	10-12%	54	36	47**	17**
LKW	CP	Riverine	Dorovan	Typic Haplosaprists	25%	100	50	100	51
MUN	P	Riverine	Chewacla	Fluvaquentic Dystrudepts	10-12%	56	62	48**	49**
NSC	CP	Riverine	Wehadkee	Fluvaquentic Endoaquepts	12-16%	30	38	50	51
RDU	P	Riverine	Chewacla	Fluvaquentic Dystrudepts	10-12%	23*	**7***	10	**2**
RKR	P	Riverine	Wehadkee	Fluvaquentic Endoaquepts	12-16%	37	18	27*	26*

P: Piedmont CP: Coastal Plain
*Above average yearly precipitation determined with local WETS table
**Below average yearly precipitation determined with local WETS table

In three sites, the period of continuous saturation was not as long as the NCIRT standards, meaning they were drier than the NCIRT recommends for restoration sites. In all the other sites, the period of continuous saturation exceeded the standards by a factor of two to six, meaning they were much wetter than the NCIRT recommended for restoration sites.

This comparison gives some indication of the appropriateness of these standard, but longer-term monitoring would provide a better assessment of the average conditions and the range of variability of these natural sites. In the sites where the standard was not met, there was typically some anthropogenic disturbance. For example, the RDU site (riverine) failed to meet the IRT standard for that soil type. This site had a deeply incised stream channel, which likely caused the floodplain to drain faster following precipitation events. The well location that failed to meet the standard was nearer to the stream. The BLK site (headwater) was very dry at the downslope well, but this could also be the result of below average rainfall totals in 2016 and 2017 at this site. The downslope well at the HOG site failed to meet the IRT hydrology standard in 2017. However, this site received below average rainfall in 2017. In addition, the water level in this site was very close to -30 cm for much of this period; however, a brief drop below -30 cm broke up the period of continuous saturation and was the reason that the standard was not met.

There were multiple sites underlain by the Chewacla soil series (Fine-loamy, mixed, active, thermic Fluvaquentic Dystrudepts) and the geographically associated Wehadkee soil series (Fine-loamy, mixed, active, thermic Fluvaquentic Endoaquepts). The Chewacla soil series is often referred to as a "problem soil" because it can be located in wetland landscapes but is not considered a hydric soil. Often time Wehakdee and Chewacla soils are closely associated on floodplains (NCIRT, 2016). There was a wide range of continuous saturation percentages observed over the two years of monitoring at these sites. Median percent of the growing season that was continuously saturated among these sites was around 40%, although, the percentage ranged from less than 5% to 100%. Observations of continuous saturation were generally above 30% for the minimally disturbed sites. This variability again illustrates the need for longer-term monitoring or modeling to determine average ranges and the typical variability in these

saturation percentages. The very long saturation durations for the basin wetland may be indicative of the need to consider wetland type as part of the standard and likely expand upon the simple definition of continuous saturation duration.

Based on our initial observations and comparisons, if wetland restoration projects are designed to meet the current threshold from NCIRT, they may be substantially dryer than their natural wetland counterparts. More investigation and comparisons to natural wetlands hydrology over the long term is needed.

Table 2.6: Continuous saturation percentages during the growing season observed for wetlands with Wehadkee and Chewacla soil series.

Site	Type	Soil Series	NCIRT %	2016		2017	
				Wet (%)	Dry (%)	Wet (%)	Dry (%)
DRG	Basin	Wehadkee	12-16%	100	39	80**	34**
DEN	Basin	Wehadkee	12-16%	55	39	48**	46**
NSC	Riverine	Wehadkee	12-16%	30	38	50	51
RKR	Riverine	Wehadkee	12-16%	37	18	27*	26*
HNK	Riverine	Chewacla	10-12%	54	36	47**	17**
MUN	Riverine	Chewacla	10-12%	62	56	49**	48**
RDU	Riverine	Chewacla	10-12%	23*	**7***	10	**2**

*Above average yearly precipitation determined with local WETS tables
**Below average yearly precipitation determined with local WETS tables

Long-term Hydrology and Restoration Implications

To examine the changes in the year-to-year saturation durations, the daily water level output from the two modeled sites in Chapter 5 was analyzed. As previously discussed, these two sites were not part of the 16 sites described thus far in this chapter. The two modeled sites were coastal forested wetlands underlain by Deloss (Typic Umbraquults) (NRF site) and Belhaven (Terric Haplosaprists) (GDS site) soil series. The NCIRT recommended continuous saturation percentage thresholds were 12% and 20% of the growing season for Deloss and Belhaven soil series, respectively (NCIRT, 2016). The GDS site (Belhaven soil series) met the 20% threshold

in 27/30 years and the GDS site (Deloss series) met the 12% threshold in 19/30 years. However, there was considerable variability in the length of the periods of continuous saturation from year-to-year (Figure 2.22). For the GDS site (Belhaven series), the duration ranged from about 15% to 100% with a median of around 40% of the growing season, which was double the recommendation for restoration projects from NCIRT (2016). For the NRF site (Deloss series), the duration ranged from 0 to 65% with a median around 14% of the growing season, which was very close to the NCIRT recommendation (12% of the growing season). Although this was a very limited number of sites, the large range of variability in saturation durations observed over the 30-year period indicated that there may not be narrow ranges of continuous saturation to which the different soil series correspond. This raises further questions regarding the appropriateness of these ranges that should be further investigated. Rather, a more detailed description of the hydrology might be needed to differentiate between soil series or wetland types for restoration criteria.

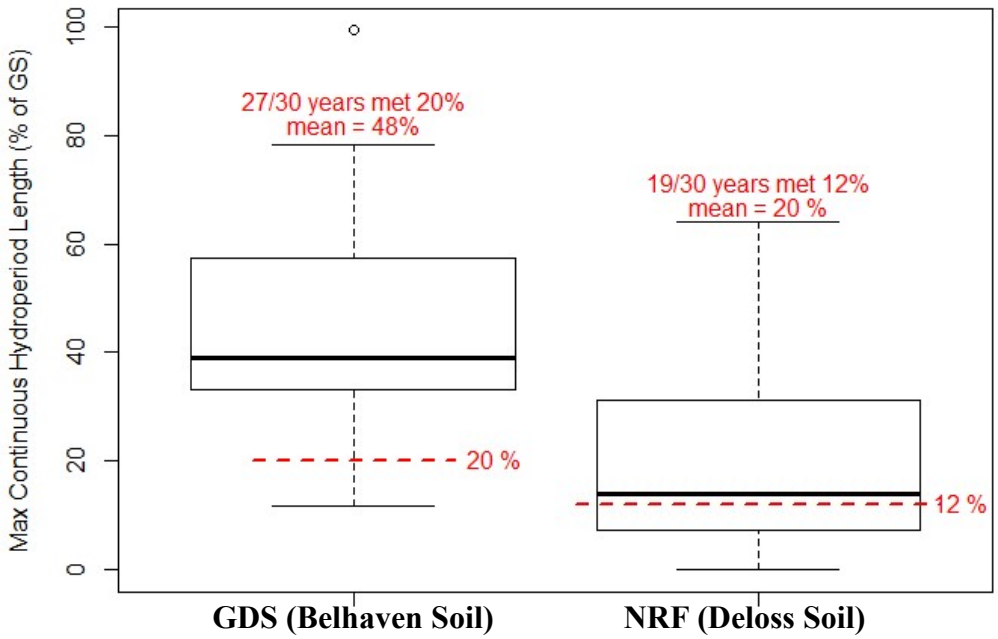

Figure 2.22: Length of continuous hydroperiod above -30 cm obtained from hydrologic data generated through DRAINMOD simulations for 30 years (see Chapter 5). Note: NC IRT recommendations for wetland restoration are indicated with dotted lines.

Any proposals to improve the hydrologic success criteria for wetlands beyond the current USACE saturation duration standard should try to focus on recreating the hydrology of natural wetlands. These criteria should strive to emphasize characteristics of the wetland hydropattern that directly relate to recreating functions that are important in the restoration project or engineering hydrology that is similar to high quality natural wetlands. One possible approach would be to use a range of ECDF curves to ensure that the entire hydropattern could be addressed (e.g. 30th and 70th percentiles). This may increase the likelihood that along with a period of saturation, the wetland would be required to become drier during the growing season, which is a characteristic of most types of natural wetlands. This might help address the problem observed in some previous studies of mitigation projects that show they are actually too wet (e.g. Brooks and Gebo, 2013; Cole et al., 2006; National Research Council, 2001).

To examine the potential of using long-term data to develop more comprehensive hydrologic criteria, 30-year of modeled daily water level for two sites was analyzed (the same model results from Chapter 5 that were used in the previous section). The yearly hydropattern and ECDFs from modeled daily water level were plotted for each year (Figure 2.23 and Figure 2.24). These plots clearly illustrated a wide range of the year-to-year variability that is ostensibly a characteristic of precipitation-dominated wetlands. The 30th and 70th percentiles of the water table depth (modeled on WETS table percentiles) were used as a representation of "average" conditions for the wetland hydropatterns and ECDFs.

The 30 and 70th percentile ECDFs and hydropatterns indicated that there does not appear to be a narrow range that represented wetland hydropatterns and there was substantial variability outside the the "average" range (30th-70th percentiles of the daily water table depth over the 30-year period). However, the 30th and 70th percentiles were somewhat arbitrary given they were based on the WETS table percentiles for average rainfall conditions and may represent too narrow a range for "average" hydropatterns. A wider range might be more appropriate given the variability. The 20th and 80th percentiles were also plotted to illustrate this.

In addition, the hydropatterns were grouped based on the yearly total rainfall (i.e. <30%, avg., and >70%) to determine if yearly rainfall could be used as a general indicator of the depth of the overall hydropattern. This analysis can be found in Appendix E. This analysis showed that, in general, the highest rainfall years corresponded to hydropatterns closest to the surface; however, there was substantial overlap among the rainfall-grouped hydroperiods. This indicated that it may not be possible to set target hydroperiods for restoration based on yearly rainfall (e.g. wet year, average, dry year). This was intended as a first cut at examining the potential use of long-term data to improve restoration criteria. While this preliminary analysis did not yield

conclusive results, this approach should be pursued further with more long-term data and comparisons to restoration sites.

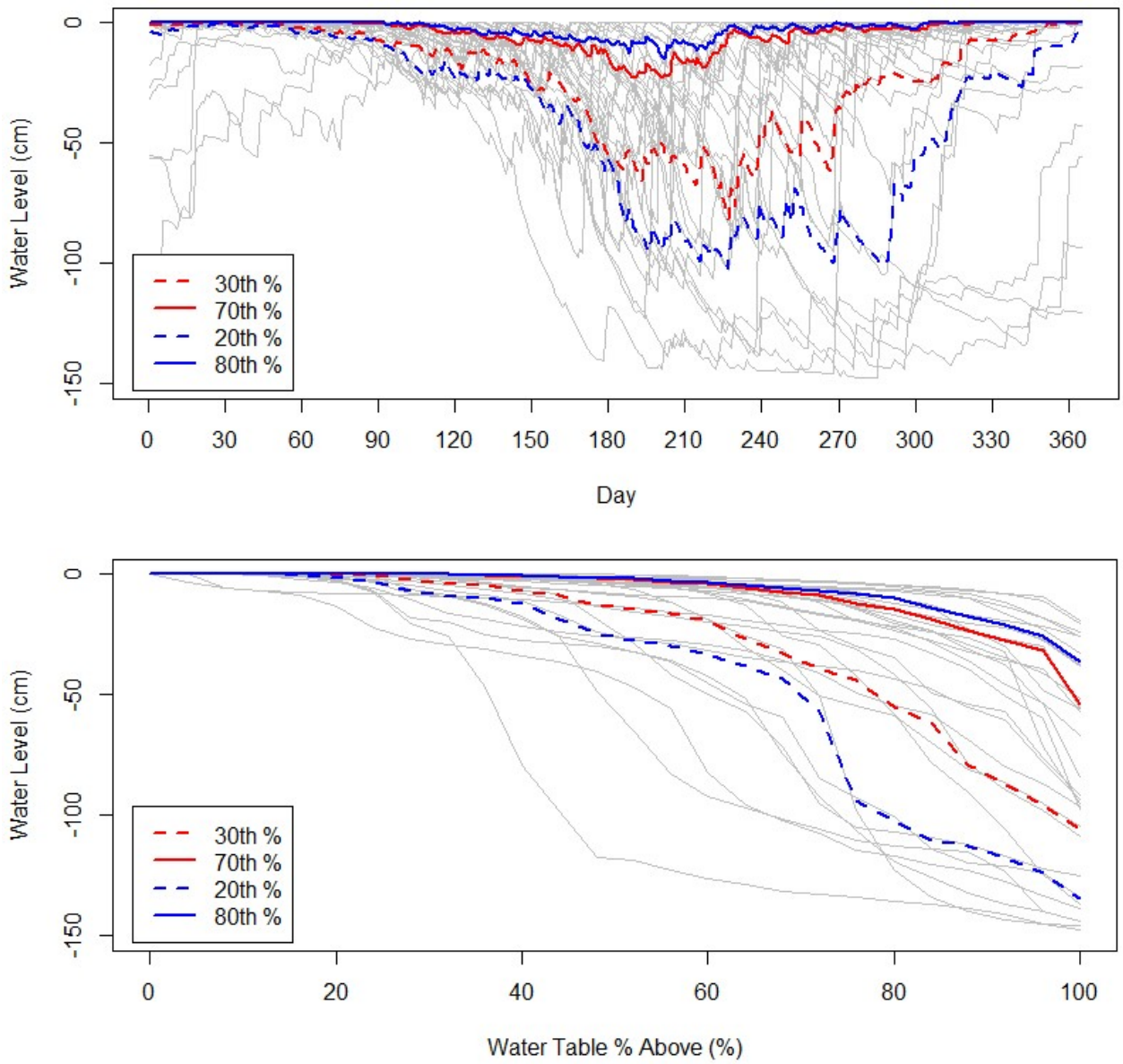

Figure 2.23: ECDF of water level with potential percentiles representing "average" conditions plotted (bottom) and yearly hydropatterns with percentiles (top) for the GDS site (Belhaven soil).

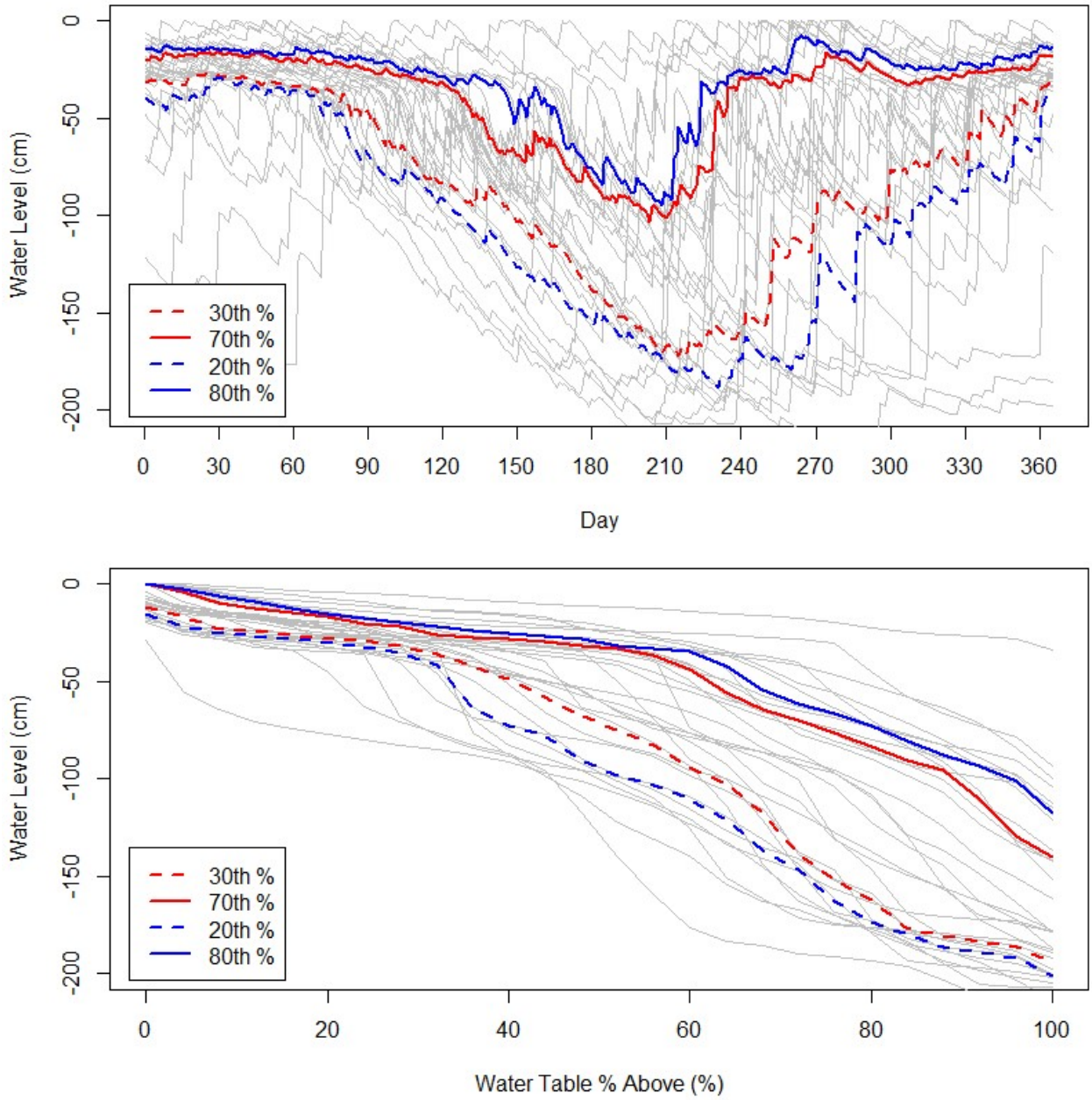

Figure 2.24: ECDF of water level with potential percentiles representing "average" conditions plotted (bottom) and yearly hydropatterns with percentiles (top) for the NRF (Deloss soil).

Conclusion

This project was started with the goal that 8 to 10 years of water level data would be available for each wetland site, and the addition of 3 years of data would allow for an in-depth analysis of the long-term hydrology of natural wetlands, an area for which information is critically lacking (Cole and Kentula, 2011), as well as an identification of any trends in the

hydrologic conditions of a group of 16 natural wetlands in NC. However, due to the poor quality of the long-term data, these goals were not fully realized. Instead, more limited but informative characterizations of general wetland hydropatterns and comparison to standards for wetland restoration were completed. The variability in the water level between the two years at many of the sites (mainly the precipitation dominated sites) clearly illustrates the need for long-term water level data to fully characterize wetland hydrology for different wetland types and physiogeographic locations.

While the riverine wetlands generally had higher maximum water levels due to overbank flooding events, the summary statistics did not differentiate the wetlands by type. This is not necessarily surprising given the large geographic range, variability in wetlands size, different ecoregions, and varying levels of anthropogenic disturbance. A more focused future study might consider analyzing minimally disturbed wetlands of the same type and similar size over a smaller geographic range. However, some general characteristics were identified that were representative of specific wetland types and generalized hydropatterns for minimally disturbed sites were proposed for the three wetland types.

The impacts of anthropogenic disturbance were evident on the hydrologic regimes of some of these wetlands. Some disturbances likely contributed to wetter conditions (e.g. road construction) while others likely contributed to lower water levels (e.g. ditching, upstream development) (e.g. US EPA, 2008).

The observed saturation durations typically far exceeded the recommended ranges from NCIRT except for a few disturbed sites; however, revised ranges were not proposed because of the short sampling period, large geographic range, small sample size and different wetland types. The long-term reference wetland modeling results (from Chapter 5) showed that a wide range of

saturation durations occurred in these wetlands and a narrow range that was similar to the proposed thresholds was not likely to occur. In addition, there was a wide range of hydropatterns observed over the 30-year simulation and a generalized range for hydropatterns should be compared to data from restoration sites to evaluate the applicability of an approach like this.

The analysis of long-term wetland data should be pursued further. Long-term hydrology data or hydrologic models to predict long-term hydrology for natural reference wetlands would be required. Long-term data or model output could then be used to compare against current hydrologic criteria and restoration projects to develop new guidelines based on natural reference systems. This should be done for the most frequently restored wetland types. Because of the variability in climate, it is critical that long-term water level data are used to develop standards. While an effort like this may be time and resource intensive initially, it should be pursued to improve our understanding of wetland hydrologic regimes (the primary driver of structure and function) and thus potentially improve restoration guidance and outcomes. Researchers should propose and agencies like NC DEQ and US EPA should prioritize supporting future focused studies in this area.

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Chapter 3: Characterizing background water quality of natural wetlands to support the possible future development of wetland nutrient criteria in North Carolina

Abstract

Nearly thirty years ago the U.S. Environmental Protection Agency directed states to begin developing water quality standards specific to wetlands. However, minimal progress has followed, in part due to the intrinsic complexity and heterogeneity among wetland ecosystems. A first step towards the development of numeric water quality criteria for wetlands would be the characterization of background concentrations in wetlands and impacts of external stressors on the ambient conditions. Characterizing background water quality could enhance the understanding of wetland ecosystems, provide goals for regulatory programs, and enable earlier detection of degradation, which would help strengthen state wetland narrative standards. The main objectives of this research were to (1) compare the background water quality of wetlands to that of streams, (2) to define thresholds for reference conditions for nutrients for natural wetlands in North Carolina, and (3) to compare the results for toxic metals to surface water quality standards for the protection of aquatic life. Beginning in 2015, quarterly grab samples were collected in 16 natural wetlands in the Piedmont and Coastal Plain regions of North Carolina. The sites span a range of anthropogenic disturbances, and consist of headwater, riverine, and basin wetlands. This data was combined with historical data collected at these same sites from 2005 to 2013 by the NC Department of Environmental Quality Division of Water Resources. The observed values for designated reference wetlands were similar to previous research in natural wetlands; however, the small sample size and infrequent sampling combined with the heterogeneity in wetland ecosystems limited the inferences that could be drawn from the data. Nutrient thresholds were compared to the published background levels for streams. Wetland

background levels should be set higher than streams for TN, TP, NH_4^+, but likely lower than streams for NO_3^--N. A method developed by the USEPA to establish nutrient criteria was modified to define draft thresholds for reference conditions for nutrients. The thresholds were calculated as 0.07 mg/L NO_3^--N, 0.15 mg/L NH_4^+-N, 0.3 mg/L TP, 2.9 mg/L ON and 3.2 mg/L TN. However, ON and TN thresholds did not seem to be robust indicators of anthropogenic disturbance. Thresholds for NO_3^--N and NH_4^+-N were more indicative of disturbance and could likely be applied across ecoregions and wetland types, while wetland type and ecoregion specific thresholds would be more appropriate for TN, ON and TP. Conclusions regarding the comparison of metals (Cu, Pb, Zn) results to the state surface water standards were limited because of the difference in the form of results and criteria (total vs dissolved). However, exceedance of the criteria did generally occur in more developed land uses. Overall, this research presents a reasonable first attempt at defining nutrient reference thresholds for undisturbed natural wetlands, despite the limitations imposed by the sampling frequency and site selection.

Introduction

Wetlands are often lauded for their pollutant removal capability and other important ecosystem services they provide. However, water quality standards for protecting aquatic life do not explicitly apply to wetlands in most states, and numeric nutrient criteria have not been developed for wetlands. Seeking to expand the regulatory protection of wetland ecosystems, nearly three decades ago (1991) the U.S. Environmental Protection Agency (US EPA) issued *National Guidance Water Quality Standards for Wetlands* (US EPA, 1994). This document provided guidance for the states to develop state water quality standards specific to wetlands. Extending water quality standards to wetlands would strengthen the regulatory framework to better ensure the overall goal of conserving wetland resources, while also protecting the important natural functions wetlands provide (US EPA, 1994). While some states have extended designated uses and narrative water quality criteria to wetlands, few states have made progress towards developing numeric criteria. The lack of progress in implementation by the states is likely the result of a critical lack of historical water quality data from wetlands and the costs associated with large-scale data collection efforts. Another impediment is the variable nature of wetland water chemistry that is dependent on physiochemical dynamics and seasonality (US EPA, 2008), and the wide range of hydrologic regimes that often result in an absence of surface water for long periods (Kusler and Christie, 2012). In addition, many of the environmental factors present in wetlands (US EPA, 2008) make wetland water chemistry during normal conditions fall outside the range of typical surface water conditions (Kusler, 2011). Furthermore, while studies on stream water quality have been published for decades (e.g. Schröpfer 1942), there are very few published, long-term studies on surface water quality in natural wetlands.

In North Carolina, there are currently no numeric water quality standards for wetlands; however, narrative criteria are described in the North Carolina Administrative Code- 15A NCAC 2B .0231 Wetland Standards. The North Carolina regulations designate uses for wetland waters, and the narrative criteria language focuses on avoiding "adverse impacts" to the wetland designated uses. Specifically, avoiding "concentrations… of substances which are harmful to human, animal or plant life… which individually or cumulatively may cause adverse impacts on existing wetland uses" and maintaining the hydrologic conditions necessary to avoid adverse impacts to the chemical, nutrient, DO, and pH conditions in the wetland (NCAC, 2015). However, without any numeric values or background ranges that would allow for the identification of changes indicative of "adverse impacts," these narrative standards would be very difficult to effectively implement. While state surface water standards do not explicitly include wetlands, it is suggested that selected surface water criteria may be applicable to some wetlands on a limited basis (Kusler, 2011). However, comparisons between ambient wetland water quality and surface water standards have not been published to date. The development of completely defensible numeric water quality criteria would require an extensive data collection and analysis effort. This would require years of data collection and substantial financial commitments from the states. However, the simple development of general background concentrations would be a valuable first step in this process and would help support the rather vague narrative criteria.

To help start this process, the overall goal of this study was to develop general nutrient background conditions for wetlands, using the data from 16 North Carolina natural wetland sites collected for various periods over the last twelve years. The primary objectives of this effort were to:

1. Compare observed concentrations in wetlands to background conditions published for streams.
2. Develop thresholds for reference conditions for TN, TP, ON, NO_3^--N, and $NH4^+$ -N using a modified US EPA nutrient criteria methodology.
3. Compare the metals concentrations to background levels for surface waters.
4. Compare metal concentrations to state standards for metals to assess the risk to the wetland biota.

Materials and Methods

Study Sites

Sixteen natural wetland sites located in the Coastal Plain and Piedmont physiographic regions of North Carolina were sampled for this project (see Figure 3.1). These sites were selected by the North Carolina Department of Environmental Quality Division of Water Resources (NC DEQ DWR) as part of a larger assessment of the condition of natural wetlands in North Carolina dating back to 2005. The sites were made up of three different wetland types: riverine, headwater and basin wetlands. The sites were located in a wide range of surrounding land uses and displayed varying degrees of anthropogenic disturbance, including logging, road construction, ditching, agriculture, and upstream urbanization. There was a wide range of underlying soil series and soils textures that ranged from mineral to muck.

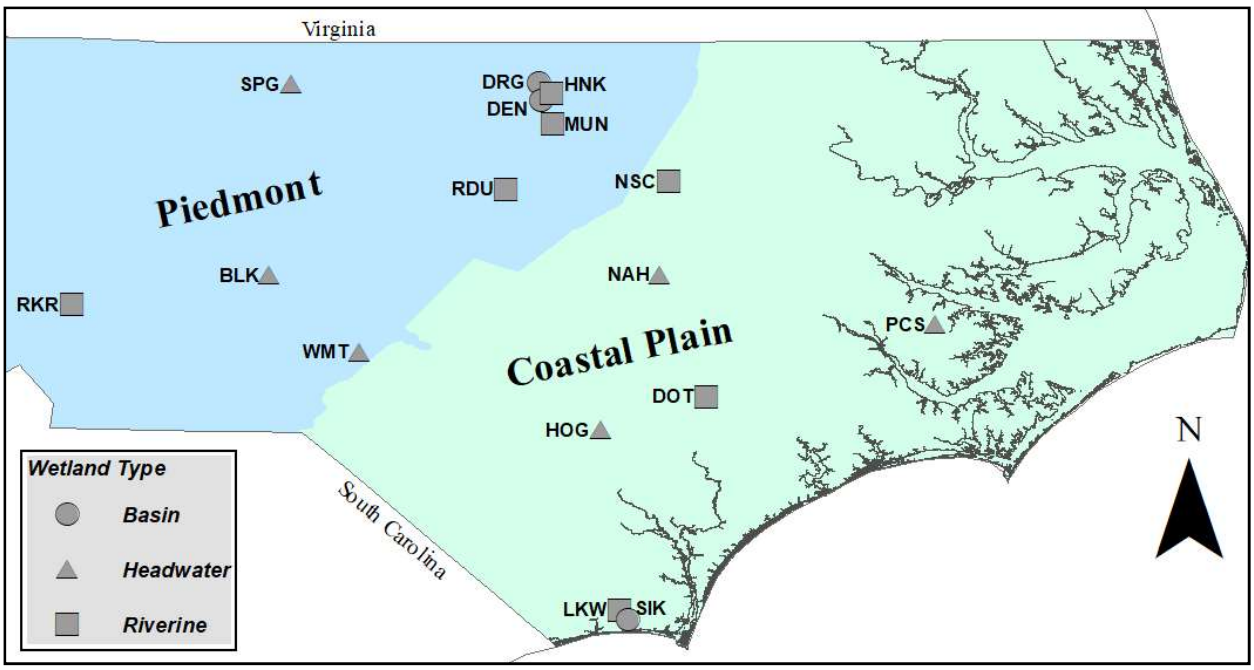

Figure 3.1: Site map with wetland type and site identifier code.

Table 3.1: Wetland site characteristics.

Site Code	County	Region	Wetland Type	Likely Anthropogenic Pressure
BLK	Montgomery	Piedmont	Headwater	Pasture; highway
SPG	Rockingham	Piedmont	Headwater	Minimal
WMT	Moore	Piedmont	Headwater	Urban
RDU	Wake	Piedmont	Riverine	Upstream Development
RKR	Cabarrus	Piedmont	Riverine	Upstream Development
HNK	Granville	Piedmont	Riverine	Upstream Development; highway
MUN	Granville	Piedmont	Riverine	Pasture; row crops
DEN	Granville	Piedmont	Basin	Row crops
DRG	Granville	Piedmont	Basin	Minimal
PCS	Beaufort	Coastal Plain	Headwater	Minimal
HOG	Sampson	Coastal Plain	Headwater	Pasture; spray field
NAH	Wayne	Coastal Plain	Headwater	Row crops; highway
DOT	Duplin	Coastal Plain	Riverine	Row crops, drainage
NSC	Nash	Coastal Plain	Riverine	Upstream agriculture + development
LKW	Brunswick	Coastal Plain	Tidal Riverine	Upstream Development; highway
SIK	Brunswick	Coastal Plain	Basin	Minimal

Sampling Design and Methods

Two sampling locations were established at each site, generally located in an upslope – downslope orientation, when possible. For the current project, grab samples were collected from the surface water during site visits four times per year: Spring (March/April), summer (June/July), fall (September/October) and winter (December/January). Samples were collected in HDPE bottles, acidified (see Table 3.2), and transported to the lab on ice following US EPA sampling guidelines (US EPA, 1982). During the summer and winter site visits, samples were analyzed at the NC DEQ DWR Water Sciences Section Chemistry Laboratory for NH_4^+-N, NO_3^--N, TKN, TP, Cu, Zn, Pb, Mg. Ca, TOC and DOC using standard methods (see Table 3.2). For the spring and fall site visits, the samples were analyzed at the Department of Biological and Agricultural Engineering Environmental Analysis Lab (EAL) for NH_4^+-N, NO_3^--N, TKN, and TP using standard methods (see Table 3.2). Total nitrogen (TN) was calculated as the sum of TKN and NO_3^--N and organic nitrogen (ON) was calculated as the difference between TKN and NH_4^+-N. In addition, pH, dissolved oxygen (DO), specific conductivity (spec. cond.), and temperature were measured during each site visit using an YSI Professional Plus Multiparameter Instrument (YSI Incorporated, Yellow Springs, OH).

Table 3.2: Sample preservatives and laboratory analysis methods.

Constituent	Preservative	EAL methods	NC DEQ methods
NH_4^+-N	H_2SO_4	SM 4500-NH3 G.	EPA 350.1 Rev. 2.0
NO_3^-+NO_2^--N	H_2SO_4	EPA 353.2.	EPA 353.2 Rev. 2.0
TKN	H_2SO_4	EPA 351.2	EPA 351.2 Rev. 2.0
TP	H_2SO_4	SM 4500-P F.	EPA 365.1 Rev. 2.0
Cu, Zn, Pb	HNO_3	----------	EPA 200.8 Rev. 5.4
Ca, Mg	HNO_3	----------	EPA 200.7 Rev. 4.4
DOC/TOC	H_3PO_4	----------	SM 5310 B-2000

Because of the variability in hydrologic regimes of wetlands (most wetlands dry out in the summer and fall because of higher evapotranspiration during the summer), there were many

site visits on which samples were not collected because of the absence of surface water. In addition, samples were not collected from stagnant, turbid pools during low water conditions to avoid artificially elevated nutrient and metal results (see Appendix J).

Long-term Samples Collected by NC DEQ

The NC DEQ DWR Wetlands Group collected quarterly grab samples at these same sites for various periods spanning the years of 2005 to 2013 (see Appendix G). Samples were analyzed at the NC DEQ DWR Water Sciences Section Chemistry Laboratory for the same parameters as the current sampling. Different sampling strategies and protocols were implemented depending on the project for the long-term sampling completed by NC DEQ DWR. For example, when no surface water was present at a site, pits were excavated to collect samples during some periods of the NC DEQ DWR sampling. These samples were eliminated so that only surface water samples were included in this dataset. In addition, if NC DEQ DWR field notes indicated samples were collected from stagnant, isolated pools, those samples were also eliminated from the analysis.

Analysis of Nutrient Concentrations

Because of the small sample size, it was not appropriate to group the sites based on wetland type and ecoregion as recommended by USEPA (2008). This was not suitable for this analysis because doing so would have resulted in a lack of replication (due to the small number of sites) within each type and ecoregion. This would have seriously limited any of the inferences drawn from the results and may have resulted in a biased analysis. Therefore, sites were grouped based on level of disturbance. A single metric such as Landscape Development Intensity (LDI) index (Brown and Vivas, 2005) was not used because different wetland types had different contributing areas and hydrologic regimes that influenced pollutant transport. Instead, the sites

were grouped based on best professional judgment (BPJ), as recommended by USEPA (2008) as an alternative to type and ecoregion groupings. The groupings were based on surrounding land use information from aerial photography and site visits. The land use groupings consisted of three groups: “Reference”, “Ag. Impacted”, and “Average Disturbance (Avg. Dist.).” “Reference” was defined as the best available, least disturbed sites. “Ag. Impacted” was defined as sites located adjacent to agriculture land uses. “Avg. Dist.” was defined as sites that were located in impacted areas but lacked intensive agricultural land uses adjacent to the wetland sites (see Table 3.3). Water quality data from the sites and disturbance classes were compared using boxplots. These comparisons were based on site average water quality values (average of two sample locations). Site averages were used because there were no discernable upstream and downstream locations at all the sites (as intended in the study design) due to the differences in hydrologic regimes among the wetland types. The observations were compared to the national background levels for streams (USGS, 2010) shown in Table 3.4. Dissolved oxygen and pH were compared by wetland type in order to compare pH values to previous research and because DO is dependent on hydrologic regimes (i.e. flow vs. no flow).

Table 3.3: Disturbance class (BPJ), wetland type and region for each site.

Site Code	Region	HGM Wetland Type	Disturbance Class
BLK	Piedmont	Headwater	Avg. Dist.
WMT	Piedmont	Headwater	Avg. Dist.
MUN	Piedmont	Riverine	Avg. Dist.
RKR	Piedmont	Riverine	Avg. Dist.
HNK	Piedmont	Riverine	Avg. Dist.
LKW	Coastal Plain	Riverine	Avg. Dist.
NSC	Coastal Plain	Riverine	Avg. Dist.
DEN	Piedmont	Depressional	Ag. Impacted
DOT	Coastal Plain	Riverine	Ag. Impacted
NAH	Coastal Plain	Headwater	Ag. Impacted
HOG	Coastal Plain	Headwater	Ag. Impacted
DRG	Piedmont	Depressional	Reference
SPG	Piedmont	Headwater	Reference
PCS	Coastal Plain	Headwater	Reference
RDU	Piedmont	Riverine	Reference
SIK	Coastal Plain	Depressional	Reference

Table 3.4: Draft nutrient criteria and background conditions applicable to North Carolina streams and lakes.

Water Body	TP (mg/L)	TN (mg/L)	NO_3^--N (mg/L)	NH_4^+ -N (mg/L)	Source
Lakes and Reservoirs-Ecoregions IX, XIV-63	0.02	0.40 – 0.46	0.02- 0.10	-	(US EPA, 2000b, 2001)
Rivers and Streams-Ecoregions IX, XIV-63	0.04-0.05	0.42 -0.55	0.04- 0.13	-	(US EPA, 2000c, 2000a)
National Background for Streams	0.03	0.58	0.24	0.025	(USGS, 2010)

Background on Developing Nutrient Criteria Values

The US EPA provides guidance on methods to develop nutrient criteria specific to wetlands. In 2008, the US EPA released *Nutrient Criteria Technical Guidance Manual for Wetlands*, with the goal of better protecting wetlands from nutrient enrichment (US EPA, 2008). Methods for developing nutrient criteria for wetlands include the use of reference sites to develop criteria based on minimally disturbed natural systems, frequency distribution-based

approaches, ecological thresholds utilizing predictive relationships between nutrient concentration and wetland function, and developing standards based on peer reviewed literature (US EPA, 2008). For wetlands, nutrient criteria should be established for each wetland type or class within a given ecoregion as wetlands of the same class typically function similarly and have similar hydrologic regimes. For example, riverine wetlands typically have pulses of influent floodwater that continually imports nutrients and sediment (Mitsch and Gosselink, 2007), while basin wetlands (e.g. Carolina Bays) may only receive inputs from rainfall (Sharitz, 2003).

The US EPA's nutrient criteria document provides guidance for three reference-based methods for developing proposed criteria for wetlands. These approaches require water quality data for a group of wetlands over an extended period, but do not require in-depth information about specific wetland conditions. The least complicated approach to establish a starting point for nutrient criteria put forward by US EPA (and used in their draft nutrient criteria for streams and lakes) is to define reference conditions based on a percentile of the distribution of observations. This defined percentile is assumed to approximate relatively undisturbed conditions. The percentile can vary based on the population of wetlands monitored. For a large population, which best represents a range of conditions, US EPA recommends a percentile ranging from the 5th to 25th (typically the 25th) for an approximation of reference conditions.

When the population is large enough to identify a subset of minimally disturbed reference wetlands, US EPA recommends using the 75th to 95th percentile of the reference subset for the development of criteria. This percentile can be extended closer to the 95th percentile when there is high confidence the population is made up of high-quality reference sites. In theory, the 25th percentile of the overall population of sites should be approximately equal to the 75th percentile of the subset of reference sites (see Figure 3.2). The support for these approaches has been

mixed. Studies from New York and Tennessee showed close agreement between the 75th percentile of the reference population and the 25th percentile of all the sites (USEPA 2000a). Suplee et al. (2007) found that the 86th percentile of the reference was equivalent to the 25th percentile of the entire population for streams in Montana, but also that these percentiles vary considerably between ecoregions and among target constituents.

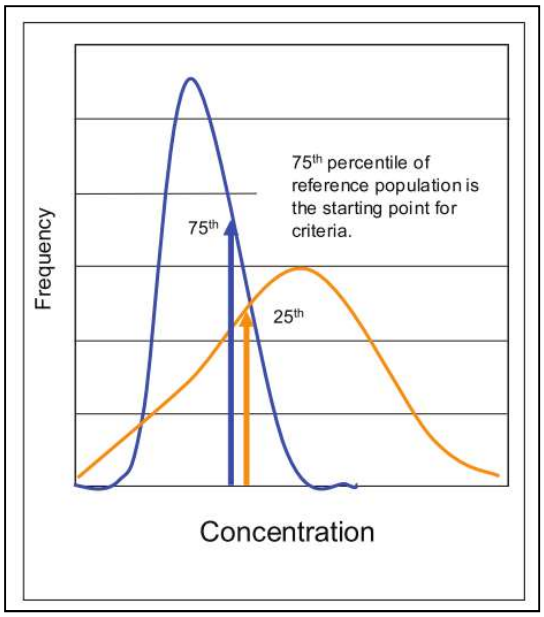

Figure 3.2: Schematic of frequency-based approaches to approximate reference conditions. From US EPA (2008). Blue represents distribution of reference sites while gold represents all sites.

Another approach summarized by USEPA (2008), relies on identifying reference wetlands in an ecoregion that exhibit the least disturbed conditions, typically based on best professional judgment, and then defining the water quality criteria based on monitoring these reference sites (US EPA, 2008). Setting the criteria as some approximation of undisturbed sites and maintaining water quality concentrations below this threshold will theoretically protect the biological integrity of wetlands (US EPA, 2008).

Other studies have used the percentile-based approach for defining background conditions. For example, the USGS used the 75th percentile of a large dataset of streams in undisturbed areas to define background conditions (USGS, 2010). The USEPA has published

draft recommended nutrient reference conditions for lakes and reservoirs, and rivers and streams for most Level III Ecoregions in the United States (e.g. USEPA, 2000). The draft criteria levels are based entirely on the 25th percentile of the distribution of the population in each ecoregion. A lack of sufficient data has prevented the same approach from being implemented for wetland reference conditions. However, reference conditions for streams and lakes may serve as a starting point for characterizing the water quality of wetlands.

Proposed Modified US EPA Approach

A modified reference, frequency distribution-based approach for developing background conditions is proposed herein. This modified method seeks to address some of the shortcomings of the frequency-based approaches by developing a reasonable first cut for thresholds for reference conditions that accounts for the small sample size, limited sampling frequency, difference in sampling periods and lack of site replication that is common in many wetland-monitoring programs.

For the US EPA's frequency-based method, as applied to streams and lakes, the median for each site is initially calculated and then some percentage of the site medians is selected (based on the population of interest) (e.g. USEPA, 2000b). By initially calculating site medians, half the samples are disregarded, so even in reference sites, 50% of the samples would be above the selected threshold for that given site. It is hypothesized that this approach risks setting the threshold prohibitively low and disregards the natural variability in undisturbed wetland sites. This was also identified as a concern during external peer review of the US EPA's proposed methods (US EPA, 2000d).

The proposed approach relies on a combination of the US EPA methods and some elementary nonparametric statistical methods. In the initial step, the wetlands were categorized

by disturbance class as previously discussed (see Table 3.3). Next, the individual samples were combined based on the site's designated disturbance class. This approach relies on the uncertain assumption that the individual samples from a given site are independent. However, for the data available from this study, this was a necessary assumption given the small sample size, lack of replication and limited sampling frequency.

Next, for a given target nutrient, 95% bootstrap confidence intervals were calculated for percentiles ranging from the 20th to the 90th for each disturbance class. Bootstrap confidence intervals allow for an illustration of the variability of the data and an estimate of the potential range of the calculated percentiles given longer-term sampling. The estimation of bootstrap confidence intervals relies on the assumption that the observed samples are a good representation of the distribution of the population (Efron, 1982). The confidence intervals for the percentiles were then plotted for each group. Wider confidence intervals represent more within class variability in the distribution. Separation between the confidence intervals for the reference group and the other group indicates that the particular constituent may be a good indicator of disturbed conditions. Bootstrap sampling was previously used to identify differences in groups for stream water quality by MTDEQ (2008) and is a widely used method in the field of statistics (Davison and Hinkley, 1997).

The upper 95% confidence interval of the reference sites at the 90th percentile was selected as the threshold for reference conditions (see Figure 3.3). The 90th percentile provides sufficient protection to cover most of the undisturbed sites while protecting against the inclusion of outliers that may elevate the threshold. Outliers were considered values that were outside the typical range for a site and could not be easily explained due to the very infrequent sampling. Outliers may result from sampling or laboratory errors, infrequent, anomalous events not

explained by surrounding land use, or possible high organic or sediment content in the samples (see Appendix J). The 90th percentile is a reasonable choice given that it is commonly used to set recreational water quality criteria (US EPA, 2012) and has been used previously for setting draft nutrient criteria for streams (MTDEQ, 2008).

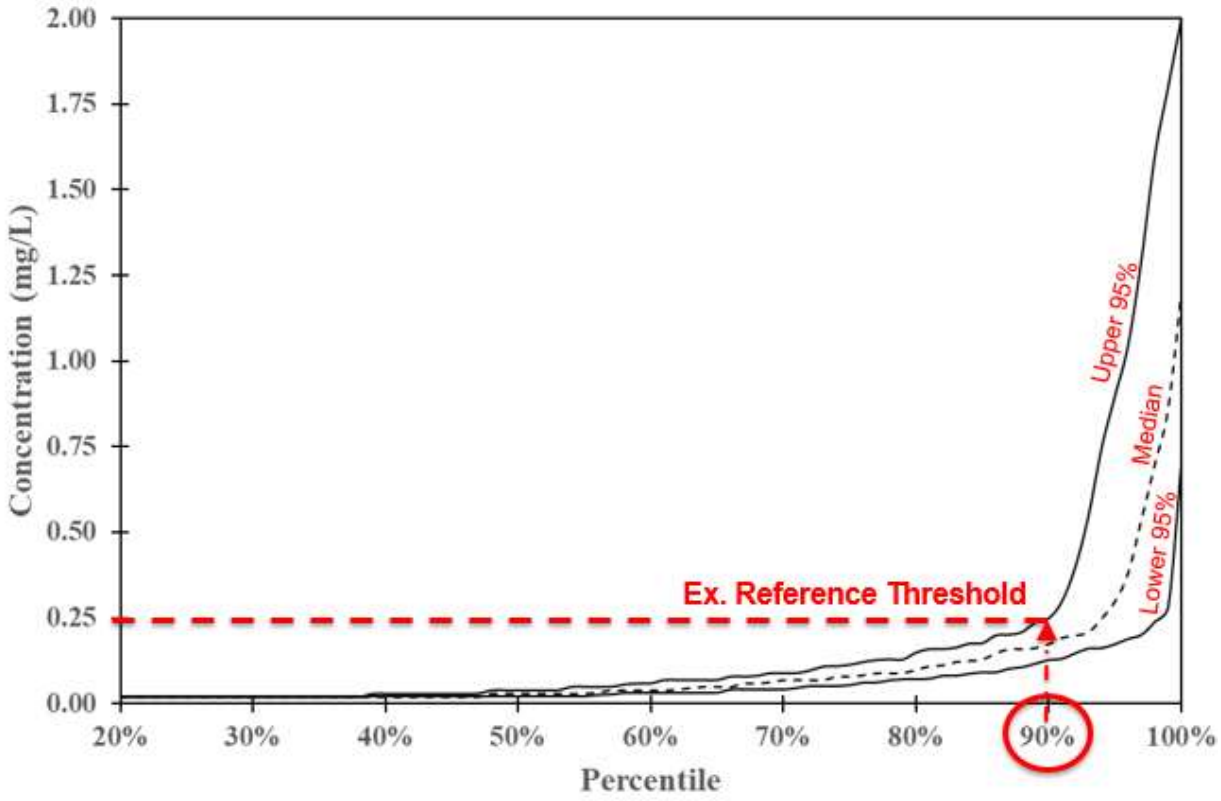

Figure 3.3: Example schematic of proposed approach for estimating threshold for reference conditions using bootstrap confidence intervals of exceedance percentiles.

US EPA (2008) stresses that their frequency-based approach for defining reference conditions should only be used as a starting point and the estimated values for criteria can be adjusted based on the distribution of sites or other environmental or anthropogenic factors. Best professional judgement (BPJ) needs to be employed in the selection of threshold concentrations, as these rudimentary approaches do not account for any external variables that may influence the distribution of nutrient levels.

This modified method was an attempt to overcome the issues of small sample size, limited sampling frequency, combination of different wetland types and sites located in multiple ecoregions. The resulting suggested reference conditions are meant as a first step towards the further development of criteria. All calculations were completed in R 3.4 (R Core Team, 2017) and bootstrap sampling was completed with the Simpleboot package in R (Peng, 2015).

Comparing Metals to State Surface Water Standards

Water quality results for metals were compared to typical surface water concentrations using boxplots. In addition, the results were compared to the surface water standards for the protection of aquatic life. These standards were developed for freshwaters but are not intended for wetlands. Samples were compared to two different versions of the standards for Cu, Zn and Pb. First, comparisons were made to the historical surface water standards for North Carolina that were used prior to 2015. These standards were expressed in total metals. Second, comparisons were made to the current North Carolina standards for dissolved metals. A conversion factor from USEPA (1996) was used to transform the dissolved standards to an estimated total metals equivalent.

Current dissolved standards for metals have two thresholds: chronic and acute levels. The current aquatic life water quality standards are based on dissolved metal concentrations (NC DEQ 2017), because dissolved metals are more bioavailable and thus potentially more toxic to aquatic life (US EPA, 1996). Dissolved metals are operationally defined as the portion passing through a 0.45 micron filter. These standards for dissolved metals (Cu, Pb, Zn) are hardness-dependent; meaning the threshold concentration increases for increasing hardness levels. Metals are less toxic to aquatic organisms at higher hardness levels because aquatic organisms more readily uptake magnesium and calcium (primary contributors to hardness) than the toxic metals

(NCDENR, 2015). The hardness dependent standards are calculated using equations (Table 3.5) developed by (US EPA, 2004). Hardness was calculated for each sample using equation 3.1 below from NC DEQ (2015).

Table 3.5: Hardness dependent standards for dissolved metals and historical total metal standards.

Metal	Equations for Hardness-Dependent Dissolved Metals Standards (µg/L)	Total Metal Standard (Pre 2015) (µg/L)
Copper, acute	$WER*(0.960*e^{0.9422*\ln(hardness)-1.700})$	7.0
Copper, chronic	$WER*(0.960*e^{0.8545*\ln(hardness)-1.702})$	
Lead, acute	$WER*[(1.46203-\ln(hardness)*0.145712)*e^{1.273*\ln(hardness)-1.460}]$	25
Zinc, acute	$WER*[0.978*e^{0.8473*\ln(hardness)+0.884}]$	50
Zinc, chronic	$WER*[0.986*e^{0.8473*\ln(hardness)+0.884}]$	

For water hardness of 25 to 400 mg/L
WER= Water Effects Ratio- Factor used to adjust laboratory developed water quality standards to site specific criteria due to differences in site water chemistry. A value of 1.0 was used for this study.

$$\text{Hardness} = 2.497 * Ca^{2+} + 4.118 * Mg^{2+} \qquad \text{Equation 3.1}$$

Because the sampling protocol for metals was to analyze for total recoverable metals (dissolved plus particulate bound) per NC DEQ historical studies and later this study, conversion factors from USEPA (1996) were used to translate the current criteria values for dissolved metals to total recoverable metal criteria thresholds (Table 3.5). The conversion factors were estimated using default-partitioning coefficients for streams from USEPA (1996). While, this is likely not the most accurate conversion due to the differences in pH, DOC, TSS, and other parameters between wetlands and streams that could affect the relative particulate and dissolved fractions, this provides an estimate for the purposes of this project (Table 3.6). See Appendix I for the calculations for these conversion factors.

Table 3.6: Factors for converting from dissolved metals criteria to total recoverable criteria values.

Metal	Conversion Factor
Cu	2.87
Pb	6.71
Zn	3.47

Results and Discussion

The following section includes the results in boxplot format and comparisons to background conditions for streams from USGS (2010). More detailed results can be found in Appendix H.

Nitrate-N

Overall site median values were low, with 14/16 site median values below 0.1 mg/L and 9/16 site median values below the detection limit of 0.02 mg/L (Figure 3.5). The influence of land use was apparent on NO_3^--N concentrations. For example, one headwater wetland site (HOG site) downslope from a waste disposal spray field had a median value of 26 mg/L and samples were recorded above 40 mg/L. The lowest value recorded at this site was 10 mg/L. In other agricultural sites, the median values were substantially less than 0.2 mg/L, but occasional values above 3.0 mg/L were recorded. More frequent sampling would be required to adequately characterize the nitrate fluctuations in these wetland systems. In the agricultural impacted sites, higher values were generally observed in the winter (Figure 3.4), which was likely due to lower denitrification rates because of lower temperature and decreased microbial activity (Reddy and DeLaune, 2008).

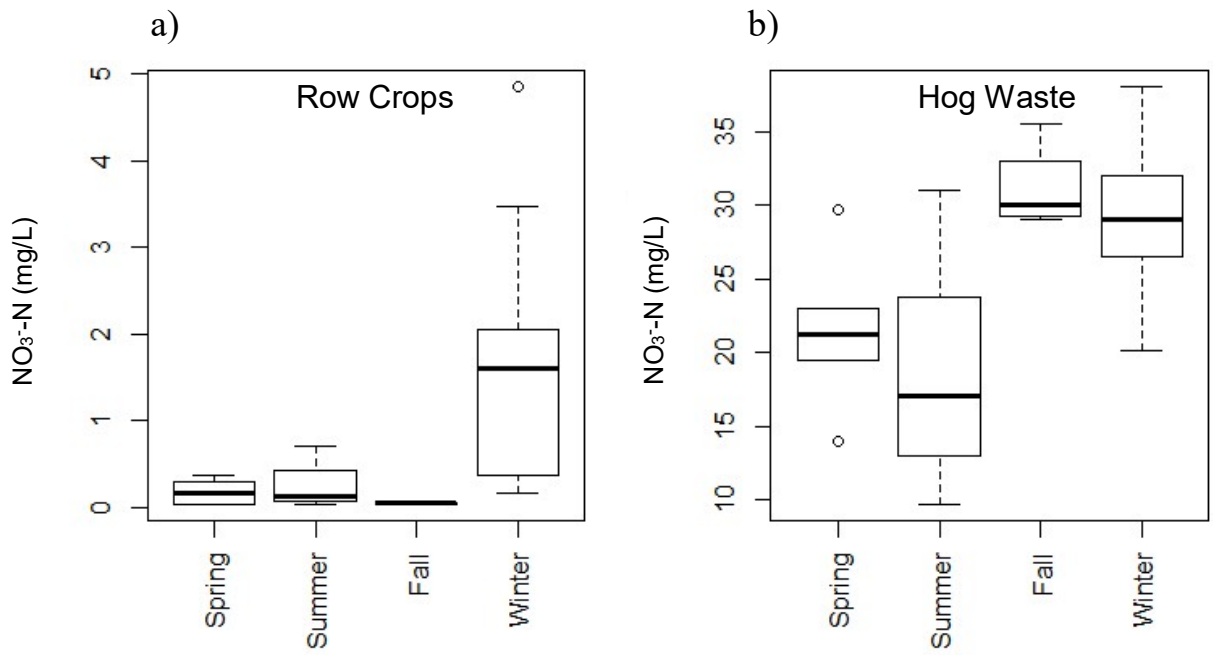

Figure 3.4: Seasonal differences in nitrate-N concentrations for two "Ag. Impacted" headwater sites at a) downslope from row crops (NAH site) and b) downslope from hog waste spray field (HOG site).

Most of the samples collected were below the laboratory's limit of detection. For the "Reference" and "Avg. Dist." sites 73% and 61% of the samples, respectively, were below the detection limit. Even in the "Ag. Impacted" sites, 25% of the samples fell below the detection limit. This indicates that a single sample may not indicate disturbance and longer-term sampling is required to detect elevated concentrations. The very low values documented in the "Reference" sites were similar to values observed in previous studies from natural wetlands. For example, Bruland et al. (2003); Chow et al. (2016); Justus et al. (2016); Lane et al. (2015); Reiss (2006); Skaggs et al. (1980) all observed mean values below 0.1 mg/L in undisturbed natural wetlands, and mean values were often well below 0.05 mg/L.

A large majority of the samples were below the national background level for stream of 0.24 mg/L from USGS (2010). For "Avg. Dist." and "Reference" sites, over 95% of the samples were below the background level. For the "Ag. Impacted" the sites, 60% of the samples were

below the background level for streams. This was the first indication a lower background level was likely applicable for wetlands. The US EPA's draft nutrient criteria for streams and lakes across North Carolina's two ecoregions ranged from 0.02 to 0.13 mg/L. This was closer to the range of values observed in the reference sites. A more detailed site-by-site analysis of the number of samples exceeding the detection limit and the values below the background level for streams can be found in Appendix H.

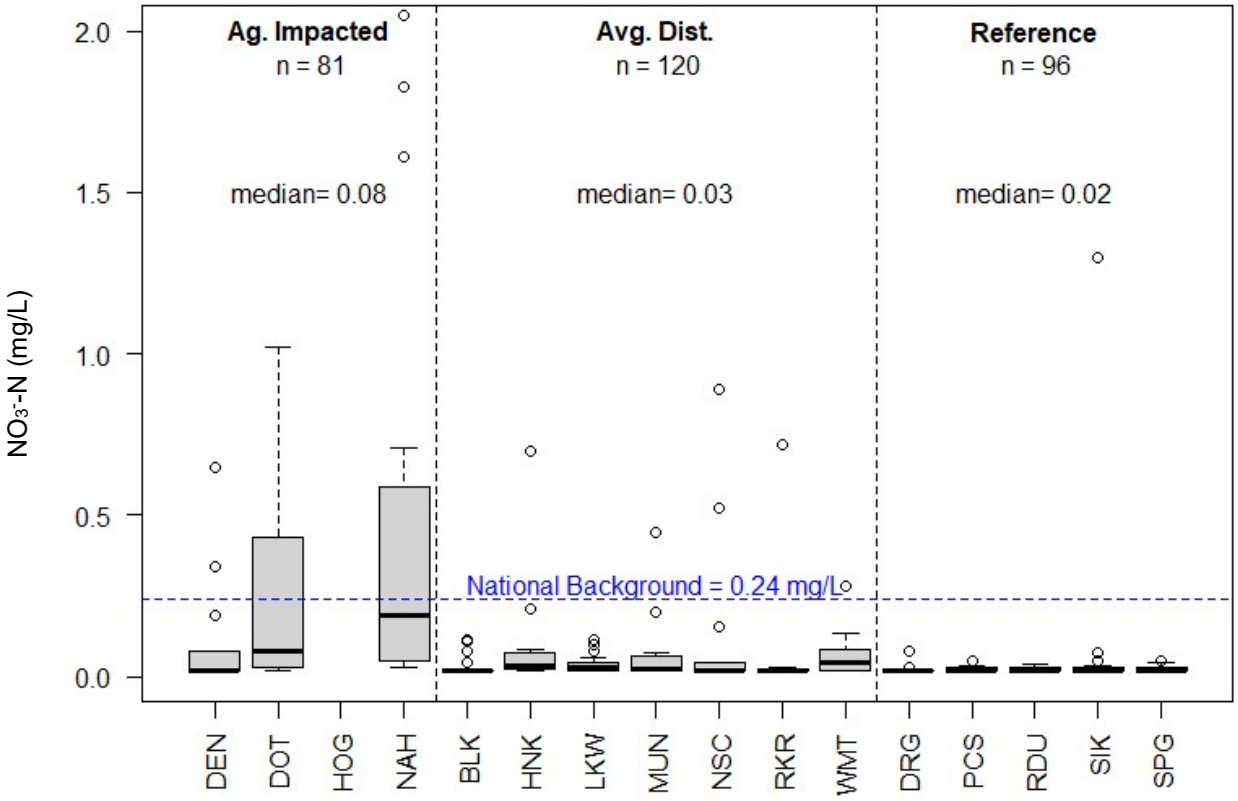

Figure 3.5: Nitrate-N by site and disturbance class. Vertical axis was adjusted to eliminate extreme values and extreme outliers.

Ammonium-N

Ammonium-N values were also low overall, with a quarter of wetland site medians below the detection limit. Median values in 15/16 sites were below 0.15 mg/L. The maximum median value observed was 0.26 mg/L; however, this was in the tidal riverine site (LKW site). The elevated ammonium levels in this site may be the result of the unique hydrologic regime, which

resulted in highly reduced conditions, and elevated salinity (median specific conductivity of 1100 µs/cm). Previous studies of wetlands in the southeastern U.S. also reported median ammonium values of generally less than 0.15 mg/L for undisturbed sites (Reiss, 2006; Skaggs et al., 1980) and up to 0.33 mg/L for sites in agricultural and urban areas (Reiss 2006). In an interesting exception, Walbridge and Richardson reported values above 1.2 mg/L in extremely acidic, ombrotrophic North Carolina pocosins. However, these elevated values were not necessarily surprising given wetlands often have internal sources of ammonium due to the decomposition of organic matter when mineralization outpaces immobilization (Reddy and DeLaune, 2008). In wetlands ammonium can accumulate in the water column due to the low rate of nitrification in the low oxygen environment. These anaerobic or low oxygen conditions are the result of stagnant water which limit the resupply of oxygen to the water column. In addition, nitrification can be inhibited by low pH (Reddy and DeLaune, 2008), which is typical of organic soil wetlands in North Carolina.

Overall, 45% of the samples were below the laboratory practical quantitation limit. In the "Reference" and "Avg. Dist." sites 51% and 35% fell below the detection limit, respectively. Just under 40% of the samples were below the detection limit in the "Ag Impacted" sites. A similar percentage of samples fell below the national background level for streams from USGS (2010) of 0.025 mg/L. This data indicated that a higher background level for ammonium would likely be more appropriate for wetlands.

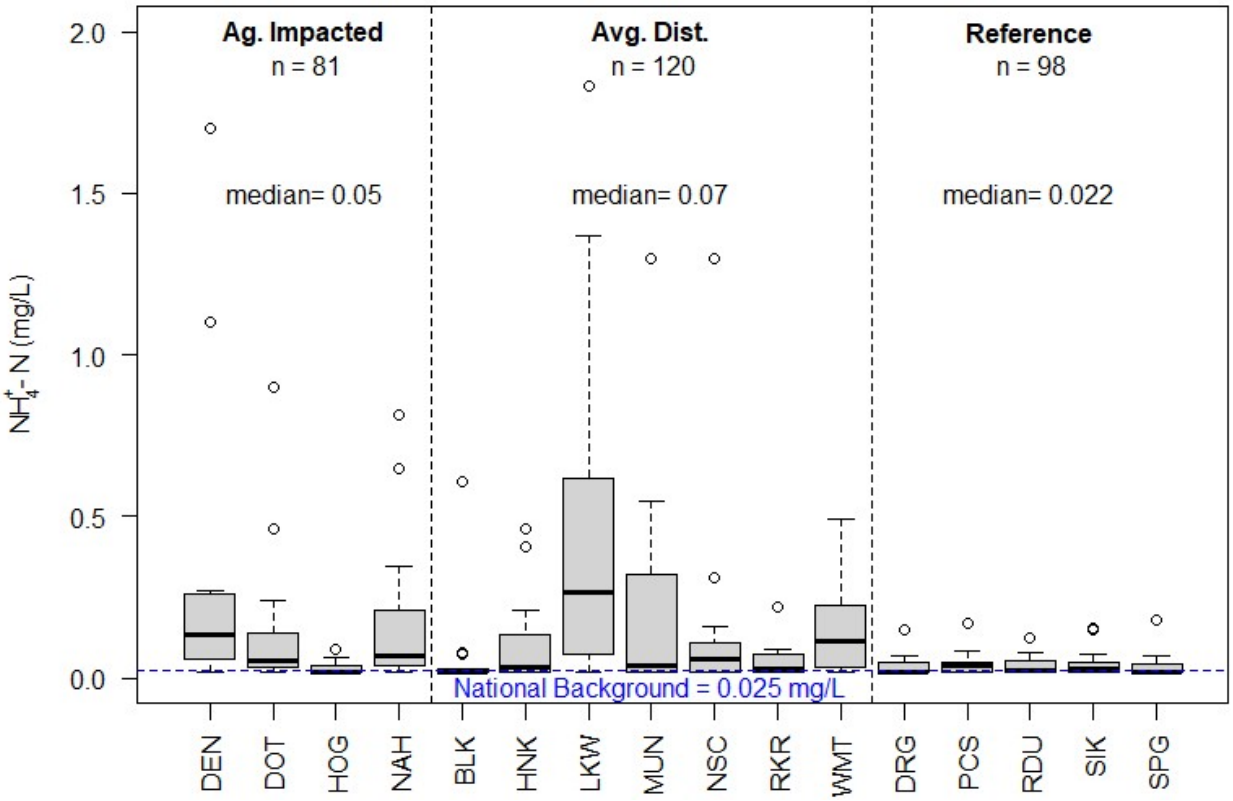

Figure 3.6: Ammonium by site and disturbance class. The vertical axis was adjusted to eliminate extreme outliers.

Total Phosphorus

Site medians for TP ranged from 0.06 to 0.38 mg/L. Overall medians increased from "Reference" to "Ag. Impacted" and the sites in the agricultural land use all had median values in the upper 50th percent of the sites (Figure 3.7). Some previous studies reported mean values ranging from 0.025 to 0.08 mg/L in undisturbed wetland sites (Bruland et al., 2003; Daniel, 1981; Lane et al., 2015; Reiss, 2006; Skaggs et al., 1980), while agriculture-impacted wetland sites may exhibit mean values up to 0.8 mg/L Reiss (2006). The median values for the "Avg. Dist." and "Ag. Impacted" sites reported here were similar to values reported by (Bruland et al., 2003) for a restored wetland and Skaggs et al. (1980) for Ag Drainage. When compared to ammonium or nitrate-N, there were very few values measured below the detection limit; only 5% for the "Ag. Impacted" and "Reference" samples and 9% for the "Avg. Disturbance" samples.

TP concentrations measured in these wetlands appear to be much higher than values set for background conditions for other aquatic ecosystems. For example, USGS (2010) sets TP background for streams at 0.03 mg/L. The US EPA draft nutrient criteria for lake and streams ranged from 0.02 to 0.05 mg/L for North Carolina ecoregions. All site medians exceeded these values. Under 10% of the values were below the national background level for streams. This again indicated that wetlands likely require a higher background level than streams for TP. Outliers were present across all of the sites and more frequent sampling would likely be required to adequately characterize the variability of TP concentrations in these systems.

TP concentrations can fluctuate based on water level, redox potential, and vegetation processes. For example, P can be released from wetland soils under inundated, anaerobic conditions but rebind to the soil matrix under aerobic conditions (Mitsch and Gosselink, 2015). Due to these dynamic processes affecting TP, wetlands can function both as sources and sinks for phosphorus (Reddy and DeLaune, 2008). According to the observed median concentrations, these wetland ecosystems would either qualify as eutrophic or hypereutrophic systems (Reddy and DeLaune, 2008). However, based on the very low inorganic nitrogen levels observed, these wetlands may not be phosphorus limited, which is typically the case for lakes and streams (Reddy and DeLaune, 2008).

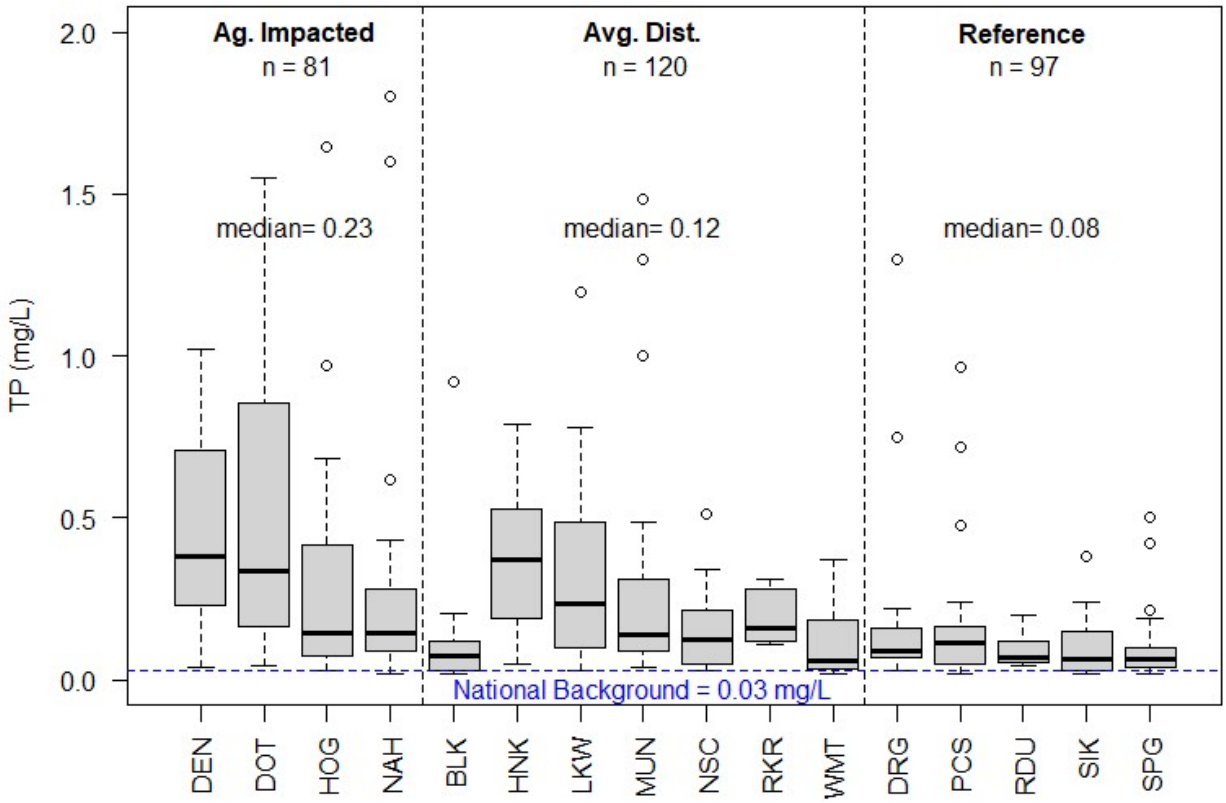

Figure 3.7: Total phosphorus by site and disturbance class. Vertical axis was adjusted to exclude extreme outliers.

Organic Nitrogen

Median site organic nitrogen values ranged from 0.36 to 2.0 mg/L (see Figure 3.8). The highest values tended to be in the sites with long retention times and stagnant conditions where recalcitrant organic material builds up over time. The lowest values tended to be in the Piedmont headwater and riverine wetland sites, which had shorter periods of inundated conditions. The site with the highest median and most variability (LKW site) was a tidal riverine site with very high suspended sediment levels that may have contributed to the higher ON and thus TN results. On average, organic nitrogen made up over 80% of nitrogen in the water samples in a majority of these sites (Figure 3.9). For the "Reference" sites, the relative percentage of ON was typically near 90% and above. For the "Avg. Dist." sites, the mean relative ON percentage was typically

above 80%. However, where significant loading of inorganic nitrogen occurs (e.g. agricultural settings) the relative percentage of organic nitrogen can be much lower.

Results from previous studies of natural wetlands also indicate that ON makes up a majority of the total nitrogen in wetlands (Skaggs et al., 1980) and this is the case for most undisturbed aquatic ecosystems. Natural wetlands can serve as a source of organic nitrogen to downstream aquatic ecosystems, with ON increasing as percentage of wetland area in the watershed increases (e.g. Pellerin et al., 2004). Therefore, production of ON is a natural process and higher levels of organic nitrogen in some wetlands should not be interpreted as an indicator of contamination.

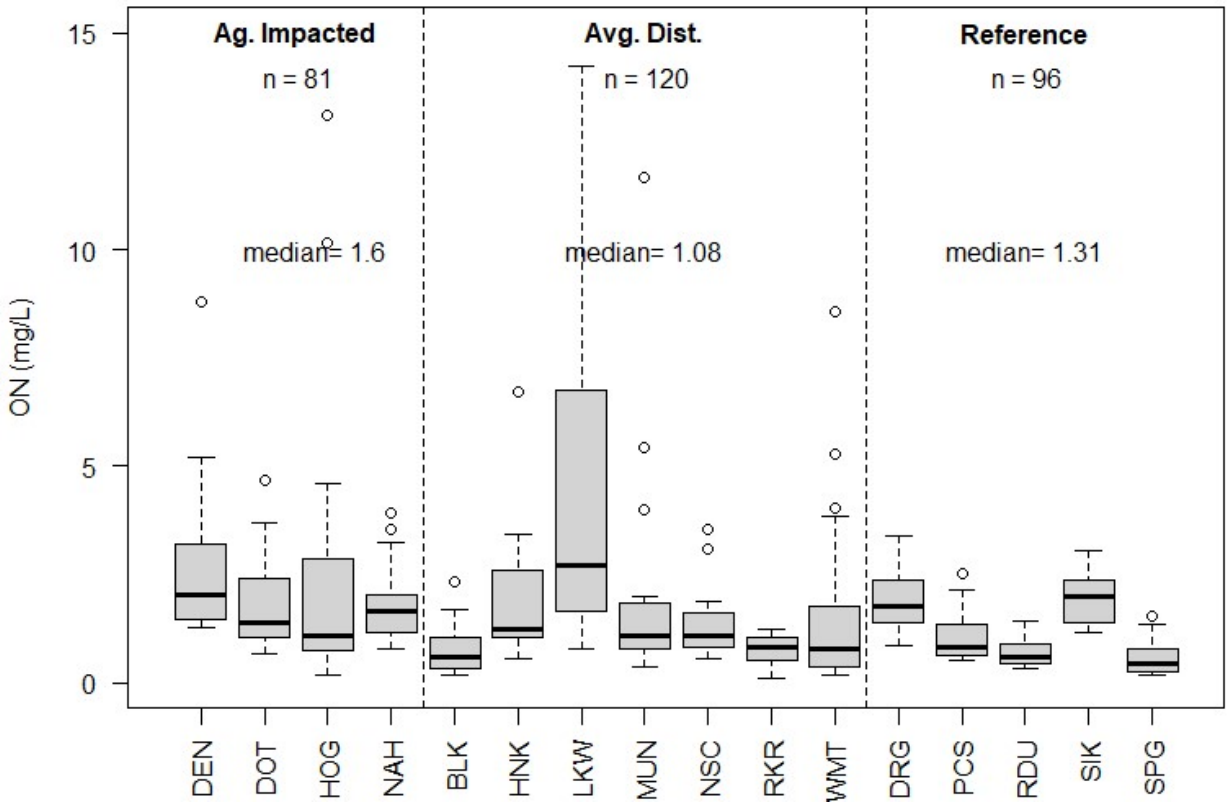

Figure 3.8: Organic nitrogen by site and disturbance class.

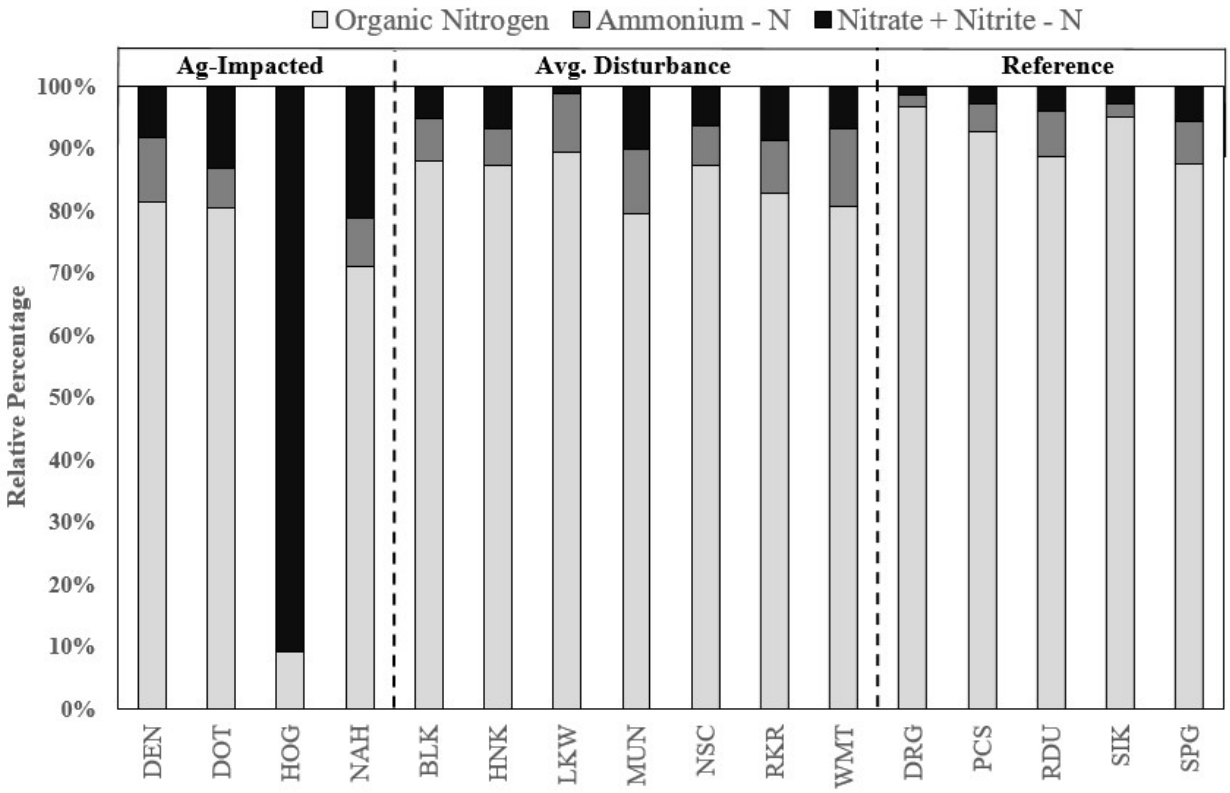

Figure 3.9: Relative percentage of ON, NH_4^+-N and NO_3^--N for each site grouped by disturbance category. Values shown are mean relative percentages from all samples.

Total Nitrogen

Site medians for total nitrogen ranged from 0.4 mg/L to 25 mg/L; however, 15/16 site medians were below 2.4 mg/L. The median value of 25 mg/L was the result of the very high nitrate-N values. Because of the low inorganic nitrogen results documented in this study, total nitrogen results were largely a reflection of the organic nitrogen values. Overall, in undisturbed wetlands TN will generally be approximately equivalent to ON. Median values fell in the range of previous research in natural wetlands (Bruland et al., 2003; Lane et al., 2015; Skaggs et al., 1980). Overall, about 25% of the "Avg. Dist." and "Reference" samples were below the national background level for streams (USGS, 2010). None of the samples from the "Ag. Impacted" sites were below the national background level. This indicates that the background levels for streams is too low to apply to wetland systems. This was also the case for the draft nutrient criteria for

streams and lakes from US EPA for North Carolina ecoregions, which were lower than the background levels from USGS (2010). However, the large range in median values of about 0.4 to 2.4 mg/L and the apparent differences among wetland types seems to indicate that wetland type and perhaps ecoregion specific levels for ON and TN are more appropriate as recommended by USEPA (2008).

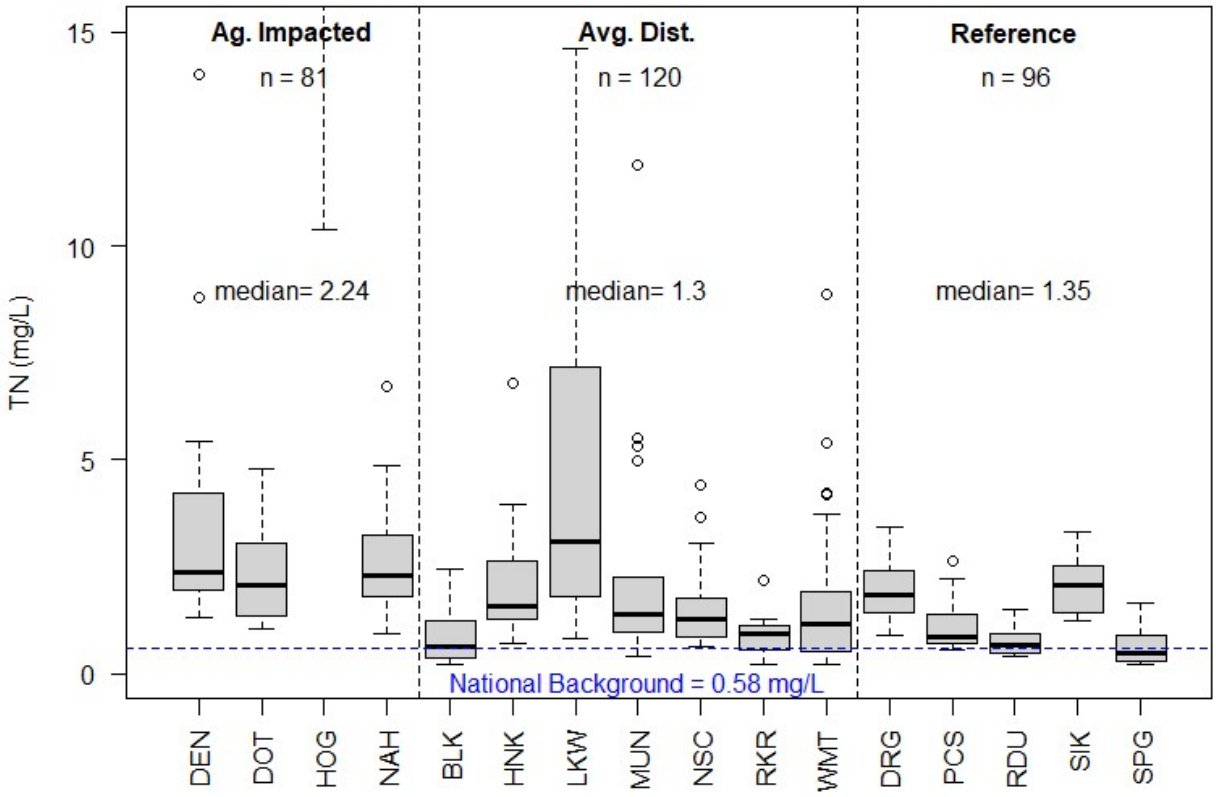

Figure 3.10: Total nitrogen by site and disturbance class. The vertical axis scale was adjusted to eliminate extreme outliers and very high values.

Specific Conductivity

Previous research shows specific conductivity is generally low in undisturbed freshwater wetlands (Reiss, 2006; Walbridge and Richardson, 1991); however, there are differences in wetland type, underlying geology and the source of nutrients and minerals. Elevated specific conductivity levels can result from wastewater discharges, agricultural runoff, stormwater runoff, and other industrial sources (US EPA, 2016). Site medians ranged from 30 to 460 µS/cm for the

freshwater sites (Figure 3.11); the one tidally influenced site (LKW) had a median value above 1000 µs/cm. For the most part, values for undisturbed sites were below 150 µs/cm with median values below 110. Sites with the greatest level of disturbance had the highest specific conductivity levels. For example, one riverine site that was traversed by a sewer pipe and received stormwater runoff had median specific conductivity of 208 µs/cm and a site downslope from a spray field where nitrate-N regularly exceeded 30 mg/L had a median value of 460 µs/cm. In addition, the impacts of land use were also apparent on two basin wetland sites in Granville Co. For the largely undisturbed reference site (DRG), the median specific conductivity value was 43 µS/cm, while the other basin site (DEN), which is adjacent to an agricultural field, had a median value of 90 µS/cm.

Previous research has shown that specific conductivity can fluctuate over short periods and significant diurnal changes have even been observed (Bosserman, 1984). For these reasons, it would be very difficult to develop criteria for specific conductivity without an extensive data collection effort. While more research is needed, specific conductivity is potentially a good indicator of anthropogenic disturbance, as specific conductivity is the measure of the water's ability to conduct electrical current and increases with increased concentration of inorganic ions. US EPA is currently considering developing specific conductivity criteria for streams in some ecoregions (US EPA, 2016).

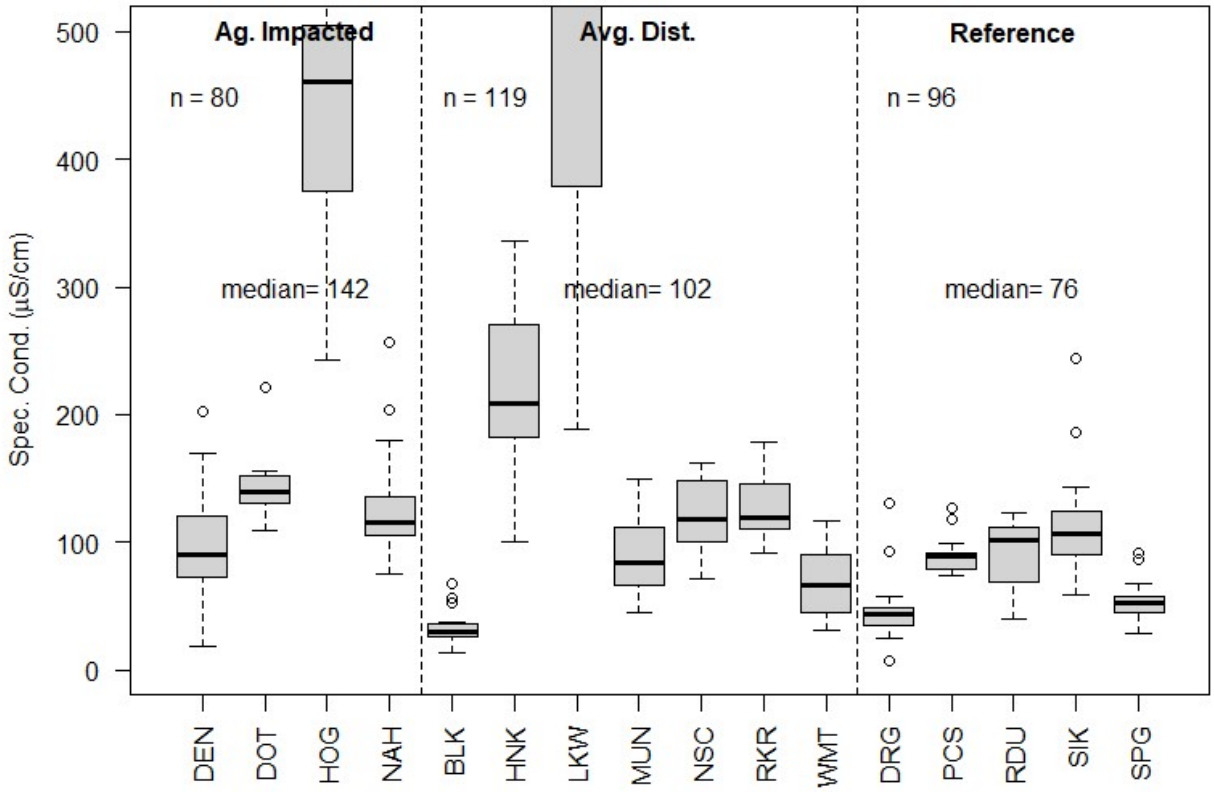

Figure 3.11: Specific conductivity by disturbance class. The vertical axis scale was adjusted to demonstrate the variability in a majority of the sites. A full-scale plot can be found in Appendix L.

pH

pH ranged from very acidic to near neutral (Figure 3.12). For the riverine wetlands, pH was generally in the range of 5.5-7, similar to values reported by Mitsch and Gosselink (2015). Basin wetlands had lower pH that generally fell in the range of 3.5 -5.5, which is also comparable to the conditions described by (Mitsch and Gosselink, 2015). In the headwater wetlands, pH ranged to 4.5 – 6.5, with the exception of one organic soil wetland where the median value was 3.5. The pH of the waters in this deep organic soil wetland (PCS site-headwater) was very low (<4), which falls in the range previously reported for organic soil wetlands (Walbridge and Richardson, 1991). Within site variability of pH was about 1–2 pH units during the study period. pH can vary in wetland ecosystems depending on hydrologic regime, redox potential, chemical inputs, flooding, and algal productivity (Reddy and DeLaune,

2008). It is clear that state surface water standards for pH of 6.0 to 9.0 (NC DEQ, 2017) should not be applied to wetland systems and that region and wetland type specific ranges would need to be developed if the goal was to create pH background ranges or criteria for wetlands. Changes in pH could be used as an indicator of pollution as long as initial ambient concentrations are known. For example, inputs of caustic or acidic chemicals might cause a large shift in pH that could be detrimental to wetland biota.

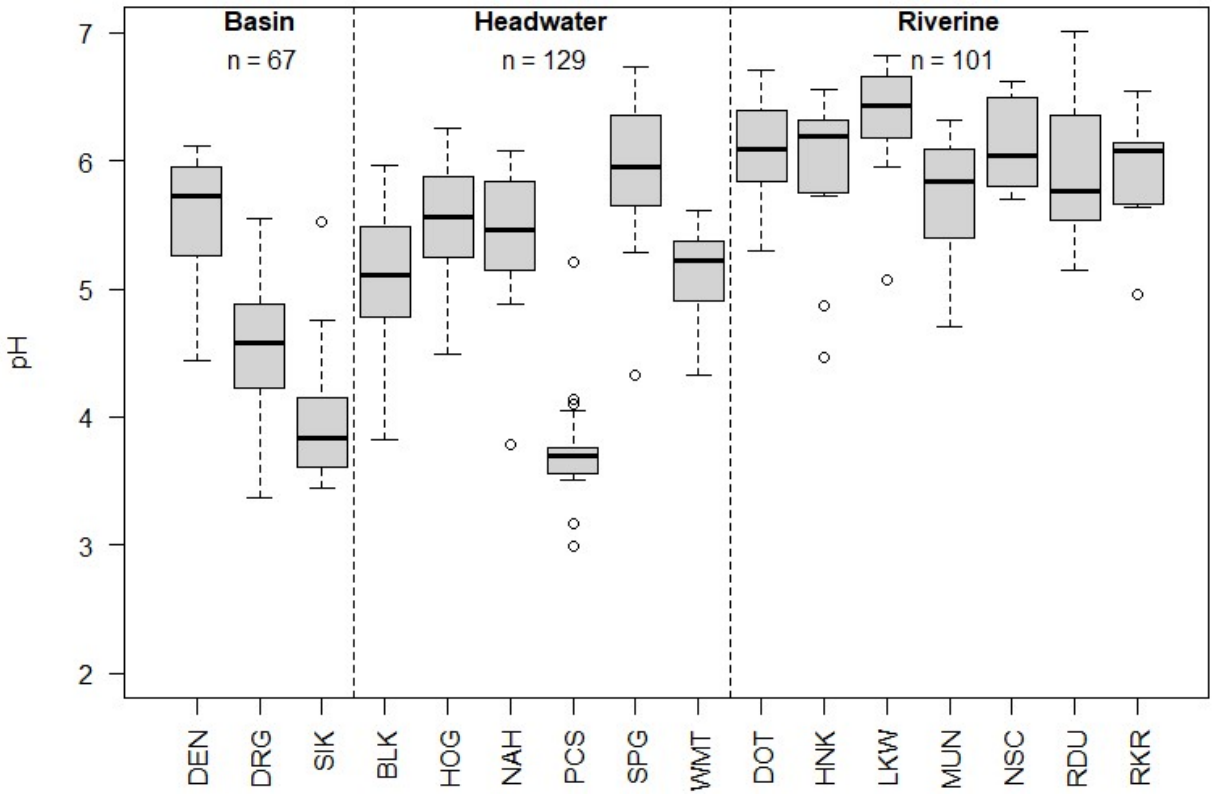

Figure 3.12: pH by wetland type.

Dissolved Oxygen

As expected, it was apparent that the state surface water standard for DO (>5.0 mg/L) was not applicable to wetlands, and standards for DO would not be appropriate for wetlands. Very low DO conditions are a natural occurrence in wetland ecosystems, and the low oxygen levels drive many biogeochemical processes unique to wetlands. In this study, DO varied across a wide range from near anaerobic to above 100% saturation, with median values in the range of

2-4 mg/L. DO is a highly variable parameter that is dependent on temperature, and is influenced by flow vs. no flow conditions. In addition, DO can vary by several mg/L diurnally when algae is producing oxygen through photosynthesis (Reddy and DeLaune, 2008) and there was some evidence of this observed during this study. Lower DO values were generally observed in the summer months, when temperatures were highest. For wetlands, DO is not an indicator of pollution or degradation as it is in other surface water bodies.

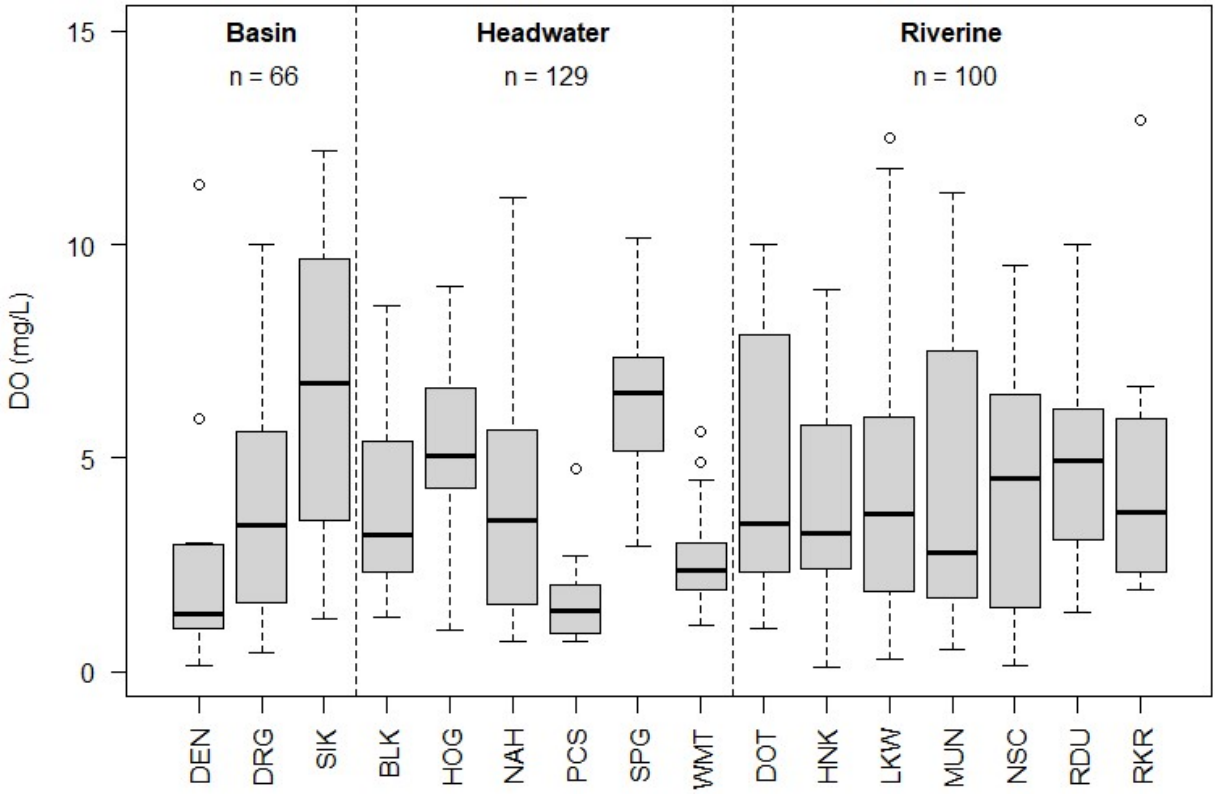

Figure 3.13: Dissolved oxygen by wetland type.

Proposed Nutrient Thresholds for Reference Conditions

Nitrate-N

The bootstrap confidence intervals for nitrate-N indicated there was clear separation between the disturbance classes for the upper percentiles (Figure 3.14). This separation can be interpreted as evidence of the impact of land use on nitrate-N levels. The narrow confidence intervals for the "Reference" sites indicated that there was little variability and that nitrate-N may

be a strong indicator of disturbance because of the consistent levels measured in undisturbed sites. The calculated threshold was 0.07 mg/L. This value is similar to the upper limit of background or irreducible concentration of 0.1 mg/L for constructed wetlands from USEPA (2000e). This threshold value only separates "Reference" sites from the other land uses at the upper percentile. For example, 70% or more of the samples collected in "Avg. Dist." site would likely fall below this level and almost half of the samples collected in an "Ag. Impacted" site may fall below this level. Therefore, this threshold can only be used as an upper threshold above which some anthropogenic disturbance may be present. No inferences can necessarily be inferred regarding the long-term water quality based on a sample below the threshold.

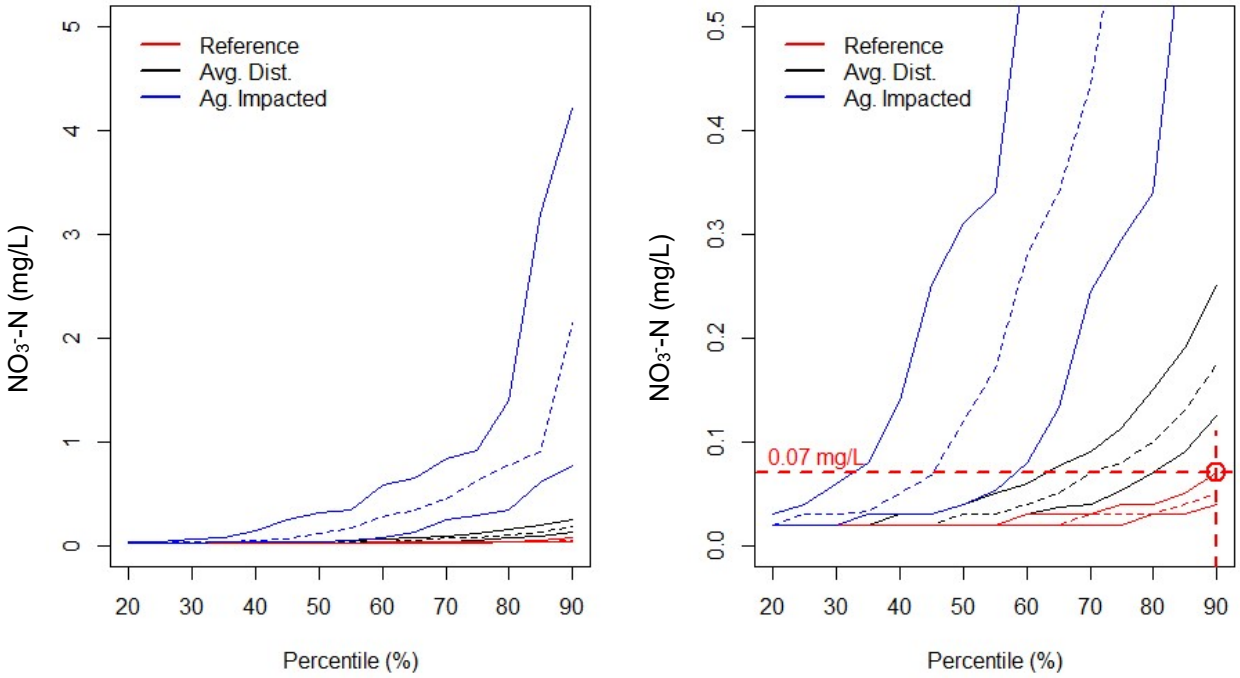

Figure 3.14: Bootstrap confidence intervals by percentile for each disturbance group for nitrate-N. Plot on the right is same data with y-axis scaled for clarity.

Ammonium-N and Organic Nitrogen

For ammonium-N, there was again some separation between the "Reference" sites and the other disturbance classes at the upper percentiles indicating the impact of land use on

ammonium-N concentration extremes (Figure 3.15). Again, there was little separation between any of the disturbance classes at the lower percentiles. The calculated reference threshold was 0.15 mg/L. This again only represents a level above which some anthropogenic influences are likely affecting the wetland water chemistry. For the other disturbance classes, 60-70% of the samples were below this threshold. The proposed reference threshold was slightly higher than the upper end of the background concentrations for constructed wetlands of 0.1 mg/L from USEPA (2000).

For organic nitrogen, there was much less separation between the confidence intervals for the different disturbance classes and the confidence intervals were narrow for the "Reference" sites (Figure 3.15). This is likely the result of including wetlands of different types and regions in the analysis. For example, depressional wetland organic nitrogen medians averaged about 1.9 mg/L, while it was only 1.1 for riverine wetlands, and only 0.75 for headwaters, although there was a large range among the wetland types and regions. Therefore, wetland type specific thresholds are likely necessary for ON. However, ON is generally not considered a pollutant at these levels and threshold values may not be appropriate for ON, given the large fluctuations observed in natural systems.

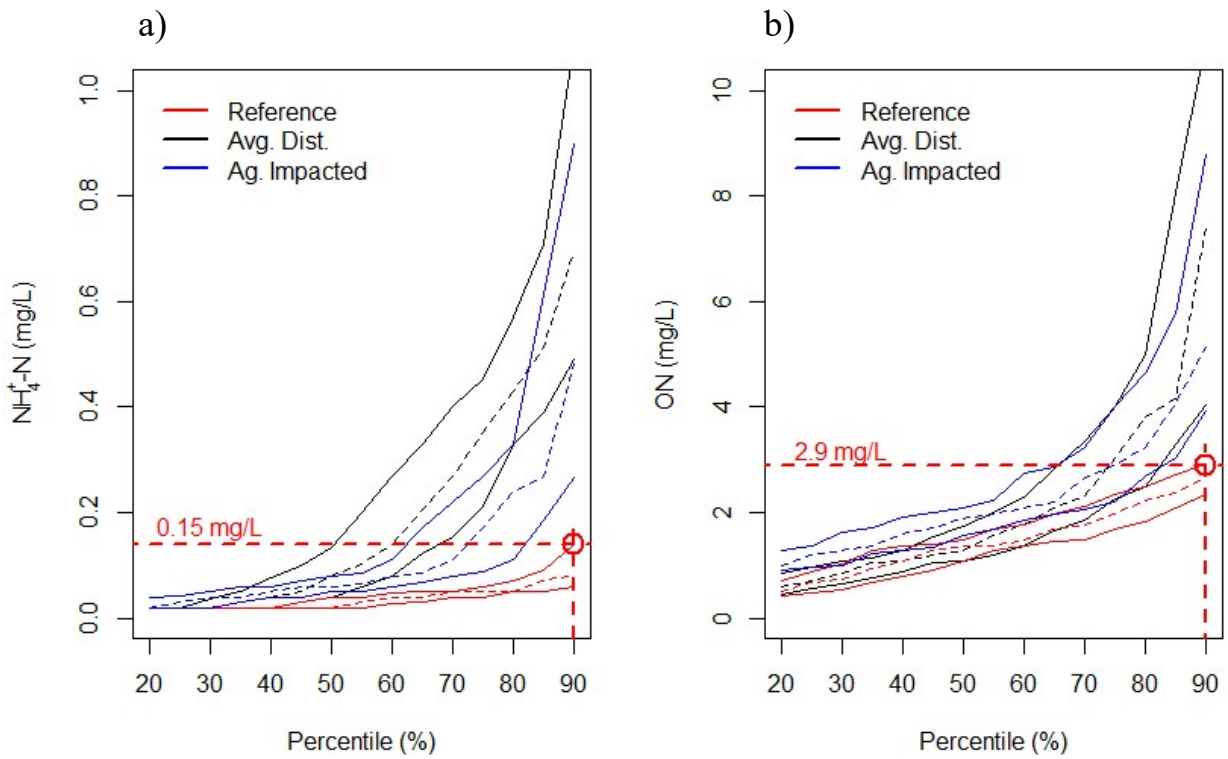

Figure 3.15: Bootstrap confidence intervals and reference condition thresholds for a) ammonium-N and b) organic nitrogen.

Total Nitrogen and Total Phosphorus

For total nitrogen, there was some separation between the confidence intervals for the "Reference" group above the 70% percentile. However, based on the range of TN values observed, this may not be an accurate representation of differences between the groups. Total nitrogen may not be a good indicator of wetland disturbance, especially at the levels typically observed in natural wetlands. For example, a TN result of 2.4 mg/L could be the result of 2.4 mg/L ON while NH_4^+-N and NO_3^--N values were below the detection limit, or 1.4 mg/L NO_3^--N, NH_4^+-N below detection and 1.0 mg/L ON. These would be two completely different indications of inputs to the wetland. The first scenario might be commonly observed in an undisturbed depressional wetland, while the second might occur in an agriculture or urban-development affected area. Analysis indicated the threshold value calculated for TN was 3.2 mg/L, which is

very similar to the upper range of the background (irreducible) concentrations for constructed wetlands (3.0 mg/L) from US EPA (2000). For comparison, the background (reference) levels for streams and lakes from US EPA and USGS range from 0.4 to 0.58 mg/L for North Carolina.

For total phosphorus, a number of outliers contributed to a very wide upper confidence interval for the "Reference" group, which eliminated any separation between the disturbance classes. The calculated upper threshold for TP was 0.6 mg/L (Figure 3.16), which is three times the upper threshold for background conditions for constructed wetlands from (US EPA, 2000e). However, US EPA suggests that BPJ must be employed when selecting thresholds using such a rudimentary approach, and percentile used for threshold development should be adjusted based on the population of wetlands sampled (US EPA, 2008). For example, MTDEQ (2008) used a range of percentiles when developing criteria for streams. Thus, from Figure 3.16 it is apparent that there is considerable variability of the upper confidence interval above the 85th percentile and the threshold should likely be lowered. The threshold was lowered to 0.3 mg/L, which corresponds to the 85th percentile of the distribution. This level is still higher than the background for constructed wetlands (0.2 mg/L) and would be higher than about 50-60% of the samples from "Ag. Impacted" or "Avg. Dist." samples. The 0.6 mg/L threshold would have been higher than 70-85% of those samples.

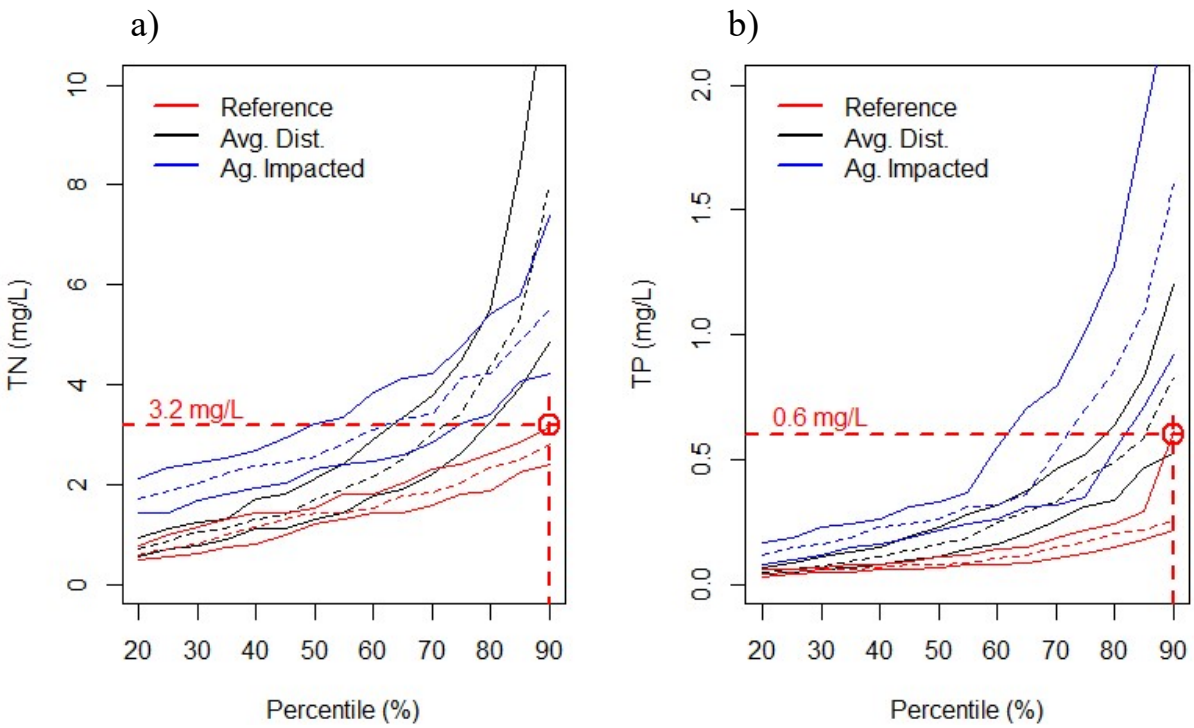

Figure 3.16: Proposed reference thresholds for a) total nitrogen and b) total phosphorus.

Overview of Nutrient Thresholds

These proposed upper limits or thresholds for reference conditions (Table 3.7) are not intended to give any indication of contamination or need for corrective action, only that concentrations above this level may be indicative of some anthropogenic disturbance upstream that is impacting the wetland water chemistry in such a way to alter it from ambient background conditions. These thresholds are based on the theory that in undisturbed, natural systems there should be limited variability in nutrient concentration and some upper limit will define the maximum concentration that is theoretically similar for most undisturbed sites. The proposed thresholds are intended to be used for comparison with a single sample. If criteria were developed for site means or medians (for longer-term sampling as opposed to a single grab sample), thresholds would be much lower, likely closer to the detection limit for inorganic nitrogen.

This study validated the hypothesis that the 25th percentile method used by US EPA for stream and lakes may produce prohibitively low thresholds and that would be overprotective (Table 3.7). The background values for streams from USGS (2010) were lower than the proposed thresholds for ammonium-N, TP, and TN, and higher for nitrate-N. The proposed values were very similar to the background (irreducible) concentrations for constructed wetlands from US EPA (2000) (Table 3.7), which was promising.

It is apparent from the large number of samples below the detection limit, even in the more disturbed landscapes, that several samples may be required to adequately characterize the site conditions. This study is a demonstration of the concept of developing draft thresholds as an initial step in a much longer process. These values should not be adopted for any regulatory or legal uses, but may serve as a starting point if wetland water quality standards are pursued further in the future. When evaluating if a given wetland exceeds the reference conditions, it is important to evaluate the thresholds for all constituents concurrently.

Table 3.7: Summary of estimated reference thresholds.

Source	NO_3^--N (mg/L)	NH_4^+-N (mg/L)	ON (mg/L)	TN (mg/L)	TP (mg/L)
Bootstrap CI -90th Percentile (this study)*	0.07	0.15	2.9	3.2	0.3
USEPA (2008) 25th percentile method (this study)	0.02	0.02	0.8	0.91	0.07
Background for FWS constructed wetlands (USEPA, 2000)	<0.1	<0.1	-	1-3	<0.2
Background for streams (USGS, 2010)	0.24	0.03	-	0.58	0.05

*$NO3^-$-N, NH4+-N, ON and TN are based on the 90th percentile. The threshold for TP was adjusted based on best professional judgment to the 85th percentile.

Metals

Metal contamination in wetlands is typically associated with urbanization and industrial runoff. Sources of Zn, Cu, and Pb in the environment include building and infrastructure, cars,

industrial sources, mining and impervious surfaces (ATSDR, 2005). Sources of metals can also include runoff from livestock operations (Nicholson et al., 1999; Zhang et al., 2012).

Elevated levels of heavy metals pose risks to many aquatic macroinvertebrates (Malaj et al., 2012) and amphibians (Vitt et al., 1990) and can cause shifts in species composition to more tolerant species as well as physiological reactions (Malaj et al., 2012). In addition, Zn and Cu can bioaccumulate in fish and other species that consume macroinvertebrates causing further adverse impacts at higher levels of the food chain (Simon, 2002). In addition, at high enough levels, toxic metals can inhibit some microbial and enzymatic processes. This can affect or inhibit many of the significant biogeochemical processes in wetlands, such as mineralization, nitrification, and denitrification (Reddy and DeLaune, 2008).

Site medians for Zn ranged from <PQL to 28 µg/L. There was very little difference between the disturbance class medians (range of <PQL (10 mg/L) to 13 µg/L) (Figure 3.17 and Appendix I). For comparison, mean zinc levels in surface waters across the United States were below 50 µg/L and can be below 5 µg/L in some cases (ATSDR, 2005) and median stormwater concertation from the National Stormwater Quality (NSWQ) Database was 116 µg/L (Maestre and Pitt, 2005). Site median were below 15 µg/L, with the exception of two sites: HNK (riverine) and PCS (headwater). The HNK site was downslope from a developed area, located along a sewer line corridor, received stormwater runoff from a nearby highway and it appeared the floodplain was used as a refuse disposal site (encountered during soil sampling). Therefore, there were several sources of potential contamination at this site. In addition, the zinc levels in the soils (see Chapter 4) were also relatively elevated at this site. Conversely, the elevated levels at the PCS site were not as easily explained. The PCS site was relatively undisturbed, with no obvious sources of metals. However, the downstream sample site may be influenced by a

highway downslope from the site. The elevated levels were not reflected in the soil sampling at this site (see Chapter 4). Elevated samples could be the result of discarded galvanized metals, which was observed around this site. Outliers (with respect to boxplots) were observed at most of the sites and the possible causes are discussed below.

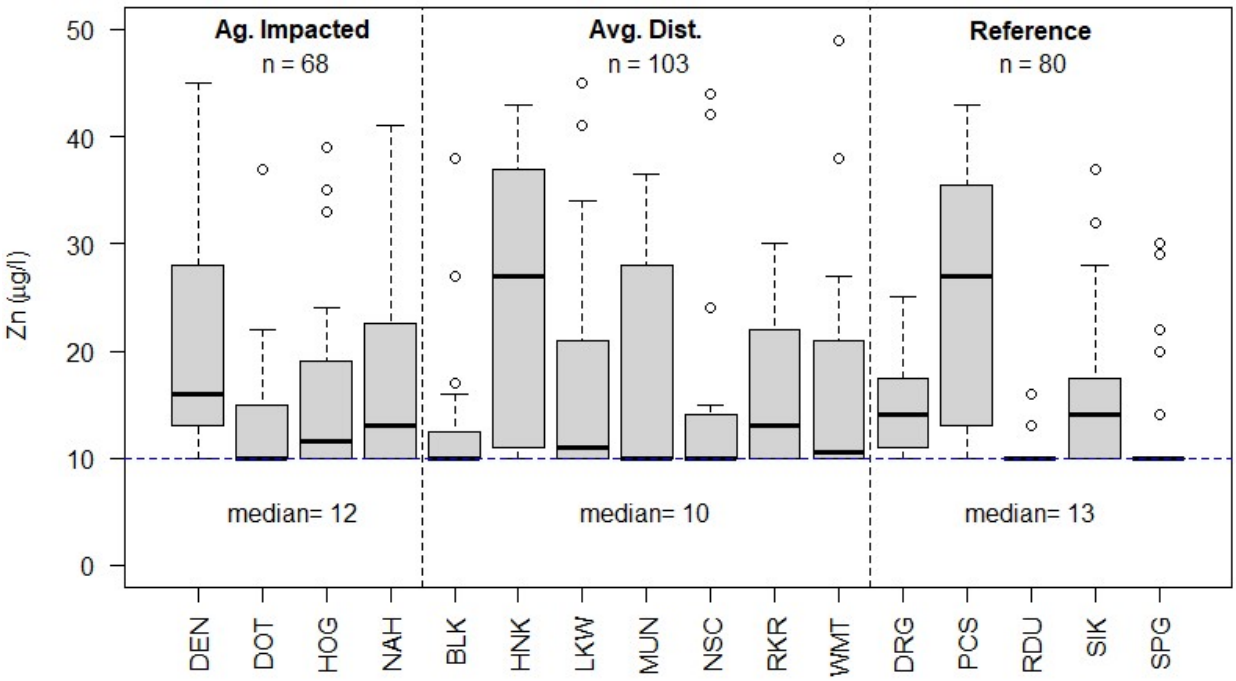

Figure 3.17: Boxplot for Zn. Horizontal dashed blue lines PQL (10 µg/L). The vertical axis was scaled to eliminate extreme values for clarity.

Site medians for Cu ranged from <PQL to 6 µg/L with an overall median of 2 µg/L (<PQL). Again, there was no apparent difference in the disturbance class medians (Figure 3.18 and Appendix I). Copper is also detected in most surface water bodies in the US. A large study of surface waters (lakes and streams) across the country reported a mean concentration of 4.2 ug/L for total copper (Eckel and Jacob, 1988). The background level for reference waters is about 3.0 µg/L (Reddy and DeLaune, 2008).

Site median were low again, with many at or near the detection limit. The HNK site again had the highest median Cu concentration. The RKR site (riverine) also had a slightly elevated median value. This site is a riverine wetland with a highly developed watershed that experienced

frequent overbank events. Outliers were again spread across most of the sites. The "Reference" sites had relatively low values with the exception of outliers at the PCS site.

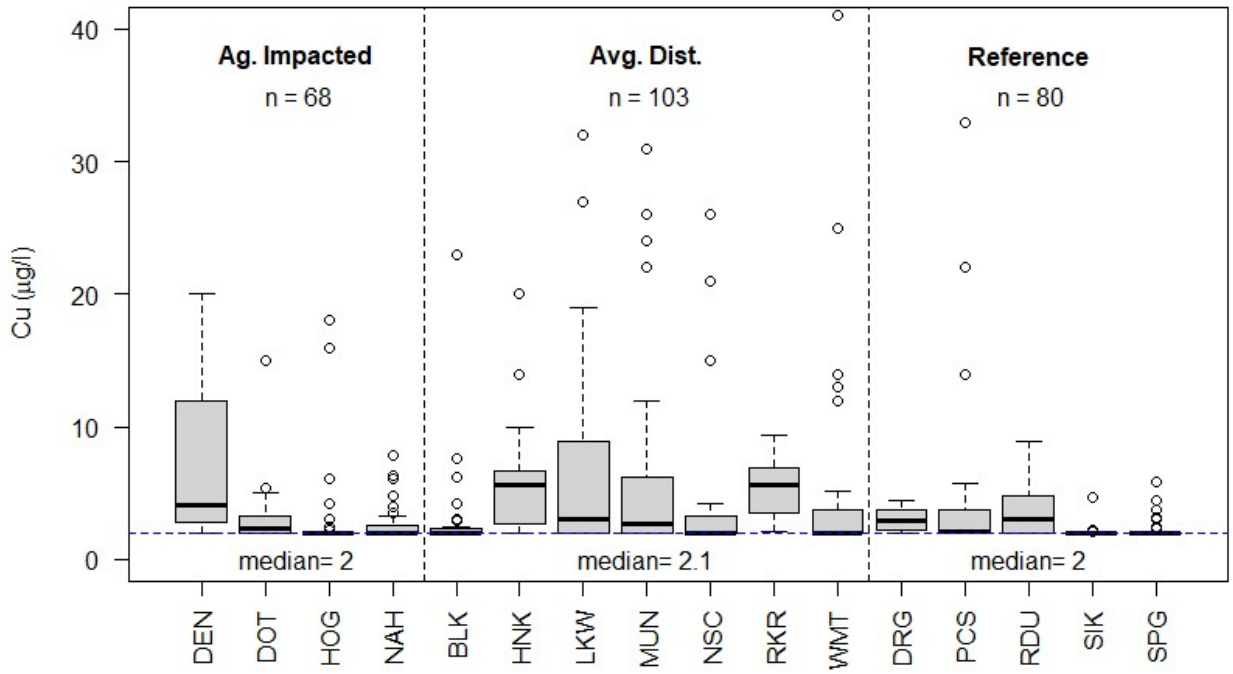

Figure 3.18: Boxplot for Cu. Horizontal dashed blue lines PQL (2 µg/L). The vertical axis was scaled to eliminate extreme values for clarity.

For Pb, all site medians were less than the highest PQL of 10 µg/L (Figure 3.19 and Appendix I). Pb is present in most surface water in the United States, concentrations range from below the detection limit to several hundred micrograms per liter in contaminated areas. Undisturbed areas typically have low concentrations. The median stormwater concentration from the NSWQ Database was 17 ug/L (Maestre and Pitt, 2005). Eckel and Jacob (1988) reported mean lead levels below 4.0 ug/L based on a large study of surface water sampling stations across the United States. Again, there was considerable variability among the outliers at most sites.

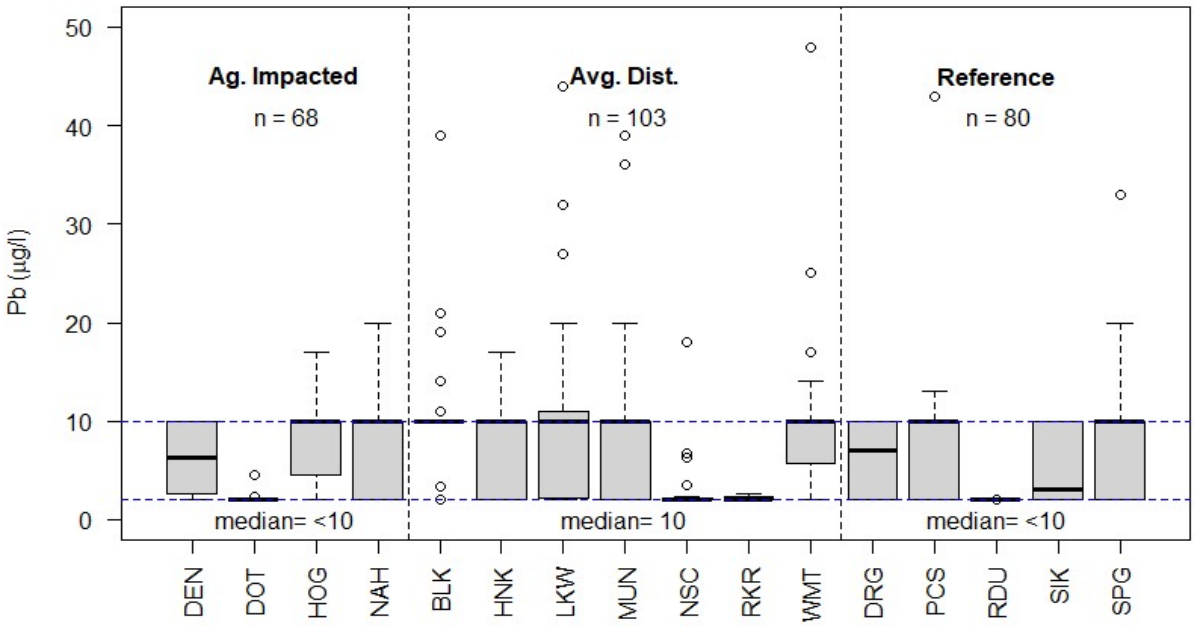

Figure 3.19: Boxplots for Pb. Horizontal dashed blue lines are PQLs (pre-2011: 10 µg/L, post-2011: 2 µg/L). The vertical axis was scaled to eliminate extreme values for clarity.

It was difficult to assess the differences between sites because of the large number of sample results below the detection limit (Table 3.8). Samples were analyzed for total recoverable metals in order to maintain continuity with the long-term data. The plots were adjusted to eliminate extreme outliers that were likely the result of anomalous conditions or possible sample error. It was hypothesized that some of the extreme values for metals was the result of high TSS waters or where soil was collected with the sample. The particulate bound metals in very small volume could increase the analytical results, without representing the actual concentration in the wetland. Dissolved metals were not measured so this could not be verified. See Appendix J for more detail and examples.

Table 3.8: Percentage of samples below the detection limit.

Site	Zn % Below PQL	Cu % Below PQL	Pb % Below PQL
BLK	70	67	70
DEN	11	17	50
DOT	56	38	94
DRG	20	14	71
HNK	19	14	62
HOG	31	9	76
LKW	49	37	51
MUN	61	38	77
NAH	31	66	71
NSC	60	60	70
PCS	9	44	65
RDU	85	39	85
RKR	56	0	78
SIK	39	79	75
SPG	78	78	78
WMT	47	74	84

Zn PQL = 10 µg/L
Cu PQL = 2 µg/L
Pb PQL = 10 µg/L (prior to 2011), 2.0 µg/L (2011 to present)

Comparison to State Standards for Surface Waters

The standards for Cu were exceed more frequently than the samples for Zn and Pb (Table 3.9). The converted thresholds for acute and chronic levels were exceeded in seven and ten of the sites, respectively, over the entire sampling period. The total Cu standard (pre-2015) was exceeded in thirteen sites. The thresholds for acute Pb and chronic and acute Zn were exceeded in two sites. The total Pb and Zn criteria were exceeded in eight and nine of the sites, respectively.

For the "Reference" sites, Pb, Zn and Cu criteria were exceeded in two sites. However, interestingly criteria exceedance was much less for the current sampling period (Table 3.10), in which only the standard for chronic lead was exceeded. This potentially points towards

decreasing metals levels in the wetlands due to settling, binding to organic matter, and adsorption and complexation (Reddy and DeLaune, 2008), removal of metal sources, or possible differences in sampling protocols (see Appendix J).

For the current sampling period, one site exceeded the converted dissolved metal criteria for Zn. This site was located in a developed area and received runoff from large parking lots. Only two sites exceeded the historical total metals criteria for Zn during the current sampling period. Pb samples never exceeded the current acute criteria. The chronic Pb criteria was exceeded in samples from half the sites. However, in most cases these were relatively low values that just exceeded the criteria because of low hardness values (threshold decreases with increasing hardness). Samples from two sites exceeded the total Pb criteria for the current sampling period. For Cu, samples from three sites exceeded the chronic and/or acute standard for dissolved metals for the current sampling period. Only one of the "Ag Impacted" sites exceeded the criteria during the current sampling period.

Table 3.9: Percent exceedance of surface water aquatic life standards by site for entire sampling period.

Disturbance Class	Site	# Site visits	Cu Acute	Cu Chronic	Total Cu*	Pb Acute	Total Pb*	Zn Acute	Zn Chronic	Total Zn*
Ag. Impacted	DEN	15	27%	33%	33%	0%	0%	0%	0%	0%
	NAH	18	0%	0%	6%	0%	0%	0%	0%	11%
	HOG	22	0%	0%	18%	0%	18%	0%	0%	18%
	DOT	12	0%	8%	8%	0%	0%	0%	0%	0%
Avg. Dist.	BLK	15	7%	7%	13%	7%	13%	0%	0%	7%
	HNK	12	8%	17%	42%	0%	8%	0%	0%	17%
	LKW	20	0%	0%	50%	0%	20%	0%	0%	5%
	MUN	16	13%	31%	31%	0%	19%	0%	0%	19%
	NSC	12	8%	8%	17%	0%	0%	0%	0%	0%
	RKR	5	0%	20%	40%	0%	0%	0%	0%	0%
	WMT	22	23%	23%	23%	0%	5%	9%	9%	23%
Reference	DRG	21	0%	0%	0%	0%	0%	0%	0%	0%
	PCS	13	31%	31%	31%	8%	15%	8%	8%	15%
	RDU	8	0%	13%	13%	0%	0%	0%	0%	0%
	SIK	22	0%	0%	0%	0%	0%	0%	0%	5%
	SPG	17	0%	0%	0%	0%	6%	0%	0%	0%
Total Exceedance**			7/16	10/16	13/16	2/16	8/16	2/16	2/16	9/16

*total recoverable limit used prior to 2015 in North Carolina
**number of sites where standard was exceeded in at least once sample

Table 3.10: Percent exceedance of surface water aquatic life standards by site for current sampling period (2015-2018).

Disturbance Class	Site	# Site visits	Cu Acute	Cu Chronic	Total Cu*	Pb Acute	Pb Chronic	Total Pb*	Zn Acute	Zn Chronic	Total Zn*
Ag. Impacted	DEN	5	20%	20%	20%	0%	40%	0%	0%	0%	0%
	NAH	5	0%	0%	0%	0%	0%	0%	0%	0%	0%
	HOG	6	0%	0%	0%	0%	0%	0%	0%	0%	0%
	DOT	6	0%	0%	0%	0%	0%	0%	0%	0%	0%
Avg. Dist.	BLK	3	0%	0%	0%	0%	66%	0%	0%	0%	0%
	HNK	3	0%	0%	33%	0%	0%	0%	0%	0%	0%
	LKW	5	0%	0%	20%	0%	0%	20%	0%	0%	20%
	MUN	4	0%	0%	0%	0%	0%	0%	0%	0%	0%
	NSC	6	0%	0%	0%	0%	17%	0%	0%	0%	0%
	RKR	4	0%	25%	50%	0%	0%	0%	0%	0%	0%
	WMT	6	50%	50%	50%	0%	50%	17%	17%	17%	50%
Reference	DRG	6	0%	0%	0%	0%	17%	0%	0%	0%	0%
	PCS	6	0%	0%	0%	0%	17%	0%	0%	0%	0%
	RDU	5	0%	0%	0%	0%	0%	0%	0%	0%	0%
	SIK	6	0%	0%	0%	0%	50%	0%	0%	0%	0%
	SPG	6	0%	0%	0%	0%	17%	0%	0%	0%	0%

*total recoverable limit used prior to 2015 in North Carolina

It is important to recognize that the surface water aquatic life standards were developed under pH, DOC and TSS conditions that may be very different from the wetland environment. These factors can affect the bioavailability of metals in aquatic ecosystems. Therefore, due to the unique conditions in wetlands, exceedance of these standards may not be indicative of risk to the wetland biota and should not be interpreted as a sign of contamination. The total metals criteria (pre-2015) used for comparisons in this study is no longer used and the dissolved criteria were converted to a total metals equivalent using estimated conversion factors for streams. In addition, these comparisons did not comply with the time component required for metals standards (i.e. Acute- average of two samples within 24 hours, Chronic- average of four samples over 96 hours) because of the infrequent sampling regime. These comparisons were made to establish a general

idea of the relative metals concentrations in wetland surface waters, and should only be interpreted as indication of possible risk.

Conclusion

The primary objectives of this research involved comparing wetland water quality to background levels for streams and developing thresholds based on reference wetlands that could be used to identify anthropogenic disturbance. However, the inferences drawn and confidence in the conclusions was limited due to the small sample size, infrequent sampling, the combination of different wetlands types, the large geographic range and variability among wetland ecosystems. When comparing the observed data to background levels for streams, it was apparent that background levels for wetlands should likely be higher than the background levels for streams for TN, TP and NH_4^+-N, but should likely be set lower than streams for NO_3^--N.

This research indicates that reference levels for inorganic nitrogen may be applicable across ecoregions and wetland types. Wetland and ecoregion specific reference levels should be further investigated for TN, TP, and ON. However, thresholds for TN and ON may not be informative given they did not appear to be robust indicators of disturbance. Inorganic nitrogen, particularly nitrate, is a better indicator of anthropogenic disturbance. The draft thresholds developed are not proposed as standards or criteria and are not indicative of adverse impacts to wetland ecosystems or problematic levels of nutrient enrichment. Instead, these thresholds only indicate that a concentration above this level may be indicative of some anthropogenic influences, thus providing some support for narrative criteria. It should be noted that this is only a preliminary attempt at the process of developing reference thresholds for wetlands and there is a need for future validation and improvement. However, the underlying theory that setting backgrounds at the upper limit of the reference population will preserve the biological and

chemical integrity aquatic ecosystems is widely accepted (e.g. US EPA, 2001; US EPA, 2008; USGS, 2010), and the similarity between the results and the upper limit of background (irreducible) concentrations in constructed wetlands from USEPA (2000e) was promising.

The sampling scheme implemented in this project (quarterly grab samples) was only able to provide a very general overview of the water quality parameters in natural wetlands. A focused future study would implement more intensive sampling to adequately characterize any seasonal variability and better identify outliers, and focus on a single ecoregion with adequate replication of sites of the same wetland type in different land uses. One of the major challenges associated with wetland water quality is the variability of wetland ecosystems, and the reality is that many wetlands lack surface water for substantial periods of the year. This raises significant questions regarding how water quality criteria could be implemented and under what conditions the criteria would apply (i.e. seasonal, minimum water depth, etc.).

While EPA recommends the use of ecoregion specific nutrient criteria for wetlands and streams, research suggests that for streams the use of ecoregion grouping does not always appropriately account for the natural variation that exists among nutrient concentrations (Herlihy and Sifneos, 2008; Robertson et al., 2006). The grouping of sites within ecoregions for nutrient criteria will inevitably lead to the overprotection of some sites and the under protection of others (Olson and Hawkins, 2013). As a result, sub-ecoregion, or ecosystem specific criteria may be more appropriate as a starting point for wetlands in select cases. This could ensure the protection of wetland ecosystems that are at risk of degradation due to nutrient inputs. For example, TP criteria has been adopted for the Everglades in Florida. The average concentration limit was set at 10 μg/L and the maximum concentration at 15 μg/L. These values were based on negative impacts on sensitive fauna in the Everglades (US EPA, 2008).

While nutrient criteria for wetland ecosystems may help identify anthropogenic disturbance, the best way to protect wetland ecosystems is by protecting and preserving adjacent uplands. The likelihood of encountering elevated nutrient levels can generally be determined by examining surrounding land use. In addition, the presence of slightly elevated concentrations of nutrients may be a good indication that these wetlands are providing an ecosystem service of assimilating and removing nutrients. It should be considered an advantage to have wetlands in these landscape positions to buffer downstream waters.

This project demonstrated a method to set thresholds for reference conditions that could be generally used to support narrative criteria, but was not intended to infer contamination because these thresholds were not developed based on concentrations that would result in adverse impacts to the wetland ecosystem. Overall, wetland specific water quality criteria are probably not needed for all wetlands, particularly for dryer or intermittently flooded wetland systems. Wetland water quality criteria development should likely be approached on a case-by-case basis depending on the potential benefits to the specific wetland ecosystem. For example, if wetlands ecosystems are threatened by specific pollutants in an ecoregion, then criteria should be developed based on observed adverse impacts on ecosystem condition (e.g. TP criteria in the Florida Everglades). This type of targeted implementation could likely ensure the greatest benefits. The variability in wetland ecosystems would make large-scale development and implementation of numeric criteria very difficult. The characterization of background conditions to support narrative wetland criteria would likely be the most realistic step to advance wetland science and support wetland protections.

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Chapter 4: Trace metal levels and organic matter content in North Carolina wetland soils

Abstract

Wetlands are often located in landscape positions where they receive and attenuate runoff, which can contain toxic trace metals and other pollutants from anthropogenic sources. Wetland soils provide a medium for the long-term storage of some pollutants. Wetlands soils have the ability to capture and store trace metals at a greater capacity than upland soils due to the relatively high surface area of fine sediments and the elevated binding capacity of soil organic matter humic functional groups. The objectives of this chapter were to: (1) compare the organic matter content of the wetlands and the relationship between soil organic matter and metals, (2) compare the extractable metal concentrations in wetlands to the adjacent upland, (3) compare the observed Cu and Zn concentrations to plant toxicity and ecological risk assessment thresholds, (4) compare the samples from a larger set of wetlands across the state to plant toxicity levels, and (5) determine if a relationship exists between Mehlich 3 and total metals that can be used to accurately predict total metals from Mehlich 3 concentrations for wetland soils. Sampling was conducted at 16 natural wetlands across the Piedmont and Coastal Plain of North Carolina. Samples were collected from the active root zone (0-30 cm) along transects from the upland to the wetland at each site. Samples were analyzed for Mehlich 3 extractable Cu and Zn, soil organic matter, and then a subset of the samples was analyzed for total recoverable Cu and Zn. No significant differences were observed between the wetland, wetland edge and upland for Cu or Zn. Zn was significantly higher in the surface than the subsurface samples at all locations in both the Piedmont and Coastal plain. Cu concentrations were significantly higher in the Piedmont wetlands and at the wetland edge locations. No significant difference was observed for Zn between regions. The site means for Mehlich 3 Cu ranged from 0.1 to 5.9 mg/kg and Zn

ranged from 0.2 to 13.3 mg/kg. Overall, the site means were much lower than previously published average values for North Carolina soils. Results at only one site exceeded the plant warning levels for Zn. Neither extractable Cu nor Zn concentrations ever exceeded the critical levels for plant toxicity. Cu levels were not observed above the plant toxicity warning levels. The lowest Ecological Soil Screening Levels (Eco-SSLs) were exceeded in about half the sites, but higher more relevant Eco-SSL thresholds were only exceeded in one site. The Eco-SSLs are intended to be conservative, and this was apparent in this study. The mean ratios of Mehlich 3 extractable to total metals were lower than previous studies and averaged about 0.15 for both Cu and Zn. While the relationships between Mehlich 3 and total Cu and Zn were significantly correlated, a predictive relationship that yielded accurate estimates did not exist. Overall, some sites showed elevated levels of total metals, but this did not always translate to high levels of Mehlich 3 metals. Very low mobile Zn and Cu levels in the soils of most of these wetlands likely indicated that the metal levels in a vast majority of wetlands in the state do not pose a risk to the biota unless there is a direct anthropogenic source of metal contamination.

Introduction

Wetlands serve many important functions in the environment that include buffering downstream waters by trapping and removing pollutants. This is largely a result of their landscape position, which allows for the retention of runoff, and the unique physiochemical conditions and biogeochemical processes that occur in wetland sediments. For the pollutants that are trapped in wetlands, soil is the medium for long-term storage. Wetland soils have a higher affinity to bind metal cations than upland soils because of the fine texture of the mineral particles and higher organic matter content (Gambrell, 1994). The binding ability of wetland soils is the result of the strong negative charge and large surface area of carboxylic, phenolic, alcoholic, and amino functional groups that make up soil organic matter (SOM) (Reddy and DeLaune, 2008). In the anaerobic environment, metals bound to humic substances are effectively immobilized; however, they can be released if the soil is oxidized for extended periods. These wetland soil processes are often exploited to remediate heavy metal pollution in the environment. For example, constructed wetlands have been used to remove heavy metals from acid mine drainage (e.g. Sheoran and Sheoran, 2006), stormwater (e.g. Walker and Hurl, 2002), municipal wastewater (e.g. Chen et al., 2009) and industrial wastewater (e.g. Khan et al., 2009). However, outside of Carolina Bays (e.g. Barton et al., 2008; Ewing et al., 2012), few studies have documented the levels of metals in natural wetlands in North Carolina, and more generally, there is limited information on the background concentration of metals in natural wetland soils. Soils generally contain very low background concentrations of toxic metals derived from natural sources (e.g. mineral weathering), and elevated levels are typically only observed due to anthropogenic disturbance (Reddy and DeLaune, 2008). Therefore, absent a significant source of metals in the wetland's catchment, contamination should not be expected. For example,

Vymazal et al. (2010) reported higher metal levels in impacted wetlands, although they also observed a large range of values for trace metal concentration in unpolluted natural wetlands. Azous and Horner (2000) observed a trend of increased heavy metals in wetlands in urbanized areas compared to undisturbed wetlands, and Ewing et al. (2012) reported higher metal levels in wetlands impacted by agriculture.

The sources of Cu and Zn in the environment are primarily from industrial processes and transportation. Zn is used in galvanized steel, paints, rubbers, and other metal production. Zn enters the environment primarily though the production and use of the previously mentioned materials, and through the burning of coal and the field application of fertilizers and sludge (ATSDR, 2005). Cu is used in many alloys, and electrical wire and plumbing products. It is also important for water treatment, agriculture fertilizers and pesticides, and industrial preservatives. Cu enters the environment through the production of metals, wastewater discharges, fertilizers, and the burning of fossil fuels (ATSDR, 2004). The primary exposure of heavy metals to wetlands is from contaminated floodwaters, wastewater discharge, and stormwater runoff. Cu and Zn concentrations in stormwater can be an order of magnitude or more higher than natural, pristine waters (Maestre and Pitt, 2005; Reddy and DeLaune, 2008). Heavy metals in stormwater generally originate from exhaust, tire abrasion, brake pads wear and engine oils and fuel leaks, and metal corrosion (Wang et al., 2009). Agricultural drainage can also export residual metals from livestock waste, fertilizer, and pesticides (e.g. Savic et al., 2015).

While wetlands can be effective sinks of metals, elevated levels can pose risks to many aquatic macroinvertebrates (Malaj et al., 2012) and amphibians (Vitt et al., 1990) and can cause shifts in species composition to more tolerant species as well as physiological reactions (Malaj et al., 2012). Bioaccumulation through the food chain can also be a potential concern when

wetlands become contaminated (Goodyear and McNeill, 1999). In addition, high levels of toxic metals can inhibit microbial and enzymatic processes. This can affect or inhibit many of the significant processes in wetlands, such as mineralization, nitrification, and denitrification (Reddy and DeLaune, 2008).

The organic matter content of wetland soils in the coastal plain of North Carolina have been documented in several studies (e.g. Bridgham and Richardson, 1993; Bruland and Richardson, 2006); however, few studies have reported SOM values for wetlands in the Piedmont region. In addition, while Mehlich 3 is a common soil test and is recommended by US EPA (2008) for use in wetlands, there does not seem to be any published information on the relationship between Mehlich 3 extractable metals and total recoverable metals in wetland soils. While Mehlich 3 gives an indication of the amount of metals available for plant uptake, it does not quantify the total metals stored in the wetland soil.

The primary objectives of this chapter were:

1. Assess the differences in SOM among wetlands and determine if SOM is significantly correlated with Mehlich 3 Cu and Zn.
2. Determine if significant differences exist between the upland and wetland for Cu and Zn, if there are differences between the Coastal Plain and Piedmont, and if these relationships change with sample depth.
3. Compare the extractable Cu and Zn concentrations in wetlands to background concentrations and plant toxicity levels.
4. Compare the total Cu and Zn concentrations to the US EPA Ecological Soil Screening Levels (Eco-SSLs).

5. Determine if a relationship exists between total and Mehlich 3 metals that can be used to estimate total metal concentrations if only Mehlich 3 results were available.

Materials and Methods

Study Sites

Sixteen natural wetland sites located in the Coastal Plain and Piedmont physiographic regions of North Carolina were sampled for this project (see Figure 4.1). These sites were selected by the North Carolina Department of Environmental Quality (NC DEQ) Division of Water Resources (DWR) as part of a larger assessment of the condition of natural wetlands in North Carolina dating back to 2005. The sites were made up of three different wetland types: riverine, headwater, and basin. The sites were located in a wide range of surrounding land uses and displayed varying degrees of anthropogenic disturbance, including logging, road construction, ditching, and upstream urbanization. There was a wide range of underlying soil series and textures (see Table 4.1). The wetlands ranged in size from less than a hectare for some of the depression and headwater wetlands to hundreds of hectares for large riverine wetlands in the Coastal Plain. The evaluation areas were generally less than five hectares. For some of the smaller wetlands, the county soil surveys did not identify the sites as possessing hydric soils.

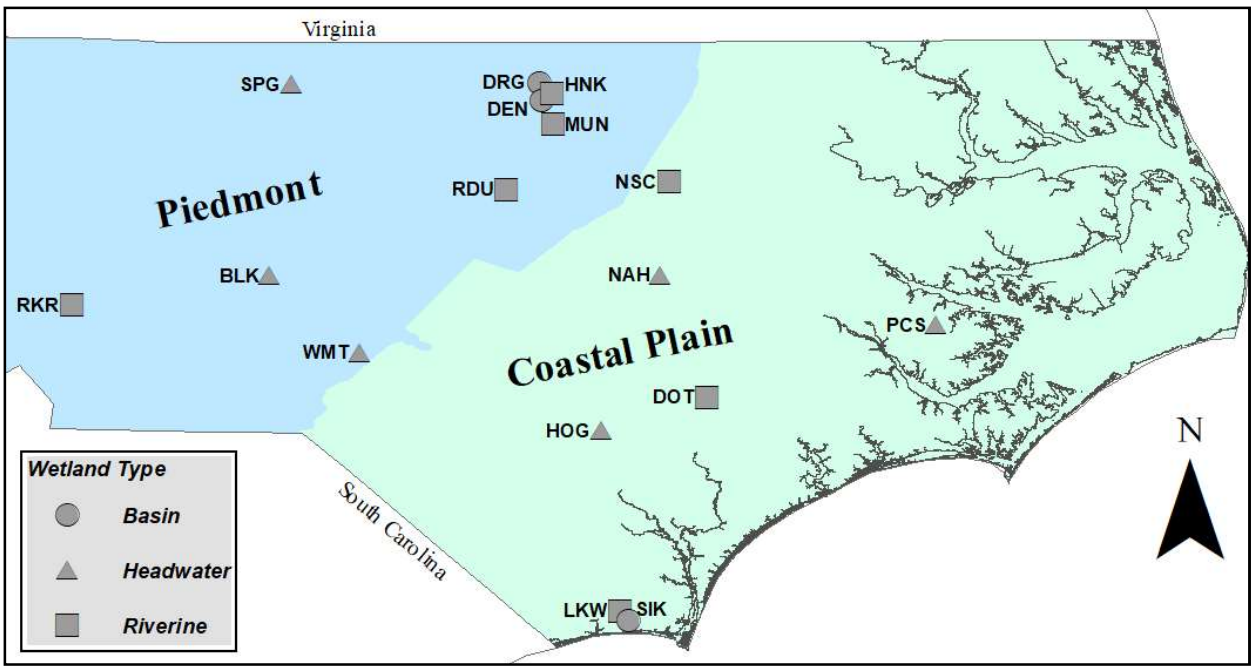

Figure 4.1: Site map with wetland type and site identifier code.

Table 4.1: Wetland site and soil series, locations, and primary anthropogenic disturbances.

Site Code	County	Region	Wetland Type	Soil Series	Anthropogenic Disturbance
BLK	Montgomery	Piedmont	Headwater	Herndon	Pasture; highway
SPG	Rockingham	Piedmont	Headwater	Codorus	Minimal
WMT	Moore	Piedmont	Headwater	Vauclose	Urban
RDU	Wake	Piedmont	Riverine	Chewacla	Upstream Development
RKR	Cabarrus	Piedmont	Riverine	Wehadkee	Upstream Development
HNK	Granville	Piedmont	Riverine	Chewacla	Upstream Development; highway
MUN	Granville	Piedmont	Riverine	Chewacla	Pasture; row crops
DEN	Granville	Piedmont	Basin	Wehadkee	Row crops
DRG	Granville	Piedmont	Basin	Wehadkee	Minimal
PCS	Beaufort	Coastal Plain	Headwater	Dorovan	Minimal
HOG	Sampson	Coastal Plain	Headwater	Coxville	Pasture; spray field
NAH	Wayne	Coastal Plain	Headwater	Torhunta	Row crops; highway
DOT	Duplin	Coastal Plain	Riverine	Bibb	Agriculture, drainage, development
NSC	Nash	Coastal Plain	Riverine	Wehadkee	Upstream agriculture + development
LKW	Brunswick	Coastal Plain	Riverine	Dorovan	Upstream Development; highway
SIK	Brunswick	Coastal Plain	Basin	Murville	Minimal

Sampling Design and Methods

At each site three to four parallel transects were established that traversed the hydrologic and topographic gradient from the upland to the wetland. Along each transect, one sample was collected at both the upland (U) and the wetland edge (E), and one to three samples were collected in the wetlands (W). Combined, fifteen to twenty locations were sampled at each site. The identification of the U, E and W locations were based on elevation changes, wetland indicators, and changes in vegetative composition. Samples in the wetland were taken at average elevations across the microtopographic gradient (not in the bottom of hollows or on hummocks). The transect layout and spacing, and the number of samples varied slightly based on wetland type and size (Figure 4.2). Transects were established prior to sampling in ArcMAP 10.4 (Esri, Redlands, CA) and adjusted based on conditions in the field. All sampling points were recorded using a handheld GPS device (Garmin, Olathe, KS).

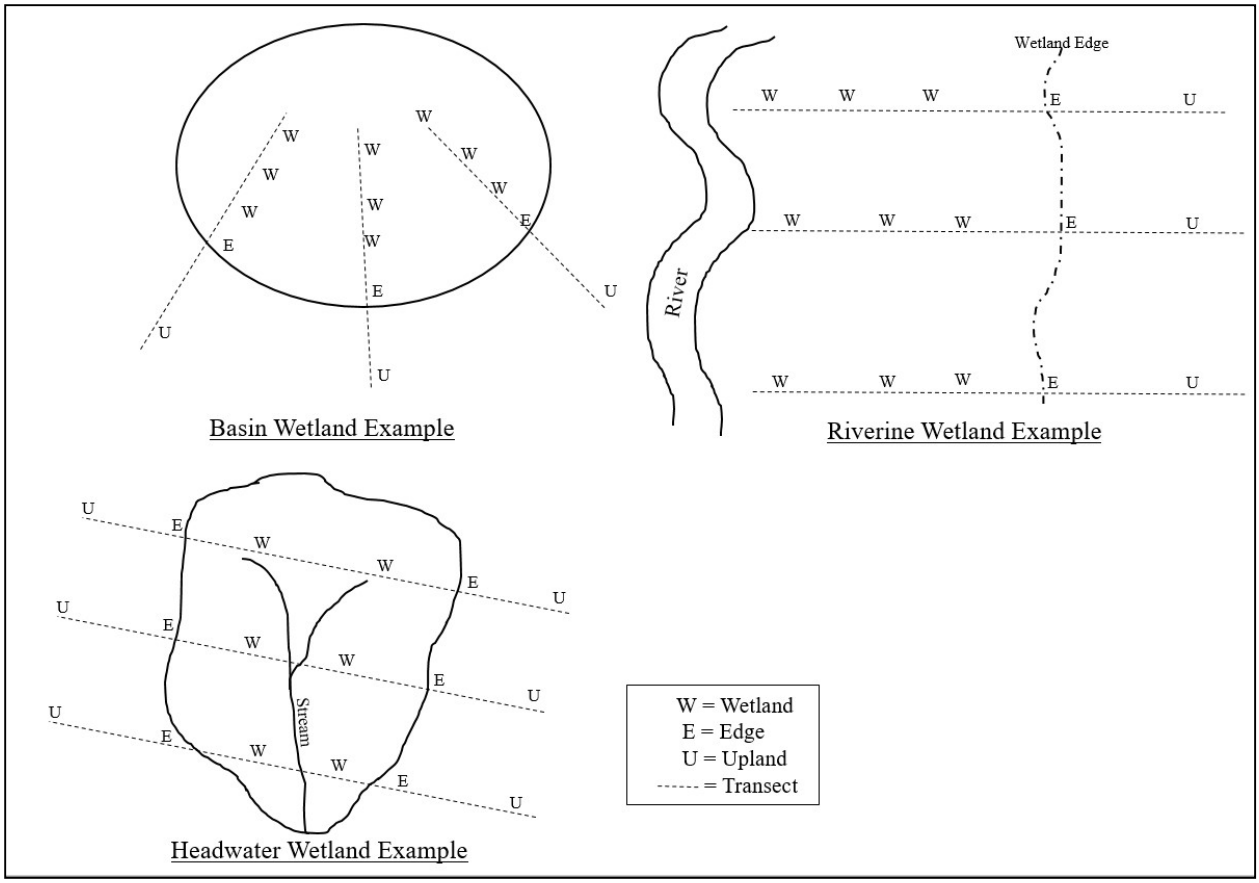

Figure 4.2: Sampling design schematics.

Samples were collected from the top 30-cm of the soil profile from May to July of 2017. A majority of the samples were collected using a 5-cm diameter stainless steel coring device (Eijkelkamp, Netherlands). When conditions prevented the use of the corer (e.g. inundated conditions or sandy soils), a stainless-steel auger was used. The samples were split into 0-10 cm and 10-30 cm increments in the field as recommended by US EPA (2008) for routine characterization of the typical root zone. The samples were placed in sealed containers and transported back to NC State University. Samples were then dried, ground, and sieved using a standard 2-mm mesh. A portion of the samples was sent to the North Carolina Department of Agriculture and Consumer Services (NCDA&CS) Soil Lab for routine analysis for TP, Cu, Zn, Na, K, Ca, Mg, base saturation (BS), pH, humic matter, and cation exchange capacity (CEC). Cu and Zn were analyzed using the Mehlich 3 extraction method (Mehlich, 1984).

The remainder of the sample was analyzed for SOM using the loss on ignition (LOI) method (Storer, 1984). For this analysis, approximately five grams of the sample was dried at 105 °C for 24 hours to remove any moisture and achieve a constant mass. The mass was recorded using Mettler Toledo AE200 Analytical Balance (Mettler Toledo, Columbus, OH) and the sample was then placed in a Fisher Scientific Isotemp Programmable Muffle Furnace (Fisher Scientific, Waltham, MA) for four hours at 450 °C, and then weighed again. The soil organic matter content (%) was assumed to be equal to the difference in mass relative to the initial mass:

$$SOM(\%) = \frac{(m_{105} - m_{450})}{m_{105}} \qquad \text{Equation 4.1}$$

where m_{105} is the mass after oven drying for 24 hours at 105 °C and m_{450} is the mass after four hours in the muffle surface at 450 °C.

In order to compare the results among the different sites in a statistical model, the values were converted to a volume basis because of the difference in bulk density between organic and mineral soils (Ewing et al., 2012). The conversion to field volume was completed by multiplying the mass-based results by the bulk density. Samples for bulk density analysis were collected from the top 30-cm of the soil using a stainless-steel coring device with plastic sleeve inserts. Bulk density samples were collected concurrently with the soil samples and then additional samples were collected in December 2017 because too few samples were originally collected at some sites. The cores were transported back to the lab in the plastic sleeves and split into 0-10 cm and 10-30 cm increments. The volume of each section was recorded, and the samples were oven dried for 24 hours at 105 °C and the mass was recorded. Bulk density was calculated using the following equation:

$$Bulk\ density = \frac{m_{105}}{vol} \qquad \text{Equation 4.2}$$

where m_{105} is the mass after oven drying for 24 hours at 105 °C and *vol* is the field volume of the sample section. All results are presented on a dry weight basis.

Total Metals Analysis

The results from the NCDA&CS lab were obtained using the Mehlich 3 extraction method, which in an approximation of plant available metals (Hardy et al., 2008). Because most non-agriculture related soil contamination threshold criteria are based on total metals concentration, a subset of the wetland samples from each site were analyzed for total recoverable metals at the Environmental Analysis Lab in the Department of Biological and Agricultural Engineering at NC State University. Four to six samples from each site were analyzed for total Cu and Zn. These samples were selected from both sample depths and represented a range of the Mehlich 3 extractable metal results. Only samples from the wetland locations were analyzed due to budget constraints. The samples were analyzed for total recoverable metals using US EPA Standard Method 220.2 (US EPA, 1994).

Statistical Analysis

Difference in depth, ecoregion, and sample location were evaluated for Cu and Zn using a generalized linear mixed model (GLMM) with a log normal distribution. The model was fit using PROC GLIMMIX Type III procedure in SAS 9.4 (Cary, NC). The PROC GLIMMIX procedure can accommodate highly variable, unbalanced, correlated data that exhibits non-Gaussian distributions. Region, sample location, and depth were set as fixed effects in the model, and site and transect were treated as nested random effects. Comparisons were made using an LS-Means approach with a Tukey-Kramer adjustment for multiple comparisons. An alpha level of 0.05 was used to establish statistical significance. The code and model output can be found in Appendix O.

The humic substances in SOM have a high capacity to immobilize metals due to the reactive functional groups (Reddy and DeLaune, 2008). High correlation coefficients between total metals and SOM have been observed in previous studies (Yuan et al., 2015). The relationship between metals and SOM was examined at each site by calculating the Spearman's Rho between SOM content and the extractable Cu and Zn concentrations. Spearman's Rho measures the correlation between two variables, is resistant to outliers and does not rely on any distributional assumptions (Helsel and Hirsch, 2002).

Linear regression was used to evaluate if there was a consistent relationship between total and Mehlich 3 extractable metals that could be used to estimate the total metal concentration from the Mehlich 3 extractable results for Cu and Zn. Individual regressions were completed for each ecoregion. The number of samples per site did not allow for site-specific regressions. A stepwise multiple regression approach was used to see if the estimates of total metals could be improved by adding predictors to the model. Additional predictors included SOM, humic matter, pH, cation exchange capacity, and lab bulk density. Variable selection was based on minimizing Akaike information criterion (AIC). Regression fitting was completed in R version 3.4 (R Core Team, 2017).

Contamination Thresholds

To assess Cu and Zn contamination and the risk to wetland vegetation the results were compared to guidelines for agricultural crops issued by the NCDA&CS (Hardy et al., 2003). The critical toxicity levels for plants are 120 mg/kg and 60 mg/kg for Mehlich 3 Zn and Cu, respectively (Hardy et al., 2003). Mehlich 3 Cu levels typically are only measured above 20 mg/kg when anthropogenic sources are present (Tucker, 1999). Others suggest that 50 mg/kg and 20 mg/kg for Mehlich 3 Zn and Cu, respectively, should be considered warning levels for

agricultural applications (RSTL, 2017). The only published thresholds available for Mehlich 3 extractable metals are related to agricultural application because the Mehlich 3 extraction method is intended to approximate plant available metals (Hardy et al., 2008). Hardy et al. (2008) reported mean background concentration for North Carolina soils of 9.2 mg/kg and 27.2 mg/kg for Mehlich 3 Cu and Zn, respectively. There are many in-depth methods to develop threshold levels that may be indicative of some impacts; however, some researchers recommend simply using the 90th or 95th percentile of the site concentrations (Reimann and de Caritat, 2017). The 90th percentile was used to propose draft background thresholds for these sites.

Thresholds that have been developed to evaluate soil contamination and risks to biota were intended to be compared to total metal concentrations, not plant available levels. For example, the US Environmental Protection Agency (US EPA) has defined Ecological Soil Screening Levels (Eco-SSLs) or concentration of metals in soils that may be indicative of potential risks to the ecological receptors of plants, soil invertebrates, birds, and mammals (US EPA, 2005). Eco-SSL are used in the initial evaluation of contamination at superfund sites (US EPA, 2005). While Eco-SSLs were developed and intended for application in upland soils, US EPA (2005) indicates that they may be useful for initial screening of wetland soils, particularly for soil invertebrate and plant Eco-SSLs. The Eco-SSLs for Cu and Zn are shown in Table 4.2 (US EPA, 2007b, 2007a). US EPA (US EPA, 2007b, 2007a) reported median background concentrations of 50 mg/kg dry weight and 20 mg/kg dry weight for total Zn and total Cu, respectively, for the eastern United States, although there was a wide range of local variability for both parameters. US EPA (2007c) reported background concentrations of 56 mg/kg and 34 mg/kg for total Zn and total Cu, respectively for North Carolina soils.

Table 4.2: EPA Eco-SSLs for Cu and Zn (total metals (mg/kg)).

Constituent	Plants	Soil Invertebrates	Avian Wildlife	Mammalian Wildlife
Cu	70	80	28	49
Zn	160	120	46	79

While some studies have used conversion factors to translate from extractable to total metals, the ratio of extractable to total varies across different soil type and conditions. McBride et al. (2009) found that the extractable fraction of Zn and Cu were both about 0.30 of the total metals in contaminated upland soils. However, Sims et al. (1991) reported that the correlation coefficients between Mehlich 3 extracted Cu and Zn and the EPA Method 3050 (total metals) were only 0.56 and 0.43, respectively for agricultural soils in Delaware. In addition, MacDonald et al. (2000) developed Sediment Quality Guidelines (SQGs) for freshwater systems which defined Threshold Effects Concentrations (TEC) and Probable Effects Concentrations (PEC). The TEC is a level below which adverse impacts are unlikely to be observed and the PEC is a level above which adverse impacts are likely to be observed. The thresholds for Cu and Zn are shown in Table 4.3.

Table 4.3: TEC and PEC for Cu and Zn from MacDonald et al. (2000) (total metals (mg/kg)).

Constituent	TEC (mg/kg)	PEC (mg/kg)
Cu	31.6	149
Zn	121	459

Evaluation of Sites across North Carolina

From 2006 to 2013, the NC DEQ DWR collected samples from 88 wetland sites across the Coastal Plain and Piedmont of North Carolina throughout a series of different US EPA funded monitoring projects (Figure 4.3). These samples were analyzed at the NCDA&CS soils lab using the Mehlich 3 extraction method. However, the number of samples, sample locations, and depths varied by the project, and sampling documentation was not always available. The results for the surface layer (approximately 0 to 30-cm) were compared to the thresholds for

plant toxicity. Sites that indicated threshold exceedance were investigated further based on surrounding land use.

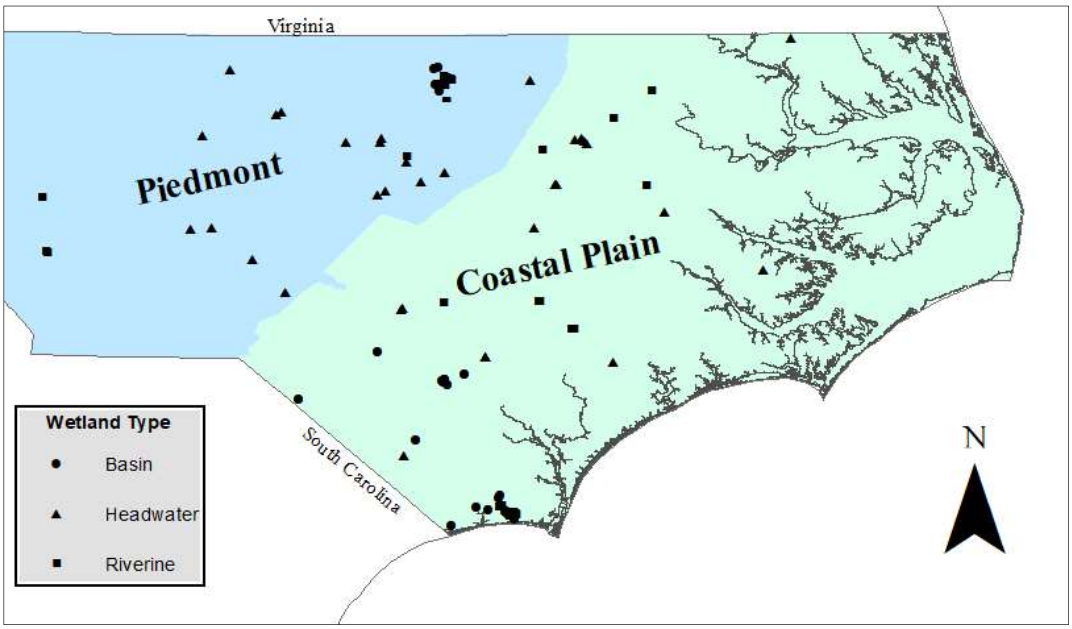

Figure 4.3: 88 sites monitored for soil chemistry by NC DEQ from 2005 to 2013.

Results and Discussion

SOM Results

Site means in the Coastal Plain for wetland samples ranged from 12 to 44% for the top 10 cm with an average of 23%. For the 10 -30 cm depth, the site means ranged from 1.6 to 22% with an average of 9.1%. These results are indicative of the variability in the wetlands that were monitored (see Figure 4.4). The conditions ranged from thick organic horizons to only a thin layer of organic matter accumulation at the surface underlain by sand in a Carolina Bay. However, the maximum SOM values observed in this study are well below the values reported in some studies in Coastal Plain wetlands. For example, several studies have reported SOM content in excess of 80% (e.g. Bridgham and Richardson, 1993; Bruland and Richardson, 2006). However, this was largely the result of site selection as none of the sites monitored for this project were organic soil flats or pocosins, which typically have very high content (Bridgham

and Richardson, 1993). Overall, the values largely fell in the lower range of a previous study of Coastal Plain wetlands completed by Bruland and Richardson (2006).

SOM content was lower in the Piedmont wetland soils that tend to have shorter durations of inundation and/or saturation due to the steeper landscape gradient in some wetland types. Site means ranged from 5.8 to 12.4% for the top 10-cm with an average of 8.7%. In the 10-30 cm depth, site means ranged from 1.4 to 8.8% with an average of 4.8%. These values were similar to previous studies from Georgia for mineral soil wetlands (Craft and Casey, 2000). While wetlands are typically associated with substantial organic matter accumulation, many of these sites had very low values they may be more typical of sites that are only inundated for short periods. However, long periods of saturated conditions did not always explain SOM content either. For example, some of the wetlands that were saturated for the longest periods in the Piedmont (MUN site-riverine and DRG site- basin wetland) only had SOM content near 10%. This was probably to be the result of low primary productivity in these sites, which appeared to be limited by large open water areas. Some of these site mean values were more similar to the typical range for SOM in forest soils of 1-5% (Osman, 2013). Therefore, it is important to recognize there is a wide range of SOM content across different wetland types and different regions. SOM accumulation is the result of hydrology, productivity, climate, nutrient levels and other factors that affect decomposition rates (Reddy and DeLaune, 2008). Overall, SOM accumulation occurs when productivity outpaces decomposition over the long term. As expected, SOM declined with depth across all sites. Because of the wide variability in geography, wetland type, soil series, landscape position, and small sample size, comparisons were not made based on wetland type, as they likely would not have provided meaningful information.

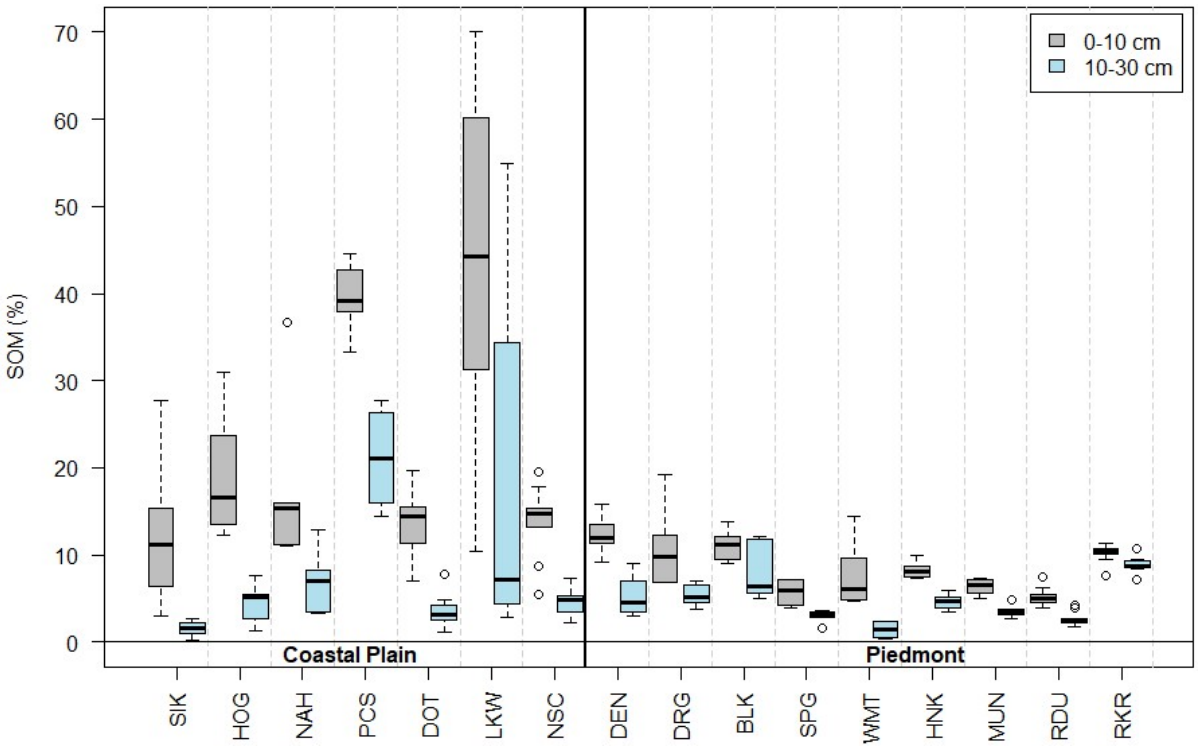

Figure 4.4: Wetland SOM results by depth for all sites.

Relationship Between SOM and Metals Content

Because SOM functional groups form strong complexes with metal cations (Reddy and DeLaune, 2008) it was hypothesized that SOM and Mehlich 3 Cu and Zn may be significantly correlated within a given site (Table 4.4). Most of the sites exhibited significant Cu and Zn correlations with SOM (11/16 sites for Cu and 14/16 sites for Zn). About half the sites showed significant correlations between humic matter and Zn and Cu. The results can be found in the Appendix N. The sites that did not show a significant correlation between SOM and Cu and Cu tended to have very low Zn and Cu concentrations, and did not display correlation between Zn and Cu.

Table 4.4: Spearman's correlation for SOM and metals. Bold values represent statistically significant correlations (0.05 level).

Site	SOM-Zn	SOM-Cu	Zn-Cu
BLK	**0.86**	-0.2	-0.02
DRG	0.36	0.31	0.2
DEN	**0.99**	**0.79**	**0.77**
DOT	**0.93**	**0.87**	**0.96**
HNK	**0.66**	**0.76**	**0.77**
HOG	**0.76**	**0.75**	**0.83**
LKW	**0.82**	**0.81**	**0.89**
MUN	**0.91**	0.39	0.29
NAH	**0.8**	**0.85**	**0.99**
NSC	**0.58**	**0.52**	**0.57**
PCS	0.48	**0.64**	**0.95**
RDU	**0.75**	**0.71**	**0.93**
RKR	**0.79**	**0.65**	**0.73**
SIK	**0.88**	**0.86**	**0.82**
SPG	**0.8**	0.11	0.09
WMT	**0.82**	0.25	0.41

Differences in Region, Sample Location and Depth for Cu and Zn

LS-Means results from the PROC GLIMMIX statistical model were low overall with values for Zn mostly below 3 μg/cm^3 and Cu below 2 μg/cm^3 (Figure 4.5). Results were presented on a volume basis because of differences in bulk density among sites. For Cu, there were no significant differences for sample depth at any location, while Zn values were significantly different in the top 10-cm at all sample locations. For Zn, there was no significant difference between the Coastal Plain and Piedmont ecoregions, while Cu results showed a significant difference between ecoregions for samples collected in the wetland and at the wetland edge. There was no significant difference between sample locations (e.g. upland vs wetland or wetland vs edge) for either Cu or Zn in either region. A summary of the statistical differences can be found in Appendix N.

Because of the very low values observed, statistical difference reported in this case are not particularly meaningful as there were no elevated levels that may indicate contamination. In addition, the variability in geography, soil type, and land use presented additional variability that could not be accounted for in the statistical model due to the small sample size and lack of replication. The result of no significant difference between the wetlands and uplands for Cu and Zn should not be interpreted as no significant difference in the amount of metals stored in the wetlands, only that there was no significant difference in the Mehlich 3 extractable (plant available) metal concentrations during the time of sampling. Ewing et al. (2012) reported Mehlich 3 Cu and Zn levels for Carolina Bays in North Carolina, which were similar to the very low values measured in the Coastal Plain for this study. They reported LS-Means for reference sites of 0.1-0.2 μg/cm^3 for Cu and 0.8-3.4 μg/cm^3 for Zn. Concentrations for impacted bays ranged from 0.6 to 1.0 μg/cm^3 for Cu and 5-12 μg/cm^3 for Zn.

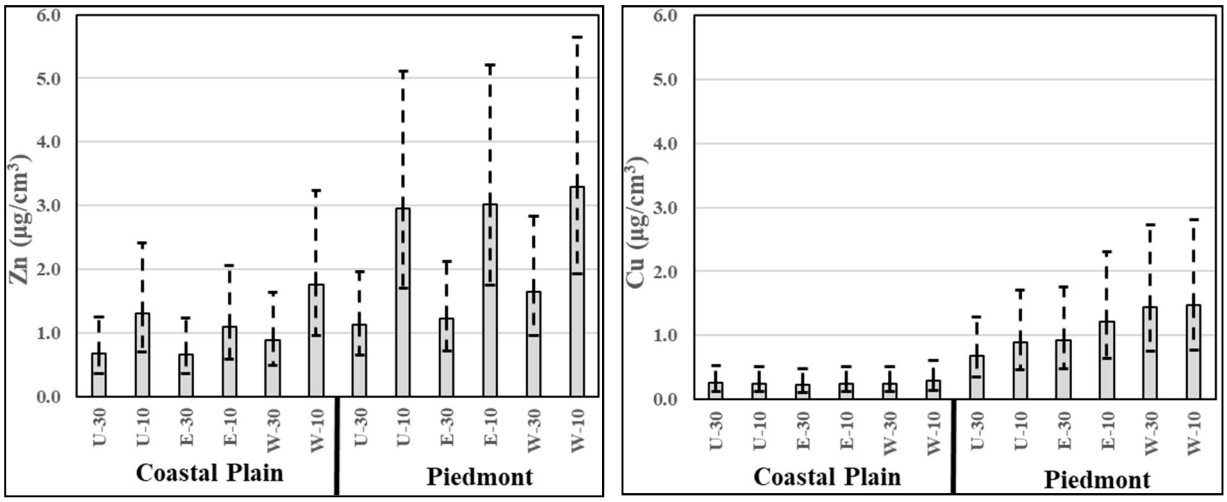

Figure 4.5: LS-Means and 95% confidence intervals for Cu and Zn by region, sample location and depth from the PROC Glimmix Model. Values were converted to volume basis for comparisons because of differences in bulk density among sites and locations (Note: U= upland, E= edge, W= wetland).

Contamination Thresholds

Plant Toxicity

The NCDA&CS critical toxicity values for plants of 60 mg/kg for Mehlich 3 Cu was never exceeded in the samples. In fact, the maximum value observed for Cu (7.8 mg/kg) was well below the warning level (20 mg/kg) from RSTL (2017) and even below the mean background level for North Carolina soils of 9.2 mg/kg (Hardy et al., 2008) (see Figure 4.6). The site means for Mehlich 3 Cu ranged from 0.2 to 5.9 mg/kg with an overall mean of 1.8 mg/kg for the top 10-cm and from 0.1 to 5.0 mg/kg with an overall mean of 1.1 mg/kg for the 10 to 30 –cm depth. The sites with the highest values (although still below the state average) were riverine wetland sites in the Piedmont with highly developed watersheds.

For Zn, the critical toxicity level of 120 mg/kg from NCDA&CS was not exceeded for any of the samples (see Figure 4.7). Only two samples from one wetland site (both from top 10-cm: 51 and 61 mg/kg) exceeded the warning level (50 mg/kg) from RSTL (2017). At this site (HNK - riverine), the upland and edge samples were higher than the wetland samples. The site mean was 13.1 mg/kg Zn. This particular bottomland area appears to have been a debris dumping area in the past, there was a sewer line that ran adjacent to the site, and highway runoff also entered this site. Therefore, all these sources could contribute to the locally elevated levels. Maximum values from only two additional sites exceeded the average background levels from Hardy et al. (2008), although the mean concentrations were well below the background levels. These samples were from sites in an agricultural setting (NAH - headwater) and an urban area (WMT - headwater). Overall, the site means ranged from 0.7 to 13.1 mg/kg with an average of 5.6 mg/kg for the top 10-cm and from 0.2 to 5.6 mg/kg with an average of 1.9 mg/kg for the 10 to 30 -cm depth. As expected, metals results appeared to be influenced by land use as was

observed in previous studies (Azous and Horner, 2000; Ewing et al., 2012). The sites with the highest mean values (although well below any level of contamination) were located in land uses such as agriculture and urban/developed- two primary sources of metals.

Overall, Mehlich 3 values were very low but similar to previous studies. Barton et al. (2008) collected samples from 22 restored and reference Carolina Bay wetlands in South Carolina, and analyzed the samples using the Mehlich 1 extraction method. Cu concentrations ranged from 0.1 to 2.2 mg/kg and Zn concentrations ranged from 0.2 to 6.6 mg/kg. These results are likely comparable to this study given Sims (1989) found coefficients of correlation near 0.90 between Mehlich 1 and 3 extractions for Cu and Zn. Results for Cu and Zn were almost universally higher in the surface layer (0-10 cm) than the subsurface (10-30 cm). This was likely partially due to the difference in bulk density, but could also be the result of the higher SOM content near the surface that has a greater affinity for binding metals (Gambrell, 1994).

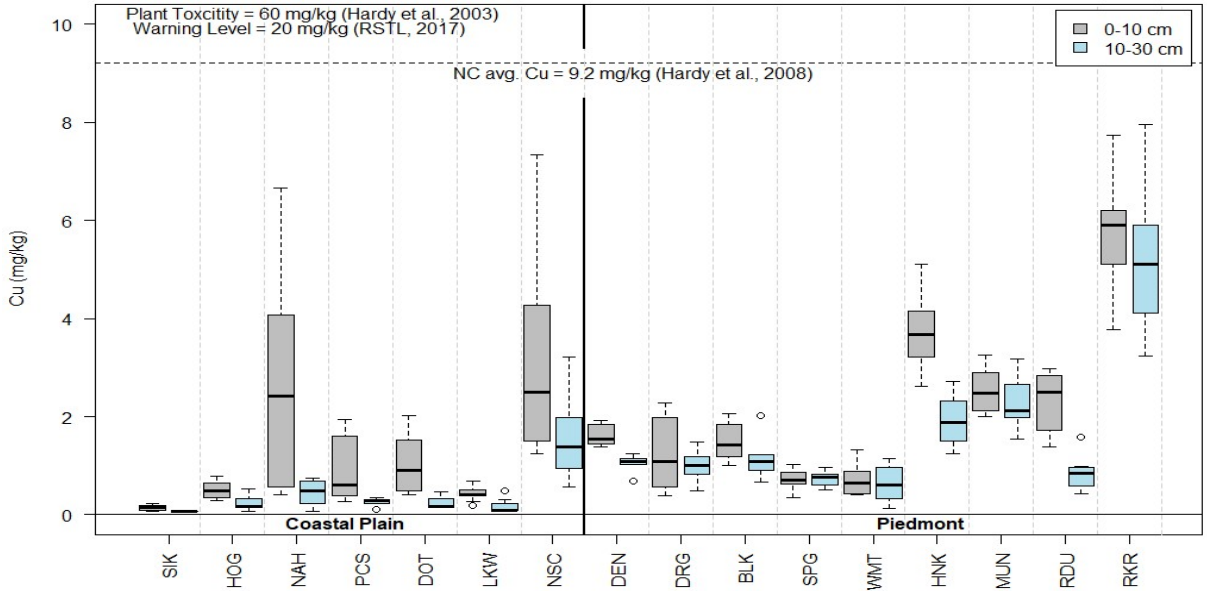

Figure 4.6: Raw data for wetland samples compared to the mean values for North Carolina soils and toxicity thresholds for Mehlich 3 Cu.

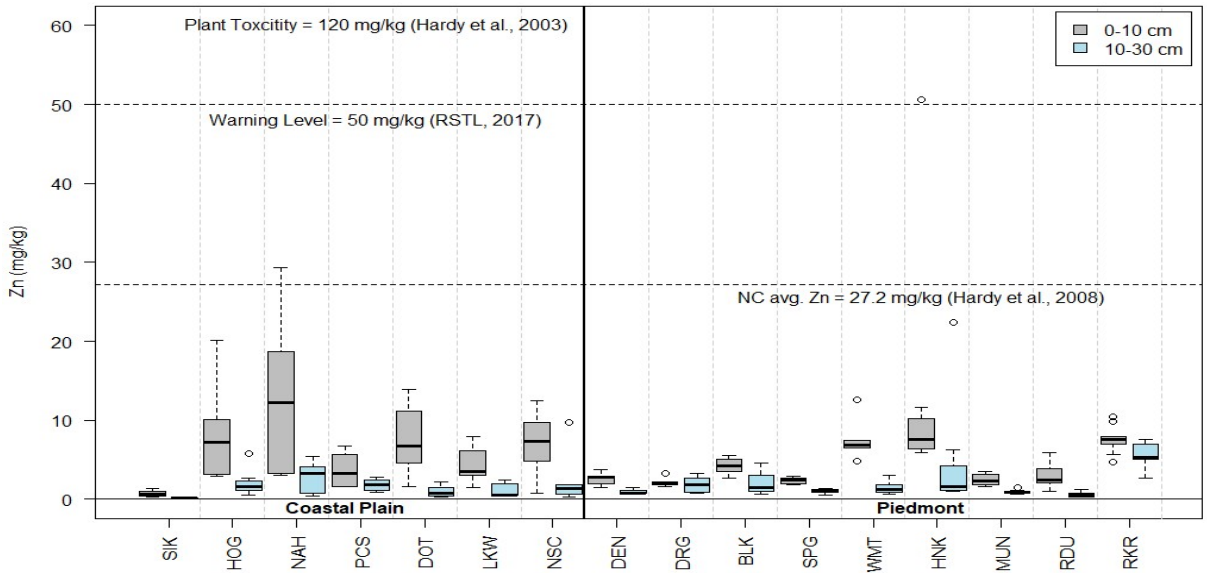

Figure 4.7: Raw data for wetland samples compared to the mean values for North Carolina soils and toxicity thresholds for Mehlich 3 Zn.

The very low values indicate that overall the level of bioavailable metals may not pose a risk to the wetland biota. However, most of these sites lacked a clear source of metals and more targeted sampling may be required to assess contamination in specific areas. For example, Johnson and Hunt (2016) collected soil samples from a stormwater bioretention cell. Overall, they found mean Mehlich 3 Cu and Zn values of 3.1 mg/kg and 26 mg/kg, respectively in the bioretention cell, which was higher than the mean levels observed in this study. However, concentrations of 11.2 mg/kg and 157 mg/kg for Cu and Zn, respectively were measured in the forebay and sample concentration generally decreased with depth. This may indicate that if metals were entering a wetland, the majority of the metals in the soil would be concentrated near the inlet source in the surface soil layer. Future sampling efforts should possibly adopt this sampling strategy to evaluate contamination and focus on sites in land uses where there is a more prominent source of metals. The results along with previous studies (Ewing et al., 2012; Barton et al., 2008) indicate that average background concentrations for wetlands in North Carolina,

specifically undisturbed wetlands, should be much lower than the values reported by Hardy et al. (2008) for upland soils.

Total Metals and Eco-SSLs Exceedance in Wetlands

Site means for total Cu ranged from 2.9 to 53 mg/kg with a mean of 15.4 mg/kg. For total Zn, the site means ranged from 7.8 to 106 mg/kg with a mean of 36.1 mg/kg (see Table 4.5

Table 4.5). These values represent and average for the top 30-cm of the soil. These values were similar to the average values for North Carolina of 56 and 34 mg/kg for Zn and Cu, respectively (US EPA, 2007c). However, these values should be interpreted with caution as they represent a small subset of the original samples, and may not always be representative of the entire site (only four to six samples per site were analyzed for total metals because of budgetary constraints). For Cu, no samples exceeded the plant Eco-SSL of 80 mg/kg. Samples from one site exceeded the soil invertebrate Eco-SSL of 70 mg/kg. This site (NSC - riverine) had some upstream development and agriculture that could contribute to these elevated total metal levels, however the Mehlich 3 results were only about 1/10 of the total metal levels for this site, which does not approach the toxicity threshold for plants. Two different site's maximum values exceeded the Eco-SSL for mammals (NSC and RKR), and both were riverine floodplain wetlands. The RKR site has significant upstream urbanization and residential development, which contributes to frequent overbank flooding, and the site received direct stormwater runoff from adjacent impervious areas. Samples from six different sites exceeded the threshold for the Avian Eco-SSL. These four additional sites either received direct agricultural runoff (NAH - headwater and DEN - basin) or were downstream from development and highways (RDU – riverine and HNK - riverine). Only two site means (RKR - riverine and NSC - riverine) exceeded the background levels for North Carolina. No maximum values for any sites exceeded

the PEC for Cu and maximum values for five sites (DEN, NAH, NSC, RDU and RKR) exceeded the TEC for Cu.

For Zn, the maximum value at one site exceeded the plant and soil invertebrate Eco-SSLs. This site (HNK) was downstream from development, adjacent to a sewer line, and received highway runoff. This same sample also exceeded the warning level for Mehlich 3 Zn. Three sites (HNK, NSC, and RKR) exceeded the mammalian Eco-SSL of 79 mg/kg and samples from six sites (HNK, NSC, RKR, DEN, RDU and MUN - riverine) exceeded the Avian Eco-SSL of 46 mg/kg. Four sites mean values (RKR, NSC, HNK, and DEN) exceeded the avian Eco-SSL. For the SQGs, no sites exceeded the PEC for Zn and one site (HNK) exceeded the TEC. The impacts of surrounding land use were again apparent as only sites in impacted areas exceeded the Zn thresholds.

The Eco-SSLs are intended to be conservative and are based on controlled lab exposure studies that indicated the greatest effects. The avian Eco-SSLs are especially conservative, as they are below background conditions for North Carolina soils (US EPA, 2007c). In addition, not as much attention should be given to the avian and mammalian Eco-SSLs as the US EPA does not recommend their application to wetland soils (US EPA, 2005). The Eco-SSLs are meant as a risk assessment tool and do not necessary indicate contamination only that further investigation may be warranted. Overall, the lower Eco-SSL were exceeded with much greater frequency (Table 4.5) than the Mehlich 3 thresholds; however, the Eco-SSL and Mehlich 3 metal thresholds for plants were only exceeded in one site for Zn. Because of variability in soil type, location and land use, it was difficult to determine if relatively elevated values at one site are indicative of contamination or simply a difference in ambient levels. US EPA (2007a and 2007b) indicated a large range in background conditions exist. However, because these sites generally exhibited

very low Mehlich 3 extractable metal concentrations, Cu and Zn levels likely no not pose a danger to the biota of these wetlands.

Table 4.5: Mean wetland site concentrations and exceedance of contamination thresholds for total and Mehlich 3 extractable Cu and Zn by at least one sample at a given site.

Site	Type	Total Cu (mg/kg)	Total Zn (mg/kg)	Eco-SSLs				SQGs		Mehlich 3	
				Plant	Soil Invertebrate	Mammal	Avian	TEC	PEC	Warning Level	Plant Tox.
BLK	H	9.2	27.4								
DEN	B	21.5	56.8				Cu, Zn	Cu			
DOT	R	4.4	19.0								
DRG	B	17.2	27.4								
HNK	R	26.9	106	Zn	Zn	Zn	Cu, Zn	Cu, Zn		Zn	
HOG	H	3.6	25.1								
LKW	R	7.1	22.2								
MUN	R	19.7	41.9				Zn				
NAH	H	12.2	28.8				Cu, Zn				
NSC	R	34.0	60.3		Cu	Cu, Zn	Cu, Zn	Cu			
PCS	H	4.9	13.4								
RDU	R	18.6	45.7				Cu, Zn	Cu			
RKR	R	53.0	87.6			Cu, Zn	Cu, Zn	Cu			
SIK	B	2.9	7.8								
SPG	H	5.5	22.8								
WMT	H	5.1	20.6								

H: Headwater Wetland R: Riverine wetland B: Basin wetland

Sites Across North Carolina

For all the 88 of the sites across the state, no single Cu sample exceeded the plant toxicity threshold or warning level. Maximum Cu values at only one site exceeded the North Carolina average of 9.2 mg/kg (see Figure 4.8). This site was a basin wetland with some agriculture in the vicinity and limited residential development. From a Level 1 assessment of the site (US EPA, 2003), it was unclear what the source of the slightly elevated levels would be, however this concentration may not be elevated and actually be indicative of the background concentration in this area. The highest site mean for Cu was 4.0 mg/kg, although about 90% of the site means were below 2 mg/kg.

Of the 88 sites, three had samples that exceeded the 50 mg/kg warning levels from RSTL (2017) for Zn. All three of these sites were located in developed areas with significant

impervious surfaces and the wetlands appear to receive stormwater runoff from these areas. However, the site means did not exceed the warning level. Only one wetland site had a sample that exceeded the plant toxicity level for Zn (120 mg/kg) from Hardy et al. (2008). The one site was located adjacent to and likely received runoff from a large parking lot, which was likely the source of contamination. The average of the site means were 0.9 mg/kg and 3.2 mg/kg for Cu and Zn, respectively for all the sites. The only sites that exceeded the warning levels were in urban landscapes. For Zn, site maximum values exceeded the warning level of 50 mg/kg at three sites and one site maximum value exceeded the plant toxicity threshold (120 mg/kg). For Cu, no site maximum values exceeded any of the warning thresholds. However, urban land use should not necessarily be used as a sole predictor of contamination, as many sites in developed areas often did not have elevated levels.

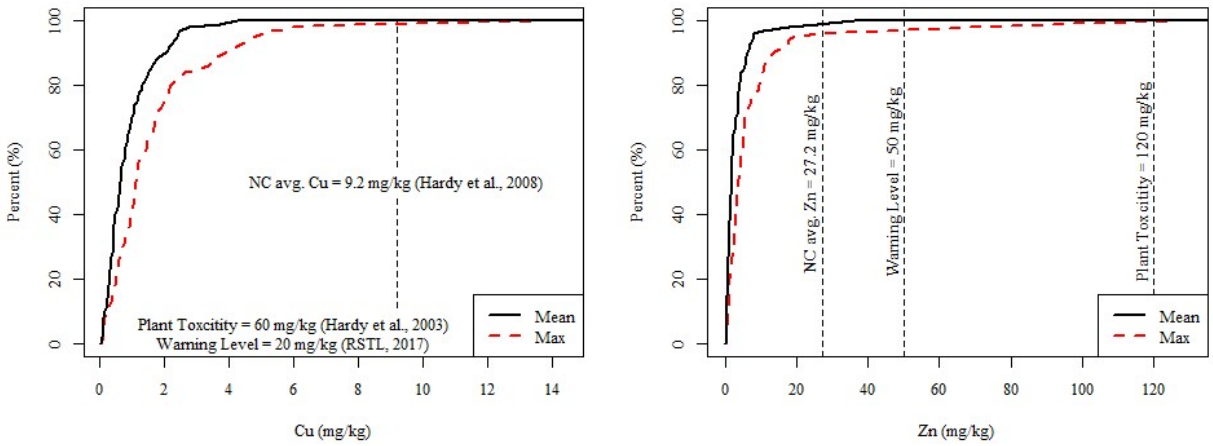

Figure 4.8: Cumulative exceedance plots for all sites with state averages, warning levels and toxicity thresholds. Site maximum values and site means are plotted (n = 88 sites).

This information combined with the more focused sampling at the 16 sites and previous studies suggests that background mean Zn and Cu concentrations for wetlands should be lower than the values for uplands published by Hardy et al. (2008). Based on these results, mean background levels for North Carolina wetlands might be set at <1 mg/kg for Mehlich 3 Cu and <4 mg/kg for

Mehlich 3 Cu. These values were based on the mean of the results from all 88 sites. In addition, the very low plant available levels may indicate that the metal concentrations in these wetland soils do not pose a risk to the wetland biota.

The 90th percentile was used to estimate thresholds for Mehlich 3 Cu and Zn in wetland soils that may be indicative of disturbance (Reimann and de Caritat, 2017). This was done for the site mean and maximum values so that these thresholds could be used to determine disturbance for both an average value and a site maximum value. The threshold for site means were about 2 mg/kg Cu and 7 mg/kg Zn. The threshold for site maximum values were 4 mg/kg Cu and 14 mg/kg Zn (Figure 4.9).

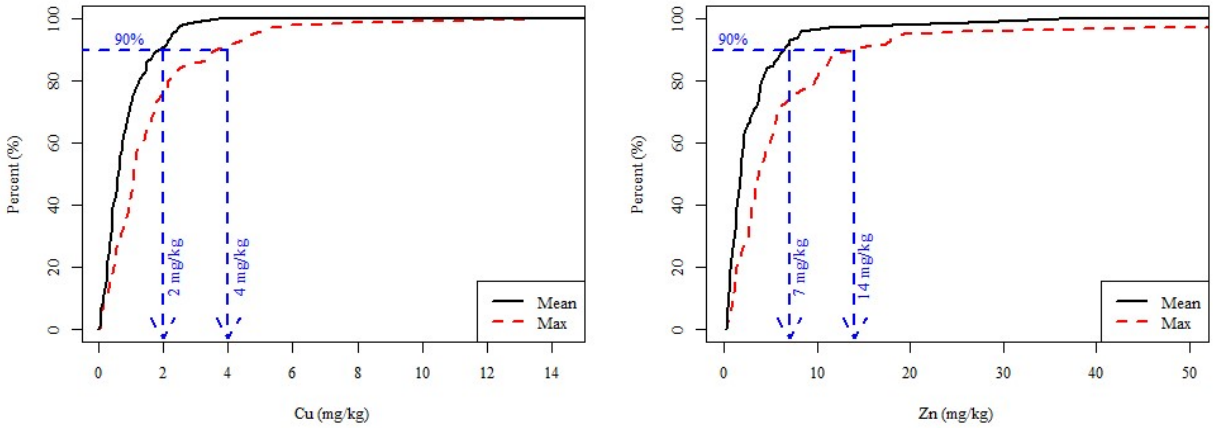

Figure 4.9: Thresholds levels that may be indicative of anthropogenic disturbance.

These calculated thresholds were compared to the results from the 16 sites using boxplots (Figure 4.10 and Figure 4.11). These thresholds do not indicate contamination, rather they were intended to indicate a potential external source causing elevation above background conditions. These thresholds appeared to be reasonable for these sites given exceedance of the mean and maximum thresholds was generally explained by some potential anthropogenic sources of metals in adjacent areas or in the wetland watersheds. The two primary sources that were identified were likely from agricultural fertilizers and sources associated with urbanization.

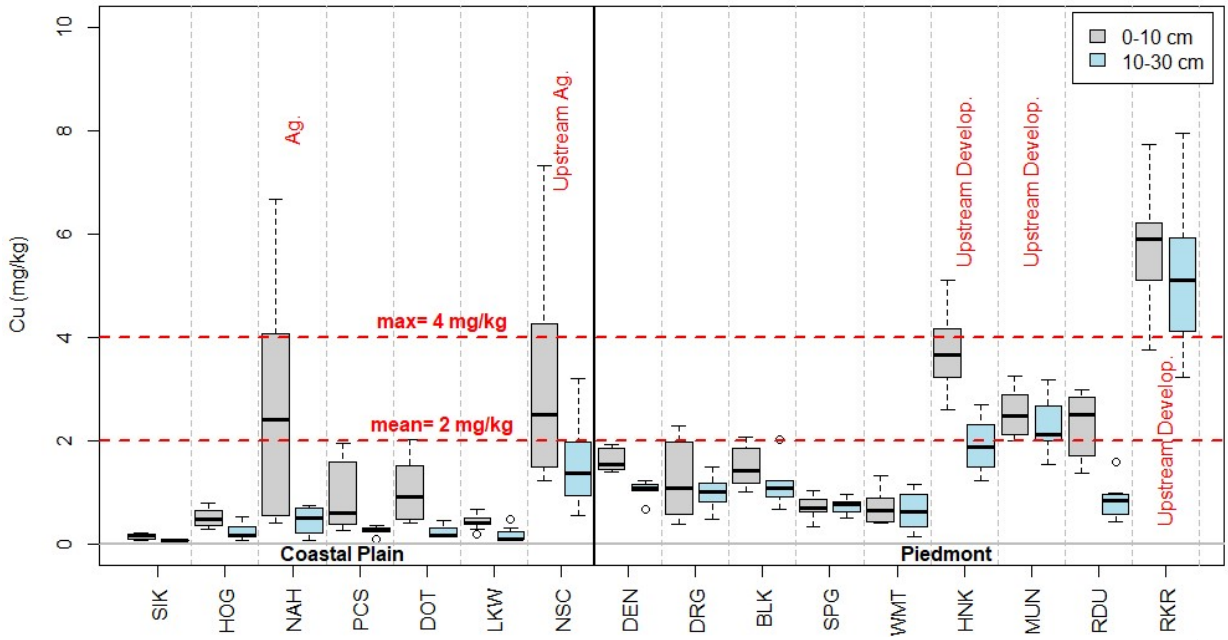

Figure 4.10: Results for Cu plotted with the calculated disturbance thresholds (based on 90th percentile). For the sites exceeding the thresholds, the primary anthropogenic disturbance (and likely sources of elevated metals levels) are included on the plot.

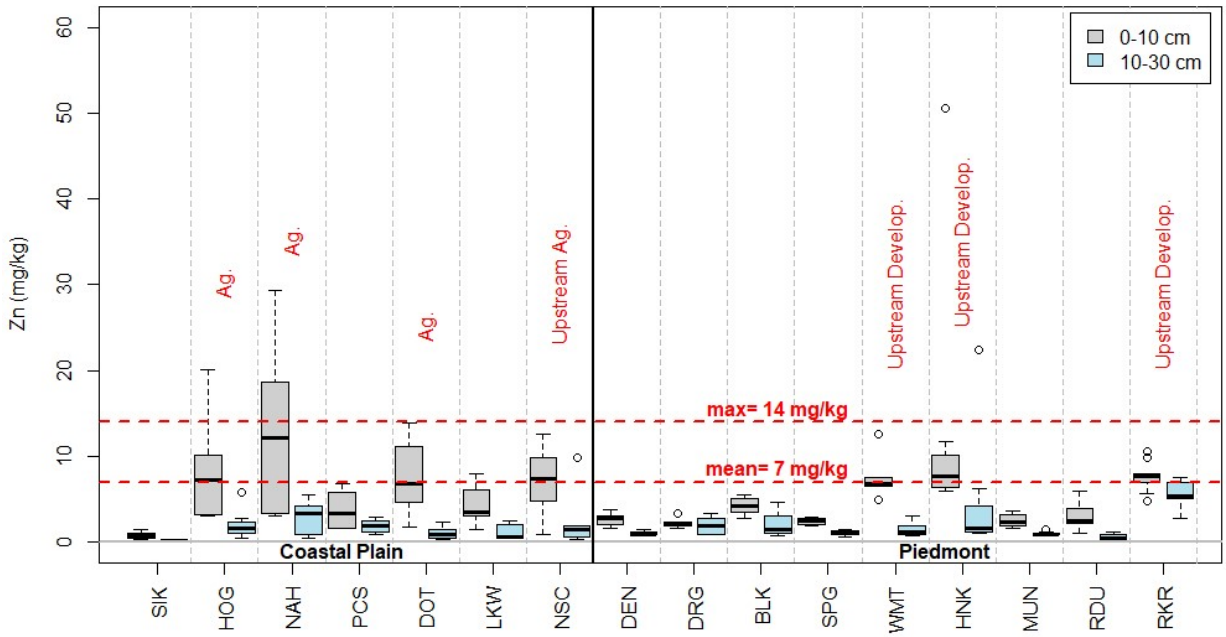

Figure 4.11: Results for Cu plotted with the calculated disturbance thresholds (based on 90th percentile). For the sites exceeding the thresholds, the primary anthropogenic disturbance (and likely sources of elevated metals levels) are included on the plot.

Mehlich 3 vs. Total Metals

The ratio of Mehlich 3 extractable metals to total metals varied widely. The ratio for Cu ranged from 0.01 to 0.51 with a mean of 0.14. For Zn, the ratio ranged from 0.01 to 0.73 with a mean of 0.15. These mean values were lower than previous studies from contaminated upland soils (McBride et al. (2009) reported ratios of approximately 0.3 for both Cu and Zn). In addition, combining the background total metal concentrations from US EPA (2007c) with the average Mehlich 3 concentrations from Hardy et al. (2008) results in ratios of Mehlich 3 extractable to total metal of 0.26 and 0.49 for Cu and Zn, respectively. However, these values were also for upland soil samples. The results from this study appear to indicate that a much lower fraction of Cu and Zn are bioavailable in wetland soils than in upland soils, although there was a large range of ratios, even within a given site.

Linear regression of total and Mehlich 3 extractable metal results was completed on a regional basis. R^2 values for the Piedmont region were higher than the Coastal Plain for both Cu and Zn (see Figure 4.12). One outlier was removed for Zn in the Piedmont region because the R^2 is very sensitive to extreme outliers. R^2 increased when the outlier was included but did not provide a better fit for a majority of the data (residuals increased). The range of values over which Mehlich 3 Cu measured was very low (0-9 mg/kg) and even with the relatively high R^2 value the residuals from the regression fit ranged from -25 to 30, which was likely not acceptable for estimating total Cu values if the goal was to compare the results to contamination thresholds. The R^2 values were lower for both regions for Zn and the residuals ranged from -76 to 69. While there was clear positive correlation between total and Mehlich 3 Cu and Zn, a simple predictive relationship did not exist that would provide accurate enough estimates of total metals to compare to contamination thresholds.

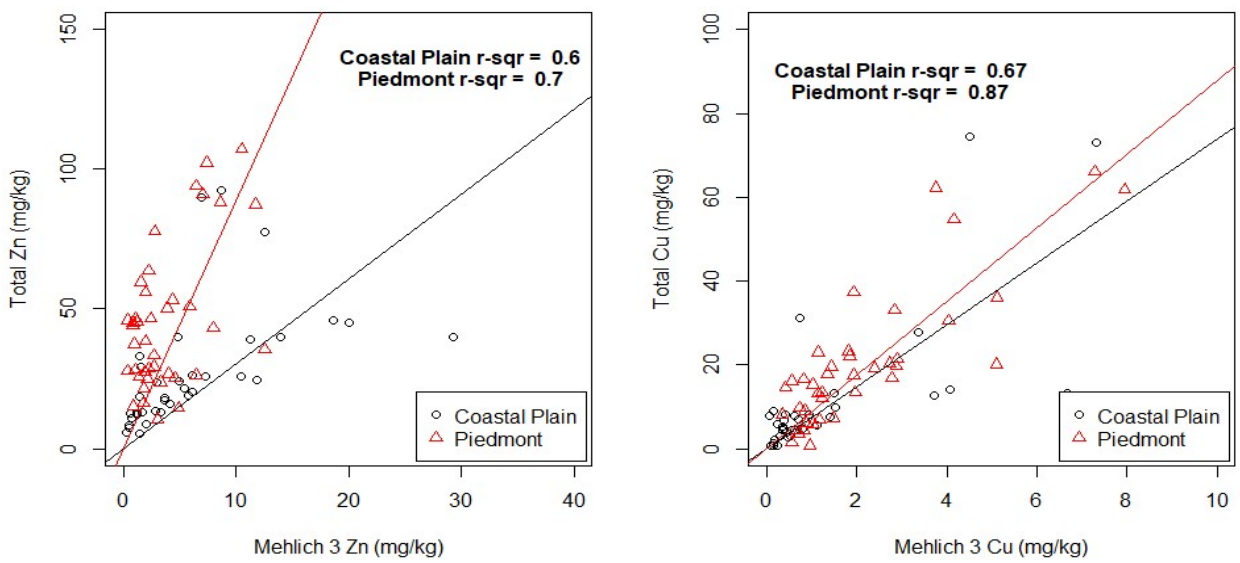

Figure 4.12: Linear regression of total metal versus Mehlich 3 extractable metals.

The variability in ratios of Mehlich 3 to total metals was likely dependent on several factors related to soil variability, and physiochemical conditions. For example, the bioavailable or mobile fraction of metals in wetlands can be affected by redox potential and pH, as well as the oxidation state of other compounds in the soils. Under reduced conditions, Cu and to some extent Zn can form complexes with sulfur, which are fairly stable in the anaerobic environment. In addition, Cu can co-precipitate with iron (Fe) in oxidized areas of the soil. Many clay minerals contain Cu that only becomes mobile after dissolution of the mineral structure through weathering processes (Reddy and DeLaune, 2008). These factors unique to wetlands combined with the different soil types, SOM content, and general spatial variability of soil chemical and physical properties (Bruland et al., 2006), as well as the small number of samples likely contribute to the large range of values observed.

Multiple linear regression (MLR) with stepwise variable selection did not markedly improve R^2 values for Cu but showed some improvement for Zn (see Table 4.6). Again, R^2 values were much higher for the Piedmont samples, which could be the result of less variability in soil organic matter content and bulk density.

Table 4.6: Results from multiple linear regression with stepwise selection.

Region	Total Metal	Final Predictors from Stepwise Selection	MLR R^2	SLR R^2
Coastal Plain	Total Zn	Zn + bulk density	0.70	0.60
Piedmont	Total Zn	Zn + pH +bulk density +humic matter	0.88	0.70
Coastal Plain	Total Cu	Cu	0.67	0.67
Piedmont	Total Cu	Cu + CEC + humic matter	0.90	0.87

Conclusion

This study involved several components related to sampling wetland soils to assess if metal contamination is a concern in natural wetlands. Overall, Mehlich 3 extractable Cu and Zn values were very low across most of the sites with exceptions that could largely be explained by surrounding land use or contaminant transport processes, and plant toxicity thresholds were rarely exceeded. Eco-SSLs were exceeded with greater frequency; however, some of the Eco-SSLs were clearly overly conservative as they were below average background concentrations for the state.

Elevated metal concentrations were largely explained by inputs from adjacent land use or upstream development. Some of the highest levels were observed in riverine wetlands, which are subject to recurrent sediment deposition that could include metals from erosion and weathering processes as well as runoff from upstream anthropogenic sources. As expected total and Mehlich 3 metal results were significantly correlated but a predictive relationship to estimate totals metals from the Mehlich 3 values that was accurate enough to compare against contamination thresholds was not obtained. Therefore, if the goal of a study was to determine the amount of metals stored in wetland soils, the total metals concentration needs to be measured rather than attempting to analyze for the mobile fraction then using a conversion factor. SOM varied across a wide range from the highly organic soils in the lower coastal plain to the mineral soils in the

piedmont. The small number of sites and lack of replication for this study prevented effective testing of differences in wetland type, region, land use and other factors.

A future study might consider evaluating wetlands in areas that have definitive sources of Zn and Cu, such as developed, urbanized areas. In addition, samples should be analyzed for other heavy metals such as lead, which has lower toxicity thresholds. The future study should be set up to focus on wetlands of the same type and similar size and landscape setting. In addition, in sites that had samples exceed the contamination risk thresholds, the exceedance did not occur at every sample location. This indicates significant spatial variability. Overall, some sites showed elevated levels of total metals, but this did not always translate to high levels of Mehlich 3 metals. Based on these results and previous studies, mean background concentrations for Mehlich 3 Cu and Zn could be established at approximately 1.0 mg/kg Cu and 4.0 mg/kg Zn for wetlands in North Carolina. Threshold values (corresponding to the 90th percentile) could be proposed as about 2 mg/kg Cu and 7 mg/kg Zn for site means and 4 mg/kg Cu and 14 mg/kg Zn for site maximum values. In addition, based on the very low mobile Zn and Cu levels in the soils of most of these wetlands, it is likely that the metal levels in a large majority of wetlands in the state do not pose a risk to the biota. As expected, elevated levels will likely not be encountered unless there is a direct anthropogenic source of metals contamination. Future studies should be targeted to areas where pollutants are most likely to aggregate across a gradient of disturbance; for example, near stormwater outfalls or near roadways.

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Chapter 5: The potential long-term impacts of climate change on the hydrologic regimes of wetlands in North Carolina

Abstract

Wetlands are especially at risk due to climate change because of their intermediate landscape position (i.e. transition between upland and aquatic environments), where small changes in precipitation and/or evapotranspiration can have significant impacts on wetland hydrology. Because hydrology is the primary factor influencing wetland establishment, structure, function and persistence, the important ecosystem services wetlands provide may be altered or lost as a result of climate change. While there is a great deal of uncertainty associated with the projected impacts of climate change on wetlands, hydrologic models and downscaled climate model projections provide tools to reduce this uncertainty. DRAINMOD is one such process-based hydrologic model that has been successfully adapted to simulate the daily water level fluctuations in natural wetlands. The objective of this project was to determine the range of possible impacts of climate change on the hydrologic regimes of non-riverine Coastal Plain wetlands in North Carolina. DRAINMOD models were calibrated and validated for two minimally disturbed, natural wetland sites in the coastal plain of North Carolina using observed water table and local weather data. Downscaled climate projections were obtained from the U.S Bureau of Reclamation. Two representative concentration pathway (RCP) scenarios were evaluated: RCP4.5 and RCP8.5. Nine models were selected from an ensemble of 32 climate models to represent the range of possible changes in mean precipitation and mean temperature. Simulations were run from 1986-2099, and results were evaluated by comparing the projected mean water table levels between the base period (1986-2015) and two future evaluation periods: 2040-2069 and 2070-2099. The model simulation results indicated projected mean water table

level may decline by as much as 25-84 cm by the end of this century (2070-2099) for the RCP8.5 scenario and may decline by 4-61 cm for the RCP4.5 scenario. Comparison of monthly mean water levels and empirical cumulative distribution functions (ECDFs) showed that water table declines were largest during the growing season when the water table was deepest in the soil profile; however, mean water table declined throughout the year for the extreme scenarios. If warming is limited to less than 1.5 degrees °C (unlikely) and rainfall does not decrease, the impacts on wetland hydrology may be limited. In Coastal Plain wetlands, declines in water tables can lead to the subsidence of organic soils, which can lead to the loss of stored carbon and increased risk of peat fires. Lower mean water levels can also lead to shifts in vegetation community composition, and loss of habitat functions for wetland dependent fauna. These results provide an overview of the potential impacts of climate change on North Carolina wetlands, and provide a range of scenarios to inform and guide possible changes to water management strategies in wetland ecosystems to limit the loss of ecosystem services over the long-term.

Introduction

The impacts of climate change on wetland structure and functions will vary across different regions (Brinson, 2006) and some wetland types are at greater risk than others. While there are uncertainties to the extent, important wetland functions will be impacted (Burkett and Kusler, 2000). The primary impacts will result from the alteration of precipitation and temperature patterns, increased temperature, and an increase in the frequency of extreme precipitation events (Erwin, 2009). As temperature and precipitation patterns change, wetlands are especially at risk because of their intermediate landscape position (i.e. transitional areas between upland and aquatic environments), where small changes in precipitation and evapotranspiration can alter the wetland hydrologic budget and result in disproportionate impacts on the area of some wetland types (Burkett and Kusler, 2000). Because hydrology is the primary factor influencing wetland structure and function (Mitsch and Gosselink, 2007), the ecosystem services that these wetlands provide may be altered or lost because of climate change.

In North Carolina, wetlands are at risk from climate change due to higher temperatures, shifting precipitation patterns, and sea-level rise. In precipitation dependent organic soil flats or pocosins in eastern North Carolina, climate change driven changes to hydrologic budgets could lower water tables and lead to oxidation and loss of the organic soils (Erwin, 2009). In some areas of Eastern North Carolina, agricultural drainage has already resulted soil subsidence of up to one meter (Ewing and Vepraskas, 2006). Lower water tables in organic soil wetlands can also increase the susceptibility to peat fires (Turetsky et al., 2011). Peatland restoration is often undertaken with the primary goal of reducing the risk of fires (Chimner et al., 2017). In addition, as societal demands for water increases in the 21st century, there may be less water available for wetland ecosystems (Middleton and Souter, 2016).

While the southeastern United States has experienced less warming than other regions of the United States during the 20th century, mean annual temperature has increased steadily since 1980, and the first decade of the 21st century was the warmest on record for the Southeast (Kunkel et al., 2013). The mean annual temperature is projected to increase between 2.2 to 4.4 °C in the southeastern United States during the 21st century (Carter et al., 2014). In addition to higher temperatures, the southeast United States will experience more extreme temperature regimes. For example, in North Carolina the number of days per year with maximum temperature exceeding 35 °C is projected to increase by 100-300% this century (US EPA, 2016). The ensemble of climate models indicate variable projections for changes in precipitation; however, a majority of models show modest increases in precipitation in North Carolina and Virginia (Kunkel et al., 2013), along with increases in extreme precipitation events (Carter et al., 2014). Higher precipitation and evapotranspiration (because of higher temperatures) may offset the hydrologic impacts on wetlands (Johnson et al., 2005), but this has not been evaluated with long-term simulations for wetlands in North Carolina.

Comparable to any long-term predictions, there is a great deal of uncertainty associated with the projected impacts of climate change on wetlands (Brinson, 2006; Erwin, 2009); however, hydrologic models that incorporate future climate scenarios have been demonstrated as a tool to reduce the uncertainty in projected outcomes (e.g. Lee et al., 2015). DRAINMOD is one hydrologic model that has been demonstrated as a useful tool for accurately simulating the long-term water level fluctuations in natural wetlands (e.g. Caldwell et al., 2007, 2011; He et al., 2002). Long-term numerical modeling of montane wetlands in California indicated that climate change will have negative impacts on wetland hydrology with some ephemeral wetlands disappearing completely and shifts in wetland type occurring for others due to changes in

hydrologic regime (Lee et al., 2015). Johnson et al. (2005) found that climate change could lead to major shifts in hydrologic regimes and regional shifts in wetland composition and wetland condition for the Prairie Pothole region of the Midwest, depending on mean changes in long-term precipitation and temperature.

The primary goal of this chapter was to evaluate the range of possible impacts on North Carolina's Coastal Plain wetlands due to climate change by addressing the following objectives:

1. Calibrate and validate DRAINMOD models for two sites in the Coastal Plain of North Carolina using observed weather and water table data.
2. Select downscaled climate model output from an ensemble of models to make the best estimate of a range of possible climate futures based on mean temperature and precipitation changes.
3. Input the selected downscaled climate projections into the calibrated DRAINMOD models and run long-term simulations to evaluate the long-term impacts of climate change on the hydrology of wetlands in North Carolina.

Methods and Materials

Site Descriptions

Two sites were selected to simulate the impacts of climate change on Coastal Plain wetlands in North Carolina (see Figure 5.1). The Great Dismal Swamp (GDS) site (36°30'15.5" N, 76°21"31.7" W) was a minimally disturbed area of the Great Dismal Swamp State Park in Camden County, North Carolina. The average annual rainfall is 124 cm and the average temperature is 16.4 °C. The area is inhabited by mature cypress (*Taxodium distichum*), black gum (*Nyssa sylvatica*), and red maple (*Acer rubrum*). The HGM wetland type is most nearly an organic soil flat. Because of the historical disturbance associated with the Great Dismal Swamp

(ditching and draining for logging) (Lichtler and Walker, 1974), there are very few undisturbed areas. The selected area was assumed to be representative of best available or minimally disturbed conditions. While there is a network of canals that were used to drain the swamp, the nearest ditch to the monitored area is over 150 meters away so it is assumed that is effects on the hydrology of the site are minimal. The mapped soil series is Belhaven muck (Loamy, mixed, dysic, thermic Terric Haplosaprists) (NRCS, 1995).

The second site is a forested non-riverine, depressional, bottomland hardwood wetland at North River Farms (NRF Site) (34°48'56.3" N, 76°34'15.8" W) in Carteret County, North Carolina. The HGM wetland type is most nearly a mineral soil flat. The average annual precipitation is 148 cm and the average temperature is 17.6 °C. This site was designated as a reference tract during wetland restoration work in the surrounding former agricultural areas. While the reference area is minimally disturbed, the surrounding areas have been ditched and drained for agriculture and forestry. This site is primarily vegetated with black gum (*Nyssa sylvatica*), red maple (*Acer rubrum*) and redbay (*Persea borbonia*). The soil series underlying the reference tract is Deloss (Fine-loamy, mixed, semiactive, thermic Typic Umbraquults) (NRCS, 1987).

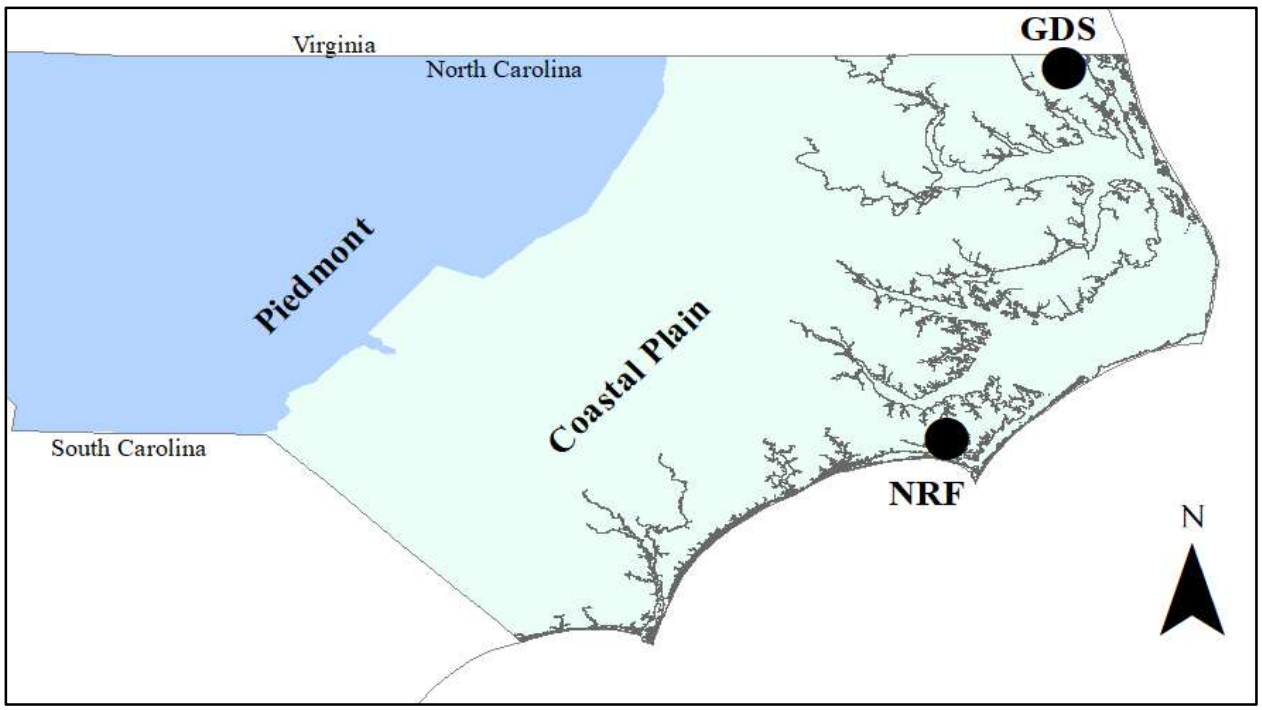


Figure 5.1: Site Map.

Water Level Monitoring

Water table monitoring wells were installed to a depth of approximately 150 cm at each site following the US Army Corp of Engineers' (USACE) monitoring well installation guidelines (USACE, 2005). The well at GDS was equipped with HOBO water level data loggers (Onset Corp., Bourne, MA) and the wells at NRF were equipped with Infinity water level loggers (Infinities USA Inc., Port Orange, FL). The data loggers recorded water level measurements every hour at NRF and every two hours at the GDS site. The data were downloaded seasonally, and manual calibration measurements were recorded. The period of record used for this project was 1/1/2015 to 12/31/2016 for GDS site and 1/1/2005 to 12/31/2008 for NRF site. The well at the NRF site was located near the edge of the depressional wetland. The data collection process and methods for the NRF site are explained in greater detail in Wright (2005) and Jarzemsky et al. (2013). The well at the GDS site was located at approximately average elevation of the wetland microtopography (not on a hummock or in a hollow). The hourly water table

measurements were converted to daily observations by selecting the last observation for each day.

DRAINMOD Model

DRAINMOD is a hydrologic model that was developed to simulate drainage and water table fluctuations on low gradient, poorly drained agricultural lands with evenly spaced parallel drains (Skaggs, 1980). The model has since been applied to successfully simulate the daily water table fluctuations in natural, non-riverine wetlands (e.g. He et al., 2002; Caldwell et al., 2007; Chescheir et al., 2008; Caldwell et al., 2011), as well as wetland restoration projects (Petru et al., 2014). DRAINMOD simulations are based on a water balance approach for a unit area of soil at the midpoint between parallel drains. Two separate water balance calculations are employed: one for the soil surface and the other in the soil profile. Water balance calculations in the soil profile rely on the following equation for each time-step.

$$\Delta V = D + ET + DLS - F \qquad \text{Equation 5.1}$$

Where ΔV is the change in drainable pore space (cm), D is drainage (cm), ET is evapotranspiration (cm), DLS is deep lateral seepage (cm) and F is infiltration (cm). Infiltration is calculated in the model using the Green-Ampt equation and drainage rate is calculated using the Hooghoudt and Kirkham equations, depending on the surface storage depth and water table height above the ground surface. ET is calculated from PET, depending on the water table depth in the profile and the upward flux. PET is calculated using the Thornthwaite equation. The change in water table depth is calculated in the model from the change in drainable pore space and the water table depth versus void space relationship (Skaggs, 1980). The water balance at the soil surface is calculated using the following equation for each time-step:

$$P = F + \Delta S + RO \qquad \text{Equation 5.2}$$

Where P is precipitation (cm), F is infiltration into the soil profile (cm), ΔS is the change in surface storage (cm), and RO is runoff (cm), which occurs when ΔS exceeds a maximum surface storage value (Sm), which is related to the surface storage provided by the microtopography.

DRAINMOD Model Inputs

Soil data

Site-specific soil parameters were not measured for this study, however, it has been demonstrated that reasonable results can be achieved using inputs that were not measured on-site (Borin et al., 1999; Messer, 2015). In addition, DRAINMOD soil inputs obtained from field measurements typically still need adjustment during the calibration process (Skaggs et al., 2012) because there is considerably spatial variability in soil properties (Skaggs et al., 1986). This is especially true of forest soils, where the hydrologic conductivity and other parameters tend to vary considerably due to tree roots. As a result of root systems, forest soils have higher hydraulic conductivity and porosity, and lower bulk density than agricultural soils (Osman, 2013). Soil inputs to the DRAINMOD model include hydraulic conductivity, layer thickness, and soil water characteristic derived parameters. For this project, layer depths and saturated hydraulic conductivity were estimated from county soil survey soil series descriptions and adjusted during the calibration process. The hydraulic conductivity was initially increased in the top layer of the profile to account for the presence of tree roots and adjusted during the calibration process. Soil water characteristic data was estimated from previous projects completed on similar soil series. Initial soil water characteristic related inputs for the NRF site were obtained from Wright (2005), inputs for the GDS site were estimated from Skaggs and Chescheir (2002). These inputs were adjusted during the calibration process within the limits recommended by Skaggs et al. (2012). DRAINMOD is very sensitive to model inputs of lateral hydraulic conductivity, and water table

depth versus volume drained relationship (Skaggs et al., 2012) therefore calibration of these parameters is nearly always required regardless of whether or not field measurements were recorded due to the spatial and vertical variability in soils. The final void volume water table depth relationships used in the models are shown in Appendix R.

Weather Data

DRAINMOD requires hourly rainfall and daily minimum and maximum temperature inputs. Daily rainfall data can be disaggregated to hourly inputs using the Weather Utility in the DRAINMOD program. The daily rainfall is typically distributed over a 4-hour period and divided over the hours of 6 am or 6 pm to minimize the impacts on the calculation of ET, which is calculated from 6 am to 6 pm in the model (Skaggs et al., 2012). Temperature and rainfall data for model calibration and past climate simulations were obtained for the nearest station in the North Carolina State Climate Office network of weather stations (State Climate Office of North Carolina, 2017). The weather station for each site are shown in Table 5.1. Onsite rainfall used for the calibration process was collected at the NRF site using a tipping bucket rain gage (Davis Instruments, Hayward, CA) by Jarzemsky et al. (2013).

Table 5.1: Weather stations used for rainfall and temperature data.

Site	CRONOS Station	Distance from Site (km)	Use
GDS	Great Dismal NWR (VGDR)	20	Long-term simulations
GDS	Wallaceton Lk Drummond (448837)	12	Calibration and validation and long-term simulations
NRF	Beaufort Smith Field (KMRH)	9	Long-term simulations
NRF	Tipping bucket rain gauge	On-site	Calibration and validation

The daily minimum and maximum temperature were used to calculate PET using the Thornthwaite method in DRAINMOD. The Thornthwaite method tends to overestimate PET during the summer and underestimate PET during the fall, winter, and spring (Skaggs et al., 2012). To account for this, monthly correction factors were used in DRAINMOD to adjust PET

estimates. The correction factors for this project were obtained from Amatya et al. (1995) (see Table 5.2). The calculation of ET in DRAINMOD is dependent on soil water availability; if sufficient water is available then ET is set equal to PET. However, if soil water is limiting, ET will be set equal to the upward flux value (Skaggs, 1980). PET monthly correction factors can be adjusted during the calibration process, but adjustment should be limited to ±15% (Skaggs et al., 2012).

Table 5.2: Initial PET correction factors.

Month	GDS site*	NRF site**
Jan.	2.27	1.9
Feb	2.84	1.99
Mar	2.12	1.88
Apr	1.67	1.69
May	1.11	1.15
Jun	1.04	0.92
Jul	0.87	0.89
Aug	0.77	0.77
Sep	0.94	0.85
Oct	1.1	1.07
Nov	1.05	1.16
Dec	1.22	1.62

*Developed for site in Washington Co, NC
**Developed for site in Carteret Co, NC

Other Inputs

Kirkham's depth (SI) and maximum surface storage (Sm) are parameters that govern the drainage rate calculation method and depth of ponding before surface runoff occurs, respectively. These parameters were initially set based on observations from the sites and then adjusted during the calibration process. Root depths were set equal to 45 cm as this is generally accepted as reasonable depth for forested wetlands in North Carolina (Skaggs et al., 1994). Deep seepage to the surficial aquifer was assumed negligible for both sites. The drainage coefficient, or the maximum drainage rate allowed in the model, is typically calculated from the size of the drain pipe or control structure capacity, but was treated as another calibration parameter for these models.

Model Calibration

The model was calibrated by adjusting parameters to minimize the average absolute difference (AAD) between the observed and simulated water levels. DRAINMOD simulates the water level between parallel drains, using the Hooghoudt and Kirkham equations, depending on the location of the water table (Skaggs et al., 2012). However, given these natural wetlands lack artificial drainage systems, the drain depth and spacing are considered as additional calibration parameters that lack a physical analog and are adjusted to best simulate the observed water table fluctuations (Caldwell et al., 2007). Additional parameters that were adjusted during the calibration process include Sm, SI, saturated hydraulic conductivity for each layer, layer depth, drainage coefficient, the water table depth versus void space relationship, and upward flux. Additional model fit measures assessed include the Nash-Sutcliffe Efficiency (NSE) and the coefficient of determination (R^2). The fit statistics where calculated using the following equations (Skaggs et al., 2012, Krause et al., 2005). Where O is observed water level, P is predicted water level and n is the number of observations.

$$NSE = 1 - \frac{\sum_{i=1}^{n}(O_i - P_i)^2}{\sum_{i=1}^{n}(O_i - \bar{O})^2} \qquad \text{Equation 5.3}$$

$$\text{AAD} = \frac{\sum_{i=1}^{n}|P_i - O_i|}{\text{n}} \qquad \text{Equation 5.4}$$

$$R^2 = \left(\frac{\sum_{i=1}^{n}(O_i - \bar{O})(P_i - \bar{P})}{\sqrt{\sum_{i=1}^{n}(O_i - \bar{O})^2}\sqrt{\sum_{i=1}^{n}(P_i - \bar{P})^2}}\right)^2 \qquad \text{Equation 5.5}$$

Climate Projections

The selection of climate models is a critical step in the climate change impacts assessment process (Lutz et al., 2016) because the uncertainty among the range of models may be the single largest source of uncertainty in the modeling process (Finger et al., 2012; Lutz et al., 2016). The latest generation of climate models from the World Climate Research

Programme's Coupled Model Intercomparison Project (CMIP5) are based on four greenhouse gas concentration scenarios or Representative Concentration Pathways (RCP) that were adapted by the Intergovernmental Panel on Climate Change (IPCC): RCP2.5, RCP4.5, RCP6.0 and RCP8.5. The RCPs represent scenarios in which future greenhouse gas concentrations increase at different rates depending on development and mitigation actions to combat emissions. RCP2.5 represents a scenario in which emissions peak around 2020 (van Vuuren et al., 2011), as a result this scenario was assumed to be unlikely because of the lack of action to curb emissions on a global scale. RCP4.5 represents an intermediate concentration scenario in which emissions peak around 2040 and RCP 8.5 represents a high-end concentration scenario in which CO_2 emissions continue to increase though the end of the 21st century (van Vuuren et al., 2011). RCP4.5 and RCP8.5 are the two RCPs often selected for climate change assessments (e.g. Hathaway et al., 2014; Lutz et al., 2016), and are the most widely available for downscaled (i.e. local) climate projections (USGS, 2014). They provide a range of possible climate futures ranging from an average mitigation scenario to a worst case or "business as usual" scenario.

A consortium of universities and government agencies headed by the U.S. Bureau of Reclamation (USBR) provided access to statistically downscaled climate projections across the United States for 32 different climate model scenarios (USBR, 2017). The downscaled daily maximum and minimum temperature and rainfall projections were obtained from the group's Downscaled CMIP3 and CMIP5 Climate and Hydrology Projections archive (https://gdo-dcp.ucllnl.org/downscaled_cmip_projections/). The downscaled projections were provided at 4 km grid units and downscaling was performed using the Localized Constructed Analogs (LOCA) statistical downscaling method (Pierce et al., 2014). While there are many statistical downscaling methods, the LOCA method better reproduces extreme events and reduces (although does not

eliminate) the problem of many days with very low rainfall totals (drizzle) that is a common issue with other downscaling methods (Pierce et al., 2014). Thirty-two different downscaled climate model projections were obtained for both RCP4.5 and RCP8.5 for this analysis. See Appendix Q for a list of all the models considered in this study.

Climate Model Selection

With the large number of climate models available, steps must be taken to reduce the total number of models for climate change impact assessments (Lutz et al., 2016). There are two primary methods for selecting a climate model for climate change assessment. The first is uncertainty-based (also referred to as envelope-based) selection in which models are selected to encompass the range of possible climate futures. The other is a performance-based approach, in which models are selected that best replicate historical climate observations (USBR, 2016). The performance-based selection method may seem like a more intuitive option, however, a link between a model's ability to simulate historical climate and its accuracy in projecting future climate change has not been demonstrated (Knutti et al., 2010). In addition, model ranking may change based on the metric used to evaluate the model in comparison to historical observations (USGS, 2014). Because climate models are not designed to recreate specific observations (Mote et al., 2011), comparisons must be made based on the long-term climate trends.

The method used in this study is a modification of the uncertainty-based approach from USBR (2016). This method aims to select models that represent the range of future uncertainty in changes to mean annual temperature and precipitation. In the first step, changes in mean annual precipitation and temperature were calculated for each model between a base 30-year period (1986-2015) and future period in which the impacts of climate change are being evaluated (2070-2099). After the change in mean temperature and precipitation were calculated for each model,

the results for all the models were then plotted as the change in mean temperature versus the percent change in mean precipitation. The 10th and 90th percentiles and the mean of precipitation change (ΔP) and temperature change (ΔT) were then calculated. The percentile pairs represent the ranges of possible climate futures. For example, the 10th percentile of ΔT and the 10th percentile of ΔP represent a "warm-dry" outcome, where temperature increase is lowest and the precipitation decrease is the highest among the models (or lowest increase). This "warm-dry" scenario does not necessarily correspond to dryer conditions than the present, but rather "dry" in relation to the ensemble of model projections. Thus, warm dry could correspond to a slight increase or decrease in relation to current mean precipitation. Conversely, the "warm" scenario always refers to an increase in mean temperature relative to the present. The 90th percentile of ΔT and the 10th percentile of ΔP represents a "hot-dry" outcome (large increase in temperature and small increase or decrease in precipitation). The 90th percentiles represent a "hot-wet" scenario in which there are relatively large increases in temperature and precipitation. The 90th percentile of ΔT and the mean of (ΔP) represents a hot-average change in precipitation scenario. The other pairs follow the same logic. The 90th and 10th percentiles were used instead of the extremes to minimize the chance of selecting possible outliers (Lutz et al., 2016). After the changes in precipitation and temperature between the two periods were calculated, the model in closest proximity to the given percentile pair (e.g. warm-dry), based on standardized Euclidian distance, was selected. In summary, the original 32 models were reduced to nine models for evaluation of climate change impacts (see Figure 5.2). All calculations and plots were completed in R 3.4 (R Core Team, 2017).

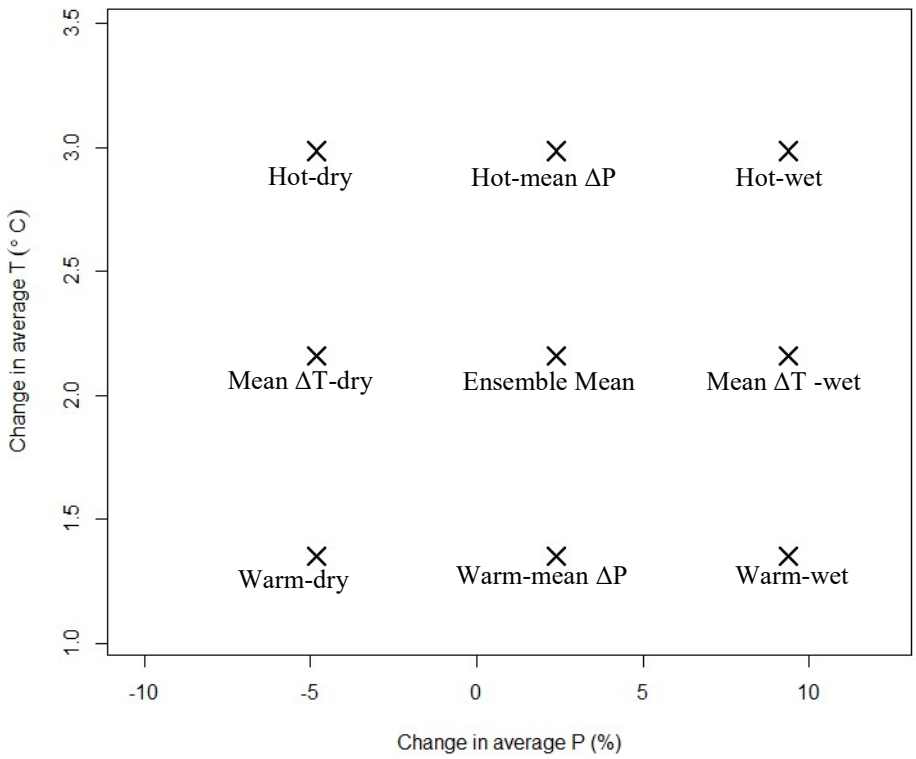


Figure 5.2: Example plot of change in average temperature versus change in average precipitation. Plotted markers (X) represent the 10th and 90th percentile and the mean values for the change in mean temperature and precipitation.

Once a final model was selected for each RCP and scenario (i.e. hot-dry, cold-wet, etc.), the corresponding projections for T_{min}, T_{max} and daily precipitation were used to create DRAINMOD input files in the DRAINMOD Weather Utility. Simulations were run using the calibrated models for each climate model scenario from 1986-2099. In addition, simulations were run from 1986-2015 using observed weather data. The impacts of climate change were evaluated based on comparing the model output for a base 30-yr period to future 30-year periods to evaluate the change in average conditions. A summary of the model simulation scenarios is provided in Table 5.3.

Table 5.3: Summary of model simulations.

Scenario	Description
Base Observed	Calibrated and validated DRAINMOD model with observed precipitation and temperature from 1986 to 2015 as inputs
Base Modeled	Calibrated and validated DRAINMOD model with climate model precipitation and temperature from 1986 to 2015 as inputs
2040-2069	Calibrated and validated DRAINMOD model with climate model precipitation and temperature from 2040 to 2069 as inputs
2070-2099	Calibrated and validated DRAINMOD model with climate model precipitation and temperature from 2070 to 2099 as inputs

Jurisdictional Wetland Status and Growing Season

The USACE wetland delineation manual states that the growing season for wetland evaluation purposes can be approximated as the "frost-free" period defined by beginning and ending dates based on 28 degrees F air temperature threshold at a 50% frequency (USACE, 2005). The growing season for the GDS site, based on the WETS table for Elizabeth City, NC, was from March 18th to November 27th. The growing season for the NRF site was March 6th to December 8th based on the WETS table for Morehead City 2WNW station (NRCS, 2017). The increase in average temperature because of climate change is projected to increase the growing season length throughout the United States (Walsh et al., 2014). The growing season length has been shown to impact the jurisdictional status of some wetlands (Skaggs, 2012). In order to evaluate the impact of growing season length on jurisdictional wetland status at these sites, the frost-free growing season was calculated from the model outputs for projected minimum temperature. An overall mean projected increase in growing season length was calculated for each RCP and divided evenly over the beginning and ending of the growing season for the evaluation. For example, if the growing season increased by 30 days, 15 days were added to the beginning and the end of the growing season.

Jurisdictional wetland hydrology is defined by the USACE as a period of continuous saturation or inundation within 30 cm of soil surface for a period of 14 days during the growing

season in most years (50%) (USACE, 2005). However, different percentages are often used in place of the 14-day duration for restoration projects. For example, a soil saturation goal of 5 to 12.5 % of the growing season has been historically used for restoration projects in North Carolina (Hill et al., 2013). In 2016, The USACE Wilmington District released new saturation duration guidelines for restoration projects based on the soil series and ecoregion (NCIRT, 2016). These standards were intended to raise monitoring standards to those set forth in the US EPA 2008 Compensatory Mitigation for Losses of Aquatic Resources; Final Rule (USACE, 2016). For the GDS site on Belhaven soil series, this duration corresponds to 20% of the growing season or 51 days for the current climate. For the NRF soil series of Deloss, the duration is 12% of the growing season or 33 days for the current climate. The 14-day saturation threshold would only correspond to about 5% of the growing for both sites.

Results and Discussion

DRAINMOD Calibration

The models were calibrated to minimize the ADD between the observed and modeled water table. Additional fit measured used to evaluate the model fit include NSE and R^2. The AAD and NSE for the calibration and validation periods were within the acceptable ranges identified by Skaggs et al. (2012). The model calibration and validation results are shown in Table 5.4 and Figure 5.3

Table 5.4: Calibration and Validation results for DRAINMOD models.

Site	Calibration Period	AAD (cm)	NSE	R^2	Validation Period	AAD (cm)	EF	R^2
GDS	Jan. 2015 – Dec. 2015	6.6	0.68	0.82	Jan. 2016 – Dec. 2016	3.9	0.73	0.68
NRF	Jan. 2006 – Jun. 2007	13.4	0.59	0.76	Jul. 2007 – Dec. 2008	16.6	0.75	0.85

Note: NSE: Nash-Sutcliffe modeling efficiency
AAD: Average absolute deviation
R^2: Coefficient of determination

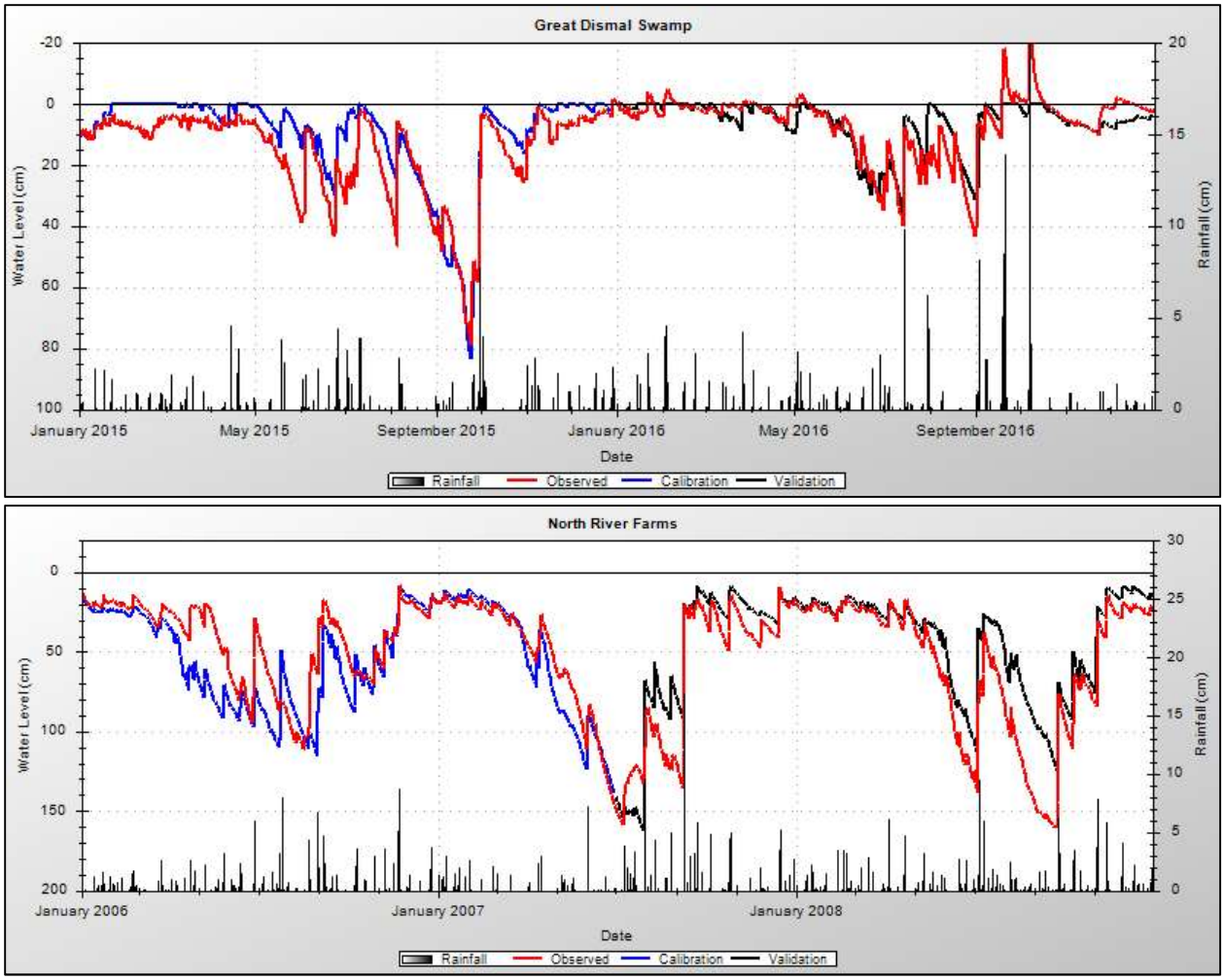


Figure 5.3: Calibration and validation results for the DRAINMOD models for the GDS site (top) and NRF site (bottom).

Overall, the GDS had much lower ADD values than the NRF site but the NSE and R^2 values were comparable between the two sites. However, the range of observed water table fluctuations was almost double at the NRF site compared to the observations at the GDS site, relative error tends to increase as the range and depth of fluctuations increase for DRAINMOD (Yang et al., 2007). The final parameter values used in the calibrated models are shown in Table 5.5.

Table 5.5: Final model parameterization.

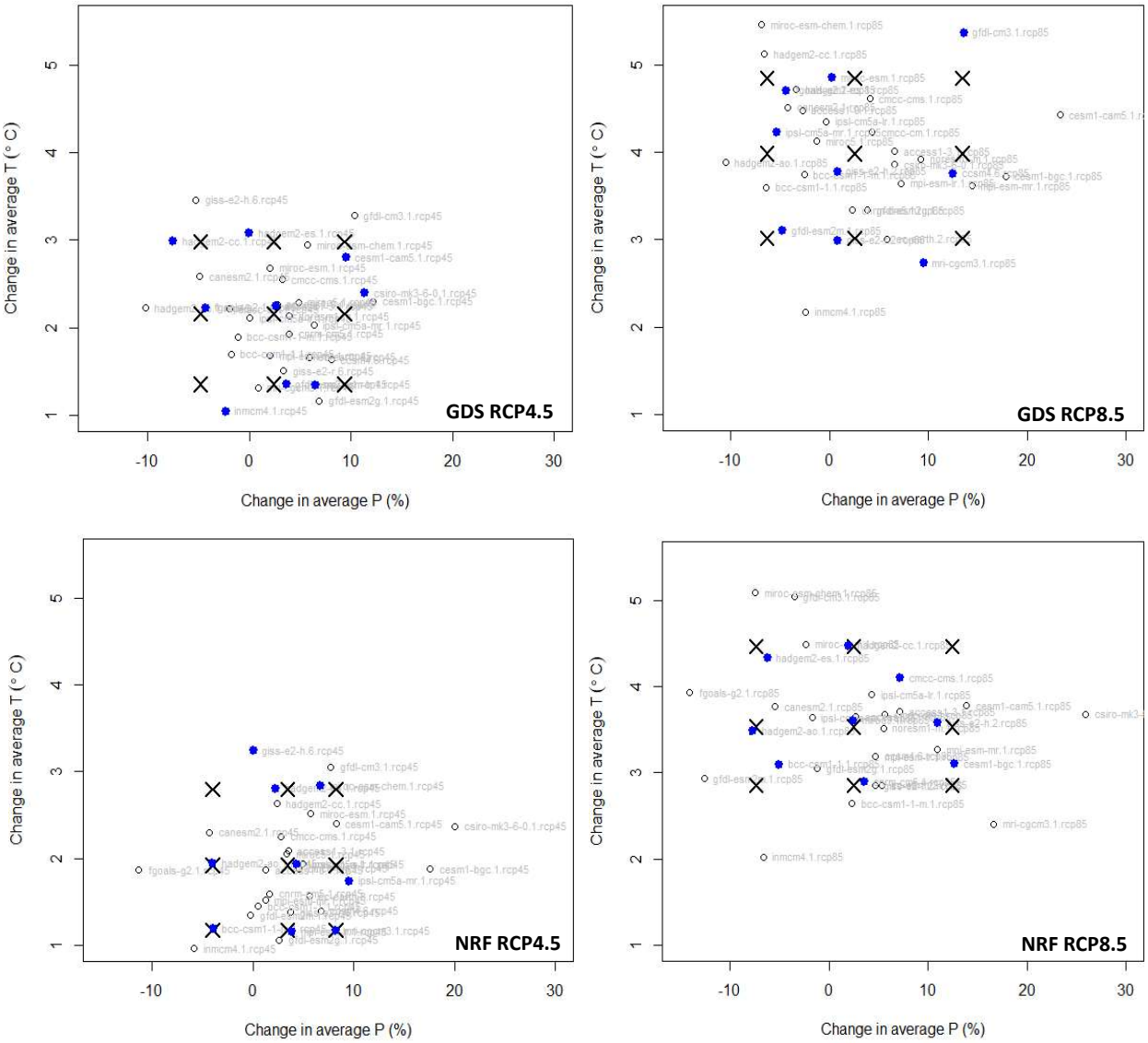
Parameter	GDS	NRF
Drain Depth (cm)*	150	150
Drain Spacing (cm)*	20000	7500
Distance to impermeable layer (cm)	200	200
Hydraulic Conductivity (cm/hr)	25 (0-30 cm) 3(30-50 cm) 2 (50-150 cm) 5 (150-200cm)	9 (0-23) 2.2 (23-51 cm) 2.8 (51-61 cm) 2 (61-165) 3 (165-200)
S_m (cm)	8	3
SI (cm)	0.5	1
Root depth (cm)	45	45
Drainage Coefficient (DC) (cm)	5	5

*Note: Drain depth and spacing have no physical analog in natural wetlands and are treated as calibration parameters in the model. Wide drain spacing is typically required to accurately simulate natural wetland water levels in DRAINMOD.

While the calibration results mostly met the criteria for good or excellent for the fit statistics from Skaggs et al. (2012), several sources of error likely contributed to the discrepancies between the modeled and observed water table. First, the lack of field-measured soil parameters was a likely source of error, especially given that forest soils differ considerably from agricultural soils. In addition, DRAINMOD relies on the Thornthwaite method for the calculation of PET. The Thornthwaite method was developed for agricultural soils and crops (Lu et al., 2005). There are differences for PET in the forest ecosystem that the Thornthwaite method cannot accurately account for, specifically transpiration related to stomatal resistance. The PET correction factors can be adjusted to account for some of the differences, and DRAINMOD has been successfully used to simulate the long-term hydrology of forested wetlands (e.g. Caldwell et al., 2011), but this is a likely source of error. In addition, for the GDS site, model calibration was completed with offsite rainfall data (approximately 12 km from the site), which could result in missing some storms and over or under predicting others, depending on the precipitation patterns during the calibration period. Based on the calibration results (Figure 5.3), it appears this was a problem for some events (May of 2015 for example).

Climate Model Selection

The ensemble of climate models selected based on the changes in mean precipitation and temperature between the base period (1986-2015) and future period (2070-2099) are shown in Figure 5.4. The bold dots represent selected models in closest proximity to the given percentile pairs. The models identified for the end of century period (2070-2099) were also used to evaluate changes for the mid-century period (2040-2069). The final models selected for each site and scenario are shown along with the projected changes in mean temperature and precipitation in Table 5.6.



○ - Model projection for ΔT and ΔP × - Percentile pairs (10th and 90th) and means
● - Selected model closest to respective percentile pair

Figure 5.4: Selection of climate models based on changes in mean temperature and precipitation. Clockwise from top left: GDS site RCP4.5, GDS site RCP8.5, NRF site RCP8.5, NRF site RCP4.5.

Table 5.6: Final models selected for each temperature and precipitation change and RCP pair.

Site	Scenario	Final model RCP4.5	ΔP (%)	ΔT (°C)	Final model RCP8.5	ΔP (%)	ΔT (°C)
GDS	Hot-dry	hadgem2-cc.1	-7.5	3.0	fgoals-g2.1	-4.4	4.7
	Mean ΔT-dry	fgoals-g2.1	-4.3	2.2	ipsl-cm5a-mr.1	-5.4	4.2
	Warm-dry	inmcm4.1	-2.4	1.1	gfdl-esm2m.1	-4.8	3.1
	Hot-mean ΔP	hadgem2-es.1	0.0	3.1	miroc-esm.1	0.2	4.9
	Ensemble Mean	access1-3.1	2.6	2.2	giss-e2-h.2	0.8	3.8
	Warm-mean ΔP	gfdl-esm2m.1	3.6	1.4	giss-e2-r.2	0.8	3.0
	Hot-wet	cesm1-cam5.1	9.5	2.8	gfdl-cm3.1	13.6	5.4
	Mean ΔT-wet	csiro-mk3-6-0.1	11.4	2.4	ccsm4.6	12.5	3.8
	Warm-wet	mpi-esm-lr.1	6.5	1.4	mri-cgcm3.1	9.5	2.7
Site	**Scenario**	**Final model RCP4.5**	**ΔP (%)**	**ΔT (°C)**	**Final model RCP8.5**	**ΔP (%)**	**ΔT (°C)**
NRF	Hot-dry	giss-e2-h.6	-0.9	3.2	hadgem2-es.1	-6.2	4.3
	Mean ΔT-dry	hadgem2-ao.1	-4.0	1.9	hadgem2-ao.1	-7.7	3.5
	Warm-dry	bcc-csm1-1-m.1	-3.9	1.2	bcc-csm1-1.1	-5.0	3.1
	Hot-mean ΔP	hadgem2-es.1	2.3	2.8	hadgem2-cc.1	1.9	4.5
	Ensemble Mean	noresm1-m.1	4.3	1.9	miroc5.1	2.5	3.6
	Warm-mean ΔP	mpi-esm-lr.1	3.9	1.2	cnrm-cm5.1	3.6	2.9
	Hot-wet	miroc-esm-chem.1	6.7	2.8	cmcc-cms.1	7.2	4.1
	Mean ΔT-wet	ipsl-cm5a-mr.1	9.5	1.7	giss-e2-h.2	11.0	3.6
	Warm-wet	mri-cgcm3.1	8.2	1.2	cesm1-bgc.1	12.6	3.1

ΔP = % change in mean annual precipitation
ΔT = Change in mean annual air temperature (°C)

There are multiple levels of uncertainty regarding the climate modeling process including the trajectories of future emissions, the response of the global climate to future emissions, and the future uncertainty regarding the range of natural climate variability (Mote et al., 2011). Different climate models take into account different processes and simulate these processes at different levels of sophistication. In addition, many models rely on similar parameterization and underlying processes so the results may not represent independent outcomes (USBR, 2016).

The uncertainty-based selection process employed here only evaluated changes in mean temperature and precipitation. Changes to precipitation patterns and extreme events were not considered as well as other variables that could influence wetland processes such as solar radiation. As a result, the outcomes presented here do not necessarily represent the full range of climate futures (USBR, 2016), and internal model variability such as differences in parameterization and process simulations represent another layer of uncertainty that cannot be

accurately assessed. However, this approach provides a credible approximation of the projected changes in mean temperature and precipitation, the primary drivers of variability in wetland hydrologic regimes and the main inputs to the DRAINMOD model. Therefore, while errors or uncertainty from other model processes may influence the outcomes, the simulations should primarily reflect the results of the changes to temperature and precipitation.

In addition, it is important to recognize there are some inherent biases in climate models. For example, climate models over-predict the number of days on which rainfall occurs (Pierce et al, 2014). So, while average rainfall totals may be similar, climate models predict many days where a very small amount of rainfall occurs (e.g. <2 mm). For example, over the observed 30-year base period at the GDS site, rainfall was recorded on an average of 123 days per year. The average number of days the climate models projected rainfall was 217 days per year, but 95% of the rainfall occurred in about 123 days. Therefore, the climate models predicted about 95 days per year with precipitation <2 mm. This is the result of inherent inaccuracies in the simulation of precipitation in GCMs (Pendergrass and Hartmann, 2014) as well as difficulties associated with downscaling precipitation from GCMs to the local scale (Pierce et al., 2014). Because this is a problem for both the base period and future simulation of precipitation in the model projections, it was assumed that it should not affect the estimation of relative changes in water level.

DRAINMOD Climate Model Results

Changes to Mean Annual Water Level

The predicted water levels for the future climate scenarios were compared based on 30-year evaluation periods, which are typically used to assess average conditions for climate change assessments (USGS, 2014). Model results were evaluated based on changes in mean water table level, relative to the base-period; not absolute depth below the ground surface to avoid errors

arising from slightly different mean water level results for the base period models. The changes in mean annual water level for each scenario are shown in Table 5.7. For the GDS site, the mean annual water level declined by 9 to 54 cm and 24 to 75 cm by the end of the century (2070-2099) for RCP4.5 and RCP8.5 projections, respectively. The mean water level declined less for the mid-century period (2040-2069), with mean declines of 0-37 cm for RCP4.5 and 14-43 cm for the RCP8.5 scenario.

For the NRF site, the range of projected declines in mean annual water level were greater than that for the GDS site. Mid-century (2040-2069) declines were projected to range from 13-54 cm for RCP4.5 and 18-50 cm for RCP8.5 models. For the end of century period (2070-2099), mean water levels were projected to decline 4-61 cm for RCP4.5 and 27-84 cm for RCP8.5. Overall, mean water level declined for all the end of century scenarios and either declined (most scenarios) or remained the same for the mid-century scenarios.

Observed water levels at the NRF site were lower than the GDS site. The water table was at or above the surface about 20% of the time for the model output based on the observed climate data at the GDS site, while the modeled water level (based on observed climate) never reached the surface for the NRF site. Because void space decreases with depth, more ET or seepage (drainage) would be required to lower the water table when it is closer to the surface. Thus, the water table tends to drawdown further at the NRF site because of lower initial conditions in the spring than at the GDS site.

Table 5.7: Decrease in mean annual water table for the future evaluation periods relative to the base period (1986-2016) for both sites. All values are in cm.

Great Dismal Swamp (GDS)										
GDS - RCP4.5: 2040-2069						GDS – RCP8.5: 2040-2069				
		ΔP						ΔP		
		Dry	Mean	Wet				Dry	Mean	Wet
ΔT	Hot	26	20	10		ΔT	Hot	43	41	19
	Mean	37	13	0			Mean	21	33	14
	Warm	14	0	10			Warm	17	14	16
GDS - RCP4.5: 2070-2099						GDS – RCP8.5: 2070-2099				
		ΔP						ΔP		
		Dry	Mean	Wet				Dry	Mean	Wet
ΔT	Hot	54	40	11		ΔT	Hot	75	63	51
	Mean	41	20	9			Mean	54	50	35
	Warm	14	9	22			Warm	38	35	24
North River Farms (NRF)										
NRF - RCP4.5: 2040-2069						NRF – RCP8.5: 2040-2069				
		ΔP						ΔP		
		Dry	Mean	Wet				Dry	Mean	Wet
ΔT	Hot	54	34	42		ΔT	Hot	42	46	50
	Mean	29	13	20			Mean	50	30	32
	Warm	27	20	15			Warm	18	32	21
NRF - RCP4.5: 2070-2099						NRF – RCP8.5: 2070-2099				
		ΔP						ΔP		
		Dry	Mean	Wet				Dry	Mean	Wet
ΔT	Hot	61	47	32		ΔT	Hot	84	67	59
	Mean	40	30	20			Mean	75	60	47
	Warm	34	20	4			Warm	66	47	27

Some of the smaller projected changes in mean water table levels were a result of the climate models that projected large increases in mean precipitation (>10%) or small mean temperature changes (<1.5°C). This suggests that a significant increase in precipitation may offset warming and limit potential impacts on wetland hydrology. A similar finding was reported by Johnson et al. (2005). In addition, if warming can be limited to less than 1.5° C the impacts may be less severe. The least projected warming and greatest increase in precipitation did not always correspond with the smallest declines in water level and the driest and hottest scenarios did not always result in the largest water table declines. For some scenario and RCP combinations, declines in water table are actually projected to be greater for RCP4.5 than for RCP8.5. This can likely be explained by internal model variability with regard to seasonal

temperature and precipitation patterns that cannot be adequately explained by mean changes between the evaluation periods. Overall, mean water level is projected to decline by the end of this century for all scenarios; however, impacts may be limited if warming is limited to the lower end of the RCP4.5 scenario and precipitation does not decrease.

It is important to recognize that these models do not represent the entire range of uncertainty, and any greater probability of occurrence cannot be associated with any particular model (Lutz et al., 2016; Sanderson and Caldwell, 2015). Therefore, the range of future uncertainty in temperature and precipitation changes is likely greater than depicted by the ensemble of models. In addition, when changes to temperature and precipitation patterns (i.e. seasonal patterns of precipitation or occurrence of heat waves and droughts) are considered in addition to changes in means (for this project the selection of climate models was based on changes in long-term mean temperature and precipitation only), the level of uncertainty only increases.

Mean Hydropattern

For the GDS site, the mean monthly water table results from the climate model projections for the base period (DRAINMOD output for climate model projected precipitation and temperature for the 1986-2015) replicated the model output from the long-term observed data (DRAINMOD output for observed precipitation and temperature from 1986-2015) well. NSE values ranged from 0.48 to 0.97 for RCP4.5 and 0.67 to 0.92 for RCP8.5 (see Figure 5.5). This indicated that the climate model temperature and precipitation were reasonable representations of the observed climate over the base period (i.e. the model output using climate model temperature and precipitation as inputs resulted in similar mean water level as the model output using observed temperature and precipitation as inputs). The climate model data did not

recreate the base period conditions as well for the NRF site, with lower overall NSE values (see Figure 5.6). The DRAINMOD model results with the climate model data inputs tended to under-predict the mean monthly water level for October through December. This may indicate there is some inherent bias in the models for this coastal location surrounded by estuaries, which could be related to difficulties some of the climate models have simulating sustained precipitation events (e.g. Pearson et al., 2015).

The model results indicated that for many of the climate change scenarios declines in mean monthly water level are, not unexpectedly, greatest during the summer and fall, which is mostly the result of higher evapotranspiration rates during the summer months (Gilliam and Skaggs, 1981). However, for the hotter, dryer scenarios water levels declined throughout the year, especially when modeled with RCP8.5. See 0 for average annual ET estimates.

In terms of wetland condition, greater declines during the growing season may not be as impactful if the water table remains at or near the surface during the winter and spring. This seems to be the case for the GDS site, with the exception of the extreme scenarios (i.e. "hot-dry"), in which water levels decline throughout the year. If water levels return to the surface for an extended period after a growing season decline (as is currently the case for the GDS site), oxidation of organic soils may be limited and long-term susceptibility to fire may be prevented for periods of the year. For example, it has been estimated that organic soils in Florida need to be saturated for 60% of the year to limit oxidation and subsidence (Lowe et al., 1984). In addition, Gilliam and Skaggs (1981) observed water table depths near the surface from November through March and then drawdowns near one meter during the growing season in undisturbed pocosins in eastern North Carolina, which is similar to the current conditions for the GDS site.

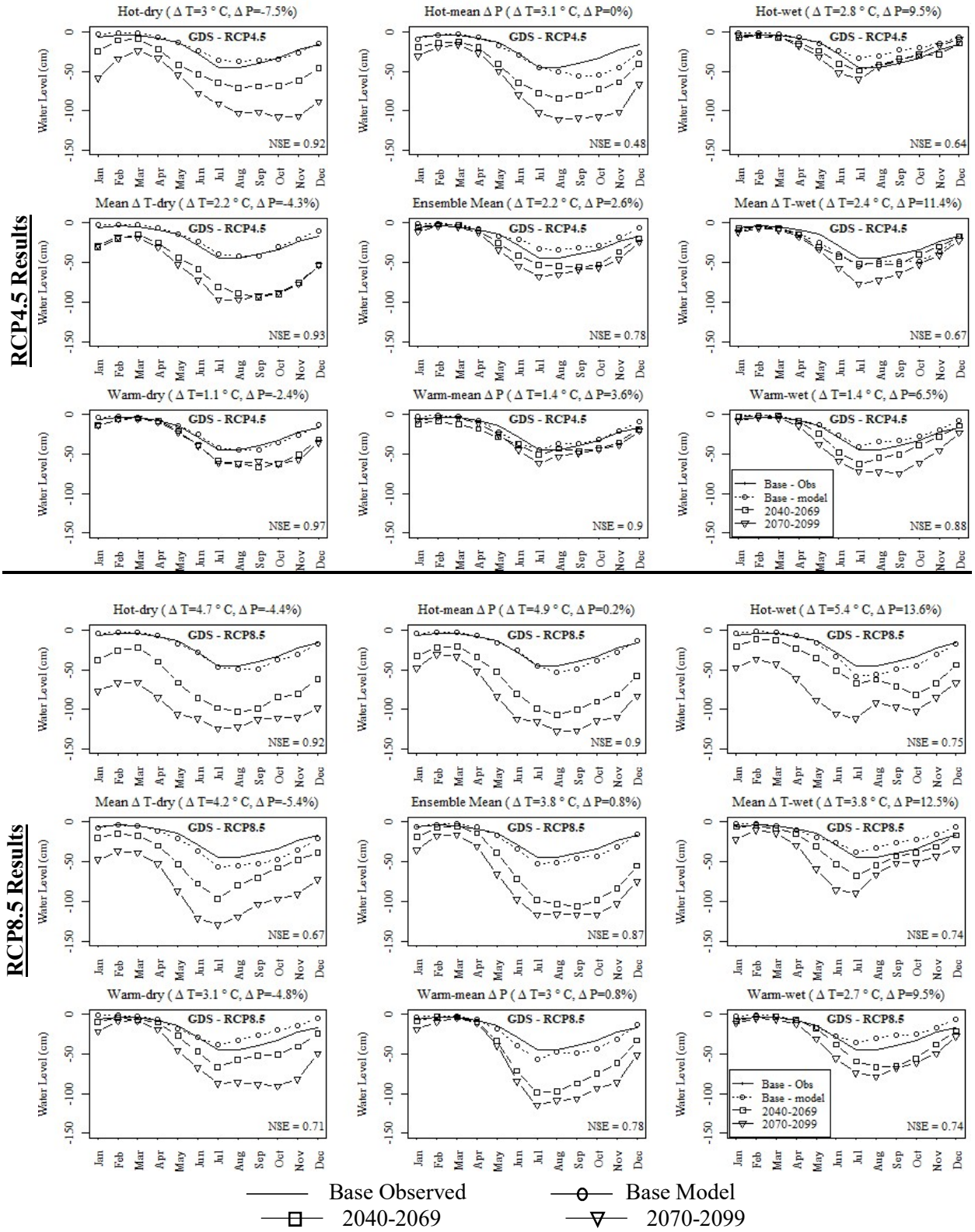


Figure 5.5: Monthly mean water levels for the GDS site. Plots are arranged with ΔT increasing from bottom to top and ΔP increasing from left to right. NSE values for the base period comparisons are shown on the bottom-right of each plot.

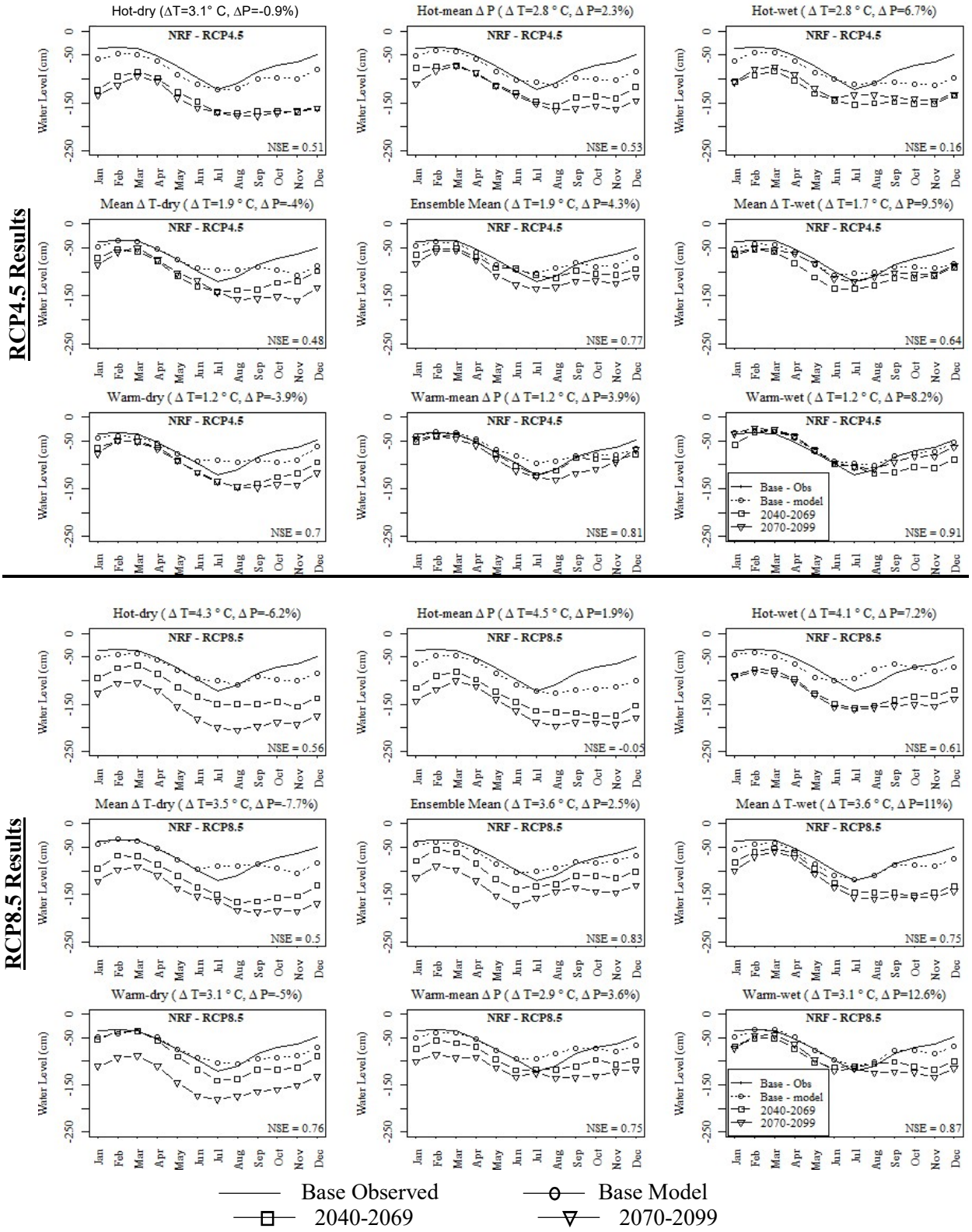


Figure 5.6: Monthly mean water levels for the NRF site. Plots are arranged with ΔT increasing from bottom to top and ΔP increasing from left to right. NSE values for the base period comparisons are shown on the bottom-right of each plot.

Water Level Empirical Cumulative Distributions Functions (ECDFs)

ECDFs are a useful tool to visualize differences between hydrologic time series data (Hychka et al., 2013). They allow for the comparison of the entire distribution of observed water level fluctuations. For this analysis, ECDFs were created for the average annual conditions for each 30-year evaluation period. The ECDF were then plotted together to compare the difference in water level distributions over time. The ECDF for the model runs of the observed temperature and precipitation data were also computed to evaluate how closely the climate model results replicated the long-term observed model results for the base period (1986-2015), and NSE values were calculated to compare the two. The ECDFs show the percent exceedance for a given water table level. For example, in Figure 5.7 the topmost right plot shows for the 2070 to 2099 period, the mean annual water level was above -100 cm 100% of the time and above -25 cm about 50% of the time.

For the most part, the ECDFs of the base modeled scenarios were relatively close matches to the base observed scenarios for the GDS site, which indicated climate models replicated the long-term observed climate relatively well, with NSE values generally above 0.85 (see Figure 5.7). That is the model output from climate model predictions of temperature and precipitation and the observed temperature and precipitation resulted in similar average water level conditions over the 30-year base period (1986-2015). This reasonable replication of base conditions indicated that despite differences in year-to- year precipitation and temperature between the climate model and observations and internal climate model variability, especially with regard to rainfall simulation, the long-term DRAINMOD simulations still result in similar average conditions. This validates the use of the DRAINMOD model and lends more credibility to the long-term results by potentially indicating that changes observed for future simulations

(i.e. 2070-2099) are largely the result of changes to precipitation and temperature and not results of the methods in which the models simulate different processes or inaccuracies in DRAINMOD inputs. The NSE values for the base period ECDF at the NRF site were again lower than for the GDS site, but still reasonable fits.

The ECDFs again illustrate that the water table declines were greatest during periods of the year when water table was furthest from the surface (during the growing season (Figure 5.5 and Figure 5.6)). However, for some of the scenarios, a decrease in depth across the entire distribution was observed, which might indicate significant warming during the winter months. Predictably, the greatest declines were for the scenarios associated with the highest increase in temperature and largest decrease in precipitation.

For both sites, it is apparent that for very low temperature increases (<1.5 °C), changes to the water level distribution may be limited if rainfall does not decrease substantially. However, it was again apparent that changes in mean temperature and precipitation are not the only causes of changes to mean water level ECDFs. For example, for the GDS site the RCP4.5 warm-wet scenario showed greater declines in water level than the warm-mean ΔP scenario; although the warm-wet scenario corresponds to a larger increase in precipitation with the same increase in temperature. Therefore, while increasing temperature and decreasing precipitation generally resulted in lower water levels, it needs to be understood that there is internal model variability that can result in models with very similar projected temperature and precipitation changes may not result in the same water level distributions. The reality is that these are model projections, and the actual future climate may be represented by a combination of models.

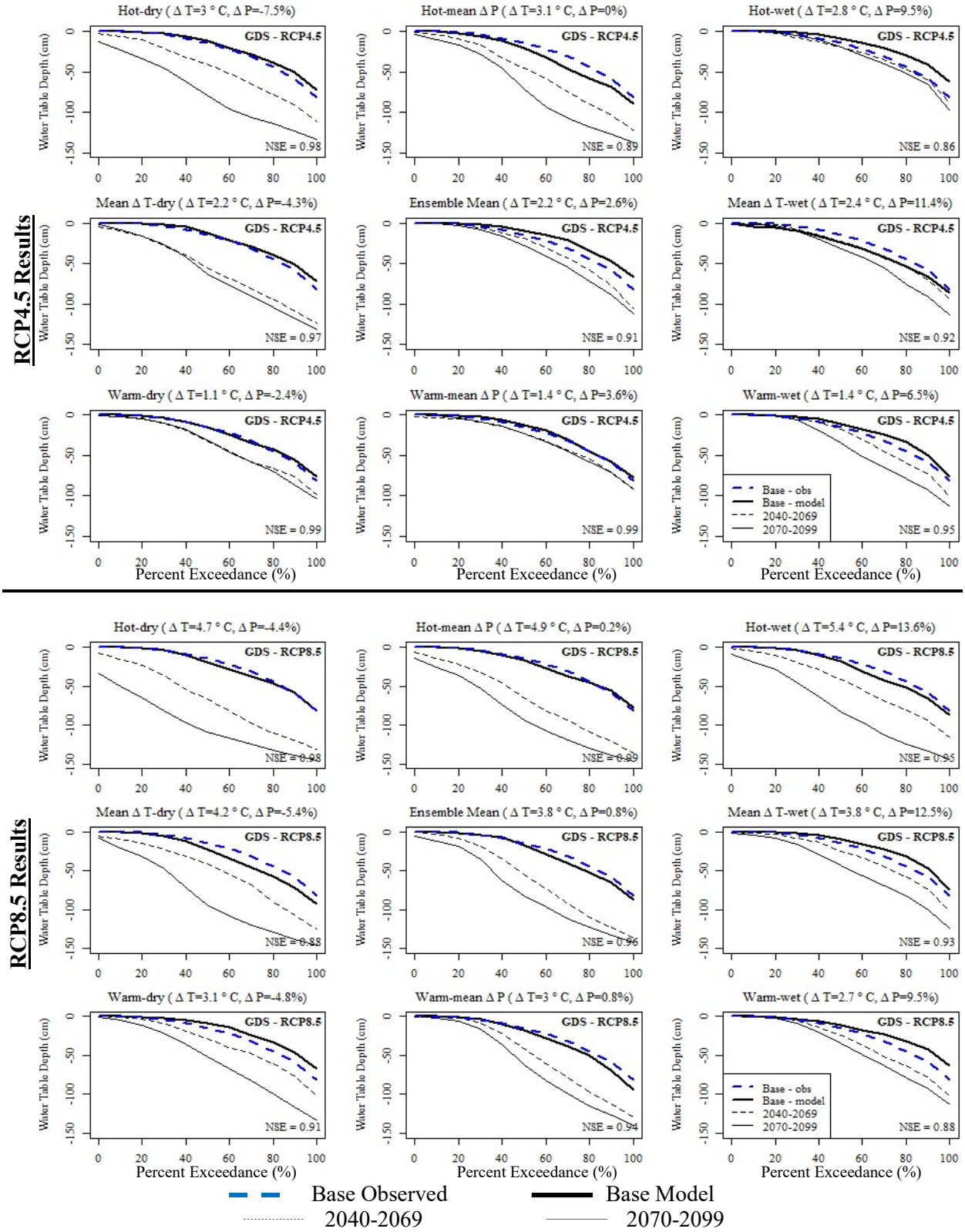


Figure 5.7: Mean annual ECDFs for the GDS Site. Plots are arranged with ΔT increasing from bottom to top and ΔP increasing from left to right. NSE values for the base period comparisons are shown on the bottom-right of each plot.

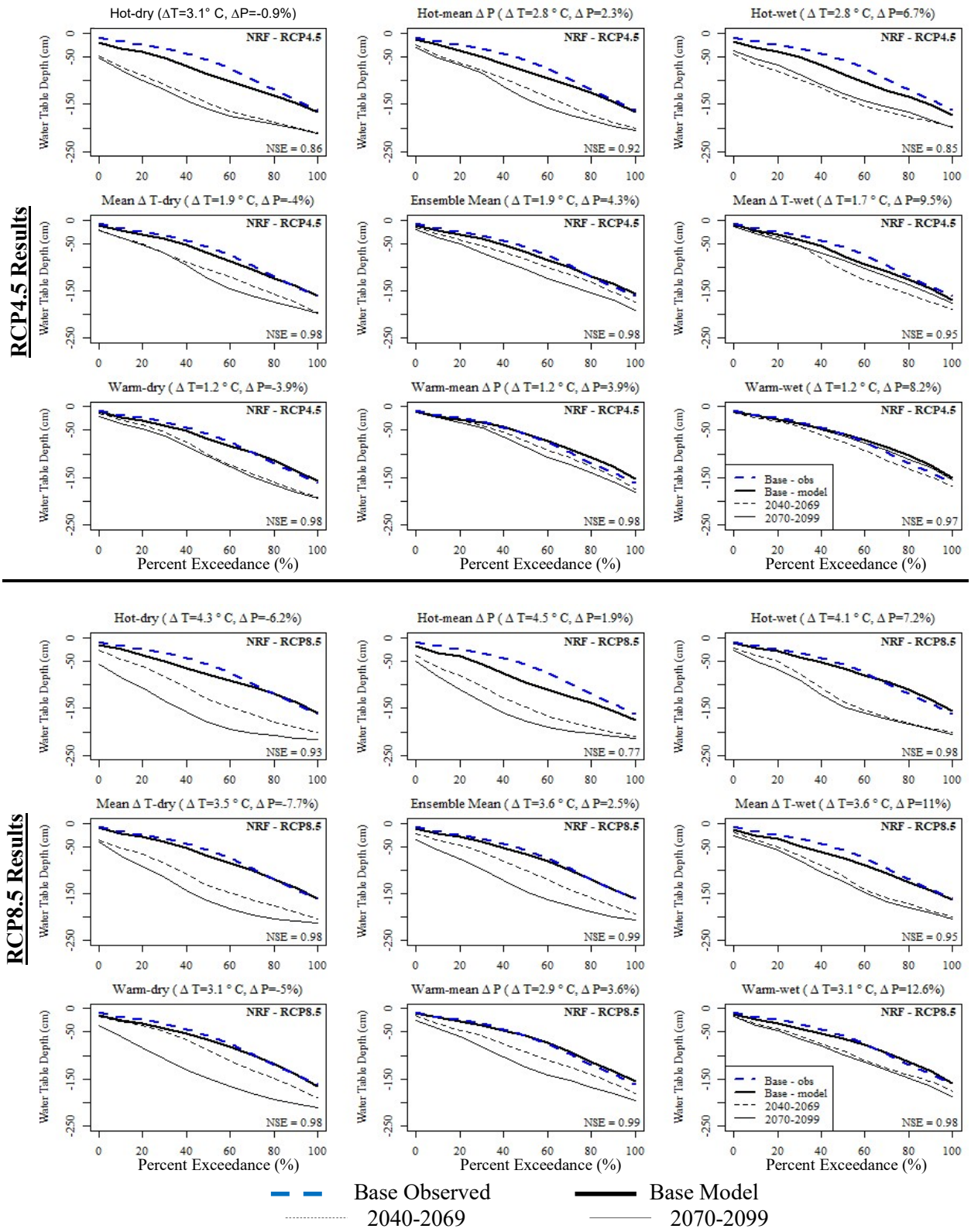


Figure 5.8: Mean annual ECDFs for the NRF Site. Plots are arranged with ΔT increasing from bottom to top and ΔP increasing from left to right. NSE values for the base period comparisons are shown on the bottom-right of each plot.

Jurisdictional Hydrology Analysis

The longest continuous period of saturation (i.e. water table within 30 cm of the soil surface) for the model results was computed and compared to the threshold saturation criteria for each site. For the base period (1986-2016), climate model saturation durations were exceeded at similar frequencies to the predictions obtained using long-term observed data (see Table 5.9). For the GDS site, the hydrologic criteria was exceeded 28-30 of 30 years for a 14-day saturation period and 26-30 of 30 years for the 51-day saturation period (20% of the growing season). Minor decreases in the number of years the criteria was satisfied for the 14-day duration was observed for the mid-century period, but more so for the end of century period. The only scenario where jurisdictional hydrologic criteria was not met was for RCP8.5 hot-dry scenario, which corresponds to the greatest temperature increase and precipitation decrease. For the longer saturation duration (20% of the growing season), the saturation criteria will be exceeded far less frequently. However, failure to meet the criteria was not observed until the end of the century period (2070-2099) for the hotter and dryer scenarios.

The NRF site was much drier than the GDS site with a deeper average water table, maximum observed water level about 10-cm below the ground surface (compared to at or above the surface at GDS), and a much greater range of observed water table fluctuations during the monitoring period. As a result, jurisdictional hydrologic criteria in future scenarios was met less often at the NRF site. For the 14-day duration threshold, failures to meet the criteria were predicted starting during the mid-century period. For the 12% saturation threshold, the model run using the observed historical data only just satisfied the threshold of 15 in 30 years. All model simulations for the base period resulted in similar exceedance percentages. For the mid and end-of century periods a majority of the models predicted that the 12% saturation period was not

satisfied in 50% of the years, with only some of the wetter scenarios for RCP4.5 meeting the criteria. These results indicate that for the wetter sites (GDS), while water levels decline, jurisdictional hydrology may still be met except for the extreme scenarios. However, for the drier borderline sites like NRF, wetland jurisdictional criteria may not be met starting at or before the middle of this century.

The longer growing season generally resulted in an increase of one to three years where the saturation criteria were met in the future (Table 5.8 and Table 5.9). However, no change or a minor decrease was also observed. For many of the models that were very close to the threshold (but still not meeting the criteria), an increase in the growing season length generally resulted in the threshold being met. This was likely because the saturation threshold was typically satisfied at the beginning of the growing season so increasing the growing season length extended the growing season into a period when the water table was near the surface (i.e. early spring), thus extending the saturation period.

Table 5.8: Increase in growing season length due to climate change.

Site	RCP4.5		RCP8.5	
	2040-2069	2070-2099	2040-2069	2070-2099
GDS	19 days	27 days	28 days	50 days
NRF	20 days	24 days	25 days	43 days

Note: The increase in growing season length was split evenly and added to the start and end dates of the growing season.

Table 5.9: Changes to jurisdictional hydrologic status for climate change scenarios. Values given are for current growing season length. Values in parentheses are for climate change adjusted growing season shown in Table 5.8. Values in bold are scenarios where wetland jurisdictional criteria was not met.

GDS (14 day saturation period)						
	1986-2016		**2040-2069**		**2070-2099**	
Scenario	**RCP4.5**	**RCP8.5**	**RCP4.5**	**RCP8.5**	**RCP4.5**	**RCP8.5**
Observed	30/30		----------		----------	
Hot-dry	30/30	30/30	28/30 (29/30)	23/30 (26/30)	19/30 (19/30)	**13/30 (13/30)**
Mean ΔT-dry	30/30	30/30	25/30 (25/30)	25/30 (25/30)	26/30 (26/30)	21/30 (23/30)
Warm-dry	30/30	30/30	29/30 (29/30)	29/30 (29/30)	29/30 (30/30)	28/30 (29/30)
Hot-mean ΔP	30/30	30/30	28/30 (28/30)	23/30 (25/30)	27/30 (28/30)	18/30 (19/30)
Ensemble Mean	30/30	30/30	30/30 (30/30)	29/30 (30/30)	30/30 (30/30)	25/30 (26/30)
Warm-mean ΔP	30/30	30/30	29/30 (29/30)	30/30 (30/30)	30/30 (30/30)	30/30 (30/30)
Hot-wet	30/30	30/30	29/30 (30/30)	27/30 (29/30)	30/30 (30/30)	21/30 (24/30)
Mean ΔT-wet	28/30	30/30	30/30 (30/30)	30/30 (30/30)	29/30 (29/30)	29/30 (29/30)
Warm-wet	30/30	30/30	30/30 (30/30)	30/30 (30/30)	30/30 (30/30)	30/30 (30/30)
GDS (20% of growing season)						
	1986-2016		**2040-2069**		**2070-2099**	
Scenario	**RCP4.5**	**RCP8.5**	**RCP4.5**	**RCP8.5**	**RCP4.5**	**RCP8.5**
Observed	27/30		----------		----------	
Hot-dry	29/30	27/30	20/30 (22/30)	**14/30** (16/30)	**11/30 (11/30)**	**2/30 (5/30)**
Mean ΔT-dry	30/30	28/30	18/30 (19/30)	19/30 (20/30)	**14/30** (17/30)	**7/30 (12/30)**
Warm-dry	28/30	29/30	24/30 (25/30)	23/30 (25/30)	25/30 (27/30)	**13/30** (18/30)
Hot-mean ΔP	27/30	26/30	17/30 (20/30)	**14/30** (15/30)	**14/30** (18/30)	**7/30 (10/30)**
Ensemble Mean	29/30	30/30	25/30 (27/30)	19/30 (21/30)	23/30 (25/30)	**10/30** (15/30)
Warm-mean ΔP	29/30	28/30	26/30 (27/30)	20/30 (24/30)	23/30 (25/30)	18/30 (26/30)
Hot-wet	30/30	26/30	26/30 (26/30)	21/30 (21/30)	25/30 (27/30)	**10/30** (15/30)
Mean ΔT-wet	26/30	29/30	21/30 (21/30)	22/30 (23/30)	23/30 (24/30)	20/30 (24/30)
Warm-wet	28/30	28/30	25/30 (28/30)	28/30 (29/30)	20/30 (21/30)	26/30 (28/30)
NRF (14 day saturation period)						
	1986-2016		**2040-2069**		**2070-2099**	
	RCP4.5	**RCP8.5**	**RCP4.5**	**RCP8.5**	**RCP4.5**	**RCP8.5**
Observed	22/30		----------		----------	
Hot-dry	20/30	22/30	**9/30 (9/30)**	15/30 (17/30)	**7/30 (9/30)**	**8/30 (10/30)**
Mean ΔT-dry	21/30	21/30	**11/30 (13/30)**	**11/30 (12/30)**	**14/30** (17/30)	**9/30 (10/30)**
Warm-dry	23/30	22/30	20/30 (22/30)	19/30 (20/30)	16/30 (17/30)	**9/30 (10/30)**
Hot-mean ΔP	22/30	19/30	16/30 (16/30)	**10/30 (10/30)**	**12/30 (14/30)**	**5/30 (8/30)**
Ensemble Mean	25/30	23/30	19/30 (21/30)	**14/30** (16/30)	15/30 (16/30)	**12/30** (16/30)
Warm-mean ΔP	26/30	26/30	24/30 (25/30)	17/30 (21/30)	23/30 (24/30)	17/30 (17/30)
Hot-wet	21/30	23/30	**13/30 (12/30)**	19/30 (20/30)	**14/30 (14/30)**	**10/30 (13/30)**
Mean ΔT-wet	22/30	23/30	20/30 (20/30)	16/30 (18/30)	19/30 (19/30)	17/30 (18/30)
Warm-wet	23/30	22/30	25/30 (27/30)	19/30 (20/30)	24/30 (25/30)	15/30 (20/30)
NRF (12% of growing season)						
	1986-2016		**2040-2069**		**2070-2099**	
Scenario	**RCP4.5**	**RCP8.5**	**RCP4.5**	**RCP8.5**	**RCP4.5**	**RCP8.5**
Observed	15/30		----------		----------	
Hot-dry	**14/30**	**14/30**	**5/30 (4/30)**	**9/30 (9/30)**	**1/30 (2/30)**	**4/30 (5/30)**
Mean ΔT-dry	18/30	17/30	**7/30 (9/30)**	**4/30 (5/30)**	**8/30 (6/30)**	**2/30 (5/30)**
Warm-dry	19/30	21/30	**7/30 (8/30)**	**13/30 (15/30)**	**9/30 (10/30)**	**3/30 (4/30)**
Hot-mean ΔP	15/30	**14/30**	**6/30 (5/30)**	**6/30 (7/30)**	**5/30 (7/30)**	**1/30 (1/30)**
Ensemble Mean	**14/30**	18/30	15/30 (15/30)	**9/30 (10/30)**	**11/30 (12/30)**	**6/30 (8/30)**
Warm-mean ΔP	18/30	23/30	17/30 (15/30)	**10/30 (11/30)**	**14/30 (14/30)**	**11/30 (11/30)**
Hot-wet	18/30	17/30	**8/30 (8/30)**	**8/30 (9/30)**	**7/30 (7/30)**	**5/30 (6/30)**
Mean ΔT-wet	18/30	**14/30**	**9/30 (13/30)**	**9/30 (10/30)**	15/30 (15/30)	**10/30 (8/30)**
Warm-wet	17/30	16/30	18/30 (18/30)	**11/30 (12/30)**	20/30 (20/30)	**11/30 (13/30)**

Model Limitations

One of the advantages of DRAINMOD is the process-based nature of the model that allows for use without specific field measured values by using inputs from a given range of reasonable values as well as the limited number of weather inputs required (Skaggs et al., 2012). However, the simplicity of the model also leads to some shortcomings for the application to climate modeling. The most problematic issue with using DRAINMOD for climate change simulations is the fact that DRAINMOD employs the Thornthwaite method to calculate PET. The Thornthwaite method is often used because the only required input is temperature. However, the Thornthwaite method was developed using past atmospheric conditions. As CO_2, temperature and solar radiation increase, the relationship between temperature and PET may become even less accurate (Shaw and Riha, 2011). This is a fundamental shortcoming of using temperature based PET methods for climate change modeling, as PET is more dependent on solar radiation, wind speed, and vapor pressure conditions than temperature (Shaw and Riha, 2011). As atmospheric CO_2 concentration increases, transpiration is actually predicted to decrease due to increased stomatal resistance (Snyder et al., 2011). This has been demonstrated in greenhouse experiments (Rosolem et al., 2010).

Therefore, the reported ranges of declines in water level may be overestimates because the Thornthwaite method will likely over-predict PET the further into the future the model is used (Shaw and Riha, 2011). However, while downscaled climate model monthly projections for PET are available, they are not vegetation specific. Downscaled projections for solar radiation are rarely available at the daily resolution. Therefore, with the publically available climate projections, the use of the Thornthwaite method for PET estimation may be the best available means of conducting a climate change modeling assessment until projections for more variables

are available. However, there is uncertainty on how ET will be impacted due to several competing factors. While transpiration may decrease due to increased CO_2, this could be offset by increased plant productivity and increased evaporation because of higher temperature (Pan et al., 2015).

The NRF site is near the coast and concurrent sea level rise will also likely affect the surrounding drainage systems. Sea level rise was not considered in this modeling approach, as the goal was to assess the impacts of projected changes to precipitation and temperature. However, a future, more comprehensive evaluation of climate change at this site would consider the impacts of rising sea level and saltwater intrusion on the wetland ecosystem.

Ecological Implications

Wetland ecosystems are dependent on prolonged periods of inundation or saturation to maintain the vegetative communities and provide suitable habitat for the wetland dependent fauna. This is especially true for rare and endangered biota in North Carolina, where approximately 70 percent of the rare and endangered plant and animal species are dependent on wetlands (Fretwell et al., 1996). Because of the history of wetland losses and degradation in Coastal Plain of North Carolina (Cashin et al., 1992), the area of high quality, minimally disturbed wetlands is relatively small and fragmented. With some of the higher end predictions (i.e. RCP8.5 – "hot-dry") these jurisdictional wetlands are predicted to possibly become legally upland environments. Even if these systems do retain jurisdictional status (as is the case for the lower end scenarios), lower water tables could alter the ecosystem structure and function because hydrology is the primary driver of these elements in wetland ecosystems.

Prolonged drawdown of the water table in organic soil wetland ecosystems can lead to primary soil subsidence due to loss of buoyant forces. The resulting aerated soil profile can lead

to oxidation of the stored organic matter, which can lower surface elevation by over 2-cm per year (Ewing and Vepraskas, 2006) and release stored carbon that accumulated over centuries (Mitsch and Gosselink, 2007). Soil subsidence exposes tree roots and likely increases susceptibility of trees to windthrow during extreme events.

Lower water tables would also influence nutrient cycling in wetland environments. In addition to increased CO_2 release from the oxidation of stored organic matter, specifically recalcitrant humic substances (Reddy and DeLaune, 2008), dryer conditions may reduce the production of methane (which could be viewed as a positive outcome) by decreasing the thickness of the anaerobic zone where methanogenesis could occur and also increasing methane oxidation to CO_2 (Mitsch and Gosselink, 2007). These same conditions could lead to changes to nitrogen cycling because of increased mineralization of organic nitrogen, increased nitrification and decreased denitrification as the result of the soil profile being aerobic for longer periods at greater depths (Reddy and DeLaune, 2008). For riparian wetlands, this could lead to increased export of nitrogen, phosphorus and carbon during high flow events.

Water table drawdowns also lead to changes to the vegetative composition of wetlands. Wetland plant species organize themselves along hydrologic gradients due to their varying tolerance to periods of saturation and inundation (Reddy and DeLaune, 2008). As wetlands become dryer and periods of saturation and inundation become less frequent and shorter, the competitive advantage enjoyed by obligate and facultative wetland plants can diminish; this can result in shifts in vegetative community structure (Mälson et al., 2007). At the GDS site, the cypress trees (*Taxodium distichum*) may be at risk long-term because of climate change. Cypress trees can survive in dryer conditions and even in upland environments; however, seed germination requires saturated conditions for one to three months (NRCS, 2002). Thus, climate

change may not eliminate the existing cypress trees immediately, but long-term, the species composition would shift to tree species that can become established and better compete under dryer conditions (i.e. upland species).

Possible Management Implications

Over the next century, ecosystem managers will be tasked with the challenge of protecting sensitive species with increasingly scarce water resources due to climate change (Middleton and Souter, 2016). The results of this research support the potential need to allocate more water resources to maintaining wetland ecosystem function in the future. For example, in highly managed wetland ecosystems such as the system of canals and water control structures in the Great Dismal Swamp, new water management strategies may need to be developed and implemented (such as maintaining water levels in drainage canals by pumping to minimize drawdowns during the growing season and installing more water control structures). However, as the demand for water resources for human consumption, agriculture, and industry increases as population grows, it will be increasingly difficult to balance the demands of society and natural ecosystems (Naiman et al., 1995).

Summary

In this project, downscaled climate models were selected from an ensemble of models using an uncertainty-based selection approach. The selected model projections for daily temperature and precipitation were input into DRAINMOD to evaluate the long-term impacts of climate change on two forested wetland ecosystems in the coastal plain of North Carolina. The results indicated that water table declines are likely for most climate scenarios by the middle of this century (2040-2069) and larger declines are projected by the end of the century (2070-2099). The only scenarios in which tables did not decline would be due to a large increase in mean

annual rainfall or a combination of limited warming and an increase in precipitation. Projected water level declines were greatest during the growing season, although the extreme warming scenarios indicated water table declined throughout the year.

This modeling effort was not meant to make exact predictions or directly quantify the possible impacts, but rather to provide a range of possible future responses to climate change in North Carolina's Coastal Plain wetlands to inform future management strategies. For both sites, if mean warming can be limited to the lower end of the RCP4.5 scenario (less than 1.5 °C by 2100) and mean precipitation does not decrease, or mean precipitation increased significantly coupled with moderate warming (<2 °C), the impacts may be limited, because the modeling results show minimal drawdowns for these scenarios. However, this scenario is probably unlikely as recent estimates put the chance of less than 2 °C in warming by 2100 at 5% (and less than 1.5 °C warming at 1%) given the current trajectory and international emission reduction commitments, and the median increase is projected to be closer to 3.2 °C (Raftery et al., 2017). While the modeling approach has its limitations, specifically with regard to PET estimates, and there is a great deal of uncertainty associated with any climate change impact assessment process, the results show that rising temperatures and changing precipitation regimes could severely impact the hydrologic regimes of some Coastal Plain wetlands.

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Chapter 6: Summary of Conclusions and Recommendations

Conclusions

This dissertation primary focused on assessing the condition of natural wetlands in North Carolina, improving our understanding of wetland processes, developing information to improve wetland management, and assessing the long-term trends of North Carolina wetlands given the increasing pressures posed by climate change. This project intended to add three years of additional data to an existing long-term dataset with the goal of assessing trends in wetland condition and identifying possible stressors. However, during the early stages of this project, analysis revealed that much of the previously collected hydrology data, and some of the water quality and soils data were not of sufficient quality or quantity to enable in-depth analyses to answer these types of questions. Therefore, the project relied more heavily on the hydrology, soils, and water quality data collected over the last three years. Despite some of the shortcomings with the previously collected data, the improved and expanded data collection methods used in this study (stricter water quality sampling protocols, more intensive soil sampling, and additional deeper water table monitoring wells) generated a high-quality dataset and produced some important information on wetlands in North Carolina including:

1. Completed general characterizations of wetland hydropatterns of different wetland types and compared observations to the new standards for wetland mitigation projects (NCIRT recommendations).
2. Compared the water quality of natural wetland to other aquatic systems and defined draft nutrient thresholds for reference conditions in North Carolina wetlands.
3. Characterized the background metals concentrations in natural wetland soils and assessed the risk posed to wetland biota.

4. Used DRAINMOD and downscaled climate model projections to model the long-term impacts of climate change on the hydrologic regimes of wetlands in North Carolina. This provided a range of possible outcomes given the changes in long-term mean temperature and precipitation.
5. Provided suggestions and observations to improve large-scale wetland research studies in the future (second half of this chapter).

The Hydrology of Natural Wetlands in North Carolina

The objectives of Chapter 2 included the general characterization of wetland hydropatterns and the comparison of observed saturation durations to the standards for wetland mitigation projects. The original plan for this portion of the study was to include 5-7 years of legacy hydrology data collected previously by NC DEQ. For various reasons, this data ultimately could not be used in this analysis. The data collected during this study (30 months) was high quality; however, this period was too short to definitively evaluate some of the hypotheses and trends of wetland hydrology that were intended at the beginning of this study. Nevertheless, there were several important observations that might help us further our understanding of hydropatterns of natural wetlands, which can also be used to improve wetland restoration projects.

1) The variability in the water level between the two years at many of the sites clearly illustrated the need for long-term water level observations to characterize average conditions and ranges.
2) While the riverine wetlands generally had higher maximum water levels due to overbank flooding events, the summary statistics did not differentiate the wetland by type.

However, this is not necessarily surprising given the large geographic range, variability in wetlands size, different ecoregions, and varying levels of anthropogenic disturbance.

3) The impacts of anthropogenic disturbance were evident on the hydrologic regimes of some of these wetlands. Some disturbances likely caused wetter conditions while others probably contributed to lower water levels.
4) Although the observed saturation durations typically far exceeded the recommended ranges from NCIRT, revised ranges were not proposed because of the short sampling period, small sample size and different wetland types.
5) The analysis of long-term simulations (from Chapter 5) revealed that there was a large range of hydropatterns and hydroperiods over the 30-year analysis period. The possibility of defining narrow ranges as targets for restoration projects appeared problematic based on this data.
6) What is needed is more long-term hydrology data or hydrologic models to predict long-term hydrology for a set of natural reference wetlands. Long-term data or model output could then be used to compare against current hydrologic criteria and restoration projects to develop new guidelines based on natural reference systems.

The Background Water Quality in North Carolina Wetlands

The primary objectives of Chapter 3 were to compare wetland water quality to background levels for streams and develop nutrient thresholds based on reference wetlands that could be used to identify anthropogenic disturbance. The results are a first cut at providing some numerical value to support US EPA and NC DEQ narrative criteria for wetland water quality. It was one of the first longer-term studies to include multiple wetland sites. This study improved sampling protocols compared to previous studies by NC DEQ in North Carolina. Some of the

hypotheses remained inconclusive because the combination of different wetland types and ecoregions and the still somewhat limited sampling frequency. Despite these limitations, several important conclusions were reached.

1) Comparisons revealed that background levels for wetlands should likely be higher than the background levels for streams for TN, TP and NH_4^+-N, but should likely be set lower than streams for NO_3^--N.
2) This research indicates that reference levels for inorganic nitrogen may be applicable across ecoregions and wetland types. Wetland and ecoregion specific reference levels should be further investigated for TN, TP, and ON.
3) Reference thresholds of 0.07 mg/L NO_3^--N, 0.15 mg/L NH_4^+-N, 0.3 mg/L TP, 2.9 mg/L ON, and 3.2 mg/L TN were calculated using a modified US EPA approach for determining reference conditions.
4) Thresholds for TN and ON are not informative and therefore not needed, given they did not appear to be robust indicators of disturbance. Inorganic nitrogen, particularly nitrate, is a better indicator of anthropogenic disturbance.
5) It should be noted that this is only a first attempt at the process of developing reference thresholds for wetlands and there is a need for future validation and improvement. However, the similarity between the results and the upper limit of background (irreducible) concentrations in constructed wetlands from US EPA (2000) was promising.

Metals in North Carolina Wetland Soils

The primary focus of Chapter 4 was an assessment of the Cu and Zn levels in the upper 30 cm of the soil at the 16 natural wetland sites. These findings provided insight on the background levels of Cu and Zn in natural wetland soils and displayed how land use and

anthropogenic disturbance can increase concentrations. To build upon previous studies, the sites had been pre-selected by North Carolina Department of Environmental Quality (NC DEQ) from previous NC DEQ/US EPA projects. This decision limited the replication in soil types, wetland types, and land use within ecoregions, which restricted the data analysis for this chapter. However, there was still some important information that was discovered.

1) Overall, Mehlich 3 extractable Cu and Zn values were very low across most of the sites. Elevated metal concentrations were largely explained by inputs from adjacent land use or upstream development. Elevated metal levels will likely not be encountered unless there is a direct anthropogenic source of metals.
2) Based on the results from this study and previous wetland studies, mean background level for Mehlich 3 Cu and Zn should be <1.0 mg/kg and <5.0 mg/kg, respectively for wetlands in North Carolina. Thresholds that might indicate disturbance could be established at 2.0 mg/kg Cu and 7.0 mg/kg Zn for site means and 4.0 mg/kg Cu and 14.0 mg/kg Zn for site maximum values.
3) In terms of contamination thresholds, plant toxicity thresholds (from agriculture recommendations) were rarely exceeded. Ecological soil screening levels (Eco-SSLs) developed by the US EPA for evaluating contaminated soils were exceeded with greater frequency.
4) Some of the Eco-SSLs were clearly overly conservative as they were below average background concentrations.
5) Total and Mehlich 3 metal results were significantly correlated, but an accurate predictive relationship to estimate totals metals from the Mehlich 3 values was not obtained.

6) Based on the very low bioavailable Zn and Cu levels in the soils of most of these wetlands, it is unlikely that the metal levels in a majority of wetlands in the state pose any risk to the wetland biota.

The Impacts of Climate Change on NC Wetlands

In Chapter 5, downscaled climate models were selected from an ensemble of models using an uncertainty-based selection approach. The selected model projections for daily temperature and precipitation were input into a calibrated and validated DRAINMOD model to evaluate the long-term impacts of climate change on two forested wetland ecosystems in the Coastal Plain of North Carolina. The results indicated that water table declines are likely for most climate scenarios by the middle of this century (2040-2069) and larger declines are projected by the end of the century (2070-2099). While the modeling approach had some limitations, the results show that rising temperatures and changing precipitation regimes could severely impact the hydrologic regimes of North Carolina of wetlands.

1) Simulated water table results for the base modeled scenario (DRAINMOD simulations with climate model projected precipitation and temperature for the base period (1986-2015)) replicated base observed scenario (DRAINMOD simulations for observed precipitation and temperature from base period) well, as indicated by high NSE values. This indicated that the climate model data was a reasonable representation of historical climate and provided some confidence in the changes predicted by future scenarios.
2) The mean annual water level was projected to decline by 0-64 cm by the middle of this century (2040-2069) and 25-84 cm by the end of the century (2070-2099). Although there is a wide range of projected drawdowns, depending on the changes in precipitation and temperature. The only scenarios in which water tables did not decline were due to a large

projected increase in mean annual rainfall or a combination of limited warming and increased precipitation.

3) Projected water level declines were greatest during the growing season, although for the extreme warming scenarios water table generally declined throughout the year.
4) These results indicate that for wetter sites, while water levels decline, jurisdictional hydrology may still be met except for the extreme scenarios. However, for the drier borderline sites, wetland jurisdictional criteria may not be met starting at or before the middle of this century.
5) For both sites, if mean warming can be limited to the lower end of the RCP4.5 scenario (less than 1.5 °C by 2100) and mean precipitation does not decrease, the impacts on hydrology may be limited.
6) This modeling effort was not meant to make exact predictions or directly quantify the possible impacts, but rather to provide a range of possible future responses to climate change in North Carolina's Coastal Plain wetlands to inform future management strategies.
7) Climate change may result in shifts in vegetative species composition and oxidation of organic soils in organic soil flat wetlands.

Recommendations for Future Large-Scale Wetland Studies

Overall Recommendations

While this project had several broad goals related to making advances in the field of wetland science and management, the problems with the previously collected long-term data combined with the observational nature of the study design (different wetland types, large geographic range, generalized data collection methods, and infrequent data collection) prevented

many of these goals from being fully realized. This section provides recommendations to improve future large-scale wetland research studies and some key factors funding agencies should consider when evaluating proposals in order to maximize the benefits of investments in state wetland monitoring and research programs.

Defining the Study Type

There are essentially two types of large-scale wetland studies: observational and experimental. Observational studies have a broad scope and limited data collection frequency. The types of questions that can potentially be answered regard general characterizations of wetland ecosystems and large-scale observations. Experimental type studies seek to address deeper, more complex questions at a more definitive level. For experimental studies, a more focused study design with more rigorous data collection is required. While both types of studies are valuable, it is very important to distinguish the type of study and set appropriate corresponding objectives because the type of study that is undertaken determines the nature of questions that can be addressed.

This project was inherited from NC DEQ with the intention of taking an observational study, expanding the monitoring intensity, and combining it with previously collected data from past U.S. EPA approved NC DEQ projects to answer some more in-depth, research type questions. However, the poor quality and quantity of the previously collected data limited the depth at which these more detailed questions could be addressed.

Funding agencies should be acutely aware of the difference in the types of studies and scrutinize proposed study design and objectives accordingly. Funding should be prioritized for projects where the study design is targeted to best answer the specific objectives of the project. For example, if the goal is to make a general characterizations and observations then a study that

covers a large geographic area with a limited number of sites may be an acceptable approach. However, if the objective is more detailed, such as to develop metrics to evaluate wetland condition, then a more detailed experimental design would be required. For example, an experimental type study to assess the impacts of anthropogenic stressors might include several components including: similar wetland types, limited geographic range, sites distributed across a gradient of disturbance in different land uses, and more intensive, higher frequency data collection. Large-scale experimental type studies are what is needed to advance the field of wetland science.

Data Management

Another crucial aspect of any project is data management. Data should be collected and documented under the assumption that personnel unfamiliar with the project may need to understand, utilize, and validate the data in the future. During the current data collection period, strict data collection protocols were adhered to and complete, accurate datasets were the result of the project. This included backing up data in multiple locations, storing hardcopies and scans of field sheets, keeping logs for site visits, photos, and recording the GPS coordinates of sample plots. All project files including the project documentation and data were stored in a structured database and backed-up continuously to online servers. Equally stringent protocols should be developed and followed in all large-scale projects. Failing to follow a strict data management plan can result in lost data or data that cannot be used due to lack of documentation. For this project, all the personnel collecting the data were familiar with the operation of the monitoring equipment, aware of how to recognize potential problems, and troubleshoot issues. Data management is a key component of any successful project. Data management and documentation issues may have been one of the problems with the previously collected data.

Hydrology

The quality of the previously collected hydrology data was a problem encountered during this project. Considering the following factors will increase the likelihood of quality data collection. Monitoring wells should be installed based on project objectives and site-specific conditions. The number of wells installed will likely be wetland type specific. While one or two wells may be enough to capture the hydrologic regimes at smaller sites, more wells may be needed at larger or more complex sites. The installation of more wells allows for some error in well placement and a better identification of the true range of conditions at a site. Well depth should also be considered, 30-60 cm well depth that is commonly used to monitor restoration projects is not appropriate for charactering the range of hydropatterns in most wetlands. For this project, 150 cm wells were installed, and wells could have even been deeper at some riverine sites. The previous collection methods used 40-60 cm deep wells, which was not adequate to capture the entire hydropattern during the growing season.

Many wetlands are underlain by fine-grained soils. As a result, sediment can accumulate in the bottom of monitoring wells and cause inaccurate measurements. During each download, the well should be checked for sediment interference with the sensor and cleaned if necessary (as was done during the current study). This was not a common practice during the previously conducted monitoring and may have contributed to some of the observed errors.

There are two basic forms of pressure transducers for water level measurement: sealed and vented to the atmosphere. The sealed version requires that two sensors be used: one for water pressure in the well and then another for atmospheric pressure. The pressure difference between the two is used to determine water depth. The vented type has a tube open to the atmosphere that allows for the direct calculation of water depth. For the vented type, the open

end of vent tube must be installed above the high-water mark in wetlands to ensure it is not submerged. If the vent tube is submerged, the sensor no longer provides accurate measurements. For riverine wetlands, where water level can commonly reach two meters above the floodplain surface, sealed type pressure transducers should be used to account for these extreme events. In addition, if the data download interval is more than 2-3 months, sealed pressure transducers should be used to avoid problems associated with saturated desiccant for the vented type pressure transducers.

Another important recommendation for hydrologic studies in wetlands is the standardization of terms used to describe different aspects of wetland hydrology. Currently different sources use the term hydroperiod differently. Mitsch and Gosselink (2007) and Reddy and DeLaune (2008) use the term hydroperiod to refer to the seasonal pattern of water level fluctuations or as the "hydrologic signature of the wetland". However, other sources such as Caldwell et al. (2011) and others in the wetland ecology field (e.g. Townsend, 2001) refer to hydroperiod as the total duration of ponding on the surface for a single event, for the entire year, or for a given season.

US EPA (2008) uses the term hydropattern to refer to the pattern of water level fluctuations through time (i.e. "the hydrologic signature of the wetland"). U.S. EPA refers to hydroperiod as the time and duration the soil is saturated to a specified depth and duration. It is recommended that the U.S. EPA terminology be used for descriptions of wetland hydrology.

Water quality

Because wetlands are very heterogeneous systems, water quality sampling methods likely need to be wetland type and project objective specific. These protocols should be established and documented prior to beginning projects. The protocols should include the conditions under

which samples are collected and the methods used. Table 6.1 includes factors that should be taken into account when developing protocols. Funding agencies should scrutinize wetland water quality monitoring projects to ensure that they include strong data collection protocols and insist on funding projects that include data collection at a sufficient frequency to develop a large enough dataset to allow for robust statistical analysis, if in-depth analysis is a project objective.

Table 6.1: Factors that should be considered when developing protocols for wetland water quality sampling projects.

Factor	Description
Sampling Frequency	Should be determined based on project objectives. Because of the variability in wetland hydrologic regimes, there can be long periods of time when there is no surface water present. Therefore, the number of samples can end up being much less than the number of site visits. Greater than quarterly sampling is needed for trend analysis or for characterizing the typical conditions. For some wetland studies event-based sampling may be appropriate. For example, for natural wetlands that are subject to stormwater inflows. Less frequent sampling is acceptable if the goal is simply a snapshot of current conditions. For trend analysis or characterization of typical conditions monthly sampling is recommended.
Water Depth	There needs to be a minimum depth established for sample collection. Collecting samples from shallow, isolated, turbid pools is probably not representative of typical conditions, and may cause anomalous results.
Sample Filtering	For wetland with high TSS waters, filtering samples at some level should be strongly considered in the sampling protocol. Samples with suspended particles may artificially elevate nutrient and metals levels if the particles make it into the small subsample taken for analysis in the laboratory. For metals, it is recommended that the samples be analyzed for both the dissolved and total form. If the budget allows, the same should be done for nutrients.
Rainfall Events	The sample collection protocol should specify whether samples should be collected during or immediately after rainfall events.
Flow vs. No Flow	For flow through wetlands, the protocols should specify under what flow conditions samples should be collected. Stagnant conditions may impact results
Wetland Type	Different wetland types have different hydrologic regimes and may need specific protocols including sample locations and the number of samples collected.
Metals Analysis	Surface water quality standards (not intended for wetlands) are expressed in dissolved form. Dissolved metals give an approximation of the bioavailable fraction. Total recoverable metals give an indication of the total metal content of the wetland. The dissolved to total metal fraction varies considerably so conversion factors should not be used if precise results are desired. It is recommended that the samples be analyzed for both the dissolved and total form.
Sample Location	Sample location should likely be wetland type specific and project objective specific. For this project an upstream – downstream approach was used. This approach is probably appropriate for wetlands where there is discernable flow through conditions. For depressional wetlands an areal composite sample may be the appropriate approach.

Soils

When conducting soil sampling in wetlands, particular attention needs to be given to sampling design and analysis methods, because this determines the types of questions that can be answered. For example, for the previous data collection, soil samples were collected from different depths at each site (by soil horizon). While this approach is appropriate for general characterizations and observations, a more uniform approach (same sample depths) may be more appropriate if the objective is to make comparisons between the sites. When analyzing soils data, it is very important to understand which analytical method was used by the lab because different metals extractions methods (e.g. Bray, Mehlich 3, EPA 3501) will yield different results. An analysis method needs to be chosen that addresses the objectives of the project. For example, if the goal is to assess the amount of metals stored in wetland soils, then a total recoverable metal analysis method should be used (e.g. EPA 3501). However, if the goal was to assess the amount of metals available for plant uptake, then a plant available extraction method should be used (e.g. Mehlich 3). In addition, when comparing results of different studies, only studies using comparable analytical methods should be used. For example, Mehlich 3 extractable metals should never be compared to results from the National Wetland Condition Assessment as they use a total recoverable analytical method. A future study might consider evaluating wetlands in areas that have sources of zinc and copper, such as developed, urbanized areas. In addition, samples could be analyzed for other heavy metals such as lead, which has lower toxicity thresholds. A future study might be set up to focus on wetlands of the same type and similar size and landscape setting. In addition, while some samples did exceed contamination risk thresholds for this study, it did not occur at every sampled location at a given the site. This indicates significant spatial variability. Future studies should be targeted to areas where pollutants are

most likely to aggregate across a gradient of disturbance; for example, near stormwater outfalls or near roadways.

Summary

While this project could have greatly benefited with more complete long-term data, more appropriate site selection, and more detailed sampling methods, there were still some important comparisons made and data presented. Future projects can avoid some of the problems encountered with the previously collected data by developing appropriate objectives that correspond to the type of project (observational versus experimental), following strict data management protocols, and specifying sampling and analysis methods to address specific project objectives. Some of the important finding of this project included: metals levels in natural wetlands were confirmed to be generally low absent some anthropogenic disturbance, thresholds were developed for nutrient concentrations in undisturbed wetlands, general characterizations of wetland hydropatterns were made, and long-term simulations were run to make estimates of the impacts of climate change on wetland hydrology.

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APPENDICES

Appendix A: Hydropattern plots and rainfall for each site

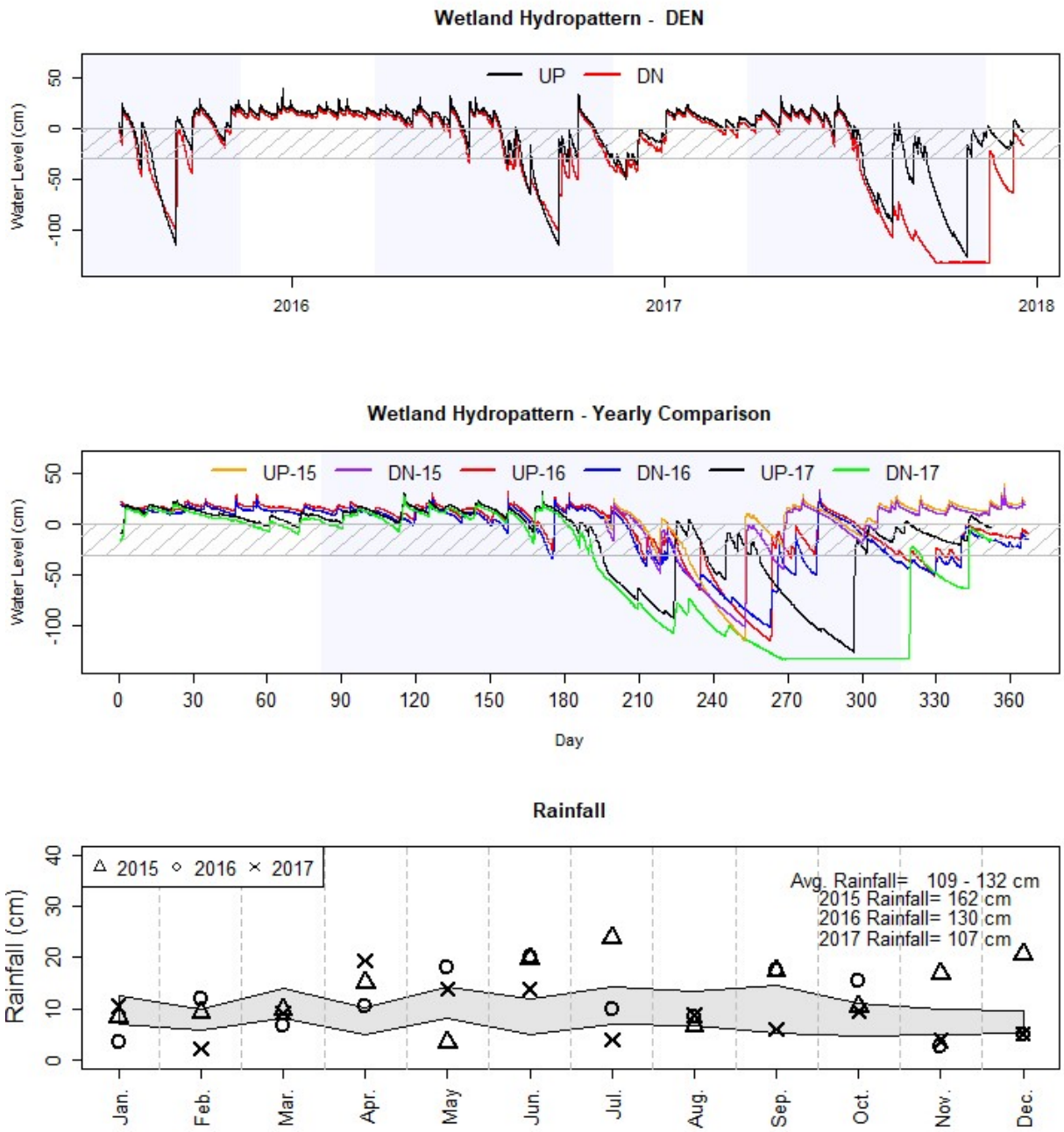


Figure A.1: Hydropattern and rainfall for DRG site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

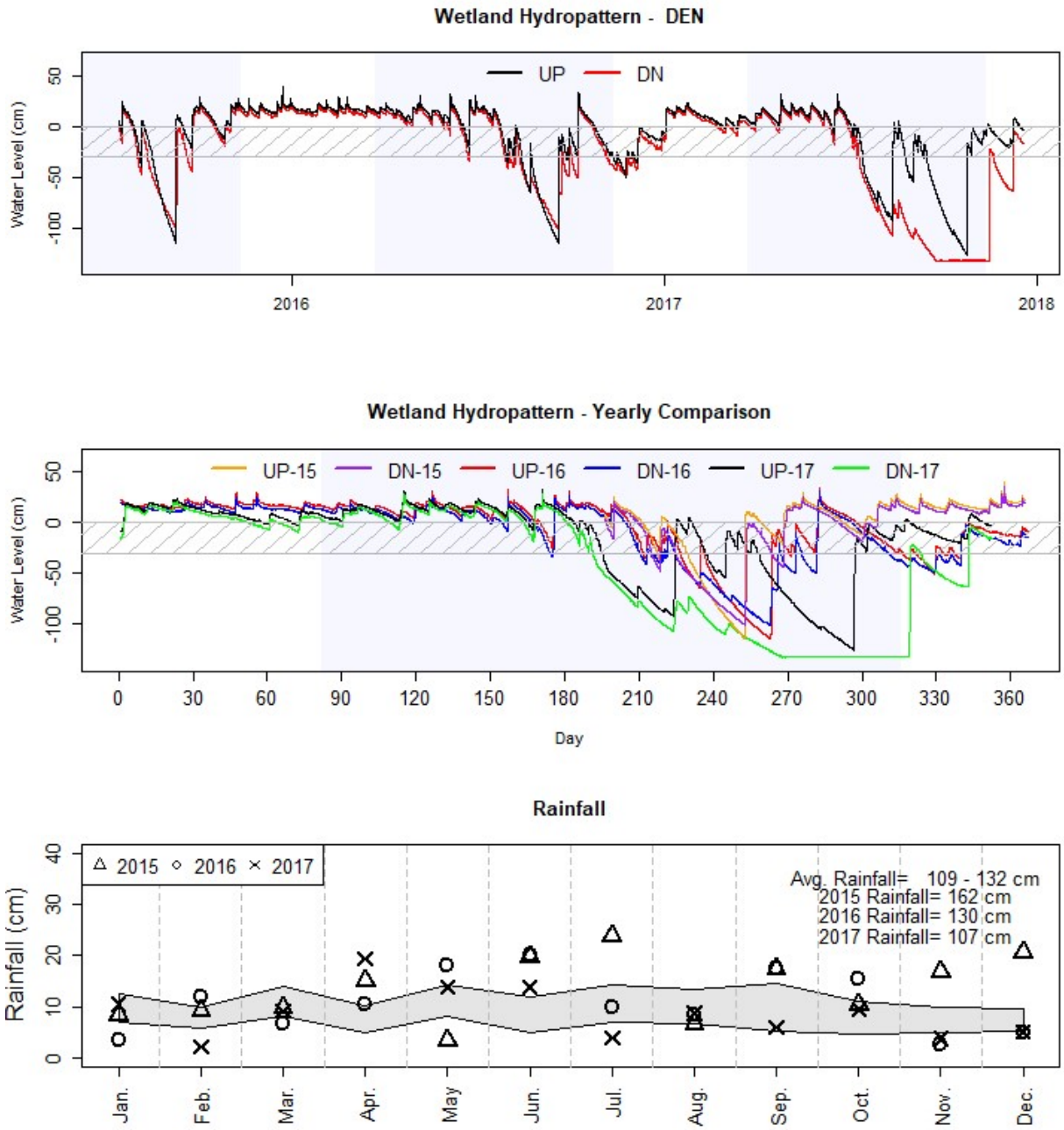


Figure A.2: Hydropattern and rainfall for DEN site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

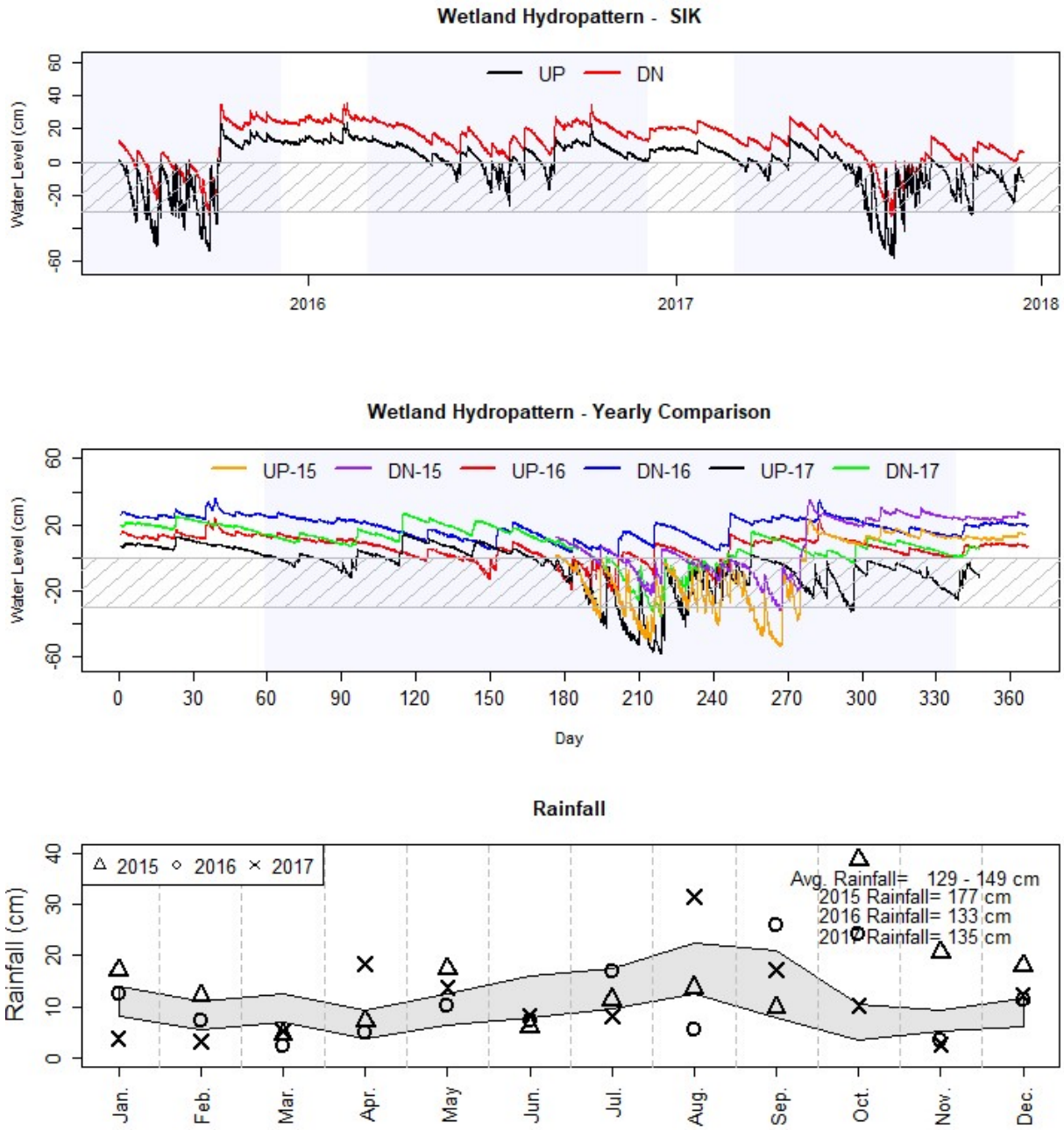


Figure A.3: Hydropattern and rainfall for SIK site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

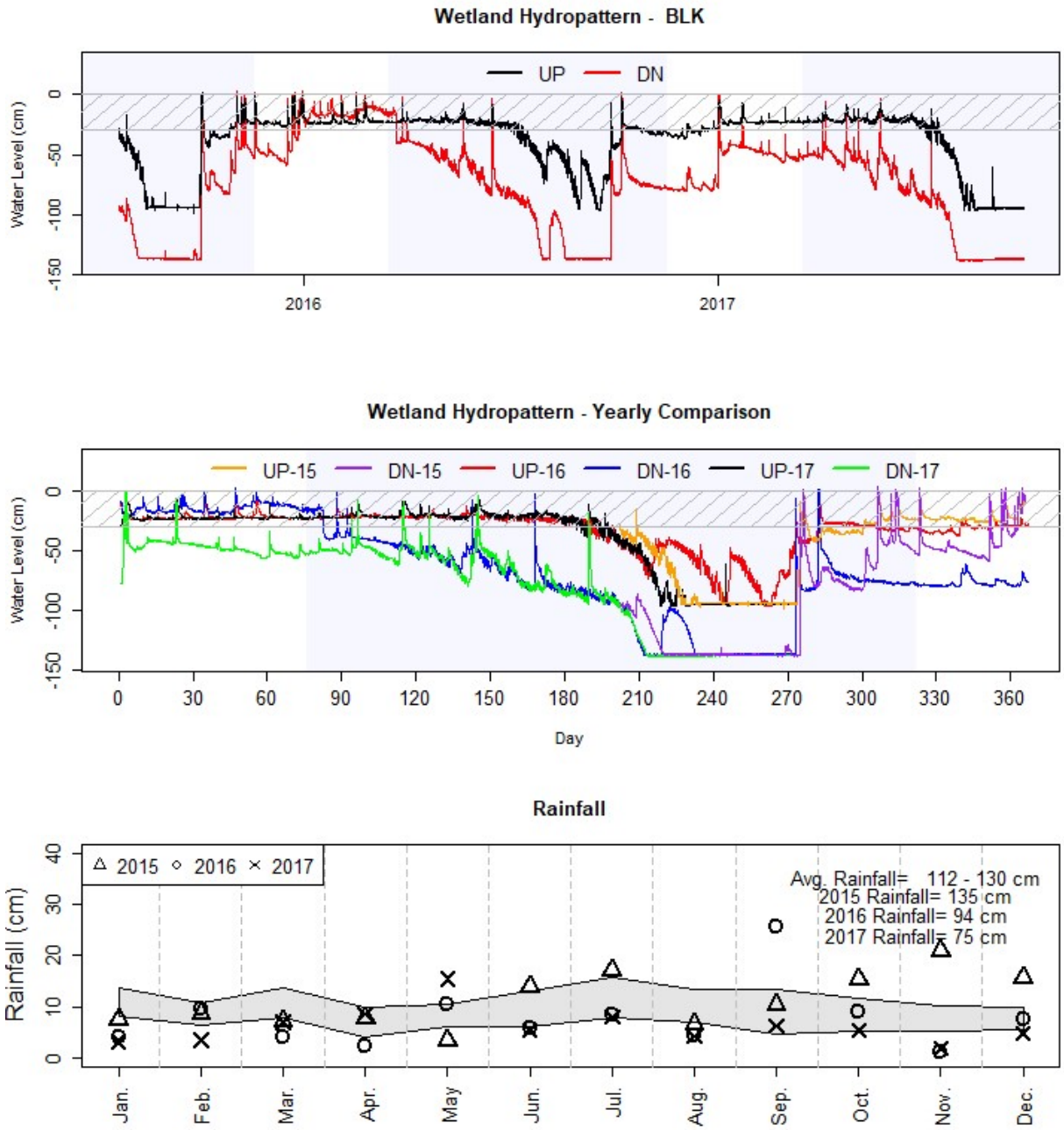


Figure A.4: Hydropattern and rainfall for BLK site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

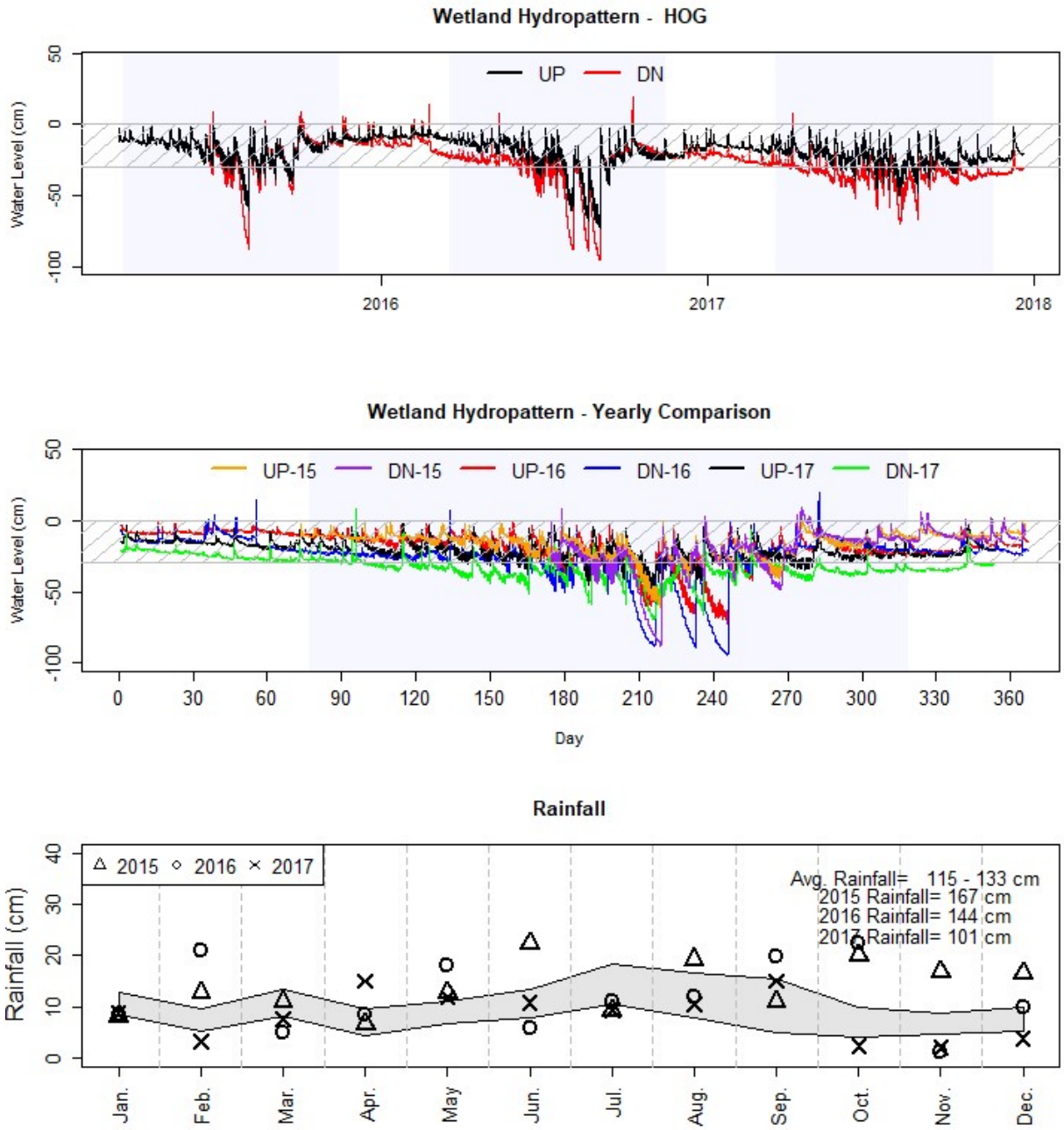


Figure A.5: Hydropattern and rainfall for HOG site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

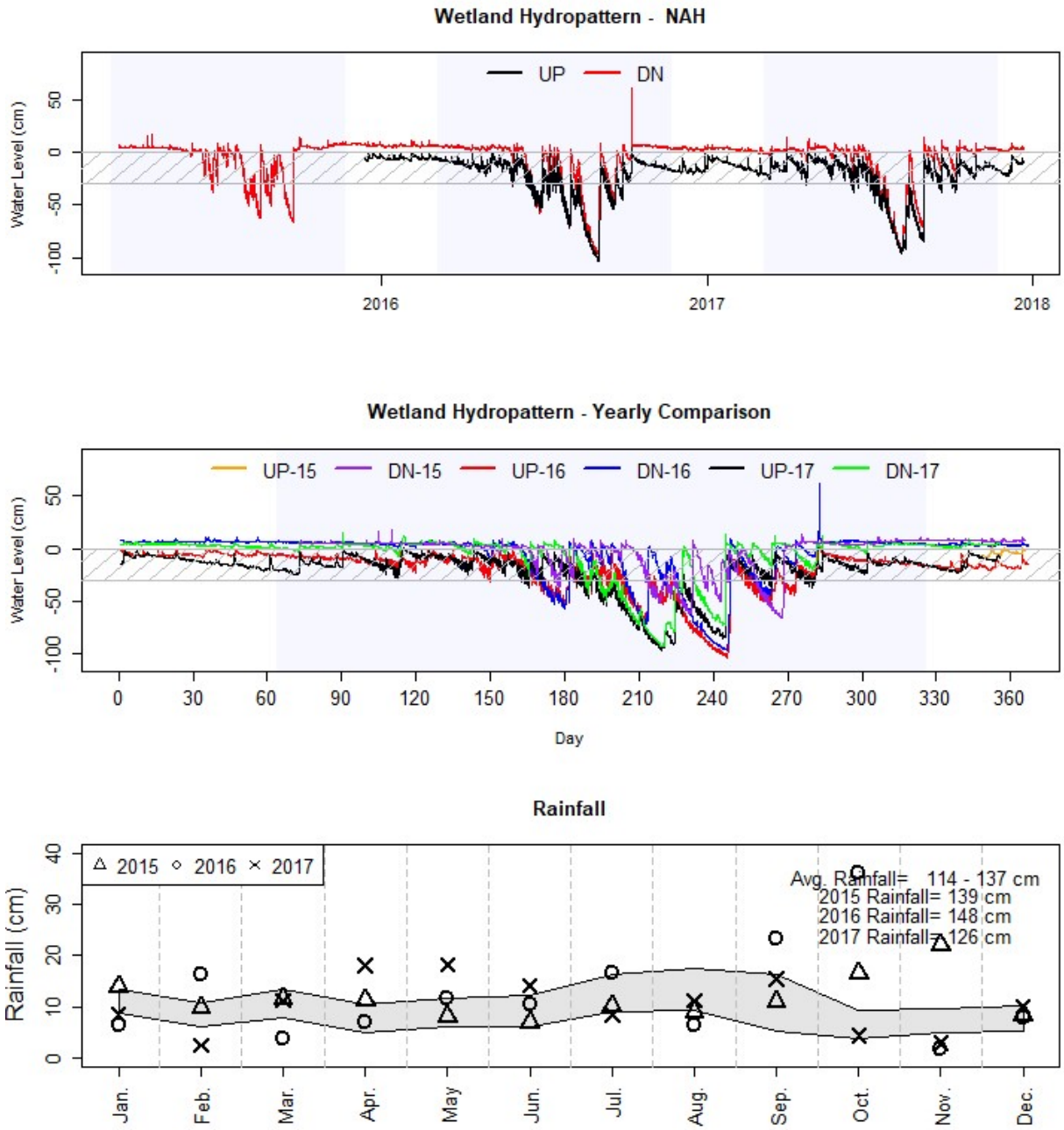


Figure A.6: Hydropattern and rainfall for NAH site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

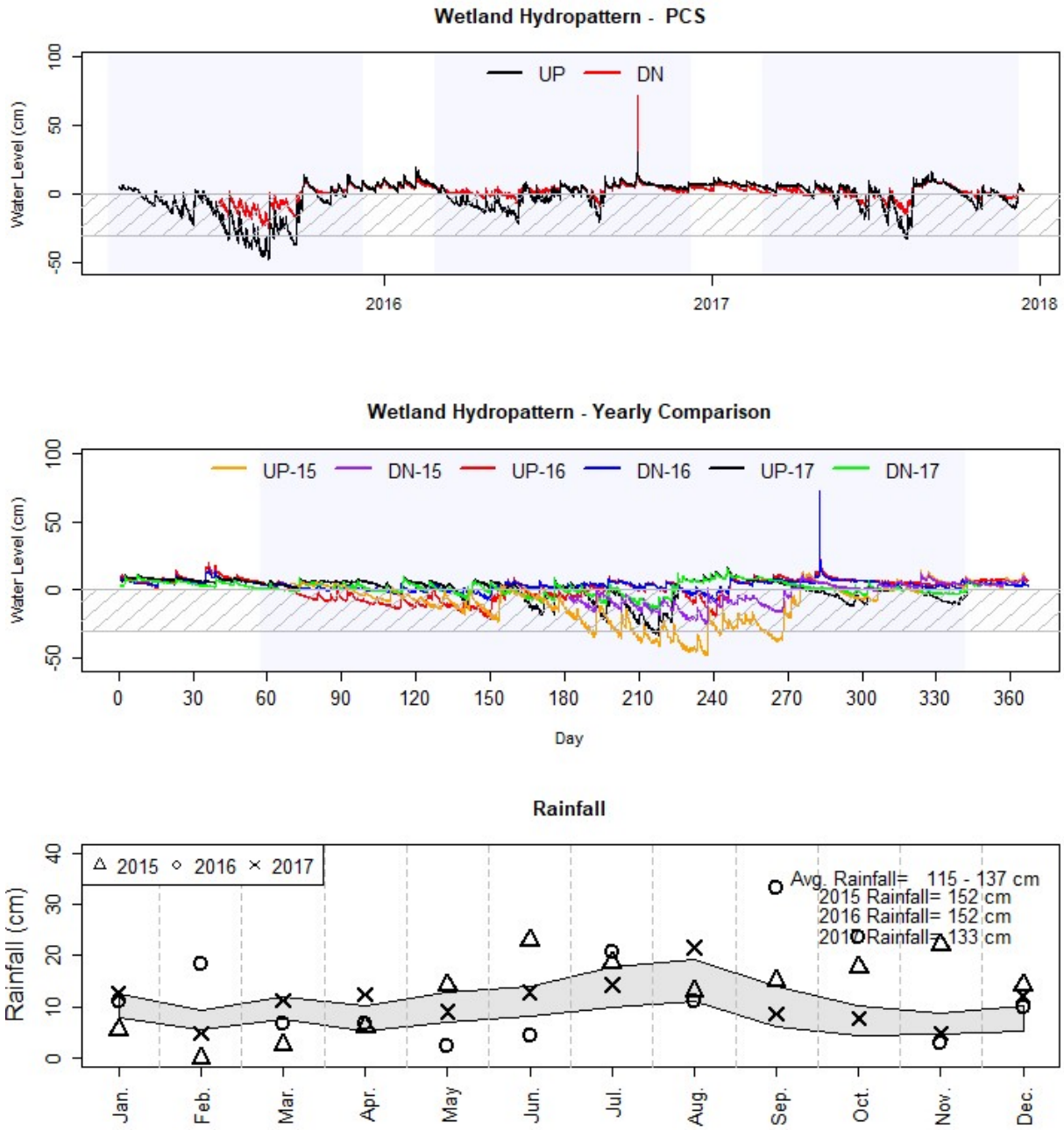


Figure A.7: Hydropattern and rainfall for PCS site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

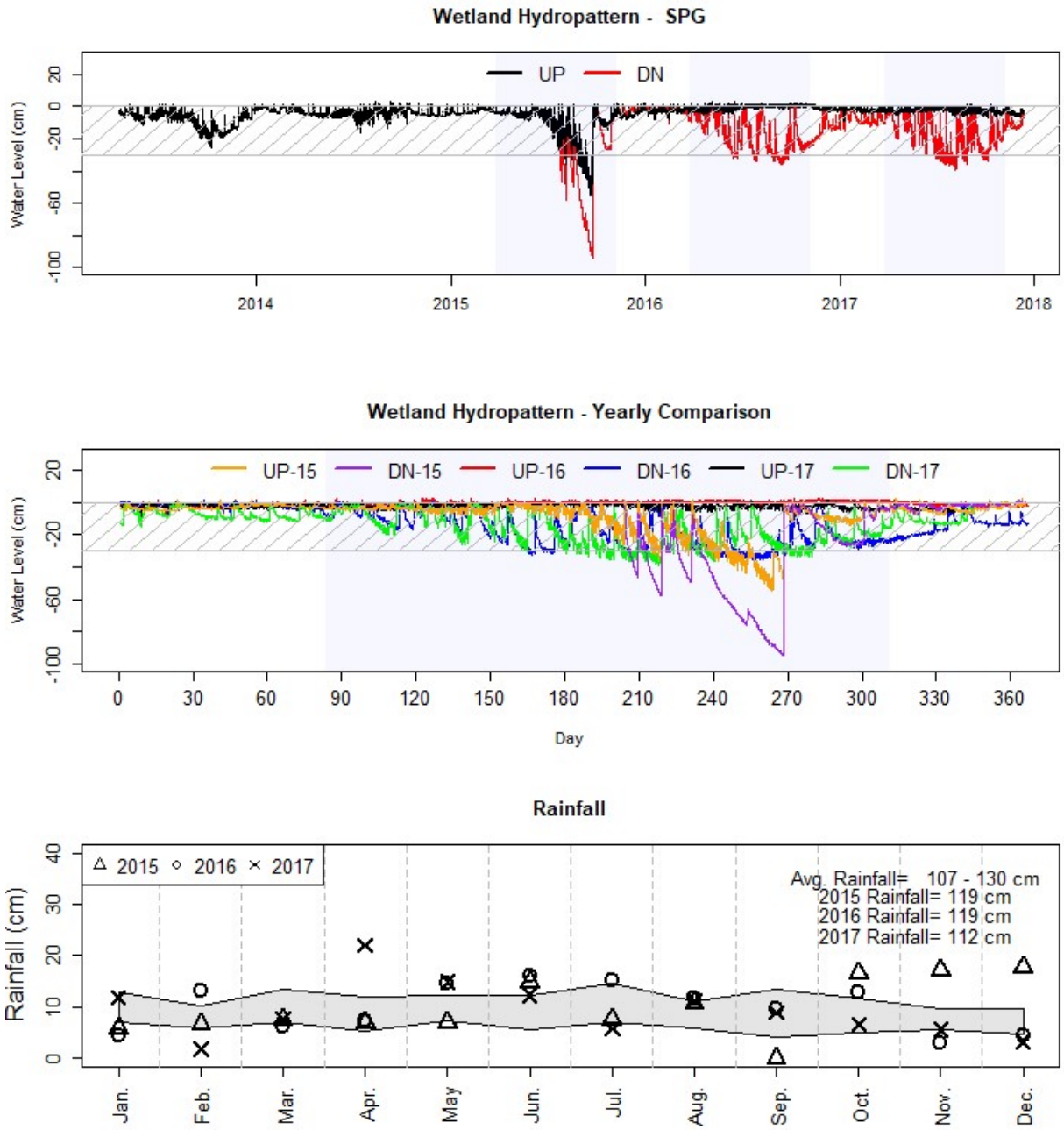


Figure A.8: Hydropattern and rainfall for PCS site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

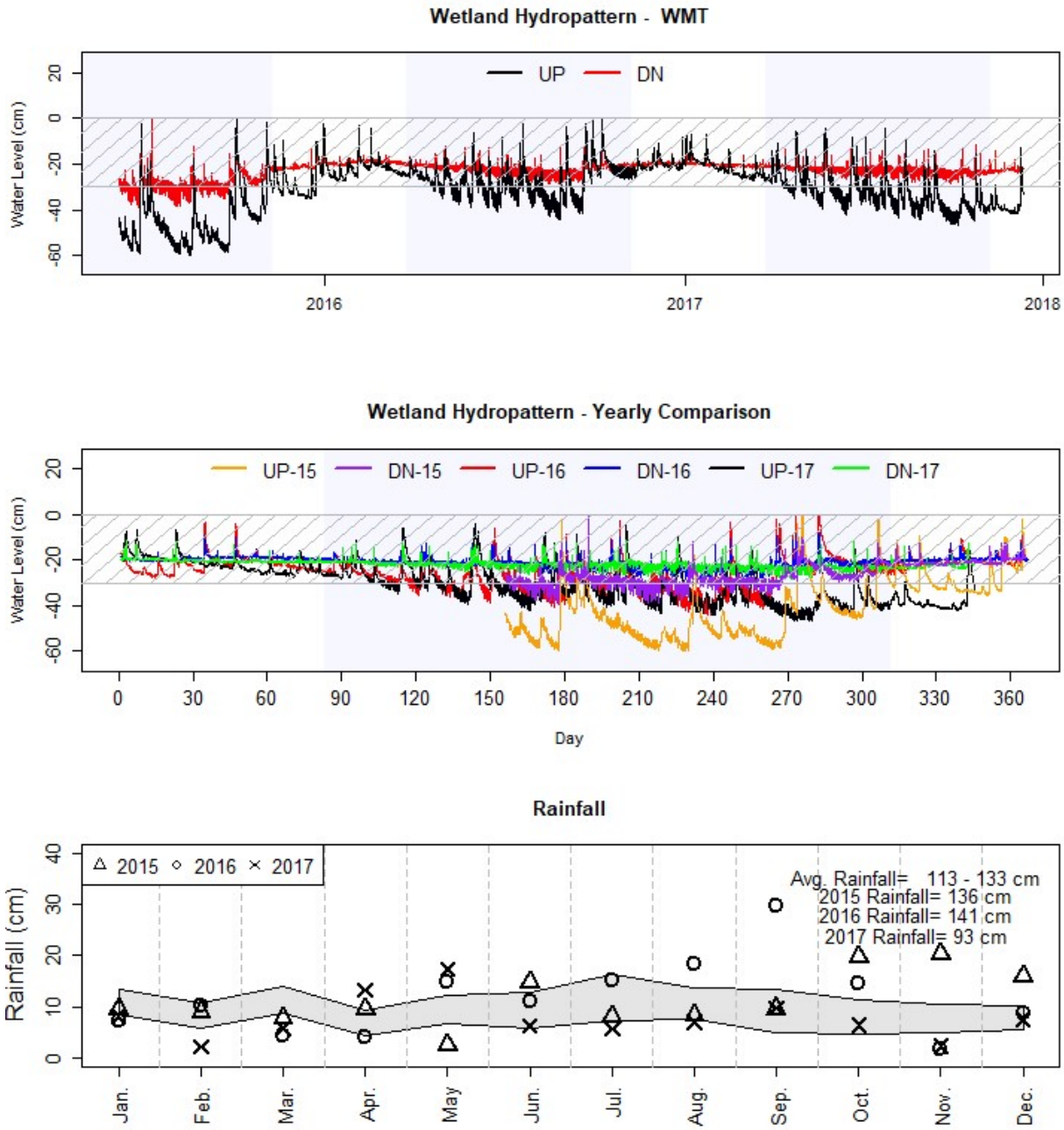


Figure A.9: Hydropattern and rainfall for WMT site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

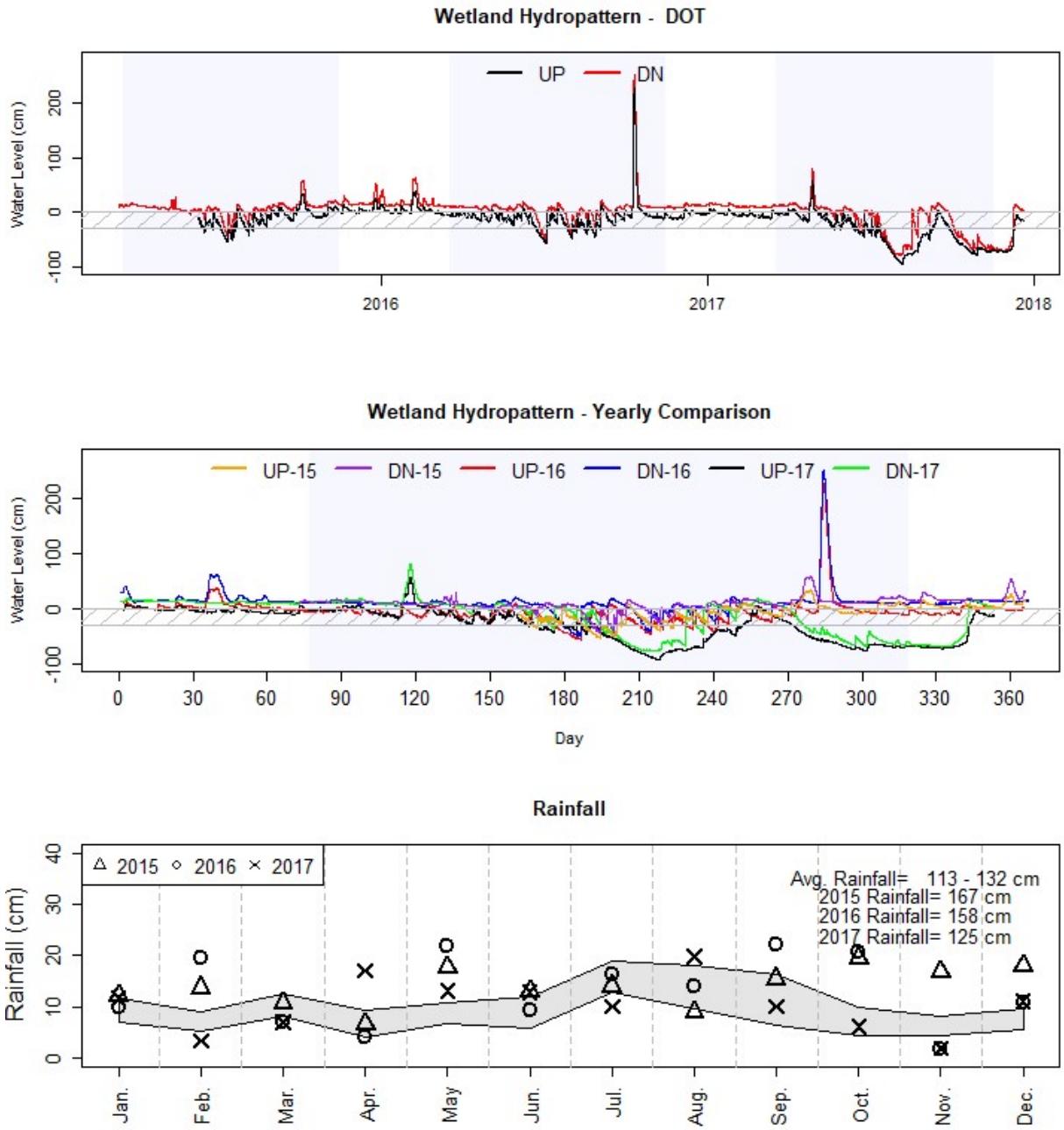


Figure A.10: Hydropattern and rainfall for DOT site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

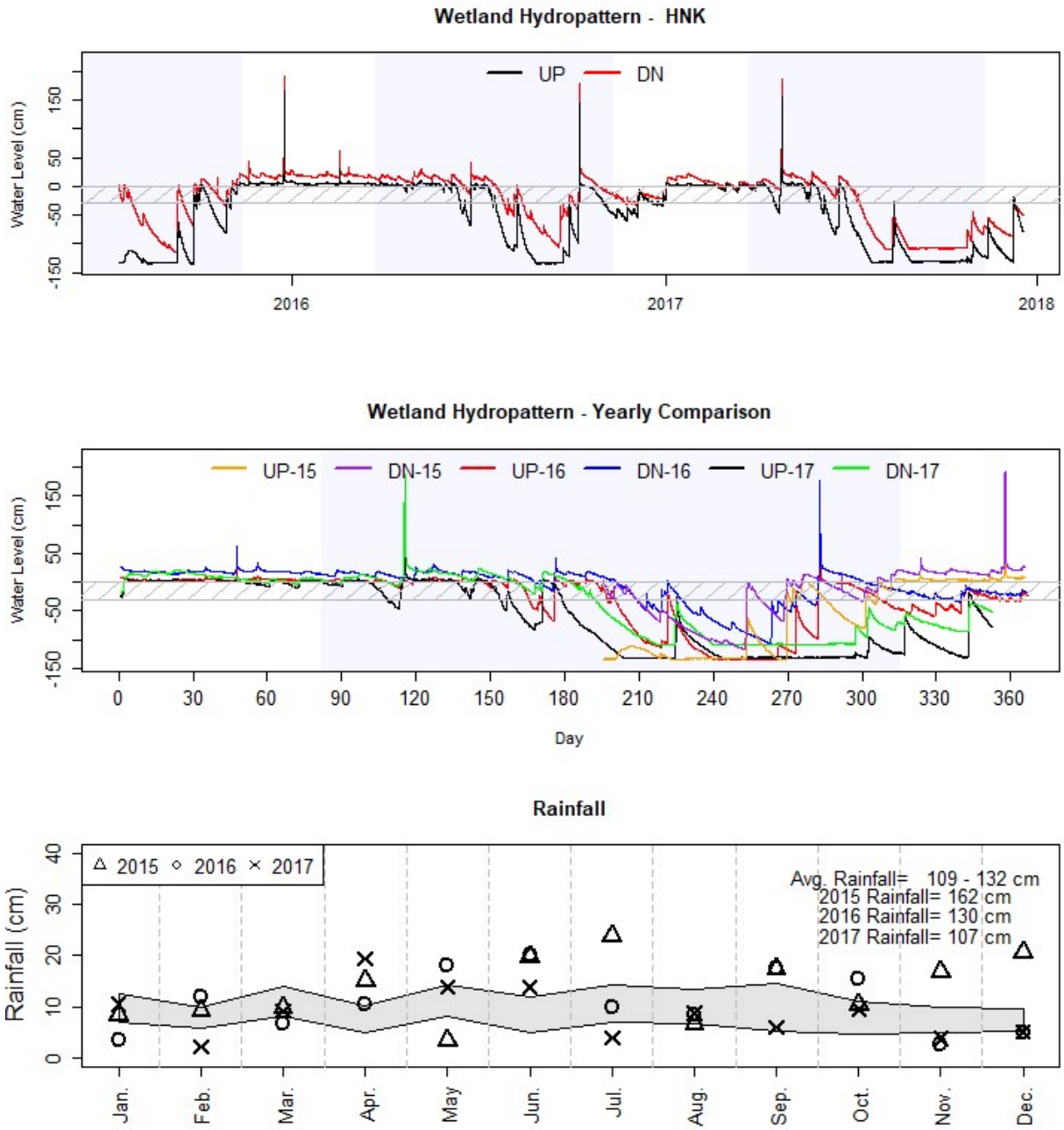


Figure A.11: Hydropattern and rainfall for HNK site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

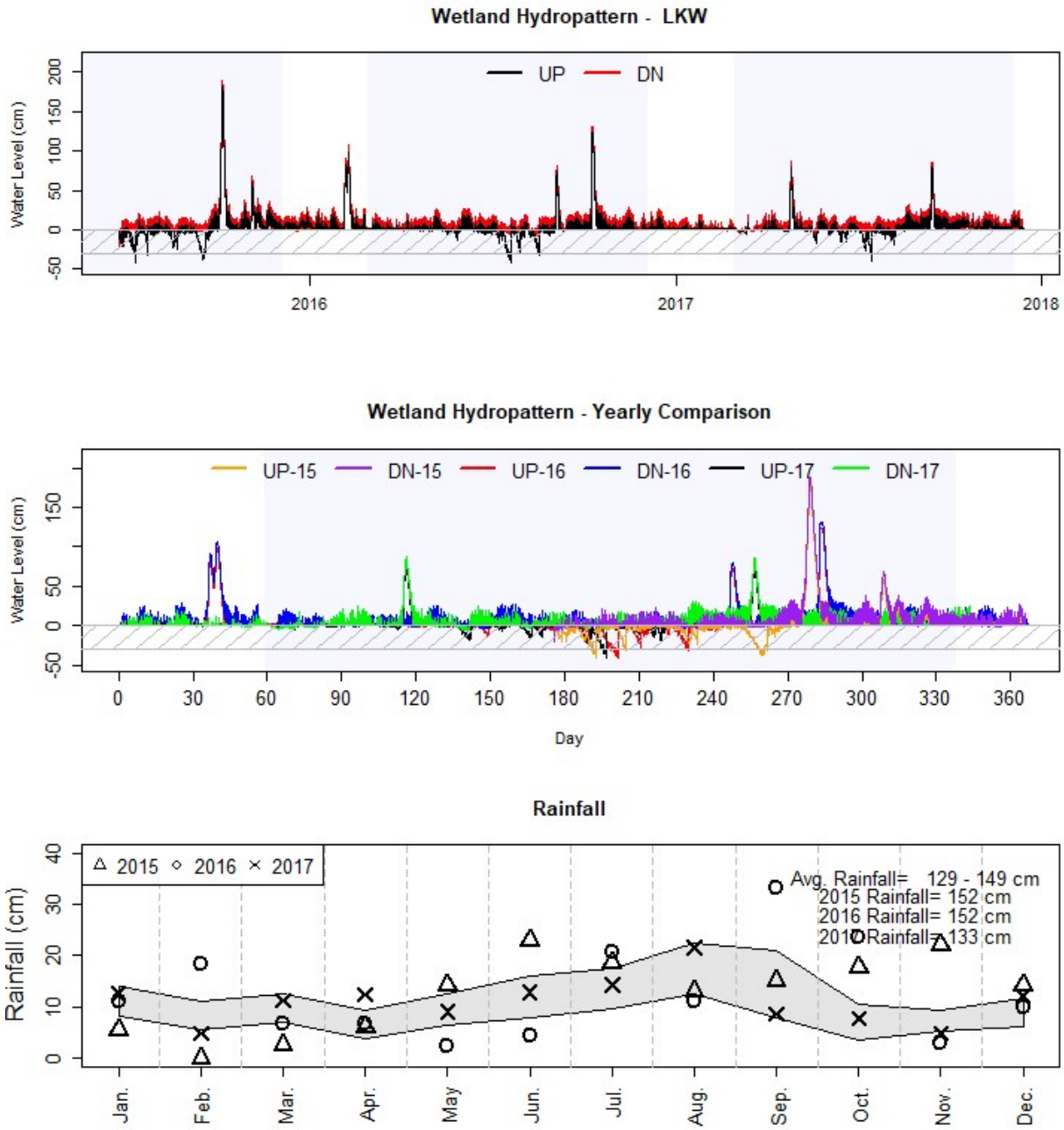


Figure A.12: Hydropattern and rainfall for LKW site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

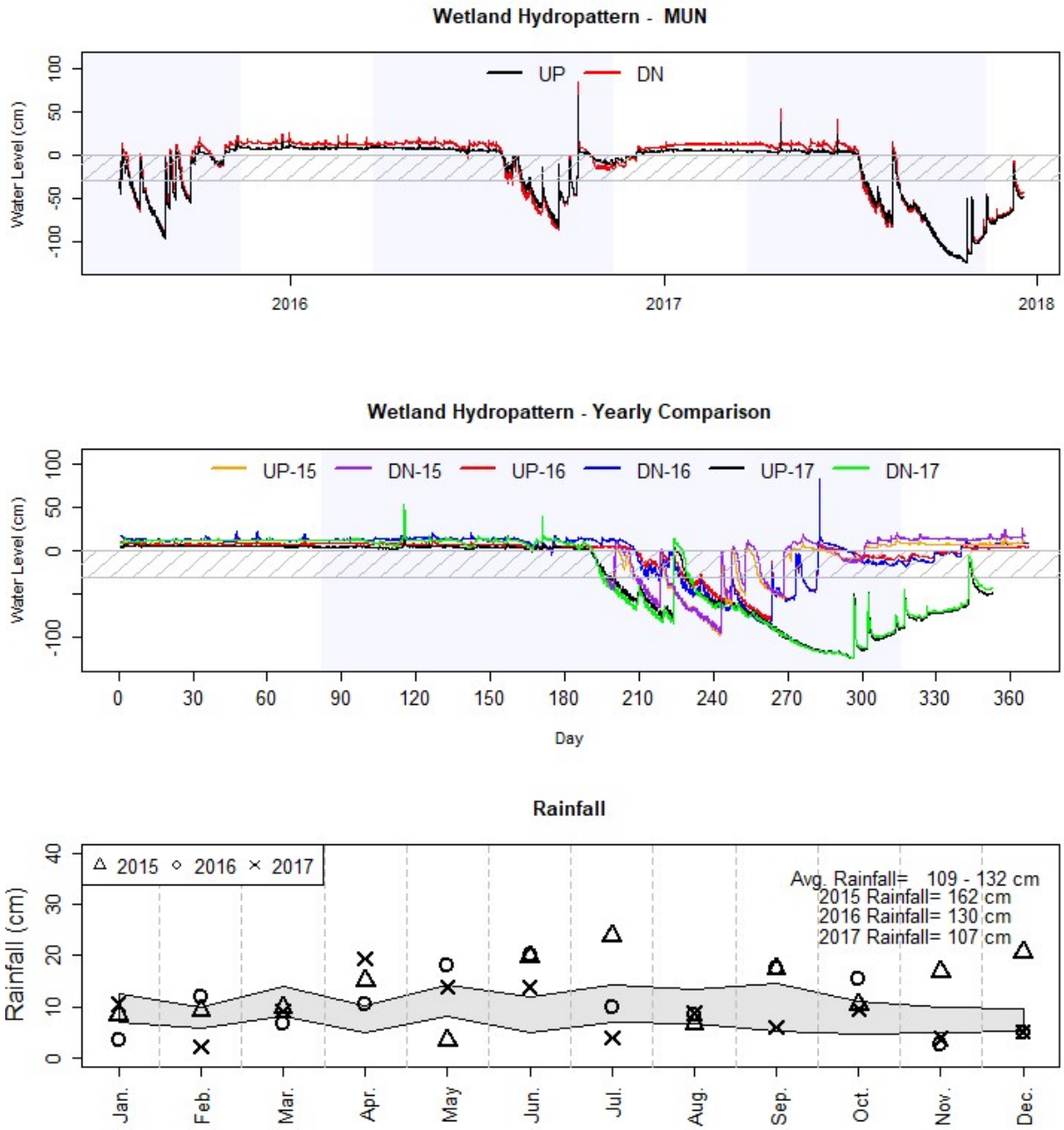


Figure A.13: Hydropattern and rainfall for MUN site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

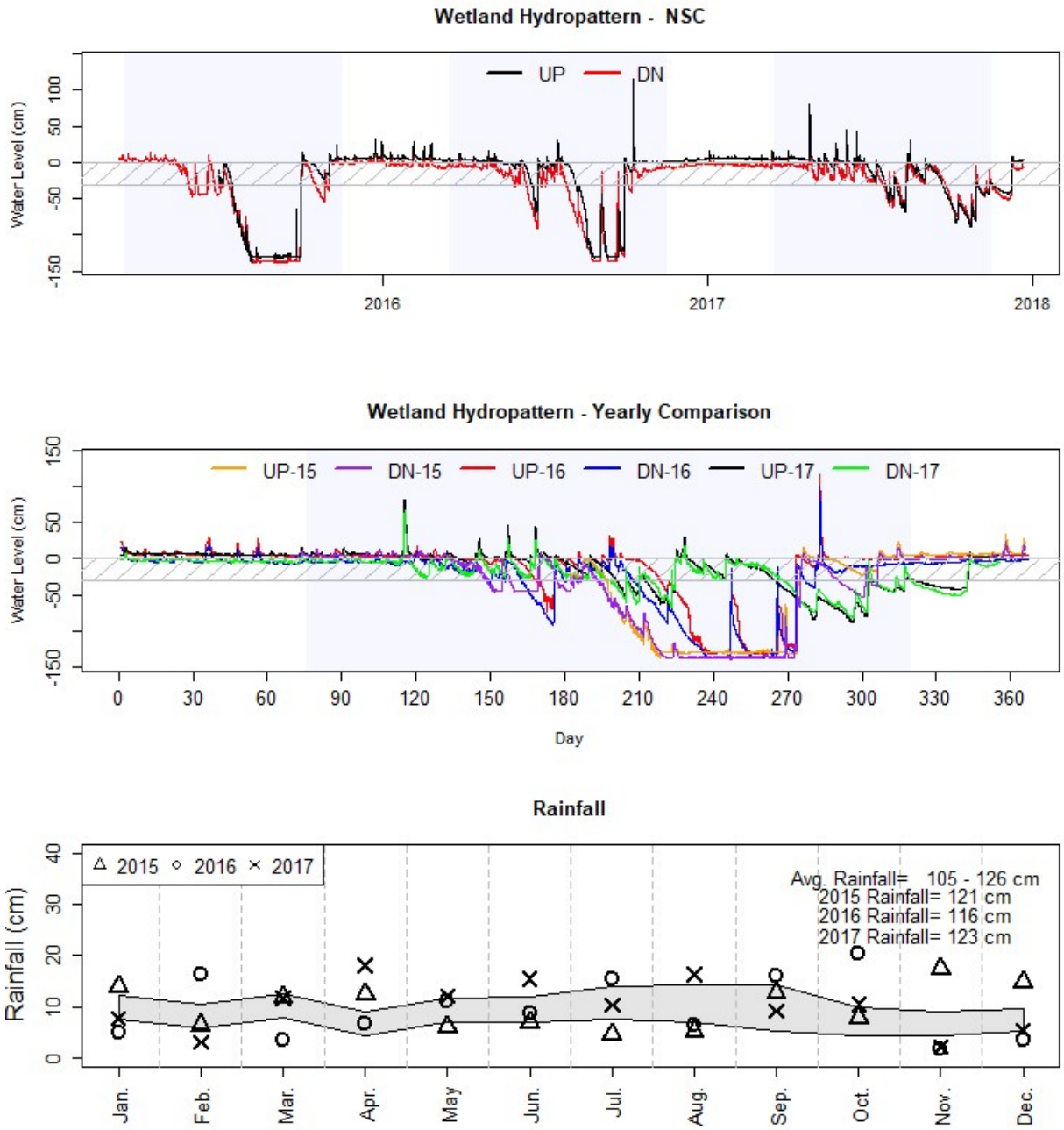


Figure A.14: Hydropattern and rainfall for NSC site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

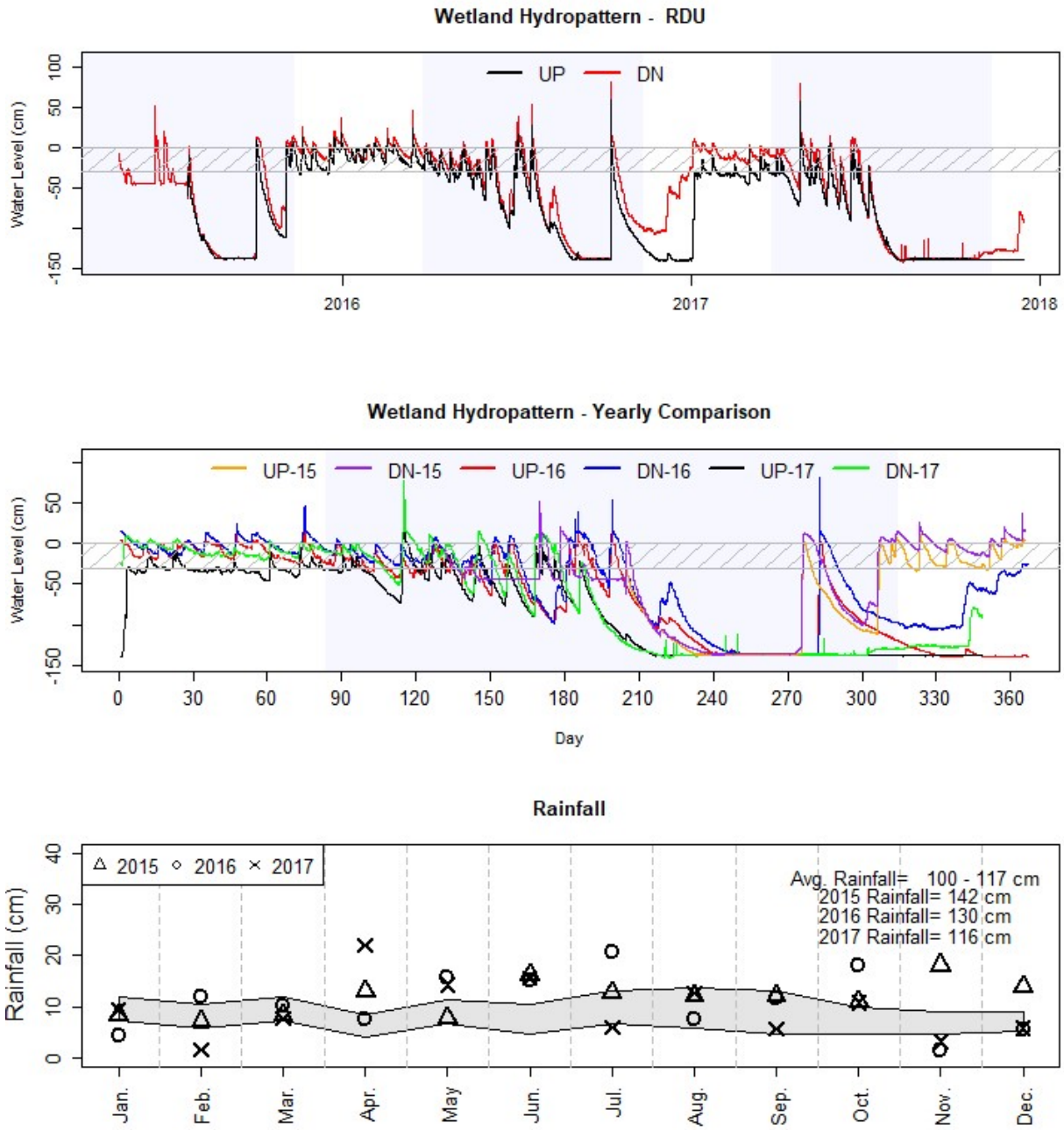


Figure A.15: Hydropattern and rainfall for RDU site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

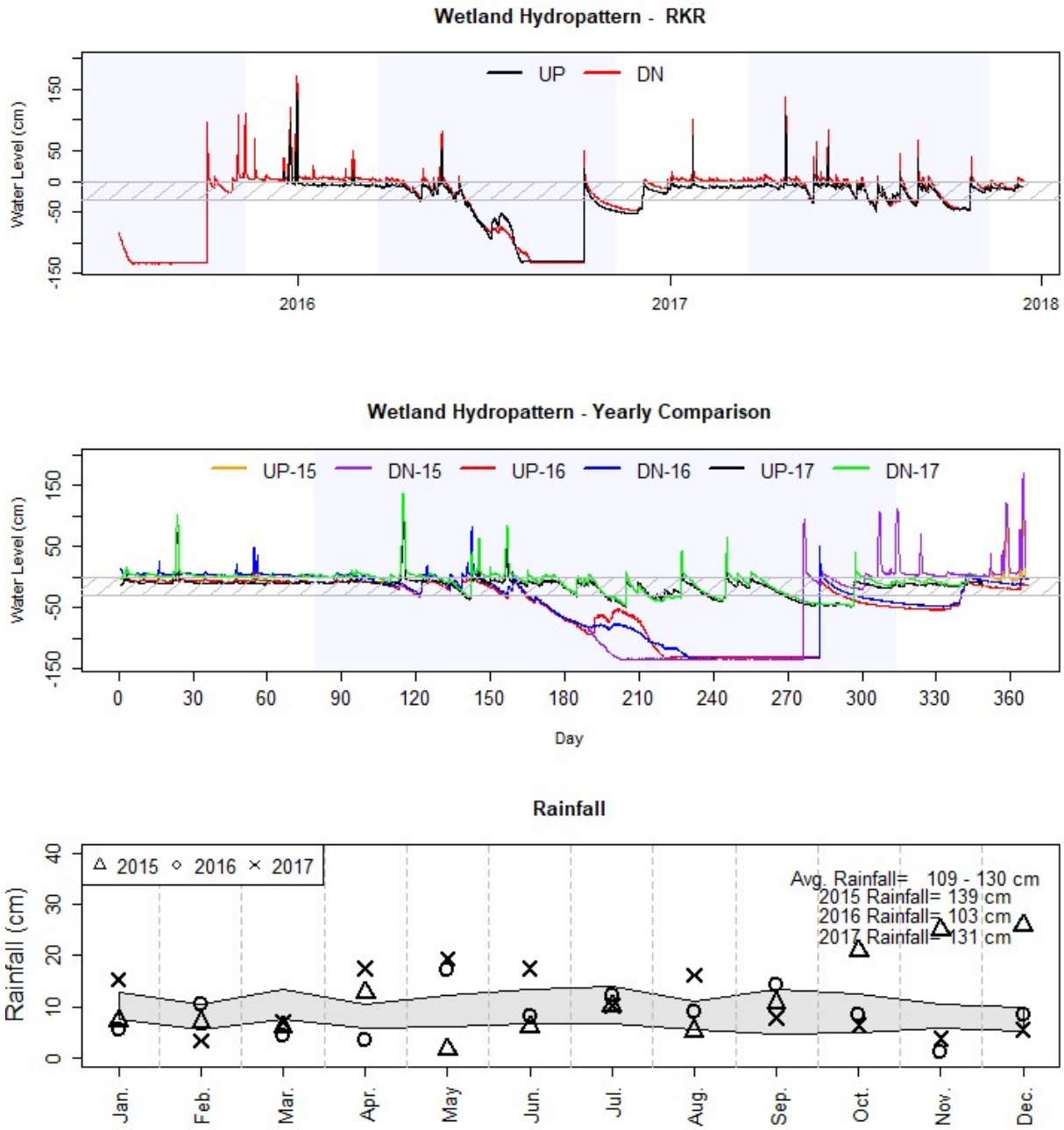


Figure A.16: Hydropattern and rainfall for RKR site. The top plot is the water level for the entire monitoring period (shaded areas are the growing seasons). The middle plot is the water level for each year overlaid on the same plot (shaded region is the growing season). Bottom plot is the monthly rainfall during the monitoring period. The shaded area is the average rainfall (30th and 70th percentiles from WETS Tables).

Appendix B: Hydropatterns by Wetland Type

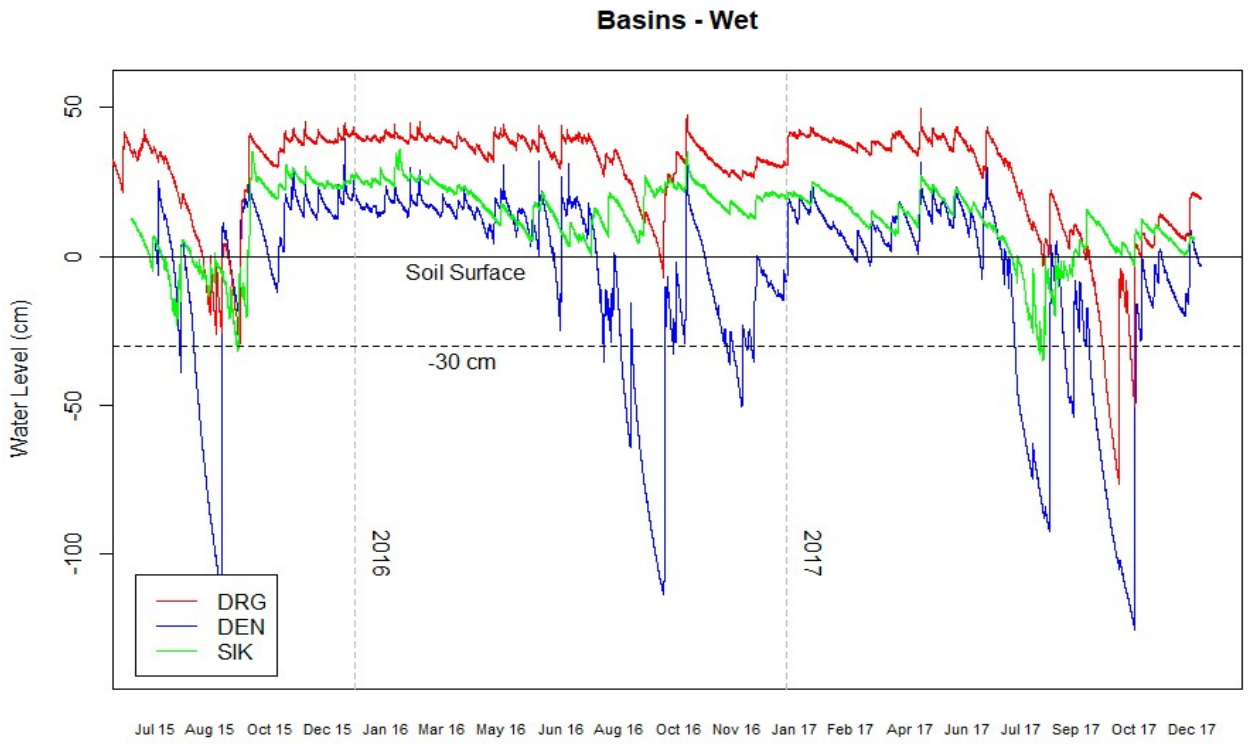


Figure B.1: Wet-end hydropatterns for basin wetlands.

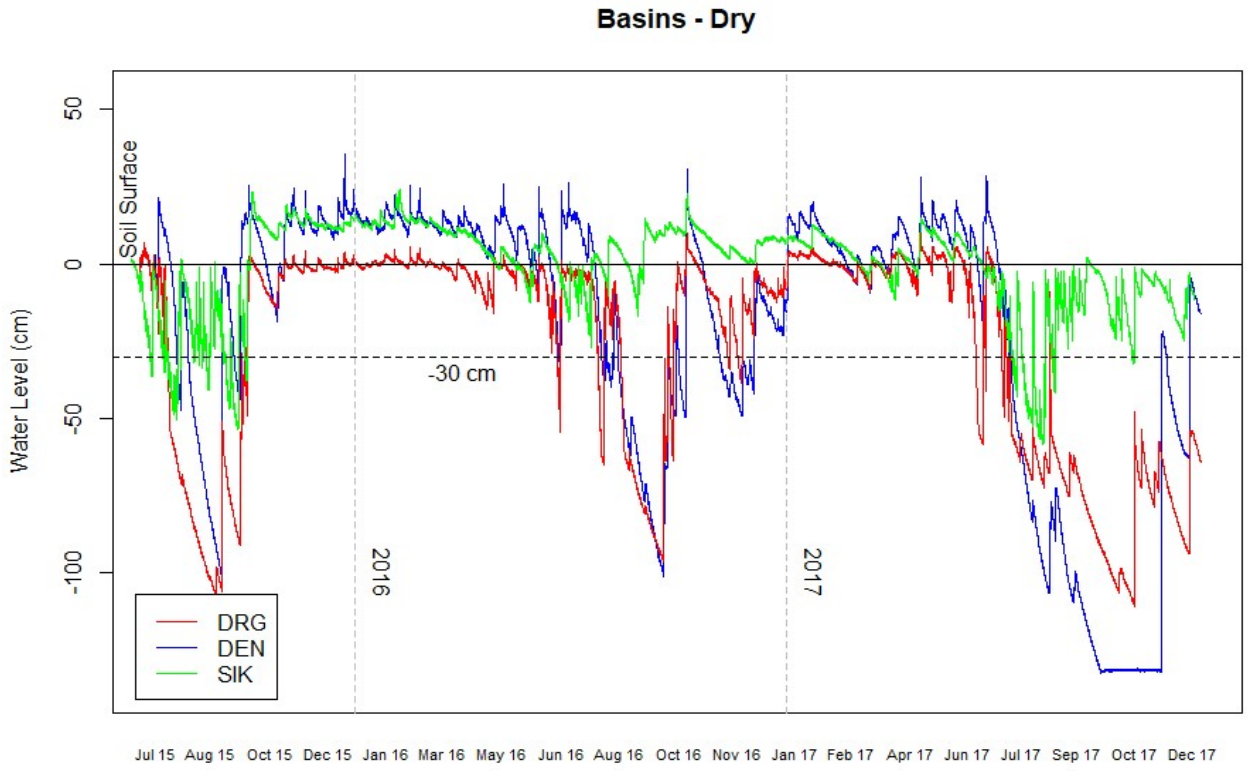


Figure B.2: Dry-end hydropatterns for basin wetlands.

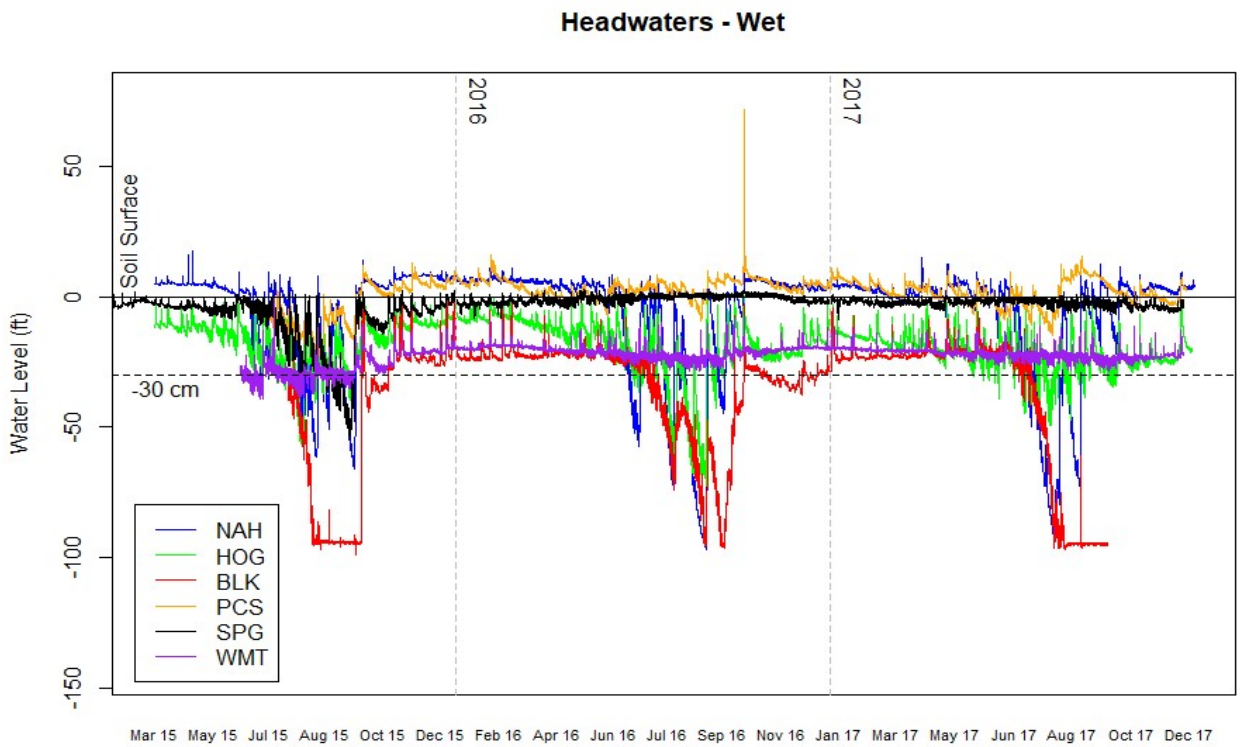


Figure B.3: Wet-end hydropatterns for headwater wetlands.

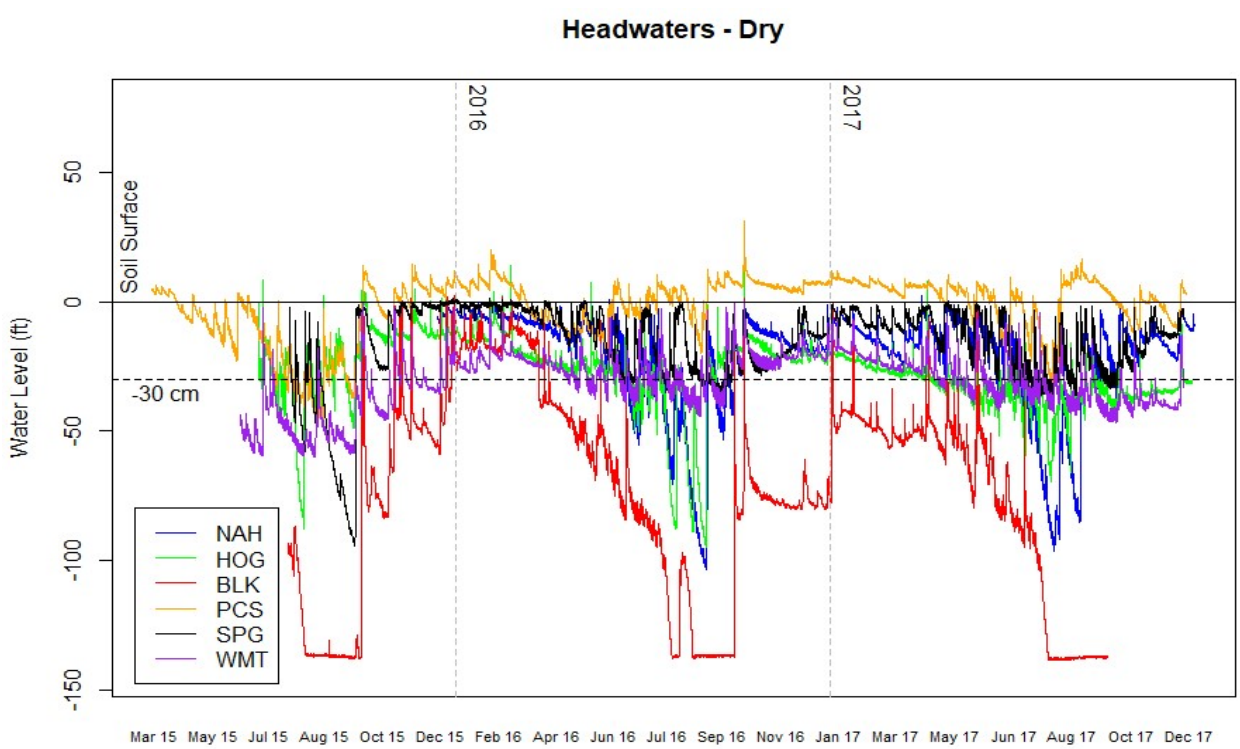


Figure B.4: Dry-end hydropatterns for headwater wetlands.

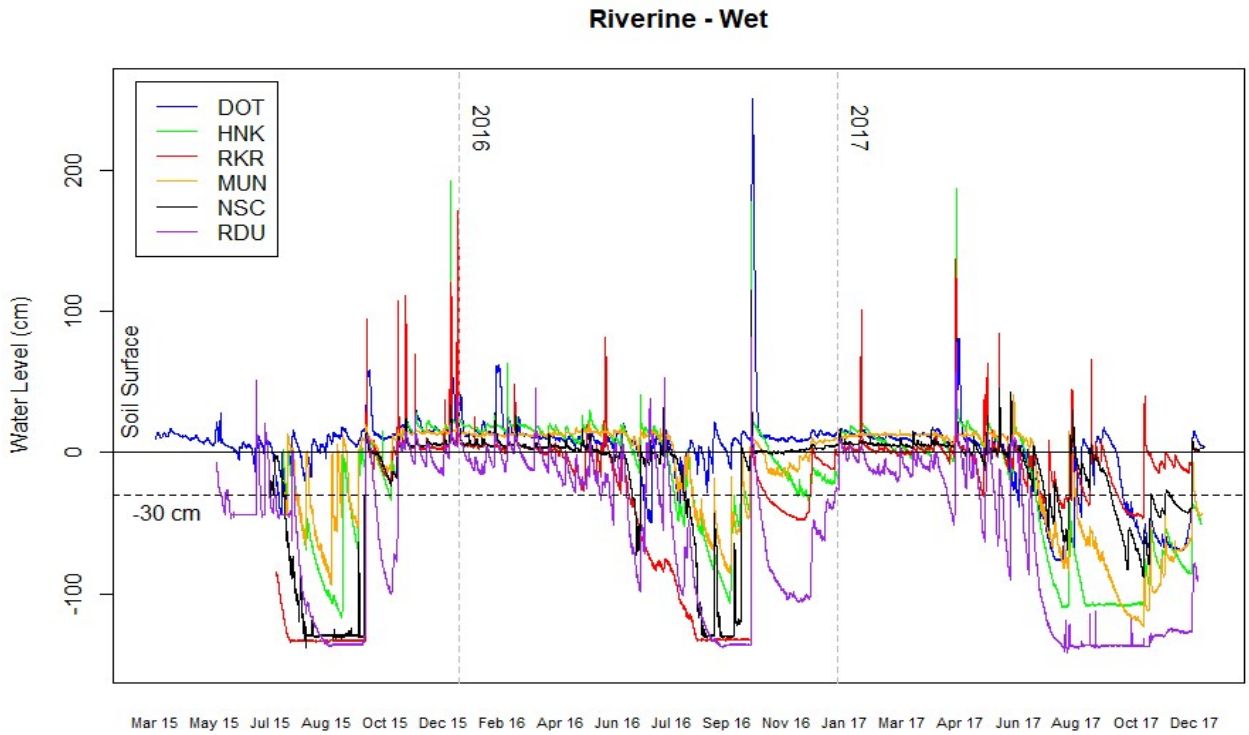


Figure B.5: Wet-end hydropatterns for riverine wetlands.

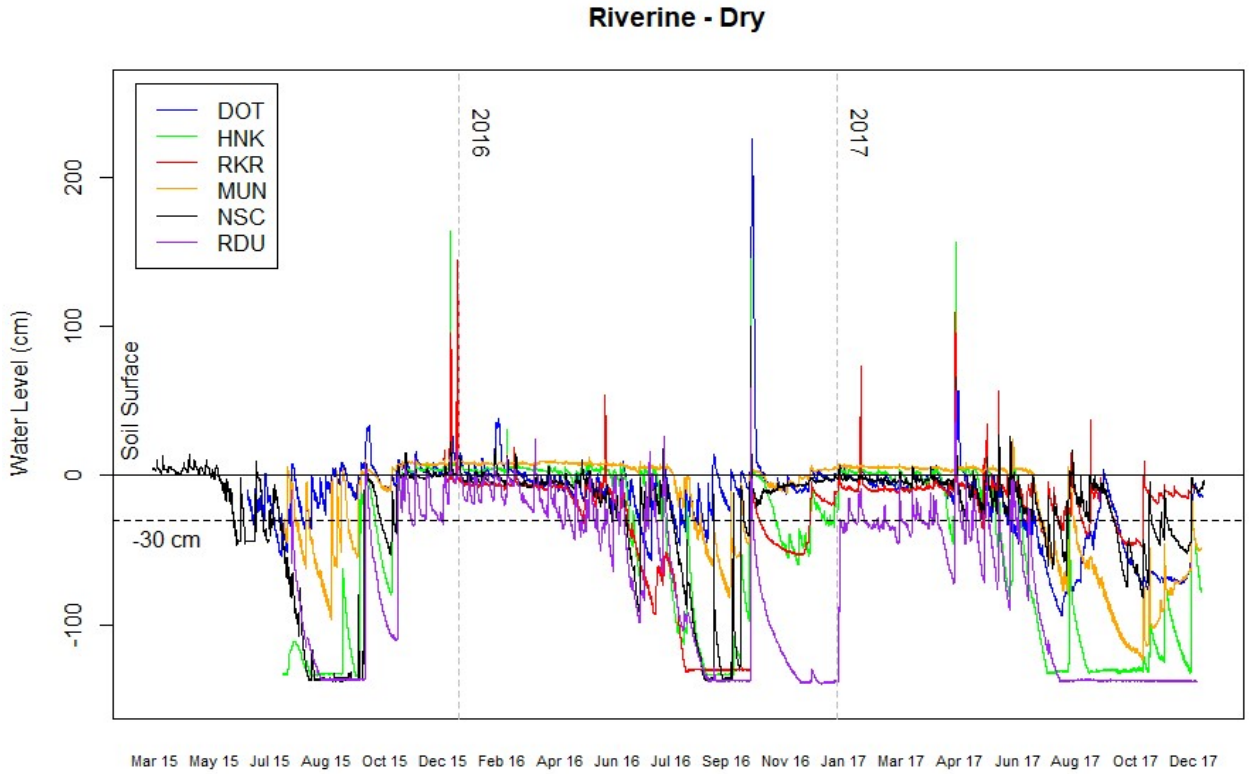


Figure B.6: Dry-end hydropatterns for riverine wetlands.

Appendix C: ECDFs and Summary Statistics

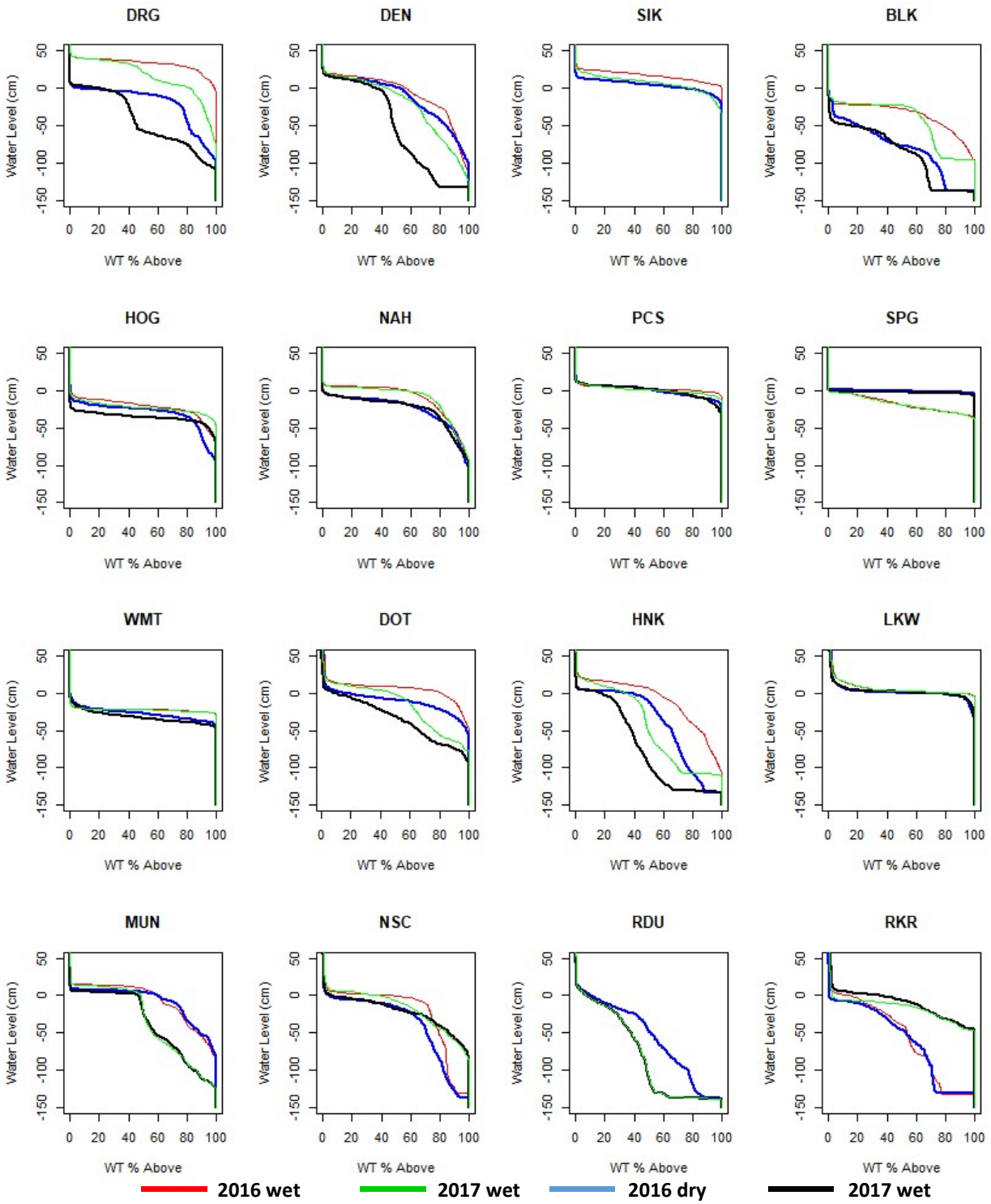


Figure C.1: Wet-end growing season ECDFs for each site.

Table C.1: Summary of growing season ECDFs for wet-end hydropatterns.

	Site	Region	Percent above surface		Percent 0 to -30 cm		Percentage above -30 cm		Percent Below -30 cm		Percent Below -60 cm	
			2016	2017	2016	2017	2016	2017	2016	2017	2016	2017
Basin	**DRG**	PD	99	82	1	10	100	92	0	8	0	3
	DEN	PD	53	36	17	11	70	47	30	53	11	50
	SIK	CP	100	80	0	19	100	99	0	1	0	0
Headwater	**BLK**	PD	0	0	56	47	56	47	44	32	14	24
	SPG	PD	0	0	100	100	100	100	0	0	0	0
	WMT	PD	0	0	100	100	100	100	0	0	0	0
	HOG	CP	0	0	84	88	84	88	16	12	3	0
	NAH	CP	58	57	23	25	81	82	19	18	8	9
	PCS	CP	84	61	16	39	100	100	0	0	0	0
Riverine	**DOT**	CP	81	48	15	22	96	66	4	34	0	16
	NSC	CP	4	5	61	67	65	72	35	28	26	8
	HNK	PD	56	34	19	14	75	47	26	53	11	48
	RDU	PD	10	7	34	23	44	30	56	70	45	59
	RKR	PD	14	35	29	43	43	78	57	22	45	0
	MUN	PD	59	48	19	3	78	51	22	49	8	38
	LKW	CP	89	90	11	10	100	100	0	0	0	0

Table C.2: Growing season summary ECDFs for wet and dry area wells.

	Site	Percent above surface				Percentage 0 to 30 cm				Percentage above -30 cm				Percent Below -30 cm				Percent Below -60 cm			
		Wet		Dry		Wet		Dry		Wet		Dry		Wet		Dry		Wet		Dry	
		2016	2016	2017	2016	2016	2016	2017	2016	2016	2017	2016	2017	2016	2017	2016	2017	2016	2017	2016	2017
Basin	DRG	99	82	7	21	1	10	71	21	100	92	78	42	0	8	22	59	0	3	17	46
	DEN	57	44	53	36	27	23	17	11	84	67	70	47	30	53	16	33	11	50	10	23
	SIK	100	80	75	29	0	19	25	62	100	99	100	91	0	9	0	1	0	0	0	0
Headwater	BLK	0	0	0	0	56	47	4	43	56	47	4	31	44	32	97	79	14	24	69	51
	SPG	0	0	0	0	100	100	87	87	100	100	87	87	0	0	13	13	0	0	0	0
	WMT	0	0	0	0	100	100	62	34	100	100	62	34	0	0	38	66	0	0	0	0
	HOG	0	0	0	0	84	88	72	18	84	88	72	18	16	12	28	82	3	0	10	2
	NAH	58	57	0	0	23	25	72	77	81	82	72	77	19	18	28	23	8	9	9	12
	PCS	84	61	55	61	16	39	45	39	100	100	100	100	0	0	0	1	0	0	0	0
Riverine	DOT	81	48	17	9	15	22	72	39	96	66	89	48	4	34	11	52	0	16	0	28
	NSC	45	38	4	5	30	32	61	67	75	70	65	72	25	30	35	28	19	10	26	8
	HNK	56	34	36	17	19	14	21	16	75	47	57	33	26	53	43	68	11	48	32	60
	RDU	10	7	2	1	34	23	25	11	44	30	27	12	56	70	73	88	45	59	52	65
	RKR	14	35	1	2	29	43	38	76	43	78	39	78	57	22	61	22	45	0	43	0
	MUN	59	48	60	47	19	3	19	5	78	51	79	52	22	49	21	49	8	38	5	34
	LKW	89	90	70	71	11	10	29	29	100	100	99	100	0	0	1	0	0	0	0	0

Table C.3: Growing season summary statistics for wet and dry area wells (all values in cm).

	site	Minimum				25th Percentile				Median				75th Percentile				Maximum			
		Wet		Dry		Wet		Dry		Wet		Dry		Wet		Dry		Wet		Dry	
		2016	2017	2016	2017	2016	2017	2016	2017	2016	2017	2016	2017	2016	2017	2016	2017	2016	2017	2016	2017
Basin	DRG	-7	-76	-96	-111	31	4	-22	-71	35	21	-7	-57	39	37	-3	-2	48	50	10	10
	DEN	-114	-126	-101	-133	-20	-53	-35	-119	7	-6	2	-56	15	12	12	9	34	32	31	28
	SIK	0	-35	-27	-58	12	2	0	-14	17	9	5	-5	22	14	10	1	35	27	23	15
Headwater	BLK	-96	-97	-138	-139	-46	-88	-105	-137	-27	-23	-87	-80	-23	-21	-54	-54	-3	-6	1	-4
	SPG	-5	-7	-37	-39	-1	-3	-26	-26	-0	-2	-19	-19	1	1	-7	-9	3	2	0	0
	WMT	-28	-28	-44	-47	-23	-23	-33	-38	-22	-22	-27	-33	-21	-21	-23	-28	-8	-11	0	-4
	HOG	-73	-49	-95	-69	-25	-26	-31	-38	-19	-23	-24	-35	-13	-18	-21	-31	0	-2	20	8
	NAH	-97	-93	-103	-96	-17	-11	-34	-27	3	1	-15	-16	5	4	-10	-12	62	15	5	2
	PCS	-9	-15	-21	-33	1	-2	-6	-4	3	1	2	2	5	4	6	6	72	16	31	17
Riverine	DOT	-50	-77	-57	-94	4	-46	-19	-65	9	-2	-9	-32	11	8	-3	-10	251	81	227	57
	NSC	-131	-89	-137	-82	-32	-38	-65	-33	-1	-7	-15	-18	2	4	-7	-5	115	81	101	66
	HNK	-107	-110	-135	-133	-31	-107	-93	-131	7	-53	-11	-95	14	7	2	-8	178	187	146	156
	RDU	-138	-142	-138	-139	-97	-137	-114	-137	-47	-109	-66	-103	-14	-20	-29	-42	82	78	59	57
	RKR	-134	-46	-132	-49	-117	-28	-130	-27	-39	-6	-46	-12	-10	2	-14	-8	82	137	55	109
	MUN	-86	-124	-82	-124	-22	-79	-15	-75	9	-23	5	-20	14	12	8	4	84	54	68	37
	LKW	-7	-7	-42	-40	1	1	-1	0	2	3	1	2	7	7	2	3	131	87	123	79

Appendix D: Growing Season Dates and Rainfall Stations

Table D.1: Growing season ates used for analysis.

Site	Start	End	Source
BLK	16-Mar	17-Nov	Jackson Springs WETs
DRG	22-Mar	11-Nov	Granville Co Soils Survey
DEN	22-Mar	11-Nov	Granville Co Soils Survey
DOT	17-Mar	14-Nov	Clinton, NC WETS Table
HNK	22-Mar	11-Nov	Granville Co Soil Survey
HOG	18-Mar	11-Nov	Sampson Co Soil Survey
LKW	28-Feb	3-Dec	Wilmington WETS Table
MUN	22-Mar	11-Nov	Granville Co Soil Survey
NAH	17-Mar	14-Nov	Wayne Co Soils Survey
NSC	18-Mar	10-Nov	Nash Co Soils Survey
PCS	26-Feb	7-Dec	Aurora WETS Table
RDU	25-Mar	10-Nov	RDU WETS Table
RKR	19-Mar	9-Nov	Cabarrus Co Soils Survey
SIK	28-Feb	3-Dec	Wilmington WETS Table
SPG	24-Mar	6-Nov	Rockingham Co Soils Survey
WMT	23-Mar	7-Nov	Moore Co Soils Survey

Table D.2: Rainfall stations used in the analysis.

Site	CRONOS Station Code	Distance from Site (km)
SIK	NNAC	6.6
LKW	NNAC	2.8
PCS	AURO	9.5
NSC	KRWI	9.5
RDU	KRDU	1.0
RKR	NC-CB-8	2.8
SPG	REID	2.4
DRG	OXFO	4.0
DEN	OXFO	2.4
HNK	OXFO	3.2
MUN	OXFO	9.2
WMT	NC-MR-9	9.0
BLK	NUWH	9.6
HOG	NC-SM-11	11.5
DOT	NC-ON-34	16
NAH	GOLD	8.7

Appendix E: Long-term DRAINMOD Simulated Water Level – Rainfall Analysis

The plotted hydropatterns corresponding to yearly total rainfall were plotted in Figure E.1 and Figure E.2. There was considerable variability between the rainfall-grouped hydropatterns. Yearly total rainfall did not appear to be a good method to define ranges for hydropatterns.

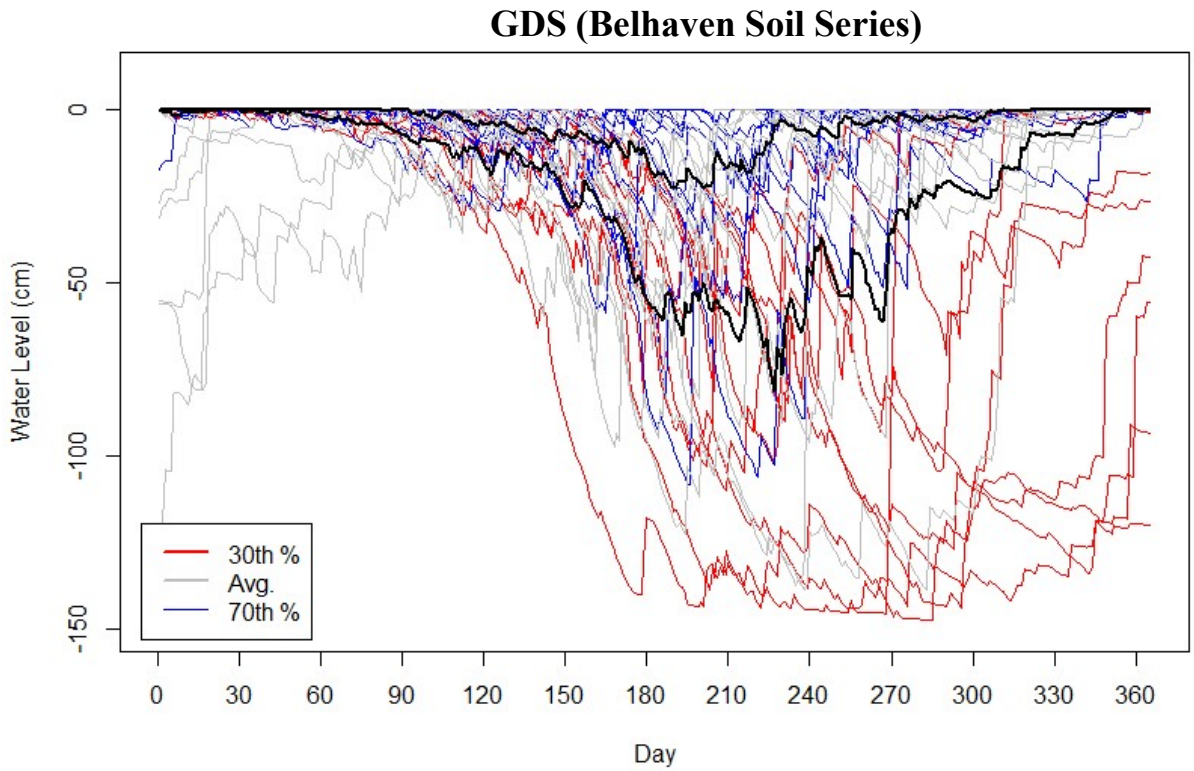


Figure E.1: Hydropatterns corresponding to yearly rainfall totals for GDS site (Belhaven soil). Bold black lines correspond to 30th and 70th percentiles of the water table depths.

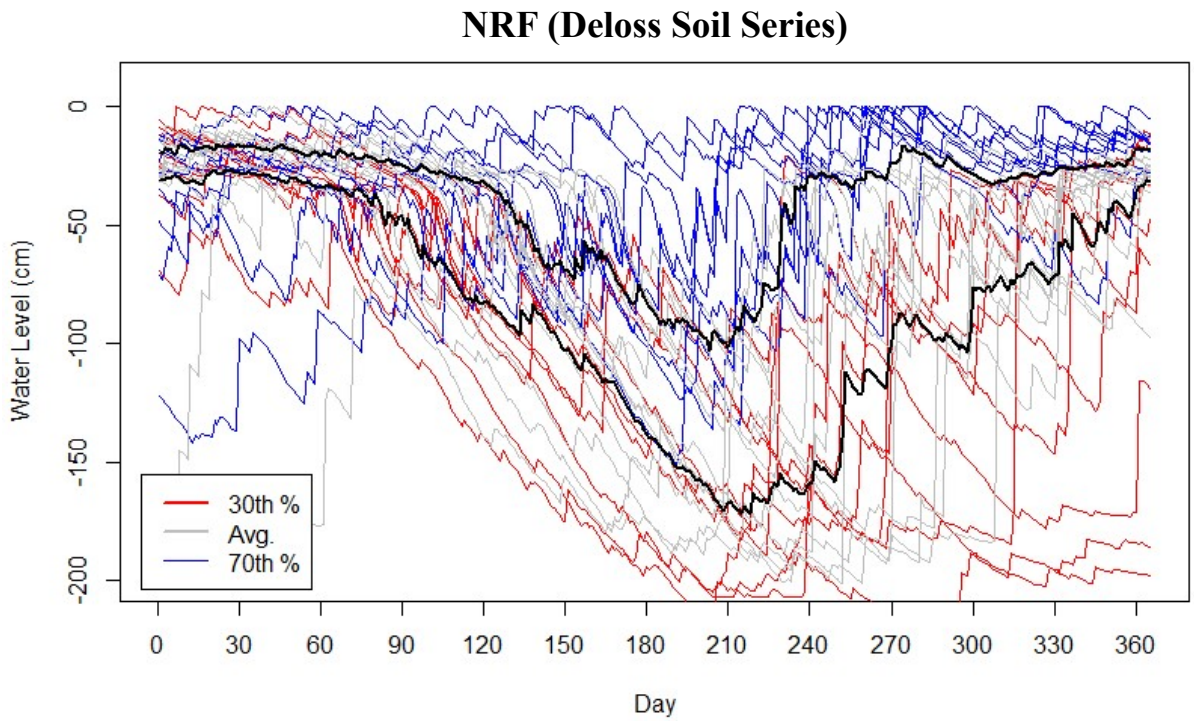


Figure E.2: Hydropatterns corresponding to yearly rainfall totals for NRF site (Deloss soil). Bold black lines correspond to 30th and 70th percentiles of the water table depths.

Appendix F: R Code for Water Level Analysis

R Code to determine longest period of continuous saturation- for hourly water level data

```
rledn=rle(GS17$well_up>-depth)  #consecutive values greater than depth (negative)
rundn=which(rledn$values==TRUE&rledn$length>=per) #consecutive runs longer than
#any(rundn)
rledn.sum=cumsum(rledn$lengths)
ends=rledn.sum[rundn]
index = ifelse(rundn>1, rundn-1, 0)
starts = rledn.sum[index] + 1
if (0 %in% index) starts = c(1,starts)
#print(sta
#calculate number of consecutive days WL above defined level and corresponding percent
days<-(ends-starts)/24       #convert from hours to days
percent<-days/(WL$g_end[1]-WL$g_sta[1])*100
#print(rundn)
#start and end of growing season from file
start<- GS17[starts,1]
end<-GS17[ends,1]
#create table of growing season calcs
GSd_up17<-data.frame(start,end,days,percent)
names(GSd_up17)[1]<-"Start"
names(GSd_up17)[2]<-"End"
GSd_up17$End<-format(GSd_up17$End,"%m-%d-%Y")
GSd_up17$Start<-format(GSd_up17$Start,"%m-%d-%Y")
GSd_up17$days<-as.numeric(GSd_up17$days)
GSd_up17$percent<-as.numeric(GSd_up17$percent)
GSd_up17$days<-round(GSd_up17$days,0)
GSd_up17$percent<-round(GSd_up17$percent,1)
GSd_up17$site<-"UP"
GSd_up17$num_s<-round(starts/24+WL$g_sta[1],0)
GSd_up17$num_e<-round(ends/24+WL$g_sta[1],0)
```

Example R code for ECDF plots

```
# Set vector bounded at upper and lower limits of ECDF plot
x<-seq(-150,150,by=1)   #vector to evaluate CDF
#generate ECDF function
u <- ecdf(GS16$well_up)
# evaluate ECDF function at x
CDF16up=1-u(x)
plot(x,CDF16up*100, type= "l",lty=4, xlab="Water Level (cm)", ylab="Percent Exceedance
(%)",col="red",main="ECDF- 2016 Water Level",xlim=c(-150,50),lwd=2))
```

Appendix G: Water Quality Sampling Periods

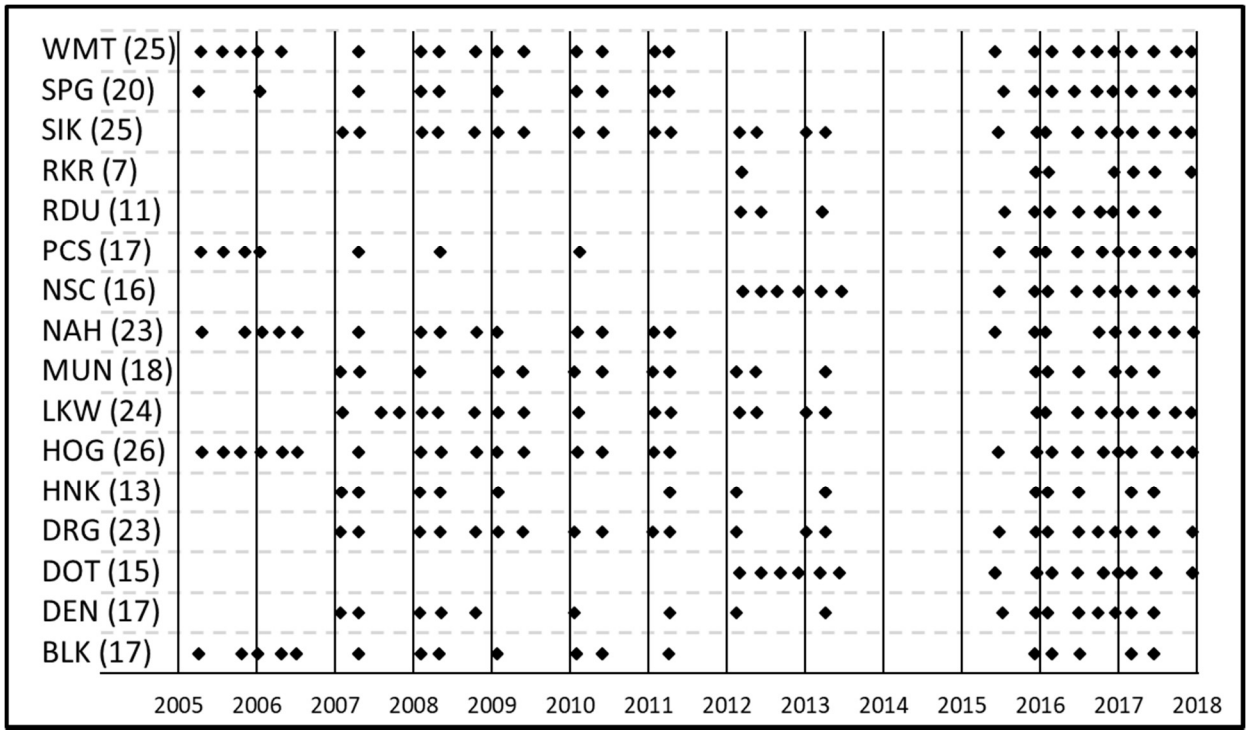


Figure G.1: Samples collected by date and site. Values in parentheses are total samples collected at each site.

Appendix H: Water Quality Nutrients

Table H.1: NO_3^--N summary.

Site	25th percentile (mg/L)	median (mg/L)	75th percentile (mg/L)	Maximum (mg/L)	% Below PQL (%)	% Below USGS Background for streams (%)
BLK	0.02	0.02	0.02	0.16	77	100.0
DEN	0.02	0.02	0.07	6.50	57	81.0
DOT	0.02	0.05	0.40	1.10	34	63.6
DRG	0.02	0.02	0.02	0.08	78	100.0
HNK	0.02	0.03	0.05	0.70	61	95.7
HOG	16.00	24.00	32.00	49.00	0	0.0
LKW	0.02	0.03	0.04	0.17	49	100.0
MUN	0.02	0.02	0.06	6.90	59	91.2
NAH	0.03	0.12	0.69	4.90	29	59.5
NSC	0.02	0.02	0.06	0.89	56	88.9
PCS	0.02	0.02	0.03	0.05	81	100.0
RDU	0.02	0.02	0.03	0.04	53	100.0
RKR	0.02	0.02	0.02	0.72	83	91.7
SIK	0.02	0.02	0.03	1.30	64	97.1
SPG	0.02	0.02	0.03	0.07	82	100.0
WMT	0.02	0.02	0.08	0.28	61	95.5

Table H.2: NH4+-N summary.

Site	25th percentile (mg/L)	median (mg/L)	75th percentile (mg/L)	Maximum (mg/L)	% Below PQL	% Below USGS Background for streams
BLK	0.02	0.02	0.03	1.20	74.2	80.6
DEN	0.06	0.12	0.25	5.20	0.0	9.5
DOT	0.02	0.05	0.08	0.90	43.2	36.4
DRG	0.02	0.02	0.05	0.15	50.0	58.3
HNK	0.02	0.04	0.14	0.46	21.7	47.8
HOG	0.02	0.02	0.04	0.11	71.4	67.3
LKW	0.03	0.13	0.59	2.80	25.6	25.6
MUN	0.02	0.03	0.32	1.30	32.4	50.0
NAH	0.02	0.07	0.19	1.00	19.0	28.6
NSC	0.02	0.04	0.10	1.30	44.4	40.7
PCS	0.02	0.03	0.05	0.18	41.9	58.1
RDU	0.02	0.03	0.05	0.14	52.6	57.9
RKR	0.02	0.03	0.05	0.22	50.0	58.3
SIK	0.02	0.03	0.05	0.17	41.7	52.8
SPG	0.02	0.02	0.04	0.27	68.4	71.1
WMT	0.02	0.05	0.21	0.91	18.2	40.9

Table H.3: ON summary.

Site	25th percentile (mg/L)	median (mg/L)	75th percentile (mg/L)	Maximum (mg/L)	% Below PQL
BLK	0.22	0.36	0.85	2.48	0
DEN	1.48	1.99	2.24	8.80	0
DOT	0.88	1.16	1.82	6.24	0
DRG	1.38	1.78	2.36	3.38	0
HNK	0.91	1.28	2.07	12.06	0
HOG	0.52	0.90	1.78	40.96	0
LKW	1.31	1.91	4.13	48.40	0
MUN	0.58	0.98	1.57	17.80	0
NAH	1.09	1.40	2.09	5.76	0
NSC	0.70	1.05	1.44	3.86	0
PCS	0.59	0.77	1.30	4.26	0
RDU	0.39	0.52	0.83	1.72	0
RKR	0.27	0.75	1.04	1.48	0
SIK	1.38	1.82	2.36	3.65	0
SPG	0.24	0.42	0.66	1.83	0
WMT	0.22	0.57	1.75	14.57	0

Table H.4: TN summary.

Site	25th percentile (mg/L)	median (mg/L)	75th percentile (mg/L)	Maximum (mg/L)	% Below PQL	% Below USGS Background for streams
BLK	0.27	0.40	0.90	3.22	0	60.0
DEN	1.66	2.23	4.14	14.02	0	0.0
DOT	1.20	1.42	2.78	6.32	0	0.0
DRG	1.42	1.82	2.42	3.42	0	0.0
HNK	1.23	1.42	2.27	12.15	0	0.0
HOG	17.82	24.91	35.70	87.00	0	0.0
LKW	1.47	2.19	4.54	51.13	0	0.0
MUN	0.85	1.32	2.06	18.04	0	9.1
NAH	1.58	2.33	3.37	7.40	0	0.0
NSC	0.73	1.22	1.70	4.42	0	0.0
PCS	0.67	0.83	1.36	4.45	0	10.0
RDU	0.46	0.61	0.89	1.80	0	47.4
RKR	0.33	0.78	1.15	2.16	0	33.3
SIK	1.42	1.94	2.51	3.84	0	0.0
SPG	0.28	0.46	0.88	1.93	0	71.1
WMT	0.40	0.69	2.13	15.05	0	43.2

Table H.5: TP summary.

Site	25th percentile (mg/L)	median (mg/L)	75th percentile (mg/L)	Maximum (mg/L)	% Below PQL	% Below USGS Background for streams
BLK	0.02	0.04	0.11	1.80	12.9	38.7
DEN	0.23	0.32	0.70	2.40	4.8	0.0
DOT	0.14	0.26	0.60	2.00	6.8	0.0
DRG	0.07	0.09	0.16	1.30	0.0	8.7
HNK	0.20	0.32	0.61	19.00	4.3	0.0
HOG	0.04	0.09	0.26	38.00	8.2	20.4
LKW	0.08	0.15	0.41	1.50	4.7	4.7
MUN	0.08	0.12	0.27	2.30	2.9	2.9
NAH	0.07	0.15	0.30	2.60	0.0	7.1
NSC	0.05	0.12	0.18	0.51	3.7	7.4
PCS	0.04	0.08	0.14	1.90	0.0	19.4
RDU	0.06	0.07	0.10	0.25	0.0	5.3
RKR	0.14	0.18	0.27	0.33	0.0	0.0
SIK	0.03	0.05	0.14	0.38	13.9	27.8
SPG	0.03	0.07	0.10	0.61	7.9	28.9
WMT	0.02	0.06	0.10	0.68	25.0	40.9

Appendix I: Water Quality Metals

Conversion factors for total metals

Procedure for calculating conversion factors for total recoverable metals. TSS was assumed equal to 10 mg/L, as a very conservative estimate. The partition coefficients and equation are from (US EPA, 1996).

$$Conversion\ factor = \frac{1}{[1+(K_{po}*TSS^{(1-\alpha)}*10^{-6})]} \qquad \text{Equation I.1}$$

Table I.1: Coefficients for calculation of metals translator factors for streams.

Metal	K_{po}	α
Cu	1.04E+06	-0.7436
Zn	1.25E+06	-0.7038
Pb	2.80E+06	-0.8

Adjusted Metals Standards

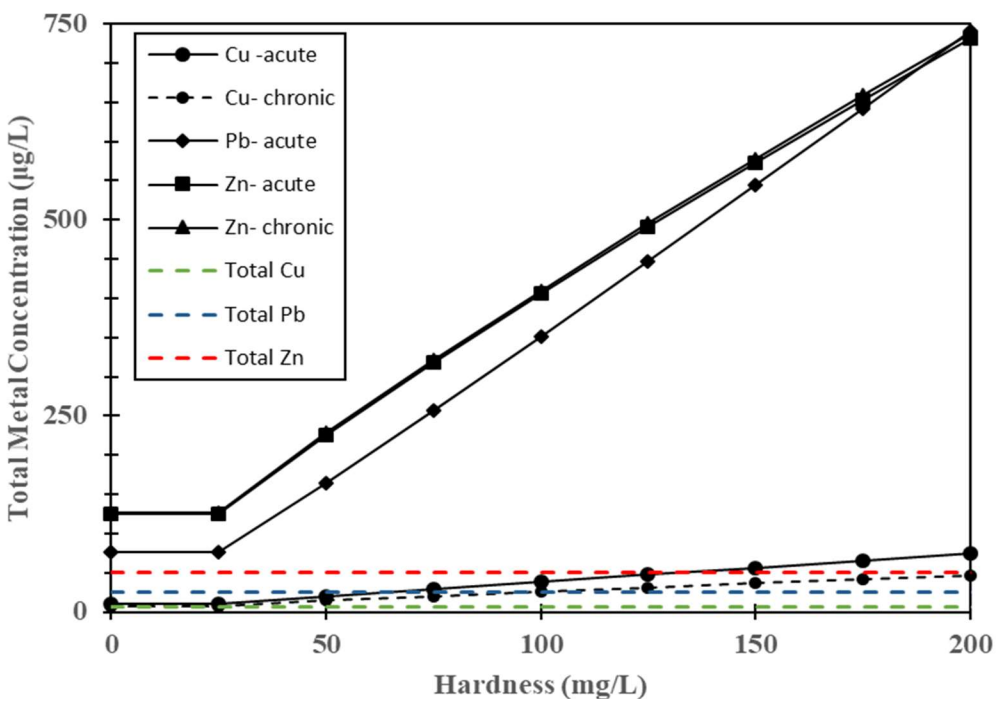


Figure I.1: Criteria for total metals (converted from dissolved criteria using conversion factors).

Table I.2: Zn summary.

Site	25th percentile (µg/L)	median (µg/L)	75th percentile (µg/L)	Maximum (µg/L)	% Below PQL (%)
BLK	<PQL	<PQL	13	76	70
DEN	<PQL	16	28	45	11
DOT	<PQL	<PQL	15	37	56
DRG	12	14	17	25	20
HNK	11	27	37	210*	19
HOG	<PQL	12	19	860*	31
LKW	<PQL	11	21	57	49
MUN	<PQL	<PQL	28	140*	61
NAH	<PQL	13	23	68	31
NSC	<PQL	<PQL	14	44	60
PCS	13	27	36	190*	9
RDU	<PQL	<PQL	<PQL	16	85
RKR	<PQL	<PQL	22	30	56
SIK	<PQL	14	17	66	39
SPG	<PQL	<PQL	<PQL	30	78
WMT	<PQL	11	20	500*	47

PQL = Practical Quantitation Limit = 10 µg/L for Zn
*See Appendix J for a potential hypothesis for extreme values

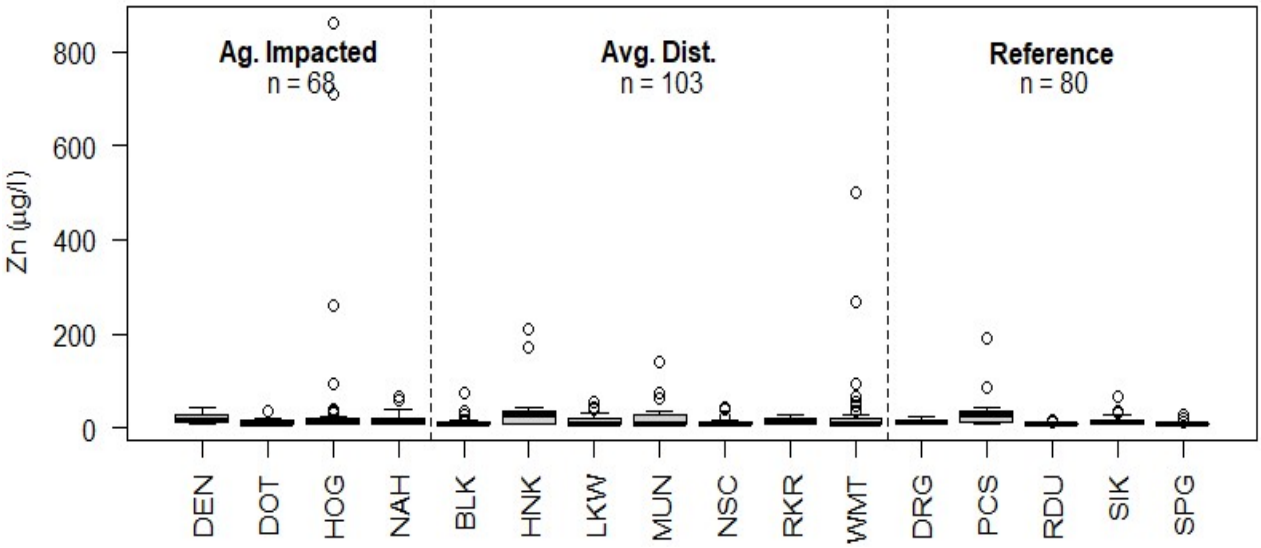


Figure I.2: Zn boxplot.

Table I.3: Cu summary.

Site	25th percentile (µg/L)	median (µg/L)	75th percentile (µg/L)	Maximum (µg/L)	% Below PQL (%)
BLK	<PQL	<PQL	2.4	23.0	66.7
DEN	2.9	4.1	10.3	20.0	16.7
DOT	<PQL	2.3	3.3	15.0	37.5
DRG	2.2	2.9	3.7	4.4	14.3
HNK	2.7	5.6	6.7	66.0*	14.3
HOG	<PQL	<PQL	2.0	81.0*	78.6
LKW	<PQL	3.0	9.0	32.0	37.1
MUN	<PQL	2.7	6.2	31.0	38.7
NAH	<PQL	<PQL	2.6	7.9	65.7
NSC	<PQL	2.0	3.1	26.0	60.0
PCS	<PQL	2.1	3.8	55.0*	43.5
RDU	<PQL	2.9	4.4	8.9	38.5
RKR	4.4	5.5	5.8	9.4	0.0
SIK	<PQL	<PQL	<PQL	4.7	78.6
SPG	<PQL	<PQL	<PQL	5.8	78.1
WMT	<PQL	<PQL	3.3	41.0*	73.7

PQL = Practical Quantitation Limit = 2.0 µg/L for Cu
*See Appendix J for a potential hypothesis for extreme values

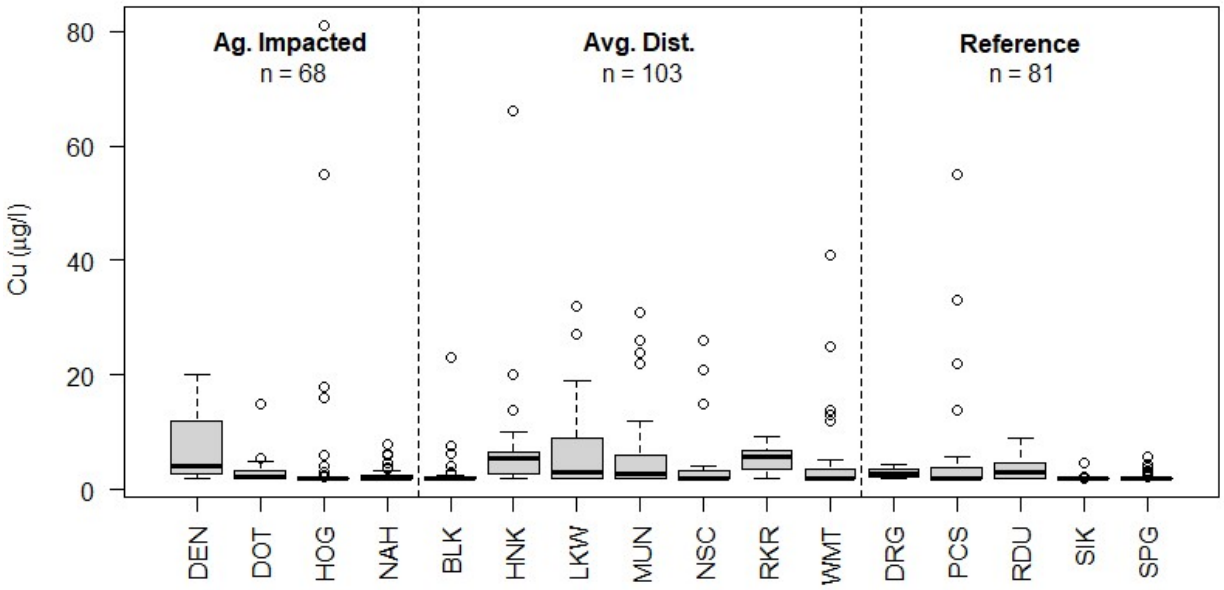


Figure I.3: Cu boxplot.

Table I.4: Pb summary.

Site	25th percentile (µg/L)	median (µg/L)	75th percentile (µg/L)	Maximum (µg/L)	% Below PQL (%)
BLK	<PQL	<PQL	<PQL	84	70
DEN	<PQL	<PQL	<PQL	10	50
DOT	<PQL	<PQL	<PQL	4.5	94
DRG	<PQL	<PQL	<PQL	10	71
HNK	<PQL	<PQL	<PQL	140	62
HOG	<PQL	<PQL	<PQL	340	76
LKW	<PQL	<PQL	<PQL	54	51
MUN	<PQL	<PQL	<PQL	39	77
NAH	<PQL	<PQL	<PQL	20	71
NSC	<PQL	<PQL	<PQL	18	70
PCS	<PQL	<PQL	<PQL	100	65
RDU	<PQL	<PQL	<PQL	2.1	85
RKR	<PQL	<PQL	<PQL	2.6	78
SIK	<PQL	<PQL	<PQL	10	75
SPG	<PQL	<PQL	<PQL	33	78
WMT	<PQL	<PQL	<PQL	48	84

PQL = 10 µg/L (prior to 2011), 2.0 µg/L (2011 to present)
*See Appendix J for a potential hypothesis for extreme values

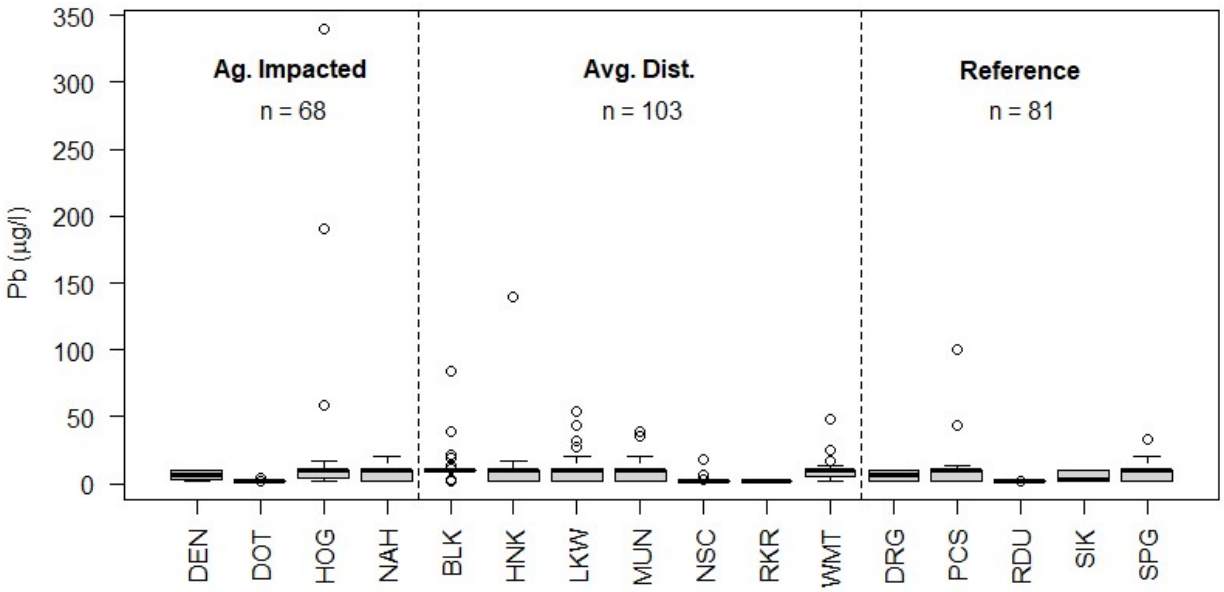


Figure I.4: Pb boxplot.

Table I.5: Percent exceedance of surface water aquatic life standards by site.

Disturbance Class	Site	Sampling Period	# Site visits	Cu Acute	Cu Chronic	Total Cu*	Pb Acute	Total Pb*	Zn Acute	Zn Chronic	Total Zn*
Ag. Impacted	DEN	long-term	10	30%	40%	40%	0%	0%	0%	0%	0%
		current	5	20%	20%	20%	0%	0%	0%	0%	0%
	NAH	long-term	13	0%	0%	8%	0%	0%	0%	0%	15%
		current	5	0%	0%	0%	0%	0%	0%	0%	0%
	HOG	long-term	16	0%	0%	25%	0%	25%	0%	0%	25%
		current	6	0%	0%	0%	0%	0%	0%	0%	0%
	DOT	long-term	6	0%	17%	17%	0%	0%	0%	0%	0%
		current	6	0%	0%	0%	0%	0%	0%	0%	0%
Avg. Dist.	BLK	long-term	12	8%	8%	17%	8%	17%	0%	0%	8%
		current	3	0%	0%	0%	0%	0%	0%	0%	0%
	HNK	long-term	9	11%	22%	44%	0%	11%	0%	0%	22%
		current	3	0%	0%	33%	0%	0%	0%	0%	0%
	LKW	long-term	15	0%	0%	60%	0%	20%	0%	0%	0%
		current	5	0%	0%	20%	0%	20%	0%	0%	20%
	MUN	long-term	12	17%	42%	42%	0%	25%	0%	0%	25%
		current	4	0%	0%	0%	0%	0%	0%	0%	0%
	NSC	long-term	6	17%	17%	33%	0%	0%	0%	0%	0%
		current	6	0%	0%	0%	0%	0%	0%	0%	0%
	RKR	long-term	1	0%	0%	0%	0%	0%	0%	0%	0%
		current	4	0%	25%	50%	0%	0%	0%	0%	0%
	WMT	long-term	16	13%	13%	13%	0%	0%	6%	6%	13%
		current	6	50%	50%	50%	0%	17%	17%	17%	50%
Reference	DRG	long-term	15	0%	0%	0%	0%	0%	0%	0%	0%
		current	6	0%	0%	0%	0%	0%	0%	0%	0%
	PCS	long-term	7	57%	57%	57%	14%	29%	14%	14%	29%
		current	6	0%	0%	0%	0%	0%	0%	0%	0%
	RDU	long-term	3	0%	33%	33%	0%	0%	0%	0%	0%
		current	5	0%	0%	0%	0%	0%	0%	0%	0%
	SIK	long-term	16	0%	0%	0%	0%	0%	0%	0%	6%
		current	6	0%	0%	0%	0%	0%	0%	0%	0%
	SPG	long-term	11	0%	0%	0%	0%	9%	0%	0%	0%
		current	6	0%	0%	0%	0%	0%	0%	0%	0%

*Pre-2015 North Carolina DEQ total metal standards - Long-term: data collected by NC DEQ from 2005 – 2013
Current: data collected by NCSU from 2015 – 2018

Appendix J: Water Quality Extreme Values and Outlier Evaluation

For the long-term sampling completed by NC DEQ, it was indicated in field notes that some samples were collected from shallow pits or stagnant isolated pools during dry conditions. While most of these samples were eliminated from the analysis when they could be identified, there were some very high metals levels that were very difficult to account for given the site conditions. Because wetlands typically have a dissolved organic carbon (DOC) to total organic carbon (TOC) around 0.9 (Reddy and DeLaune, 2008), it was hypothesized that low DOC to TOC ratios might be a good indication of samples that may have some sediment content. Samples that included sediment may result in elevated levels of metals that are not actually reflective of the metals content of the water. For example, a small subsample of several milliliters is typically used during the analysis procedure. If soil particles or organics with bound metals is captured in the subsample this could potentially elevate the results. See Table J.1 below with some example results. These extreme values with very low DOC/TOC ratios were not used in the analysis.

Table J.1: Selected sample with very low DOC/TOC ratio and elevated metals results.

Cu (µg/L)	Pb (µg/L)	Zn (µg/L)	TOC (µg/L)	DOC (µg/L)	DOC/ TOC
81	340	860	62	5.6	0.09
55	190	710	470	7	0.01
25	25	270	330	3.5	0.01
18	59	260	150	8.8	0.06
66	140	210	120	47	0.39
31	36	140	350	37	0.11
16	59	94	290	8.4	0.03
24	36	76	250	8.7	0.03
26	39	76	180	17	0.09
22	36	62	270	16	0.06
7.9	20	60	280	16	0.06
13	44	45	410	42	0.10
15	54	34	160	31	0.19

Appendix K: R Code for Water Quality Analysis

R code for bootstrap confidence intervals

```
#============================AVG DIST=============================================
Yavg=matrix(0,nrow=15,ncol=15)
g=seq(0.2,0.9,by=0.05)
WQavgD=subset(WQmax,Site=="BLK"|Site=="WMT"|Site=="HNK"|Site=="MUN"|Site=="NSC"|Site=="L
KW")
rownames(Yavg)<-g
values<-c("NO3","NH4","TN","ON","TP")
names<-c("NO3-L","NO3-m","NO3-U","NH4-L","NH4-m","NH4-U","TN-L","TN-m","TN-U","ON-L","ON-
m","ON-U","TP-L","TP-m","TP-U")
colnames(Yavg)<-names

for (j in 1:5){
  f=na.omit(WQavgD[values[j]])
  for (k in 1:15){
   h<-one.boot(f[,1],bootQ,R=1000,student=FALSE)
    Yavg[k,(j+(j-1)*2)]=quantile(h$t,0.025)
    Yavg[k,(j+1+(j-1)*2)]=median(h$t)
    Yavg[k,(j+2+(j-1)*2)]=quantile(h$t,0.975)
  }
}

#==========================REFERENCE SITES=========================================
WQref=subset(WQmax,Site=="SIK"|Site=="SPG"|Site=="PCS"|Site=="DRG"|Site=="RDU")
Yref=matrix(0,nrow=15,ncol=15)
rownames(Yref)<-g
colnames(Yref)<-names
for (j in 1:5){
  f=na.omit(WQref[values[j]])
  for (k in 1:15){
    h<-one.boot(f[,1],bootQ,R=1000,student=FALSE)
    Yref[k,(j+(j-1)*2)]=quantile(h$t,0.025)
    Yref[k,(j+1+(j-1)*2)]=median(h$t)
    Yref[k,(j+2+(j-1)*2)]=quantile(h$t,0.975)
  }
}

#===================================AG SITES=======================================
WQag=subset(WQmax,Site=="NAH"|Site=="DEN"|Site=="DOT"|Site=="HOG")
Yag=matrix(0,nrow=15,ncol=15)
rownames(Yag)<-g
colnames(Yag)<-names
for (j in 1:5){
```

```
  f=na.omit(WQag[values[j]])
  for (k in 1:15){
    h<-one.boot(f[,1],bootQ,R=1000,student=FALSE)
    Yag[k,(j+(j-1)*2)]=quantile(h$t,0.025)
    Yag[k,(j+1+(j-1)*2)]=median(h$t)
    Yag[k,(j+2+(j-1)*2)]=quantile(h$t,0.975)
  }
}
#====================Example Code for plotting================================
g=g*100
plot(g,Yavg[,3],type="l",ylim=c(0,0.5),ylab=expression(paste(NO[3]^"-"," + ",NO[2]^"-
",(mg/L))),xlab="Percentile (%)",lwd=1.6)
legend("topleft",legend=c("Reference","Avg. Dist.","Ag.
Impacted"),lty=c(1,1,1),col=c("red","black","blue"),lwd=c(2,2,2),box.lty=0,inset=0.02)
lines(g,Yavg[,2],type="l",lty=2,lwd=1.6)
lines(g,Yavg[,1],lwd=1.6)
lines(g,Yref[,3],col="red",lwd=1.6)
lines(g,Yref[,2],type="l",lty=2,col="red",lwd=1.6)
lines(g,Yref[,1],col="red",lwd=1.6)
lines(g,Yag[,3],col="blue",lwd=1.6)
lines(g,Yag[,2],type="l",lty=2,col="blue",lwd=1.6)
lines(g,Yag[,1],col="blue",lwd=1.6)
lines(c(0,100),c(0.07,0.07),lty=2,col="red",lwd=2)
lines(c(90,90),c(-1,0.11),lty=2,col="red",lwd=2)
points(90,0.07,col="red",cex=2,lwd=2)
text(30,0.09,"0.07 mg/L",col="red")
```

Appendix L: Water Quality Parameters

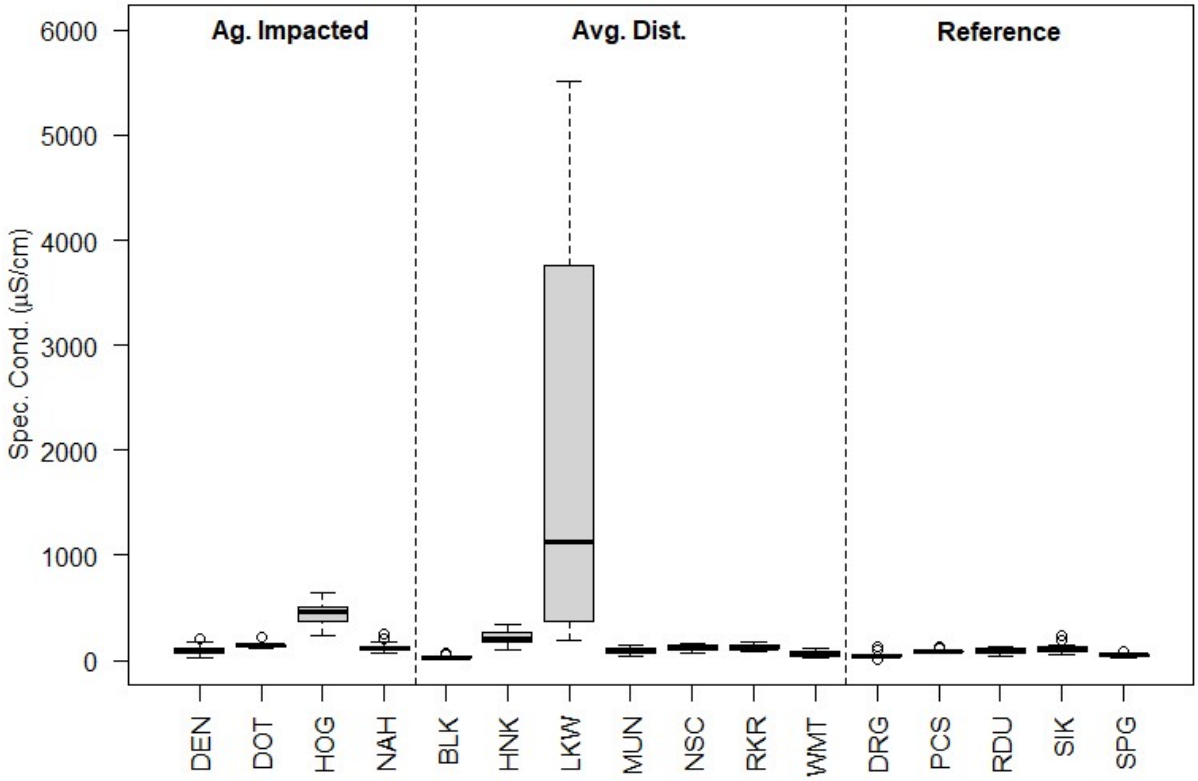


Figure L.1: Boxplot of specific conductivity.

Appendix M: Overall Soil Results by Sample Depth for Wetland Locations

Table M.1: Site means and ranges for both sample depths.

Site	Type	Region	Depth	Cu	Zn	OM
BLK	Hdw	Pd	10	1.5 (1-2.1)	4.2 (2.7-5.5)	11.2 (9-13.8)
			30	1.2 (0.7-2)	2 (0.7-4.6)	7.9 (5-12.2)
DEN	Bsn	Pd	10	1.6 (1.4-1.9)	2.6 (1.5-3.8)	12.4 (9.2-15.8)
			30	1 (0.7-1.2)	1 (0.8-1.4)	5.4 (3-9.1)
DOT	Riv	CP	10	1.1 (0.4-2)	7.5 (1.7-13.9)	13.9 (7-19.6)
			30	0.2 (0.1-0.5)	1.1 (0.3-2.3)	3.5 (1.1-7.7)
DRG	Bsn	Pd	10	1.2 (0.4-2.3)	2.2 (1.6-3.3)	10.8 (6.8-19.2)
			30	1 (0.5-1.5)	1.9 (0.8-3.3)	5.4 (3.8-7)
HNK	Riv	Pd	10	3.7 (2.6-5.1)	13.1 (5.9-50.6)	8.3 (7.3-10)
			30	1.9 (1.2-2.7)	4.7 (1.1-22.4)	4.6 (3.5-6)
HOG	Hdw	CP	10	0.5 (0.3-0.8)	8.1 (3-20.1)	19.1 (12.3-31)
			30	0.2 (0.1-0.5)	2.1 (0.5-5.8)	4.4 (1.3-7.6)
LKW	Riv	CP	10	0.4 (0.2-0.7)	4.3 (1.5-8)	44.3 (10.4-70)
			30	0.2 (0.1-0.5)	1.2 (0.4-2.5)	21.9 (2.9-55)
MUN	Riv	Pd	10	2.5 (2-3.3)	2.5 (1.6-3.6)	6.4 (5-7.4)
			30	2.3 (1.5-3.2)	0.9 (0.7-1.4)	3.6 (2.6-4.9)
NAH	Hdw	CP	10	2.8 (0.4-6.7)	13.1 (3.1-29.3)	17.6 (11-36.7)
			30	0.5 (0.1-0.7)	2.9 (0.4-5.4)	7 (3.3-12.9)
NSC	Riv	CP	10	3.2 (1.2-7.3)	7.4 (0.8-12.5)	13.8 (5.4-19.6)
			30	1.6 (0.6-3.2)	2.1 (0.3-9.8)	4.7 (2.3-7.3)
PCS	Hdw	CP	10	0.9 (0.3-1.9)	3.7 (1.6-6.8)	39.4 (33.3-44.5)
			30	0.3 (0.1-0.4)	1.8 (0.9-2.9)	21.1 (14.5-27.7)
RDU	Riv	Pd	10	2.3 (1.4-3)	2.9 (1-5.9)	5.2 (3.9-7.5)
			30	0.8 (0.4-1.6)	0.6 (0.3-1.2)	2.7 (1.7-4.3)
RKR	Riv	Pd	10	5.9 (3.8-7.7)	7.6 (4.7-10.5)	10.2 (7.7-11.4)
			30	5 (3.2-8)	5.6 (2.7-7.5)	8.8 (7.2-10.7)
SIK	Bsn	CP	10	0.1 (0.1-0.2)	0.7 (0.2-1.4)	12.1 (3-27.7)
			30	0.1 (0.1-0.1)	0.2 (0.1-0.2)	1.6 (0.2-2.7)
SPG	Hdw	Pd	10	0.7 (0.3-1)	2.4 (1.8-2.9)	5.8 (3.9-7.3)
			30	0.7 (0.5-1)	1 (0.6-1.4)	3 (1.7-3.7)
WMT	Hdw	Pd	10	0.7 (0.4-1.3)	7.5 (4.9-12.6)	7.7 (4.7-14.4)
			30	0.6 (0.1-1.1)	1.5 (0.7-3)	1.4 (0.3-2.4)

Appendix N: Results of statistical comparisons for Cu and Zn

Table N.1: Comparisons for sample depth and sample location for Cu and Zn. Statically significant differences are indicated with an asterisk (*).

Depth				Location							
Region	**Compare**	**Zn**	**Cu**	**Region**	**Compare**	**Zn**	**Cu**	**Region**	**Compare**	**Zn**	**Cu**
CP	U30 vs U10	*	ns	**CP**	W10 vs E10	ns	ns	**CP**	W30 vs E30	ns	ns
	E10 vs E30	*	ns		W10 vs U10	ns	ns		W30 vs U30	ns	ns
	W10 vs W30	*	ns		E10 vs U10	ns	ns		E30 vs U30	ns	ns
Pd	U30 vs U10	*	ns	**Pd**	W10 vs E10	ns	ns	**Pd**	W30 vs E30	ns	ns
	E10 vs E30	*	ns		W10 vs U10	ns	ns		W30 vs U30	ns	ns
	W10 vs W30	*	ns		E10 vs U10	ns	ns		E30 vs U30	ns	ns

Table N.2: Comparisons differences between the Coastal Plain and Piedmont for each sample location. Statically significant differences are indicated with an asterisk (*).

Location	Zn	Cu
W10	ns	*
W30	ns	*
E10	ns	*
E30	ns	ns
U10	ns	ns
U30	ns	ns

Table N.3: Spearman's correlation for SOM, humic matter and metals. Bold values represent statistically significant correlations (0.05 level).

Site	HM-Zn	SOM-Zn	HM-Cu	SOM-Cu	Zn-Cu	SOM-HM
BLK	**0.74**	**0.86**	-0.45	-0.2	-0.02	**0.88**
DRG	0.12	0.36	0.07	0.31	0.2	**0.69**
DEN	**0.79**	**0.99**	0.61	**0.79**	**0.77**	**0.82**
DOT	**0.8**	**0.93**	**0.74**	**0.87**	**0.96**	**0.85**
HNK	0.21	**0.66**	0.46	**0.76**	**0.77**	**0.53**
HOG	0.17	**0.76**	-0.02	**0.75**	**0.83**	0.03
LKW	-0.2	**0.82**	-0.07	**0.81**	**0.89**	0.04
MUN	**0.77**	**0.91**	0.4	0.39	0.29	**0.77**
NAH	-0.57	**0.8**	-0.53	**0.85**	**0.99**	-0.37
NSC	**0.78**	**0.58**	**0.7**	**0.52**	**0.57**	**0.56**
PCS	0.05	0.48	0.09	**0.64**	**0.95**	-0.1
RDU	**0.78**	**0.75**	**0.83**	**0.71**	**0.93**	**0.69**
RKR	**0.75**	**0.79**	**0.58**	**0.65**	**0.73**	**0.69**
SIK	**0.84**	**0.88**	**0.84**	**0.86**	**0.82**	**0.78**
SPG	-0.1	**0.8**	0.05	0.11	0.09	0.01
WMT	**0.7**	**0.82**	**0.63**	0.25	0.41	**0.7**

Appendix O: SAS Code and Results for Statistical Model

SAS Code for volume converted metal analysis

```
m 'log;clear;output;clear;results;clear'; *clear windows;
Title "Cu";

proc import datafile="C:\Google Drive\EPA Wetlands\Soils\Soil_Data_BD.csv"
dbms=csv out=soil replace; getnames=yes;
run;
ods pdf file="C:\Google Drive\EPA Wetlands\Soils\Model_Cu.pdf";
run;

proc sort data = Soil ;
by Region Site  T2  Location Depth ;
run;

*Region -coastal plain, piedmont
*Site- 16 sites
*T2- sampling transect
*Location- Wetlands, Edge, Upland
*Depth- 0-10 cm and 10-30 cm

proc glimmix data=Soil plots=all ;
  class Region Location Depth Site T2;
  model Cu_vol= Region Location Region*Location Depth  Depth*Region
Depth*Location Depth*Location*Region /dist=logn ddfm=kr;
  random Site(Region)  T2(Site Region)
       location*Site(Region)   location*T2(Site*Region)
        Depth*Site(Region)   ;
 lsmeans Region Location Depth  /  cl alpha=0.05 adjust=tukey  lines;
 lsmeans Region*Location  /  cl alpha=0.05 adjust=tukey  slice= (Region
Location)  slicediff= (region) ;
 lsmeans Region*Depth  /  cl alpha=0.05 adjust=tukey  slice= (Region Depth)
slicediff=(Region);
 lsmeans Location*Depth  /  cl alpha=0.05 adjust=tukey    slice= (location
Depth) slicediff=(location);
 lsmeans Region*Location*Depth  /  cl alpha=0.05 adjust=tukey   slice=
(Region*Depth) slicediff=(Region*Depth) ;
 lsmeans Region*Location*Depth  /  cl alpha=0.05 adjust=tukey    lines ;
 ods output lsmeans=lsmnds;
run;

ods pdf close;
```

SAS PROC GLIMMIX model output example for Cu analysis

The GLIMMIX Procedure

Model Information	
Data Set	WORK.SOIL
Response Variable	Cu_vol
Response Distribution	Lognormal
Link Function	Identity
Variance Function	Default
Variance Matrix	Not blocked
Estimation Technique	Restricted Maximum Likelihood
Degrees of Freedom Method	Kenward-Roger
Fixed Effects SE Adjustment	Kenward-Roger

Class Level Information		
Class	Levels	Values
Region	2	Cp Pd
Location	3	E U W
Depth	2	10 30
Site	16	BLK DEN DOT DRG HNK HOG LKW MUN NAH NSC PCS RDU RKR SIK SPG WMT
T2	6	1 2 3 4 5 6

Number of Observations Read	516
Number of Observations Used	506

Dimensions	
G-side Cov. Parameters	5
R-side Cov. Parameters	1
Columns in X	36
Columns in Z	382
Subjects (Blocks in V)	1
Max Obs per Subject	506

Optimization Information	
Optimization Technique	Dual Quasi-Newton
Parameters in Optimization	5
Lower Boundaries	5
Upper Boundaries	0
Fixed Effects	Profiled

The GLIMMIX Procedure

Optimization Information	
Residual Variance	Profiled
Starting From	Data

Iteration History					
Iteration	Restarts	Evaluations	Objective Function	Change	Max Gradient
0	0	4	890.15743995	.	166.7
1	0	2	845.75328612	44.40415383	24.78354
2	0	2	839.80663169	5.94665443	25.3297
3	0	2	831.45696901	8.34966267	19.59783
4	0	3	830.52373738	0.93323163	9.473141
5	0	2	830.19253866	0.33119872	26.63049
6	0	4	828.31278336	1.87975530	6.435732
7	0	3	827.1452025	1.16758086	5.985027
8	0	3	826.84238153	0.30282097	3.412492
9	0	3	826.78701102	0.05537051	1.270417
10	0	3	826.7765337	0.01047732	0.496443
11	0	3	826.77407315	0.00246055	0.202301
12	0	3	826.77335	0.00072314	0.080765
13	0	3	826.77329917	0.00005083	0.009791
14	0	3	826.7732986	0.00000058	0.00012

Convergence criterion (GCONV=1E-8) satisfied.

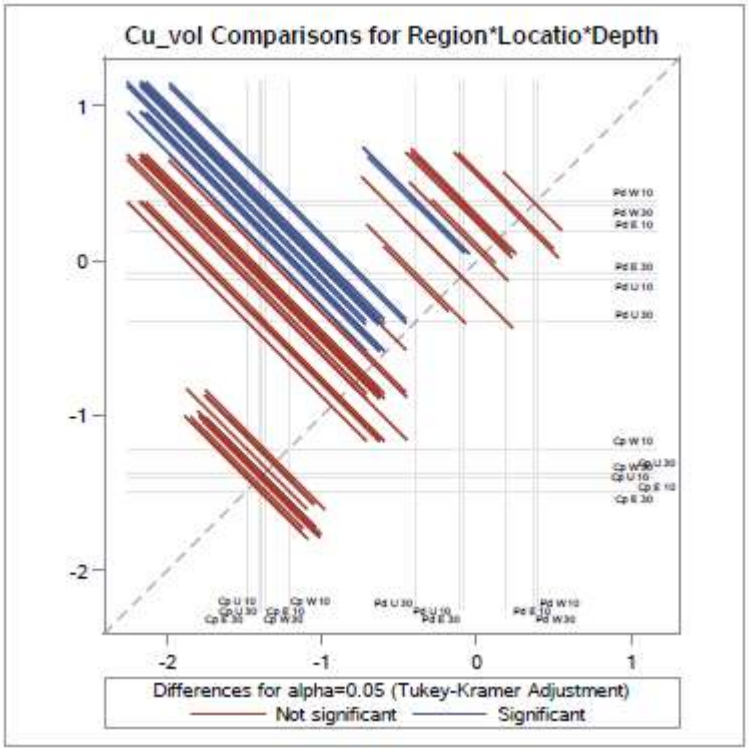
Fit Statistics	
-2 Res Log Likelihood	826.77
AIC (smaller is better)	838.77
AICC (smaller is better)	838.95
BIC (smaller is better)	843.41
CAIC (smaller is better)	849.41
HQIC (smaller is better)	839.01
Generalized Chi-Square	81.16
Gener. Chi-Square / DF	0.16

The GLIMMIX Procedure

Covariance Parameter Estimates		
Cov Parm	Estimate	Standard Error
Site(Region)	0.6535	0.2768
T2(Region*Site)	0.03321	0.01563
Locatio*Site(Region)	0.1149	0.03962
Locat*T2(Regio*Site)	0.05422	0.01867
Depth*Site(Region)	0.03476	0.01730
Residual	0.1643	0.01415

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Region	1	13.86	11.16	0.0049
Location	2	27.52	3.50	0.0441
Region*Location	2	27.52	2.11	0.1402
Depth	1	14.22	3.02	0.1038
Region*Depth	1	14.22	0.58	0.4582
Location*Depth	2	274.5	0.39	0.6772
Region*Locatio*Depth	2	274.5	3.69	0.0262

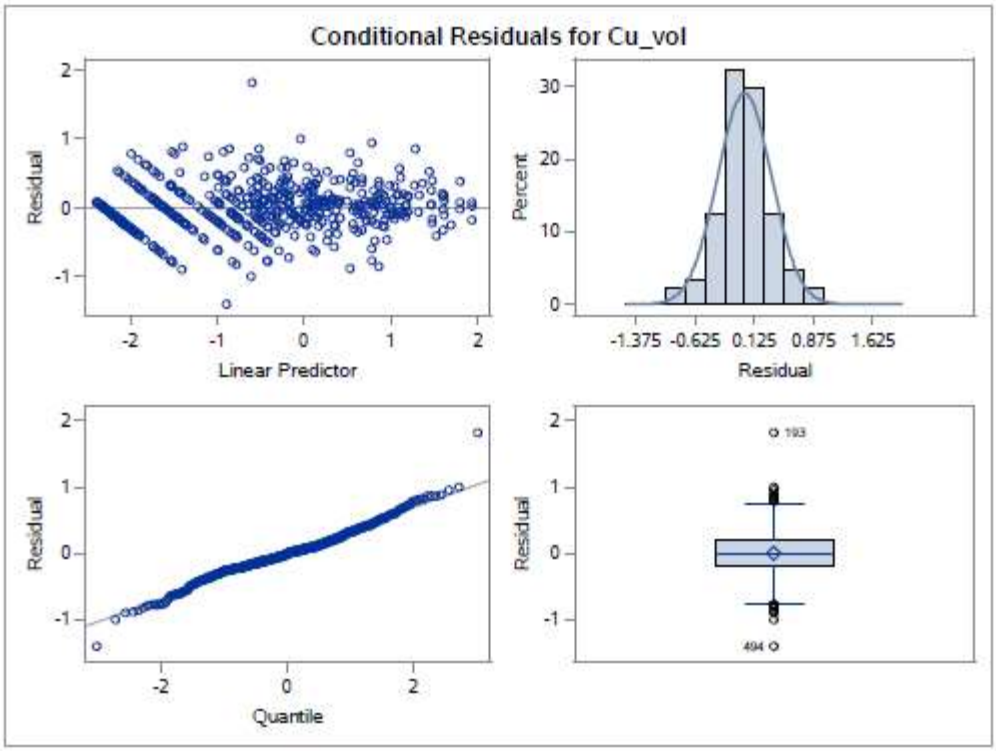
Region Least Squares Means								
Region	Estimate	Standard Error	DF	t Value	Pr > |t|	Alpha	Lower	Upper
Cp	-1.3796	0.3226	13.88	-4.28	0.0008	0.05	-2.0720	-0.6871
Pd	0.05706	0.2843	13.83	0.20	0.8438	0.05	-0.5533	0.6674



Tukey-Kramer Grouping for Region*Locatio*Depth Least Squares Means (Alpha=0.05)						
LS-means with the same letter are not significantly different.						
Region	Location	Depth	Estimate			
Pd	W	10	0.3869		A	
					A	
Pd	W	30	0.3602		A	
					A	
Pd	E	10	0.1894	B	A	
				B	A	
Pd	E	30	-0.08536	B	A	C
				B	A	C
Pd	U	10	-0.1180	B	A	C
				B		C
Pd	U	30	-0.3909	B		C
				B		C
Cp	W	10	-1.2203	B		C

The GLIMMIX Procedure

Tukey-Kramer Grouping for Region*Locatio*Depth Least Squares Means (Alpha=0.05)						
LS-means with the same letter are not significantly different.						
Region	Location	Depth	Estimate			
						C
Cp	U	30	-1.3687			C
						C
Cp	W	30	-1.3946			C
						C
Cp	U	10	-1.4017			C
						C
Cp	E	10	-1.4046			C
						C
Cp	E	30	-1.4875			C



Differences of Region*Locatio*Depth Least Squares Means Adjustment for Multiple Comparisons: Tukey-Kramer														
Region	Location	Depth	_Region	_Location	_Depth	Estimate	Standard Error	DF	t Value	Pr > |t|	Adj P	Alpha	Lower	Upper
Cp	E	10	Cp	E	30	0.08290	0.1484	38.53	0.56	0.5795	1.0000	0.05	-0.2173	0.3831
Cp	E	10	Cp	U	10	-0.00284	0.2204	36.94	-0.01	0.9898	1.0000	0.05	-0.4494	0.4437
Cp	E	10	Cp	U	30	-0.03594	0.2422	45.31	-0.15	0.8827	1.0000	0.05	-0.5238	0.4519
Cp	E	10	Cp	W	10	-0.1843	0.2148	33.54	-0.86	0.3969	0.9994	0.05	-0.6210	0.2524
Cp	E	10	Cp	W	30	-0.01002	0.2374	41.98	-0.04	0.9665	1.0000	0.05	-0.4891	0.4691
Cp	E	10	Pd	E	10	-1.5940	0.4688	19.33	-3.40	0.0029	0.0364	0.05	-2.5740	-0.6140
Cp	E	10	Pd	E	30	-1.3192	0.4688	19.33	-2.81	0.0109	0.1799	0.05	-2.2992	-0.3393
Cp	E	10	Pd	U	10	-1.2866	0.4696	19.47	-2.74	0.0128	0.2129	0.05	-2.2679	-0.3052
Cp	E	10	Pd	U	30	-1.0137	0.4696	19.47	-2.16	0.0436	0.5811	0.05	-1.9951	-0.03239
Cp	E	10	Pd	W	10	-1.7915	0.4672	19.07	-3.84	0.0011	0.0084	0.05	-2.7690	-0.8140
Cp	E	10	Pd	W	30	-1.7648	0.4672	19.07	-3.78	0.0013	0.0104	0.05	-2.7423	-0.7873
Cp	E	30	Cp	U	10	-0.08574	0.2422	45.3	-0.35	0.7250	1.0000	0.05	-0.5735	0.4020
Cp	E	30	Cp	U	30	-0.1188	0.2204	36.97	-0.54	0.5930	1.0000	0.05	-0.5654	0.3278
Cp	E	30	Cp	W	10	-0.2672	0.2366	41.51	-1.13	0.2653	0.9931	0.05	-0.7449	0.2105
Cp	E	30	Cp	W	30	-0.09292	0.2154	33.92	-0.43	0.6690	1.0000	0.05	-0.5308	0.3449
Cp	E	30	Pd	E	10	-1.6769	0.4688	19.33	-3.58	0.0020	0.0206	0.05	-2.6569	-0.6970
Cp	E	30	Pd	E	30	-1.4021	0.4688	19.33	-2.99	0.0074	0.1168	0.05	-2.3821	-0.4222
Cp	E	30	Pd	U	10	-1.3695	0.4696	19.47	-2.92	0.0087	0.1411	0.05	-2.3508	-0.3881
Cp	E	30	Pd	U	30	-1.0966	0.4696	19.47	-2.34	0.0304	0.4544	0.05	-2.0780	-0.1153
Cp	E	30	Pd	W	10	-1.8744	0.4672	19.07	-4.01	0.0007	0.0044	0.05	-2.8519	-0.8969
Cp	E	30	Pd	W	30	-1.8477	0.4672	19.07	-3.96	0.0008	0.0054	0.05	-2.8252	-0.8702
Cp	U	10	Cp	U	30	-0.03310	0.1453	35.49	-0.23	0.8212	1.0000	0.05	-0.3280	0.2618
Cp	U	10	Cp	W	10	-0.1815	0.2137	32.86	-0.85	0.4020	0.9995	0.05	-0.6163	0.2534
Cp	U	10	Cp	W	30	-0.00718	0.2365	41.29	-0.03	0.9759	1.0000	0.05	-0.4846	0.4703
Cp	U	10	Pd	E	10	-1.5912	0.4683	19.25	-3.40	0.0030	0.0367	0.05	-2.5704	-0.6119
Cp	U	10	Pd	E	30	-1.3164	0.4683	19.25	-2.81	0.0111	0.1812	0.05	-2.2956	-0.3371
Cp	U	10	Pd	U	10	-1.2837	0.4692	19.4	-2.74	0.0130	0.2144	0.05	-2.2643	-0.3031
Cp	U	10	Pd	U	30	-1.0109	0.4692	19.4	-2.15	0.0440	0.5839	0.05	-1.9915	-0.03028
Cp	U	10	Pd	W	10	-1.7887	0.4667	19	-3.83	0.0011	0.0085	0.05	-2.7655	-0.8119
Cp	U	10	Pd	W	30	-1.7620	0.4667	19	-3.78	0.0013	0.0104	0.05	-2.7387	-0.7852
Cp	U	30	Cp	W	10	-0.1484	0.2357	40.81	-0.63	0.5325	1.0000	0.05	-0.6244	0.3276
Cp	U	30	Cp	W	30	0.02592	0.2144	33.26	0.12	0.9045	1.0000	0.05	-0.4102	0.4620
Cp	U	30	Pd	E	10	-1.5581	0.4683	19.25	-3.33	0.0035	0.0456	0.05	-2.5373	-0.5788
Cp	U	30	Pd	E	30	-1.2833	0.4683	19.25	-2.74	0.0129	0.2125	0.05	-2.2625	-0.3040

The GLIMMIX Procedure

Differences of Region*Locatio*Depth Least Squares Means Adjustment for Multiple Comparisons: Tukey-Kramer														
Region	Location	Depth	_Region	_Location	_Depth	Estimate	Standard Error	DF	t Value	Pr > |t|	Adj P	Alpha	Lower	Upper
Cp	U	30	Pd	U	10	-1.2506	0.4692	19.4	-2.67	0.0151	0.2493	0.05	-2.2312	-0.2700
Cp	U	30	Pd	U	30	-0.9778	0.4692	19.4	-2.08	0.0506	0.6345	0.05	-1.9584	0.002819
Cp	U	30	Pd	W	10	-1.7556	0.4667	19	-3.76	0.0013	0.0109	0.05	-2.7324	-0.7788
Cp	U	30	Pd	W	30	-1.7289	0.4667	19	-3.70	0.0015	0.0134	0.05	-2.7056	-0.7521
Cp	W	10	Cp	W	30	0.1743	0.1283	22.02	1.36	0.1881	0.9702	0.05	-0.09179	0.4403
Cp	W	10	Pd	E	10	-1.4097	0.4655	18.8	-3.03	0.0070	0.1060	0.05	-2.3847	-0.4347
Cp	W	10	Pd	E	30	-1.1349	0.4655	18.8	-2.44	0.0249	0.3842	0.05	-2.1099	-0.1600
Cp	W	10	Pd	U	10	-1.1023	0.4664	18.94	-2.36	0.0290	0.4346	0.05	-2.0786	-0.1259
Cp	W	10	Pd	U	30	-0.8294	0.4664	18.94	-1.78	0.0914	0.8284	0.05	-1.8058	0.1469
Cp	W	10	Pd	W	10	-1.6072	0.4639	18.55	-3.46	0.0027	0.0298	0.05	-2.5797	-0.6347
Cp	W	10	Pd	W	30	-1.5805	0.4639	18.55	-3.41	0.0030	0.0357	0.05	-2.5530	-0.6080
Cp	W	30	Pd	E	10	-1.5840	0.4659	18.86	-3.40	0.0030	0.0365	0.05	-2.5596	-0.6084
Cp	W	30	Pd	E	30	-1.3092	0.4659	18.86	-2.81	0.0112	0.1816	0.05	-2.2848	-0.3337
Cp	W	30	Pd	U	10	-1.2765	0.4668	19	-2.73	0.0132	0.2150	0.05	-2.2535	-0.2996
Cp	W	30	Pd	U	30	-1.0037	0.4668	19	-2.15	0.0446	0.5870	0.05	-1.9806	-0.02679
Cp	W	30	Pd	W	10	-1.7815	0.4643	18.61	-3.84	0.0011	0.0084	0.05	-2.7546	-0.8084
Cp	W	30	Pd	W	30	-1.7548	0.4643	18.61	-3.78	0.0013	0.0103	0.05	-2.7279	-0.7817
Pd	E	10	Pd	E	30	0.2748	0.1254	32.65	2.19	0.0356	0.5570	0.05	0.01963	0.5300
Pd	E	10	Pd	U	10	0.3075	0.1931	35.81	1.59	0.1201	0.9107	0.05	-0.08424	0.6992
Pd	E	10	Pd	U	30	0.5803	0.2125	44.15	2.73	0.0091	0.2172	0.05	0.1520	1.0086
Pd	E	10	Pd	W	10	-0.1975	0.1876	32.22	-1.05	0.3002	0.9962	0.05	-0.5795	0.1844
Pd	E	10	Pd	W	30	-0.1708	0.2069	40.08	-0.83	0.4140	0.9996	0.05	-0.5889	0.2473
Pd	E	30	Pd	U	10	0.03266	0.2125	44.15	0.15	0.8786	1.0000	0.05	-0.3956	0.4610
Pd	E	30	Pd	U	30	0.3055	0.1931	35.81	1.58	0.1224	0.9143	0.05	-0.08620	0.6972
Pd	E	30	Pd	W	10	-0.4723	0.2069	40.08	-2.28	0.0278	0.4913	0.05	-0.8904	-0.05420
Pd	E	30	Pd	W	30	-0.4456	0.1876	32.22	-2.38	0.0236	0.4263	0.05	-0.8275	-0.06362
Pd	U	10	Pd	U	30	0.2728	0.1289	36.14	2.12	0.0413	0.6115	0.05	0.01141	0.5343
Pd	U	10	Pd	W	10	-0.5050	0.1898	33.6	-2.66	0.0119	0.2517	0.05	-0.8908	-0.1192
Pd	U	10	Pd	W	30	-0.4782	0.2089	41.49	-2.29	0.0272	0.4866	0.05	-0.8999	-0.05655
Pd	U	30	Pd	W	10	-0.7778	0.2089	41.49	-3.72	0.0006	0.0125	0.05	-1.1995	-0.3561
Pd	U	30	Pd	W	30	-0.7511	0.1898	33.6	-3.96	0.0004	0.0054	0.05	-1.1369	-0.3653
Pd	W	10	Pd	W	30	0.02672	0.1141	22.79	0.23	0.8170	1.0000	0.05	-0.2095	0.2629

SAS PROC GLIMMIX model output example for Zn analysis

The GLIMMIX Procedure

Model Information	
Data Set	WORK.SOIL
Response Variable	Zn_vol
Response Distribution	Lognormal
Link Function	Identity
Variance Function	Default
Variance Matrix	Not blocked
Estimation Technique	Restricted Maximum Likelihood
Degrees of Freedom Method	Kenward-Roger
Fixed Effects SE Adjustment	Kenward-Roger

Class Level Information		
Class	Levels	Values
Region	2	Cp Pd
Location	3	E U W
Depth	2	10 30
Site	16	BLK DEN DOT DRG HNK HOG LKW MUN NAH NSC PCS RDU RKR SIK SPG WMT
T2	6	1 2 3 4 5 6

Number of Observations Read	516
Number of Observations Used	516

Dimensions	
G-side Cov. Parameters	5
R-side Cov. Parameters	1
Columns in X	36
Columns in Z	382
Subjects (Blocks in V)	1
Max Obs per Subject	516

Optimization Information	
Optimization Technique	Dual Quasi-Newton
Parameters in Optimization	5
Lower Boundaries	5
Upper Boundaries	0
Fixed Effects	Profiled

The GLIMMIX Procedure

Optimization Information	
Residual Variance	Profiled
Starting From	Data

Iteration History					
Iteration	Restarts	Evaluations	Objective Function	Change	Max Gradient
0	0	4	1057.1868518	.	23.27906
1	0	2	1055.8699198	1.31693205	121.6199
2	0	7	1050.8755276	4.99439212	86.24108
3	0	5	1050.7047428	0.17078480	71.32648
4	0	3	1050.4368045	0.26793836	50.53305
5	0	2	1049.7471891	0.68961542	12.27583
6	0	3	1049.7032348	0.04395424	4.443641
7	0	2	1049.6799084	0.02332644	1.98022
8	0	3	1049.6675393	0.01236904	0.557871
9	0	3	1049.6673797	0.00015968	0.035636
10	0	3	1049.6673746	0.00000507	0.001733

Convergence criterion (GCONV=1E-8) satisfied.

Fit Statistics	
-2 Res Log Likelihood	1049.67
AIC (smaller is better)	1061.67
AICC (smaller is better)	1061.84
BIC (smaller is better)	1066.30
CAIC (smaller is better)	1072.30
HQIC (smaller is better)	1061.90
Generalized Chi-Square	143.49
Gener. Chi-Square / DF	0.28

Covariance Parameter Estimates		
Cov Parm	Estimate	Standard Error
Site(Region)	0.4163	0.1882
T2(Region*Site)	0.09545	0.03013
Locatio*Site(Region)	0.1014	0.03955
Locat*T2(Regio*Site)	0.03935	0.02432

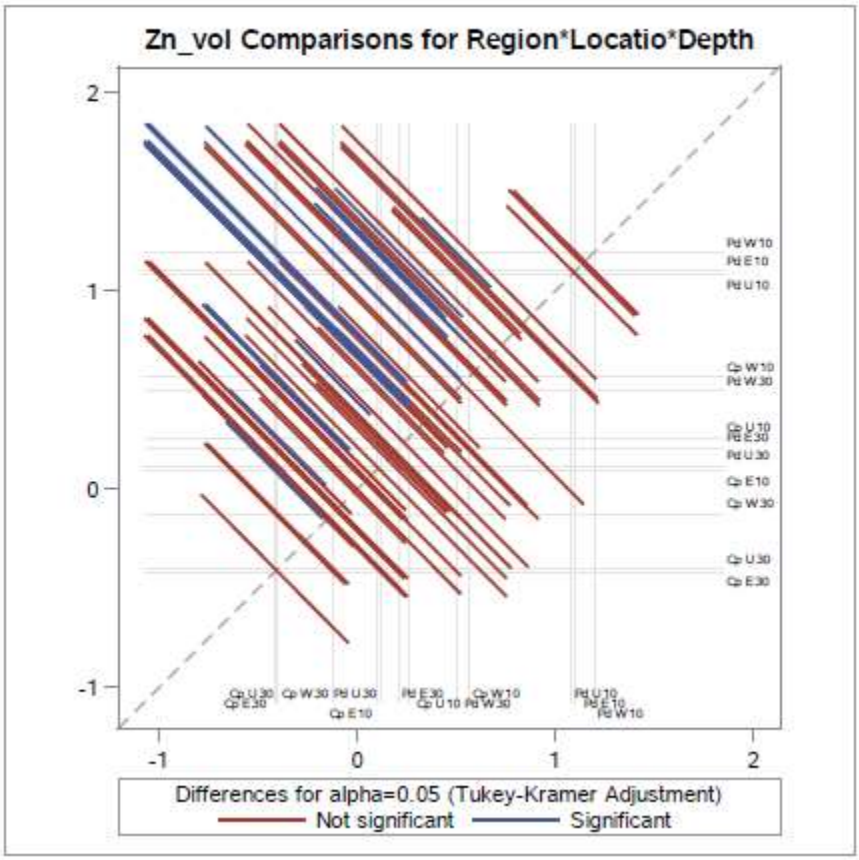
The GLIMMIX Procedure

Covariance Parameter Estimates		
Cov Parm	Estimate	Standard Error
Depth*Site(Region)	0.009123	0.01042
Residual	0.2847	0.02377

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Region	1	13.65	3.98	0.0665
Location	2	25.81	2.98	0.0686
Region*Location	2	25.81	0.28	0.7597
Depth	1	14.5	151.59	<.0001
Region*Depth	1	14.5	3.64	0.0763
Location*Depth	2	295	0.59	0.5532
Region*Locatio*Depth	2	295	1.59	0.2064

Region Least Squares Means								
Region	Estimate	Standard Error	DF	t Value	Pr > |t|	Alpha	Lower	Upper
Cp	-0.00325	0.2652	13.71	-0.01	0.9904	0.05	-0.5732	0.5667
Pd	0.7012	0.2333	13.57	3.01	0.0097	0.05	0.1994	1.2031

The GLIMMIX Procedure



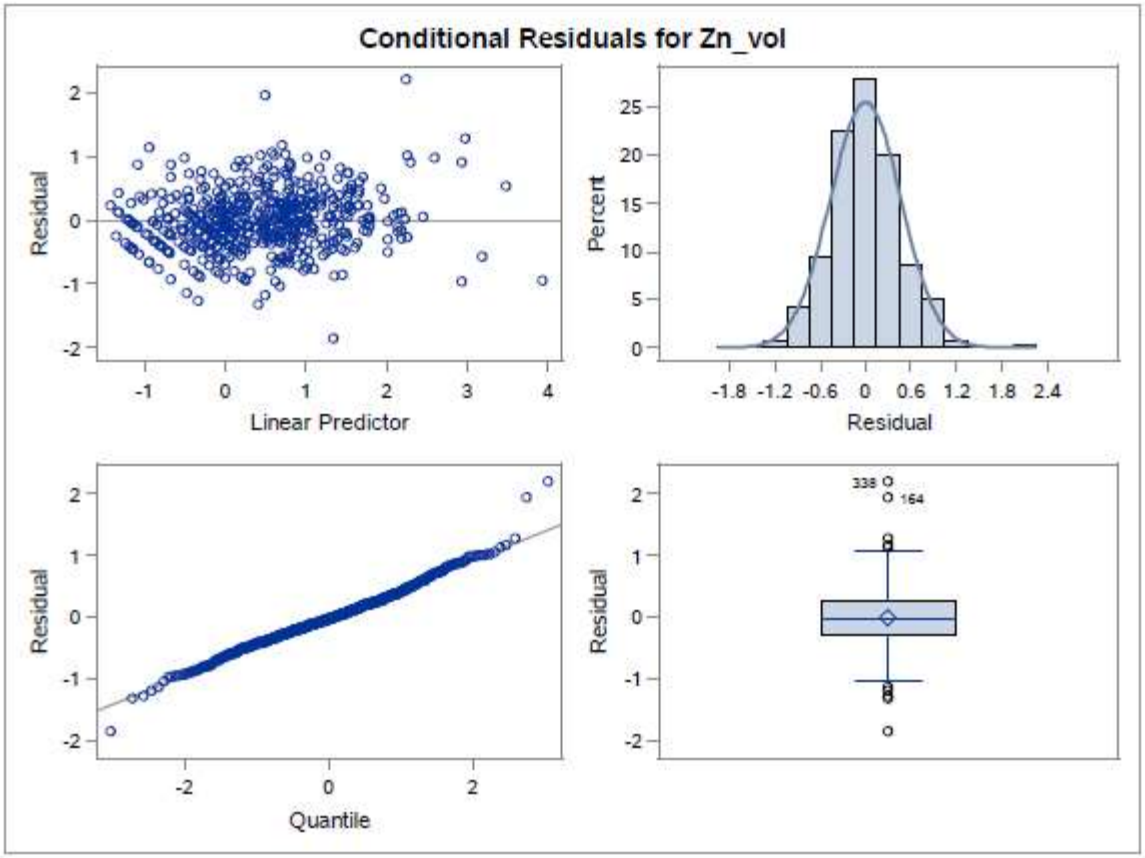
Conservative Tukey-Kramer Grouping for Region*Locatio*Depth Least Squares Means (Alpha=0.05)						
LS-means with the same letter are not significantly different.						
Region	Location	Depth	Estimate			
Pd	W	10	1.1935		A	
					A	
Pd	E	10	1.1031	B	A	
				B	A	
Pd	U	10	1.0828	B	A	
				B	A	
Cp	W	10	0.5662	B	A	
				B	A	
The LINES display does not reflect all significant comparisons. The following additional pairs are significantly different: (Pd W 10,Pd W 30), (Pd W 10,Pd E 30), (Pd W 10,Pd U 30), (Pd E 10,Pd E 30), (Pd E 10,Pd U 30), (Pd U 10,Pd E 30), (Pd U 10,Pd U 30), (Cp W 10,Cp W 30), (Cp U 10,Cp U 30), (Cp E 10,Cp E 30).						

The GLIMMIX Procedure

Conservative Tukey-Kramer Grouping for Region*Locatio*Depth Least Squares Means (Alpha=0.05)						
LS-means with the same letter are not significantly different.						
Region	Location	Depth	Estimate			
Pd	W	30	0.5001	B	A	C
				B	A	C
Cp	U	10	0.2610	B	A	C
				B	A	C
Pd	E	30	0.2083	B	A	C
				B	A	C
Pd	U	30	0.1196	B	A	C
				B	A	C
Cp	E	10	0.09622	B	A	C
				B		C
Cp	W	30	-0.1232	B		C
						C
Cp	U	30	-0.4023			C
						C
Cp	E	30	-0.4175			C

The LINES display does not reflect all significant comparisons.
The following additional pairs are significantly different:
(Pd W 10,Pd W 30), (Pd W 10,Pd E 30),
(Pd W 10,Pd U 30), (Pd E 10,Pd E 30),
(Pd E 10,Pd U 30), (Pd U 10,Pd E 30),
(Pd U 10,Pd U 30), (Cp W 10,Cp W 30),
(Cp U 10,Cp U 30), (Cp E 10,Cp E 30).

The GLIMMIX Procedure



The GLIMMIX Procedure

Differences of Region*Locatio*Depth Least Squares Means Adjustment for Multiple Comparisons: Tukey-Kramer														
Region	Location	Depth	_Region	_Location	_Depth	Estimate	Standard Error	DF	t Value	Pr > |t|	Adj P	Alpha	Lower	Upper
Cp	E	10	Cp	E	30	0.5138	0.1478	88.78	3.48	0.0008	0.0284	0.05	0.2201	0.8074
Cp	E	10	Cp	U	10	-0.1648	0.2267	41.29	-0.73	0.4713	0.9999	0.05	-0.6226	0.2929
Cp	E	10	Cp	U	30	0.4985	0.2329	42.92	2.14	0.0380	0.5942	0.05	0.02878	0.9683
Cp	E	10	Cp	W	10	-0.4700	0.2170	35.11	-2.17	0.0372	0.5755	0.05	-0.9105	-0.02959
Cp	E	10	Cp	W	30	0.2194	0.2227	36.56	0.99	0.3310	0.9979	0.05	-0.2320	0.6708
Cp	E	10	Pd	E	10	-1.0069	0.3989	21.47	-2.52	0.0195	0.3291	0.05	-1.8352	-0.1785
Cp	E	10	Pd	E	30	-0.1121	0.3989	21.47	-0.28	0.7814	1.0000	0.05	-0.9405	0.7163
Cp	E	10	Pd	U	10	-0.9866	0.4004	21.78	-2.46	0.0221	0.3669	0.05	-1.8174	-0.1558
Cp	E	10	Pd	U	30	-0.02336	0.4004	21.78	-0.06	0.9540	1.0000	0.05	-0.8542	0.8074
Cp	E	10	Pd	W	10	-1.0973	0.3957	20.81	-2.77	0.0115	0.1972	0.05	-1.9206	-0.2739
Cp	E	10	Pd	W	30	-0.4039	0.3957	20.81	-1.02	0.3191	0.9971	0.05	-1.2273	0.4194
Cp	E	30	Cp	U	10	-0.6786	0.2329	42.92	-2.91	0.0057	0.1416	0.05	-1.1483	-0.2088
Cp	E	30	Cp	U	30	-0.01523	0.2267	41.29	-0.07	0.9468	1.0000	0.05	-0.4730	0.4425
Cp	E	30	Cp	W	10	-0.9838	0.2227	36.56	-4.42	<.0001	0.0008	0.05	-1.4352	-0.5324
Cp	E	30	Cp	W	30	-0.2944	0.2170	35.11	-1.36	0.1835	0.9705	0.05	-0.7348	0.1461
Cp	E	30	Pd	E	10	-1.5206	0.3989	21.47	-3.81	0.0010	0.0090	0.05	-2.3490	-0.6922
Cp	E	30	Pd	E	30	-0.6259	0.3989	21.47	-1.57	0.1313	0.9188	0.05	-1.4543	0.2025
Cp	E	30	Pd	U	10	-1.5003	0.4004	21.78	-3.75	0.0011	0.0114	0.05	-2.3311	-0.6695
Cp	E	30	Pd	U	30	-0.5371	0.4004	21.78	-1.34	0.1936	0.9729	0.05	-1.3679	0.2937
Cp	E	30	Pd	W	10	-1.6110	0.3957	20.81	-4.07	0.0006	0.0034	0.05	-2.4344	-0.7877
Cp	E	30	Pd	W	30	-0.9177	0.3957	20.81	-2.32	0.0306	0.4654	0.05	-1.7410	-0.09434
Cp	U	10	Cp	U	30	0.6633	0.1478	88.78	4.49	<.0001	0.0006	0.05	0.3697	0.9569
Cp	U	10	Cp	W	10	-0.3052	0.2170	35.11	-1.41	0.1683	0.9616	0.05	-0.7456	0.1352
Cp	U	10	Cp	W	30	0.3842	0.2227	36.56	1.73	0.0929	0.8553	0.05	-0.06721	0.8356
Cp	U	10	Pd	E	10	-0.8420	0.3989	21.47	-2.11	0.0467	0.6153	0.05	-1.6704	-0.01365
Cp	U	10	Pd	E	30	0.05270	0.3989	21.47	0.13	0.8961	1.0000	0.05	-0.7757	0.8811
Cp	U	10	Pd	U	10	-0.8218	0.4004	21.78	-2.05	0.0523	0.6569	0.05	-1.6526	0.009048
Cp	U	10	Pd	U	30	0.1414	0.4004	21.78	0.35	0.7273	1.0000	0.05	-0.6894	0.9723
Cp	U	10	Pd	W	10	-0.9325	0.3957	20.81	-2.36	0.0283	0.4392	0.05	-1.7558	-0.1091
Cp	U	10	Pd	W	30	-0.2391	0.3957	20.81	-0.60	0.5522	1.0000	0.05	-1.0625	0.5842
Cp	U	30	Cp	W	10	-0.9685	0.2227	36.56	-4.35	0.0001	0.0011	0.05	-1.4200	-0.5171
Cp	U	30	Cp	W	30	-0.2791	0.2170	35.11	-1.29	0.2067	0.9803	0.05	-0.7196	0.1613
Cp	U	30	Pd	E	10	-1.5054	0.3989	21.47	-3.77	0.0011	0.0104	0.05	-2.3338	-0.6770
Cp	U	30	Pd	E	30	-0.6106	0.3989	21.47	-1.53	0.1404	0.9309	0.05	-1.4390	0.2178

The GLIMMIX Procedure

Differences of Region*Locatio*Depth Least Squares Means Adjustment for Multiple Comparisons: Tukey-Kramer														
Region	Location	Depth	_Region	_Location	_Depth	Estimate	Standard Error	DF	t Value	Pr > |t|	Adj P	Alpha	Lower	Upper
Cp	U	30	Pd	U	10	-1.4851	0.4004	21.78	-3.71	0.0012	0.0130	0.05	-2.3159	-0.6543
Cp	U	30	Pd	U	30	-0.5219	0.4004	21.78	-1.30	0.2060	0.9782	0.05	-1.3527	0.3089
Cp	U	30	Pd	W	10	-1.5958	0.3957	20.81	-4.03	0.0006	0.0040	0.05	-2.4191	-0.7724
Cp	U	30	Pd	W	30	-0.9025	0.3957	20.81	-2.28	0.0332	0.4928	0.05	-1.7258	-0.07911
Cp	W	10	Cp	W	30	0.6894	0.1142	37.25	6.03	<.0001	<.0001	0.05	0.4580	0.9208
Cp	W	10	Pd	E	10	-0.5368	0.3931	20.28	-1.37	0.1870	0.9690	0.05	-1.3561	0.2825
Cp	W	10	Pd	E	30	0.3579	0.3931	20.28	0.91	0.3733	0.9990	0.05	-0.4614	1.1772
Cp	W	10	Pd	U	10	-0.5165	0.3946	20.58	-1.31	0.2050	0.9775	0.05	-1.3383	0.3052
Cp	W	10	Pd	U	30	0.4467	0.3946	20.58	1.13	0.2707	0.9930	0.05	-0.3751	1.2684
Cp	W	10	Pd	W	10	-0.6272	0.3899	19.64	-1.61	0.1236	0.9048	0.05	-1.4415	0.1870
Cp	W	10	Pd	W	30	0.06609	0.3899	19.64	0.17	0.8671	1.0000	0.05	-0.7482	0.8804
Cp	W	30	Pd	E	10	-1.2262	0.3931	20.28	-3.12	0.0053	0.0827	0.05	-2.0455	-0.4069
Cp	W	30	Pd	E	30	-0.3315	0.3931	20.28	-0.84	0.4089	0.9995	0.05	-1.1508	0.4878
Cp	W	30	Pd	U	10	-1.2060	0.3946	20.58	-3.06	0.0061	0.0982	0.05	-2.0277	-0.3842
Cp	W	30	Pd	U	30	-0.2427	0.3946	20.58	-0.62	0.5452	1.0000	0.05	-1.0645	0.5790
Cp	W	30	Pd	W	10	-1.3166	0.3899	19.64	-3.38	0.0031	0.0389	0.05	-2.1309	-0.5024
Cp	W	30	Pd	W	30	-0.6233	0.3899	19.64	-1.60	0.1259	0.9085	0.05	-1.4376	0.1910
Pd	E	10	Pd	E	30	0.8947	0.1255	78.75	7.13	<.0001	<.0001	0.05	0.6448	1.1447
Pd	E	10	Pd	U	10	0.02029	0.1994	40.57	0.10	0.9195	1.0000	0.05	-0.3826	0.4232
Pd	E	10	Pd	U	30	0.9835	0.2049	42.17	4.80	<.0001	0.0002	0.05	0.5701	1.3969
Pd	E	10	Pd	W	10	-0.09041	0.1907	34.61	-0.47	0.6384	1.0000	0.05	-0.4777	0.2969
Pd	E	10	Pd	W	30	0.6029	0.1957	36.05	3.08	0.0039	0.0917	0.05	0.2061	0.9997
Pd	E	30	Pd	U	10	-0.8745	0.2049	42.17	-4.27	0.0001	0.0016	0.05	-1.2879	-0.4610
Pd	E	30	Pd	U	30	0.08874	0.1994	40.57	0.44	0.6587	1.0000	0.05	-0.3142	0.4917
Pd	E	30	Pd	W	10	-0.9852	0.1957	36.05	-5.03	<.0001	<.0001	0.05	-1.3820	-0.5883
Pd	E	30	Pd	W	30	-0.2918	0.1907	34.61	-1.53	0.1351	0.9311	0.05	-0.6791	0.09549
Pd	U	10	Pd	U	30	0.9632	0.1314	88.81	7.33	<.0001	<.0001	0.05	0.7020	1.2244
Pd	U	10	Pd	W	10	-0.1107	0.1938	36.62	-0.57	0.5714	1.0000	0.05	-0.5035	0.2822
Pd	U	10	Pd	W	30	0.5826	0.1987	38.03	2.93	0.0057	0.1352	0.05	0.1804	0.9849
Pd	U	30	Pd	W	10	-1.0739	0.1987	38.03	-5.40	<.0001	<.0001	0.05	-1.4762	-0.6716
Pd	U	30	Pd	W	30	-0.3806	0.1938	36.62	-1.96	0.0572	0.7176	0.05	-0.7734	0.01228
Pd	W	10	Pd	W	30	0.6933	0.1056	44.08	6.56	<.0001	<.0001	0.05	0.4805	0.9062

Appendix P: R Code and Results for Linear Regression

Output from SLR for total Cu as a function of Mehlich 3 Cu

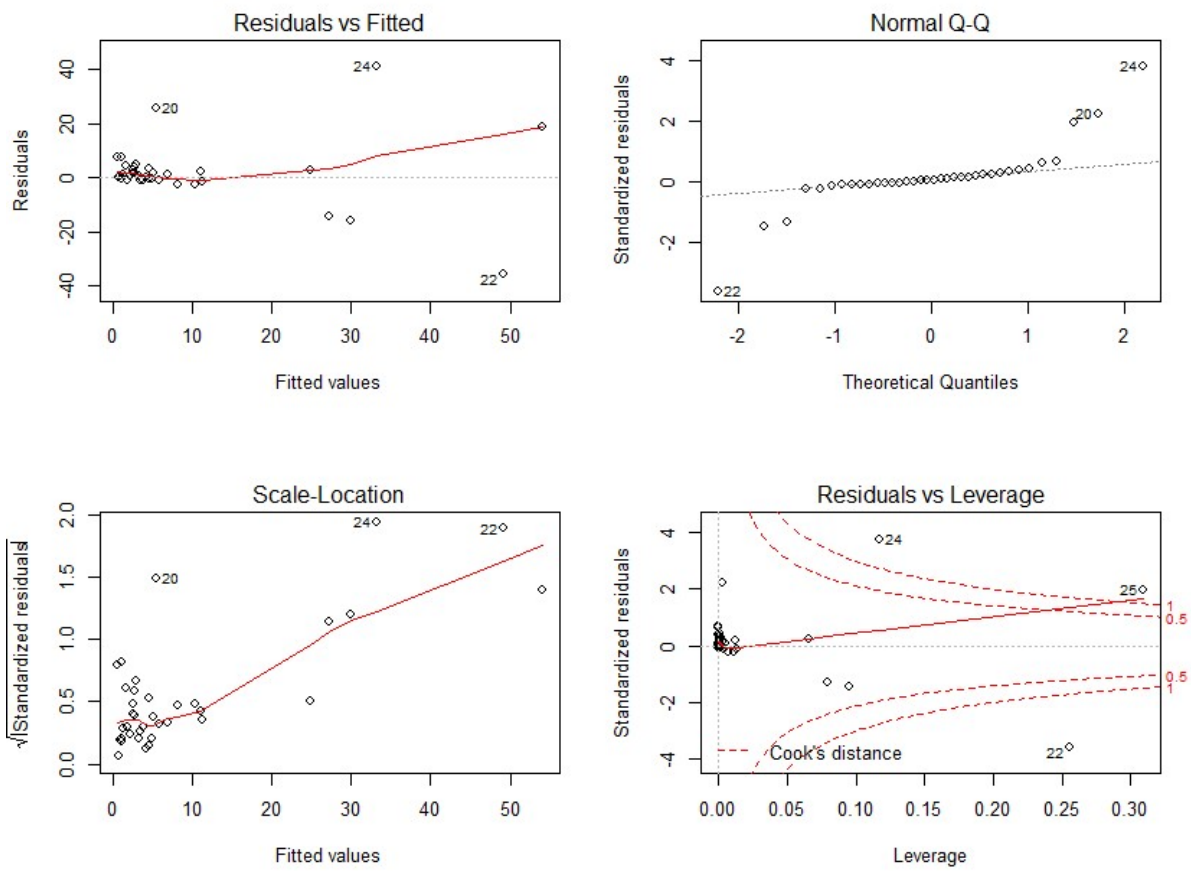
Coastal Plain

```
Call:
lm(formula = Cu_t ~ 0 + Cu, data = xc)

Residuals:
    Min      1Q  Median      3Q     Max 
-35.994  -0.841   0.576   2.912  41.284 

Coefficients:
   Estimate Std. Error t value Pr(>|t|)    
Cu   7.3657     0.8814   8.357 7.44e-10 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 11.62 on 35 degrees of freedom
Multiple R-squared:  0.6662,	Adjusted R-squared:  0.6566 
F-statistic: 69.84 on 1 and 35 DF,  p-value: 7.436e-10
```



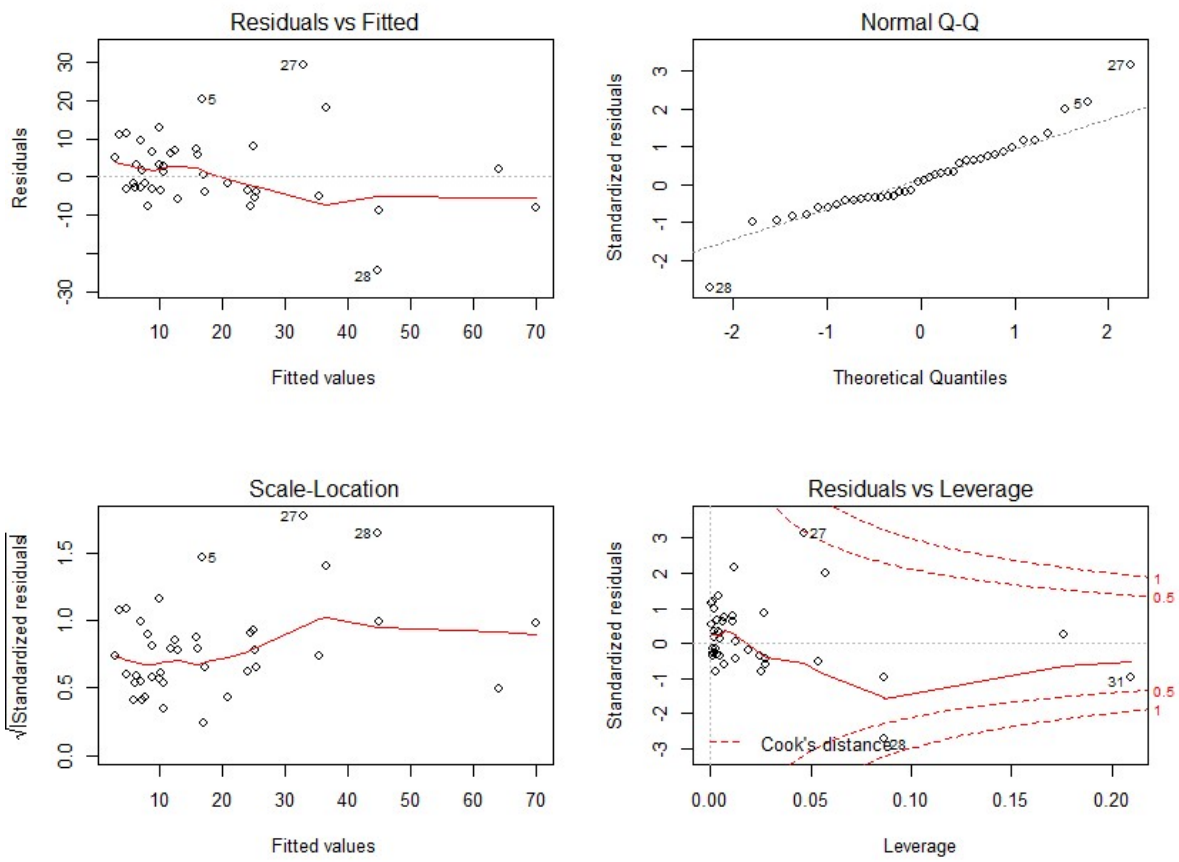
Piedmont

```
Call:
lm(formula = Cu_t ~ 0 + Cu, data = xpd)

Residuals:
     Min       1Q   Median       3Q      Max 
-24.7076  -3.7080   0.8338   6.4466  29.1935 

Coefficients:
   Estimate Std. Error t value Pr(>|t|)    
Cu   8.7806     0.5449   16.11   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 9.467 on 39 degrees of freedom
Multiple R-squared:  0.8694,	Adjusted R-squared:  0.8661 
F-statistic: 259.7 on 1 and 39 DF,  p-value: < 2.2e-16
```



Output from SLR for total Zn as a function of Mehlich 3 Zn

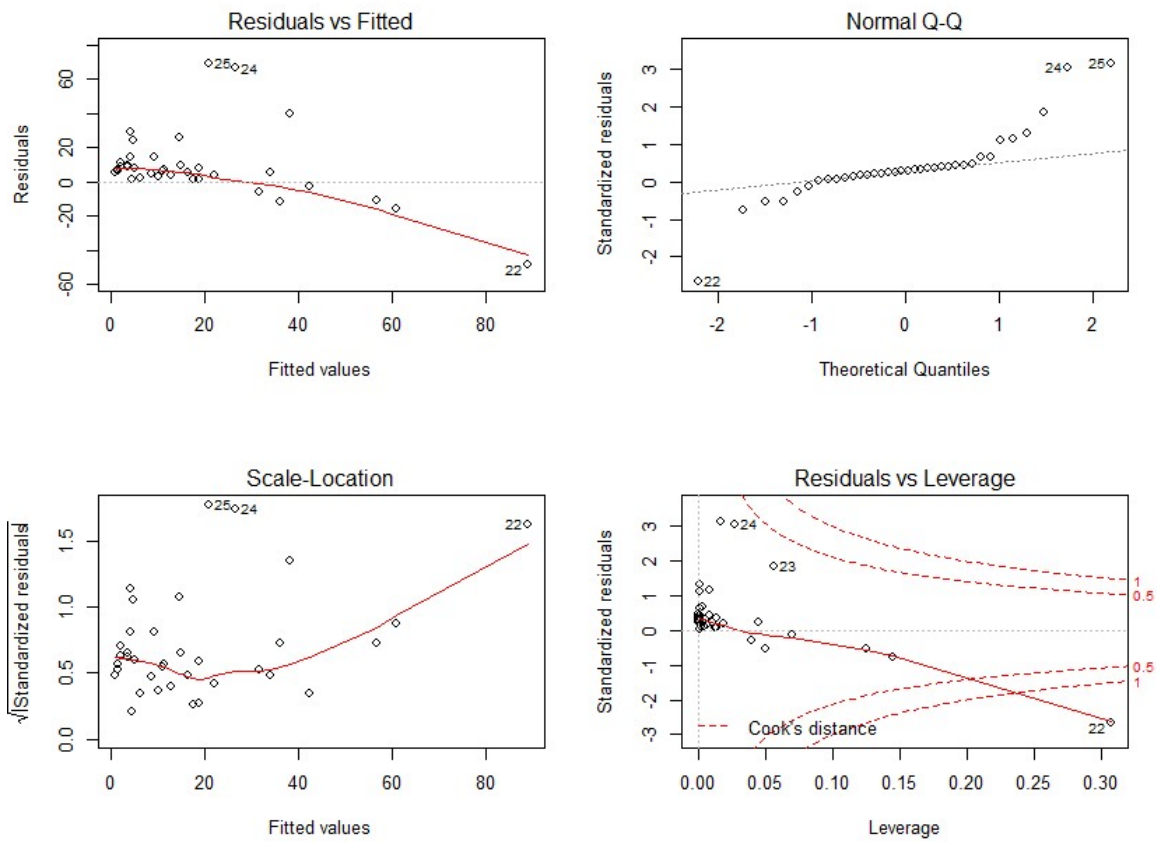
Coastal Plain

```
Call:
lm(formula = Zn_t ~ 0 + Zn, data = xc)

Residuals:
    Min      1Q  Median      3Q     Max 
-48.801   2.313   6.344   9.663  68.928 

Coefficients:
   Estimate Std. Error t value Pr(>|t|)    
Zn   3.0326     0.4174   7.265 1.74e-08 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 22.04 on 35 degrees of freedom
Multiple R-squared:  0.6013,	Adjusted R-squared:  0.5899 
F-statistic: 52.78 on 1 and 35 DF,  p-value: 1.742e-08
```



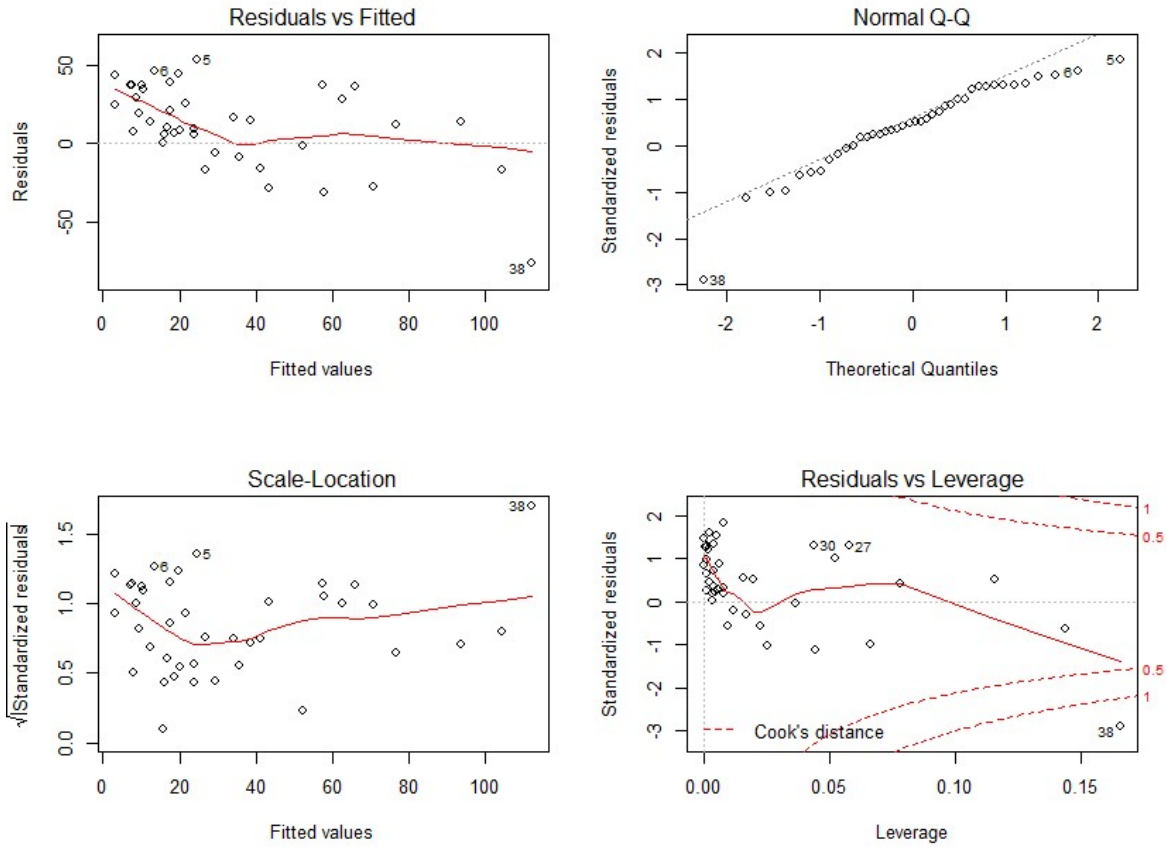
Piedmont

```
Call:
lm(formula = Zn_t ~ 0 + Zn, data = xpd)

Residuals:
    Min      1Q  Median      3Q     Max 
-76.426  -0.148  13.611  35.133  53.103 

Coefficients:
   Estimate Std. Error t value Pr(>|t|)    
Zn   8.9123     0.9377   9.504 1.06e-11 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 28.91 on 39 degrees of freedom
Multiple R-squared:  0.6985,	Adjusted R-squared:  0.6907 
F-statistic: 90.33 on 1 and 39 DF,  p-value: 1.058e-11
```



R code for MLR with stepwise variable selection

```
null<- lm(Cu_t~Cu+0,data=xc)        #null model with
full <- lm(Cu_t~Cu+OM+WV+pH+CEC+HM+0,data=xc)
step(null, scope = list(upper=full), data=xc, direction="both")
```

R output for MLR with stepwise variable selection

Cu – Coastal Plain

```
Start:  AIC=173.7
Cu_t ~ Cu + 0

       Df Sum of Sq     RSS    AIC
<none>               4727.7 173.70
+ WV    1     189.0  4538.7 174.28
+ pH    1      76.5  4651.2 175.13
+ HM    1      18.8  4708.9 175.56
+ OM    1       5.1  4722.5 175.67
+ CEC   1       1.1  4726.5 175.70
- Cu    1    9426.4 14154.1 210.08

Call:
lm(formula = Cu_t ~ Cu + 0, data = xc)

Coefficients:
   Cu
7.365
```

Cu – Piedmont

```
Start:  AIC=184.38
Cu_t ~ Cu + 0

       Df Sum of Sq     RSS    AIC
+ CEC   1     774.5  2730.2 176.14
+ OM    1     362.7  3142.0 181.90
+ pH    1     286.6  3218.1 182.88
+ WV    1     204.8  3299.9 183.91
<none>               3504.7 184.38
+ HM    1       0.4  3504.3 186.38
- Cu    1   24352.6 27857.3 267.37

Step:  AIC=176.14
Cu_t ~ Cu + CEC - 1

       Df Sum of Sq    RSS    AIC
+ HM    1     217.5 2512.8 174.74
<none>              2730.2 176.14
+ WV    1     122.4 2607.8 176.26
+ pH    1      96.6 2633.6 176.66
+ OM    1      15.4 2714.9 177.91
- CEC   1     774.5 3504.7 184.38
- Cu    1    3211.0 5941.3 206.02

Step:  AIC=174.74
```

```
Cu_t ~ Cu + CEC + HM - 1

       Df Sum of Sq    RSS    AIC
<none>              2512.8 174.74
+ WV    1     64.40 2448.4 175.67
+ pH    1     37.52 2475.2 176.12
- HM    1    217.47 2730.2 176.14
+ OM    1     33.04 2479.7 176.19
- CEC   1    991.53 3504.3 186.38
- Cu    1   2678.82 5191.6 202.49

Call:
lm(formula = Cu_t ~ Cu + CEC + HM - 1, data = xpd)

Coefficients:
    Cu     CEC      HM
 5.823   1.318  -1.818
```

Appendix Q: Climate Models Used in this Study

Table Q.1: Climate models evaluated for this project

Model Designation	Organization
access1-0.1	Australian Community Climate and Earth-System Simulator, version 1.0
access1-3.1	Australian Community Climate and Earth-System Simulator, version 1.3
bcc-csm1-1.1	Beijing Climate Center, Climate System Model, version 1.1
bcc-csm1-1-m.1	Beijing Climate Center, Climate System Model, version 1.1 (moderate resolution)
canesm2.1	Second Generation Canadian Earth System Model
ccsm4.6	Community Climate System Model, version 4
cesm1-bgc.1	Community Earth System Model, version 1 (biogeochemistry, or carbon cycle)
cesm1-cam5.1	Community Earth System Model, version 1 (Community Atmosphere Model, version 5)
cmcc-cm.1	Centro Euro-Mediterraneo per I Cambiamenti Climatici Climate Model
cmcc-cms.1	Centro Euro-Mediterraneo per I Cambiamenti Climatici Stratosphere-resolving Climate Model
cnrm-cm5.1	Centre National de Recherches Météorologiques Coupled Global Climate Model, version 5
csiro-mk3-6-0.1	Commonwealth Scientific and Industrial Research Organisation Mark 3.6.0
ec-earth.2	European Consortium Earth System Model
fgoals-g2.1	Flexible Global Ocean–Atmosphere–Land System Model, gridpoint version 1.0
gfdl-cm3.1	Geophysical Fluid Dynamics Laboratory Climate Model, version 3
gfdl-esm2g.1	Geophysical Fluid Dynamics Laboratory Earth System Model with GOLD component
gfdl-esm2m.1	Geophysical Fluid Dynamics Laboratory Earth System Model with MOM, version 4 component
giss-e2-h.2	Goddard Institute for Space Studies Model E2, coupled with HYCOM
giss-e2-r.2	Goddard Institute for Space Studies Model E2, coupled with the Russell ocean model
hadgem2-ao.1	Hadley Centre Global Environment Model, version *2 Atmosphere and land and Ocean and sea ice components*
hadgem2-cc.1	Hadley Centre Global Environment Model, version *2 Carbon Cycle (includes AO configuration with biogeochemistry, sometimes includes CCS, with S for stratospheric processes)*
hadgem2-es.1	Hadley Centre Global Environment Model, version *2 Earth System Model*
inmcm4.1	Institute of Numerical Mathematics Coupled Model, version 4.0
ipsl-cm5a-lr.1	L'Institut Pierre-Simon Laplace Coupled Model, version 5A, low resolution
ipsl-cm5a-mr.1	L'Institut Pierre-Simon Laplace Coupled Model, version 5A, mid resolution
miroc5.1	Model for Interdisciplinary Research on Climate, version 5
miroc-esm.1	Model for Interdisciplinary Research on Climate, Earth System Model
miroc-esm-chem.1	Model for Interdisciplinary Research on Climate, Earth System Model, Chemistry Coupled
mpi-esm-lr.1	Max Planck Institute Earth System Model, low resolution
mpi-esm-mr.1	Max Planck Institute Earth System Model, medium resolution
mri-cgcm3.1	Meteorological Research Institute Coupled Atmosphere–Ocean General Circulation Model, version 3
noresm1-m.1	Norwegian Earth System Model, version 1 (intermediate resolution)

Appendix R: DRAINMOD Soil Inputs

Table R.1: Volume drained and upward flux inputs for GDS site.

Water Table (cm)	Volume Drained (cm)	Upward Flux (cm/hr)
0	0	2
10	5	2
15	6	2
20	7	2
30	10	2
40	12	0.5
50	13	0.2
60	15	0.004
80	16	0.0001
100	18	0.0001
120	20	0.0001
140	22	0.0001
160	24	0.0001
180	26	0.0001
200	28	0
1000	100	0

Table R.2: Volume drained and upward flux inputs for NRF site.

Water Table (cm)	Volume Drained (cm)	Upward Flux (cm/hr)
0	0	1.5
5	2.5	1.5
10	5	1.5
15	7.5	1.5
20	10	1.5
25	12	1.5
30	15	1.5
35	17	0.733
40	18	0.733
50	19	0.0508
60	20	0.0083
70	21	0.004
80	22	0.002
90	23	0.002
100	24	0.002
110	24.5	0.002
120	25	0.002
130	25.6	0.002
140	26	0.002
150	26.5	0.002
160	27	0.002
170	27.5	0.0001
180	28	0.0001
190	29	0
200	30	0
1000	100	0

Appendix S: ET Estimates for each Scenario

Table S.1: Average annual ET from DRAINMOD simulations. All values are in cm.

Great Dismal Swamp (GDS)									
GDS - RCP4.5: 1986-2015					**GDS - RCP8.5: 1986-2015**				
Observed: 88		**ΔP**			**Observed: 88**		**ΔP**		
		Dry	**Mean**	**Wet**			**Dry**	**Mean**	**Wet**
ΔT	**Hot**	85	89	88	**ΔT**	**Hot**	90	89	90
	Mean	90	89	88		**Mean**	91	90	89
	Warm	88	88	91		**Warm**	88	90	86
GDS - RCP4.5: 2040-2069					**GDS - RCP8.5: 2040-2069**				
		ΔP					**ΔP**		
		Dry	**Mean**	**Wet**			**Dry**	**Mean**	**Wet**
ΔT	**Hot**	101	106	105	ΔT	**Hot**	109	105	116
	Mean	103	103	105		**Mean**	108	108	107
	Warm	93	98	98		**Warm**	103	101	99
GDS - RCP4.5: 2070-2099					**GDS - RCP8.5: 2070-2099**				
		ΔP					**ΔP**		
		Dry	**Mean**	**Wet**			**Dry**	**Mean**	**Wet**
ΔT	**Hot**	102	106	112	ΔT	**Hot**	113	114	128
	Mean	107	107	108		**Mean**	110	111	120
	Warm	97	98	102		**Warm**	111	106	107
North River Farms (NRF)									
NRF- RCP4.5: 1986-2015					**NRF- RCP8.5: 1986-2015**				
Observed: 100		**ΔP**			**Observed: 100**		**ΔP**		
		Dry	**Mean**	**Wet**			**Dry**	**Mean**	**Wet**
ΔT	**Hot**	102	101	99	ΔT	**Hot**	102	99	99
	Mean	102	100	101		**Mean**	101	99	101
	Warm	100	102	98		**Warm**	99	100	101
NRF - RCP4.5: 2040-2069					**NRF - RCP8.5: 2040-2069**				
		ΔP					**ΔP**		
		Dry	**Mean**	**Wet**			**Dry**	**Mean**	**Wet**
ΔT	**Hot**	125	116	118	ΔT	**Hot**	117	118	120
	Mean	113	113	113		**Mean**	112	119	120
	Warm	109	110	106		**Warm**	115	114	118
NRF- RCP4.5: 2070-2099					**NRF- RCP8.5: 2070-2099**				
		ΔP					**ΔP**		
		Dry	**Mean**	**Wet**			**Dry**	**Mean**	**Wet**
ΔT	**Hot**	123	119	123	ΔT	**Hot**	119	123	132
	Mean	113	116	117		**Mean**	117	124	131
	Warm	110	113	109		**Warm**	118	125	130