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Impact of control structures on hydrologic restoration within the Great Dismal Swamp

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ABSTRACT

The Great Dismal Swamp (GDS) is a 45,000-ha state and federally protected Coastal Plain peatland located on the border of North Carolina and Virginia that contains stands of Bald cypress and the globally threatened Atlantic white cedar. Centuries of drainage and logging have substantially altered the hydrology of the GDS, negatively affecting its ecosystem structure and function. To restore a seasonally flooded, saturated hydrologic regime to portions of the swamp, adjustable water control structures (WCS) were installed at strategic locations within existing drainage ditches. The objective of this study was to determine if the installation of the WCSs significantly altered the hydropatterns of two target restoration areas, resulting in hydrologic conditions comparable to nearby reference sites with desired forest communities. The water table (WT) was monitored for three years prior to WCS installation (pre-WCS) and three years after WCS installation (post-WCS). Comparison of WT data from the pre and post-WCS periods, using jurisdictional wetland criteria and empirical cumulative distribution functions (ECDFs), indicated increased saturated conditions within the target restoration areas following installation of the WCS. Paired Before-After Control-Impact (BACIP) statistical analysis revealed the WCS installation had a significant positive impact on WT levels in the target restoration areas relative to the reference sites. Hydrologic restoration will aid the effort to restore target forest communities within the swamp, reduce fire susceptibility, prevent peat oxidation, maintain carbon storage, and reduce non-target vegetation competition.

1. Introduction

Pocosins and associated peatlands once covered roughly 1.2 million ha of the southeastern Coastal Plain and spanned approximately 1000-km from Virginia to northern Florida (Richardson, 1983). These ecosystems have saturated, semi-permanently, intermittently, or seasonally flooded hydrologic regimes with deep peat accumulations (Cowardin et al., 1979). In this environment, *Taxodium distichum* (Bald cypress) and the globally rare *Chamaecyparis thyoides* (Atlantic white cedar) stands once formed dominant forest communities. However, extensive logging, conversion to agriculture, and commercial development in the Coastal Plain has greatly reduced the extent of these forests over the last 300 years (Drexler et al., 2017; Frost, 1987; Kuser and Zimmerman, 1995; Richardson, 2012). The remaining peatlands, although still intact, have been subjected to anthropogenic alteration via logging or drainage (Ferrell et al., 2007; Frost, 1987; Levy and Walker, 1979). One of these remaining anthropogenically altered peatlands is the Great Dismal Swamp (GDS) of North Carolina and Virginia.

Alteration of the hydrologic regime in GDS began in the late 18th century. By the end of the 19th century, five major ditches had been constructed, including the hydrologically imposing Dismal Swamp Canal (Levy, 1991; US, 2006). Since European settlement, approximately 320-km of ditches and canals have been excavated to allow for the harvest of the swamp's timber, the reclamation of land for development, and the transport of goods between the Chesapeake Bay and the Albemarle Sound (US, 2006). Extensive logging paired with the residual effects of the remaining drainage systems have hampered the regeneration of historic flora, and resulted in the transition of the GDS to a mesic forest community dominated by facultative red maple/black gum forest (Laderman, 1989; US, 2006). The once abundant stands of obligate wetland species, including Bald cypress (BC) and Atlantic white cedar (AWC), covered less than 20% of the total GDS area in 2006 (Frost, 1987; Laing et al., 2011; US, 2006). The drier conditions have also led to increased susceptibility to major peat fires, several of which occurred in the last century (1923 to 1926, 1975, 2008, and 2011). These peat fires have not only contributed to the altered ecological state of the GDS seen today (Richardson, 1982; Laderman, 1989), but also released tons of stored carbon to the atmosphere. The 2011 fire was

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estimated to have released between 0.06 and 1.10 Tg of stored carbon (Reddy et al., 2015).

Because hydrology is the primary control of wetland structure and function (Hunt et al., 1999; National Research Council, 1992, 1995; Zedler, 2000), the reestablishment of a higher water table (WT) was recognized as a prerequisite for successful wetland restoration (Chimner et al., 2017). However, the necessary magnitude of the WT increase required in the GDS was unknown. In their extensive report on the GDS, Lichtler and Walker (1974) observed, “Restoring the (Great Dismal) swamp to its original condition is impossible because that condition is unknown.” Nevertheless, a strategy to raise the WT in portions of the GDS was viewed as the most prudent and cost-effective method to reduce fire susceptibility, prevent peat oxidation, maintain carbon storage, and begin the restoration of the historically dominant forest communities within the swamp.

A plan was developed to create a hydropattern that would increase the frequency and duration of saturation in the soil profile for a portion of the park without impact to existing roads and trails. This included the installation of adjustable water control structures (WCS) at strategic locations in the existing canal system. A WCS consists of a flashboard riser attached to a culvert. Adjustable riser boards, or stop-logs, in the riser structure are used to set the canal water level and slow drainage from target restoration areas (Fig. 1). When installed and managed properly, WCS decrease the hydraulic gradient from the surrounding swamp to the ditch, slowing the drainage of groundwater and producing wetter conditions within surrounding target wetland areas.

Similar ditch blocking approaches to hydrologic restoration have been applied to other peatlands with success. Landry and Rochefort (2012) provide a detailed overview of peatland drainage impacts and several different rewetting techniques, including recommendations for the design of WCSs. Jaenicke et al. (2010) investigated the potential use of similar ditch blocking structures to dam drainage canals in two catchments that drain peat domes in Indonesia. Hydrological modelling of the two systems predicted a 50 to 70 cm rise of groundwater levels during very dry conditions and an average groundwater level rise of 20 cm over a three-year period after the installation of ditch dams. A comprehensive study on the hydrologic restoration of drained peatlands was conducted by Menberu et al. (2016). The study evaluated the impact of ditch blocking techniques on peatland hydropattern in previously drained boreal peatlands of Finland. Hydropatterns were monitored at 24 previously drained sites for one to two years before restoration and one to six years after restoration; 19 pristine peatlands were also monitored and used as control sites. The WTs in 22 of the 24 drained sites were significantly deeper than the corresponding control sites before restoration. After restoration, 12 restoration sites had WTs significantly higher than their corresponding control site WT, 10 restoration sites still had WTs significantly lower than their corresponding control site WT, and 2 restoration sites had WTs that were not significantly different than their corresponding control site WT. The average increases in WT because of restoration were 35.0 cm, 17.4 cm, and 13.3 cm, for spruce mires, pine mires, and fens, respectively.

Locally, WCS have been used in North Carolina since the 1970s to improve downstream water quality by restricting the volume of drainage water released from artificially drained agricultural fields. Gilliam et al. (1979) observed that WCS installed on tile mains or in the outlet ditches increased water table elevation, provided effective control of the water table, and reduced annual drainage volume by approximately 50%. The ability of WCSs to increase upstream WT elevation led to the application of WCSs in hydrologic restoration projects for wetlands that had been ditched, drained, and converted to agriculture (Tweedy and Evans, 2001; Jarzemsky et al., 2013). Both studies indicated that the technique of plugging drainage ditches with WCSs had the potential to restore jurisdictional wetland hydrology to previously converted wetlands.

Although success has been demonstrated, Menberu et al. (2016) noted that “restoration of peatland hydrology is still poorly documented”, WCS have been implemented in North Carolina on drainage ditches in converted agricultural lands but there is limited data on the use of WCS to restore wetland hydrology in forested peatlands that have been ditched for silviculture. In one of the only papers on hydrologic restoration in North Carolina forested peatlands, Wurster et al. (2016) detail the use of WCS for hydrologic restoration, but there was no WT data associated with this report.

This study was conducted to monitor hydrologic conditions within two target restoration sites and two reference wetland areas with desirable forest communities (AWC and BC) before and after WCS installation. The objectives were to 1) assess the hydrology of the target restoration areas, 2) compare the hydrologic conditions in the target restoration areas to those in nearby reference wetland communities and 3) evaluate the impact of WCS installation on the hydropattern of the target restoration areas. To the best of our knowledge, this is the one of the few studies to quantify, over an extended period, the impact of WCS on the WT in previously drained forested peatlands in North Carolina.

2. Materials and methods

2.1. Study site location

The GDS is comprised of the Great Dismal Swamp National Wildlife Refuge, managed by the US Fish and Wildlife Service (US FWS), and the Dismal Swamp State Park (DISW), managed by NC Department of Natural and Cultural Resources. Research was conducted at the 5840-ha

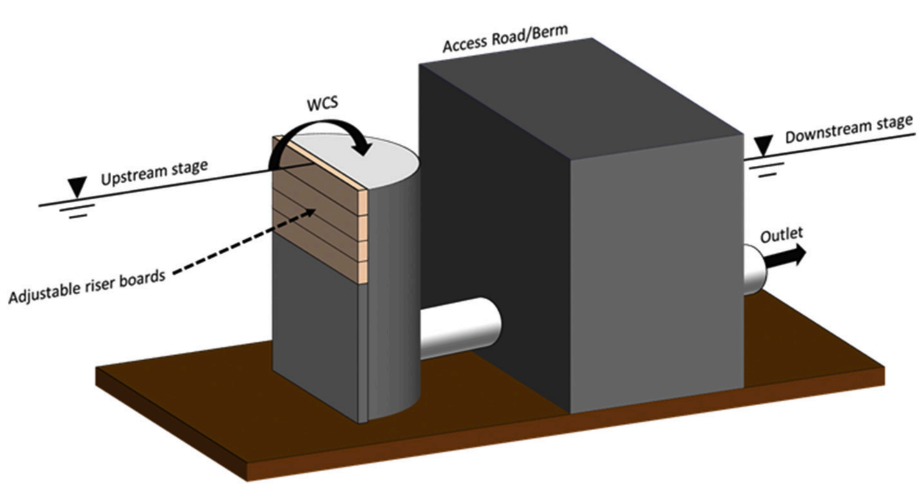

Fig. 1. Schematic of water control structures as installed in the Kim Saunders Ditch.

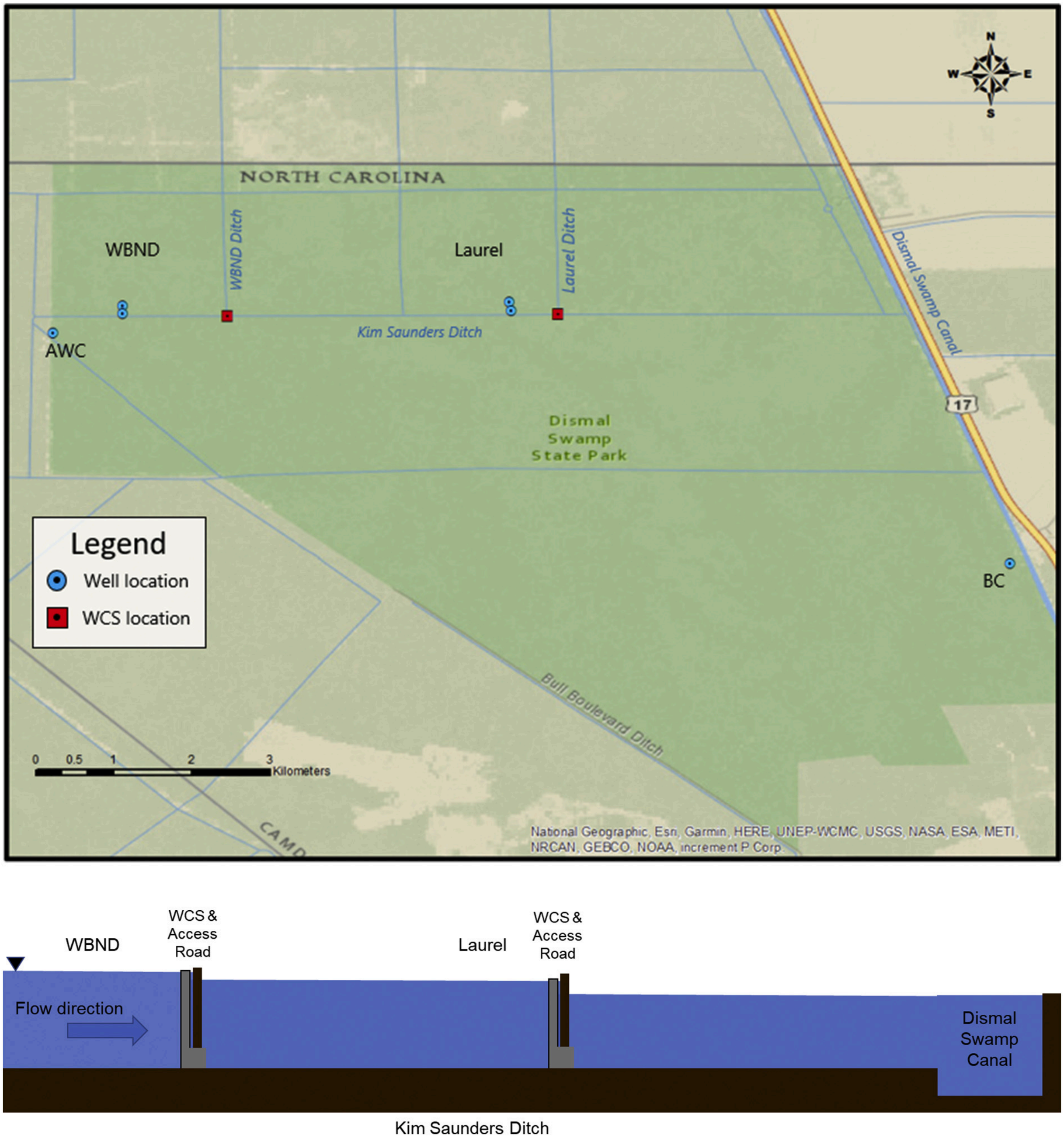

Fig. 2. Top: Well and water control structure (WCS) locations within the Dismal Swamp State Park. Bottom: General profile of Kim Saunders Ditch with approximate locations and desired effect of the installed WCS (shown in gray).

DISW located in Camden County in northeastern North Carolina (Fig. 2). The DISW experiences hot, humid summers and mild winters with an average yearly precipitation of 135 cm (54-in.) based on observations from 1987 to 2017 at the nearby Wallaceton/Lake Drummond, VA WETS station (NRCS, 2019). Two main soil types occupy 98.6% the DISW: Pungo muck (88.6%) and Belhaven muck (10%). Pungo muck is a dysic, thermic Typic Haplosaprist with an organic layer 1 to 2-m in depth, and Belhaven muck is a loamy, mixed, dysic, thermic Terric Haplosaprist (NRCS, 2018). These soil types match the descriptions from Oaks and Coch (1973), NRCS (1995) and US FWS (2013) that reported organic soil depths of 1 to 3-m within the DISW and were consistent with our soil boring observations.

The target restoration area in this study borders the northern side of a 6-km stretch of the Kim Saunders ditch, which runs west to east through the center of the DISW and drains to the Dismal Swamp Canal. The area is divided by ditches into three sections, each approximately 2.6 km^2 (1-mi^2) in extent. Two of these sections, the Western Boundary site (WBND) in the northwest corner of the state park and the Laurel site (Laurel) located further downstream and bordered on the east by the Laurel ditch, were the target restoration areas in this study. Vegetation within the restoration areas was primarily pine pocosins and maple-gum forest, which matched observations by Sleeter et al. (2017). Within WBND, a few small AWC stands were present.

In November of 2015, two WCSs were installed approximately 4-km apart in the Kim Saunders ditch (Fig. 2). The WCSs consisted of 2.4-m diameter risers attached to a 1.2-m diameter culverts. The riser, shaped as a half-circle, had a solid face from the base to a height of 1.6-m, then 1.2-m of open face for water passage that can be closed by riser boards placed into the aluminum channels. The tops of the riser structures were installed at an elevation approximately equal to the ground surface elevations of their respective target restoration areas and approximately 0.6-m below the elevation of the nearby gravel roads to protect the roads from flooding. The riser boards were managed at the top of the riser structures for the duration of the study.

2.2. Data collection

Monitoring activities were conducted from July 2012 through December 2018. To evaluate the hydrologic conditions, 5 cm diameter PVC wells were installed to a depth between 1.5 and 2 m below ground surface. The wells were installed based on guidance from the US Army Corps of Engineering (USACE) (USACE, 2005). Two duplicate monitoring wells were installed in each of the two target restoration areas in July 2012. The wells were installed to be approximately equidistant to the east and west borders of the sites. The WBND wells were installed approximately 50-m and 100-m north of the Kim Saunders ditch. The Laurel wells were installed approximately 30-m and 60-m north of the Kim Saunders ditch. In March 2013, additional wells were installed to evaluate the hydropattern in two reference wetland locations that supported desired forest communities. One reference well was installed in an AWC-dominated stand south of the Kim Saunders ditch. The other reference well was installed in a BC-dominated stand just west of the DISW visitor center. These reference wetland wells provided experimental controls and target hydrologic restoration goals. The locations of all wells are shown in Fig. 2. Each well was equipped with a HOBO® #U20L-04 Water Level Data Logger (Onset, Bourne, MA) and set to measure water pressure and temperature on 2-h intervals. An additional data logger, located above ground in the BC location, provided on-site atmospheric pressure compensation required to estimate WT. WT levels were recorded relative to the nearby peat surface. The wells within both target restoration areas, Laurel 1 & 2 and WBND 1 & 2, were treated as duplicates and the measurements at each well were averaged to provide an estimate of WT in each target restoration area.

Site visits were conducted bi-annually to download well data, complete WT calibration measurements and perform maintenance checks. During field visits, the data loggers were examined to determine if the well or the sensors had been disturbed. In December 2015, it was noted that multiple well casings had been damaged, likely by wildlife, and sensors had malfunctioned. In January 2016, new PVC wells were installed at all well locations and sensors in the target restoration areas were replaced.

The growing season is the standard period used by the USACE (2005) to evaluate wetland hydrology, which also corresponds to the most critical periods for peat fire suppression and replanting/establishment of target wetland forest communities. Therefore, the analysis focused on the impact of the WCSs during the growing season only. The growing season in Camden County, NC was from March 19th to November 30th (257 days), as defined by the Elizabeth City, NC WETS station table at 28 degrees Fahrenheit and 50% probability using data from 1978 to 2018 (NRCS, 2019). The 2013–2015 growing seasons represented pre-WCS installation years, while 2016–2018 growing seasons represented post-WCS installation years. A summary of the data collection periods and annual rainfall during those years are shown in Table 1.

Table 1
Summary of pre- and post-WCS periods with annual precipitation and the number of months during the growing season that WT measurements were obtained for all sites.

Year	Impact period	# of months measured	Annual rainfall (cm)
2013	Pre-WCS	6	132.3
2014	Pre-WCS	9	134.6
2015	Pre-WCS	9[a]	145.4
2016	Post-WCS	9	*183.6*
2017	Post-WCS	9	130.1
2018	Post-WCS	6	145.9

[a] The WCSs were installed on November 17th, 2015. Therefore, the last 10 days of the 2015 growing season were removed from the pre-WCS analysis. Annual rainfall in italics indicates non-normal precipitation totals.

2.3. Data analysis

USACE's jurisdictional wetland criteria were used as an initial method to assess the hydrology of the two target restoration areas. USACE's criteria for wetland hydrology requires a WT continuously within 30 cm of the ground surface for at least 5% of the growing season (13 consecutive days) in 5 out of 10 years (50%) to be considered a jurisdictional wetland (USACE, 2005). WT data from each growing season during the study was used to calculate the maximum consecutive days that the daily mean WT was within 30 cm of the ground surface. Since less than 10 years of data were available, the duration of saturation requirement for the USACE's jurisdictional wetland criteria was used as a metric for site hydrology rather than proof of jurisdictional wetland status. To compare the hydropattern of the restoration areas to the reference wetlands and evaluate the impact of the WCSs on the two restoration areas, the 2-h interval measurements were used to develop graphical relationships via empirical cumulative distribution function (ECDF) plots. ECDF plots were created for each restoration area and reference wetland comparison. ECDF plots represent the percentage of the period (pre-WCS or post-WCS) that the groundwater levels were at or above a certain depth during the growing seasons. The use of the WT ECDF plots to analyze hydrological changes was based on an analysis of hydrologic restoration conducted by Menberu et al. (2016). A vertical shift upward in the ECDF plots at a location indicated a rise in WT and was defined as a wetter hydropattern (i.e., WT closer to the soil surface overall). Alternatively, a vertical shift downward in the ECDF plots indicated a decline in the WT and was defined as a drier hydropattern (i.e., WT further down from the soil surface overall). ECDF plots graphically represent WCS impact.

A modified before-after-control-impact (BACI) design, known as the paired BACI or BACIP design, (Conner et al., 2016; Downes et al., 2002; Stewart-Oaten et al., 1986) was used in this study for replicated WT data collected before and after restoration at the control and impact sites. Within this statistical design, the reference wetlands were control sites and the target restoration wetlands were impact sites. Although the two restoration sites are hydrologically connected, the assumption of hydrological independence was substantiated by the placement of a WCS at the downstream edge of each site, which resulted in each site being controlled by the individual WCS. At all four sites, the 2-h measurements were averaged to provide monthly mean WT estimates for each location.

The use of monthly mean WT levels restricted the influence of autocorrelation found in continuous WT measurements. The BACIP requirement that sampling should occur at random periods was retroactively satisfied by treating the monthly mean measurements as a random set. To account for the use of mean monthly WT levels, the growing season was altered to include the entire months of March and November (275 days). The data were analyzed using a linear mixed model and evaluated using an ANOVA. The normality and constant variance of the residuals for each model were met. An $\alpha = 0.05$ significance level was used for all statistical tests. The WCS impact for each site was quantified using the ANOVA interaction effect. All analyses were conducted using R software (R core team, 2019).

3. Results

WT data from all six wells collected from March 2013 through December 2018 are shown in Fig. 3. Negative WT depths indicate water levels below the ground surface. In both target restoration areas, recorded WT levels of the two duplicate wells followed nearly identical patterns indicating the extent of the lateral influence of the stage in the

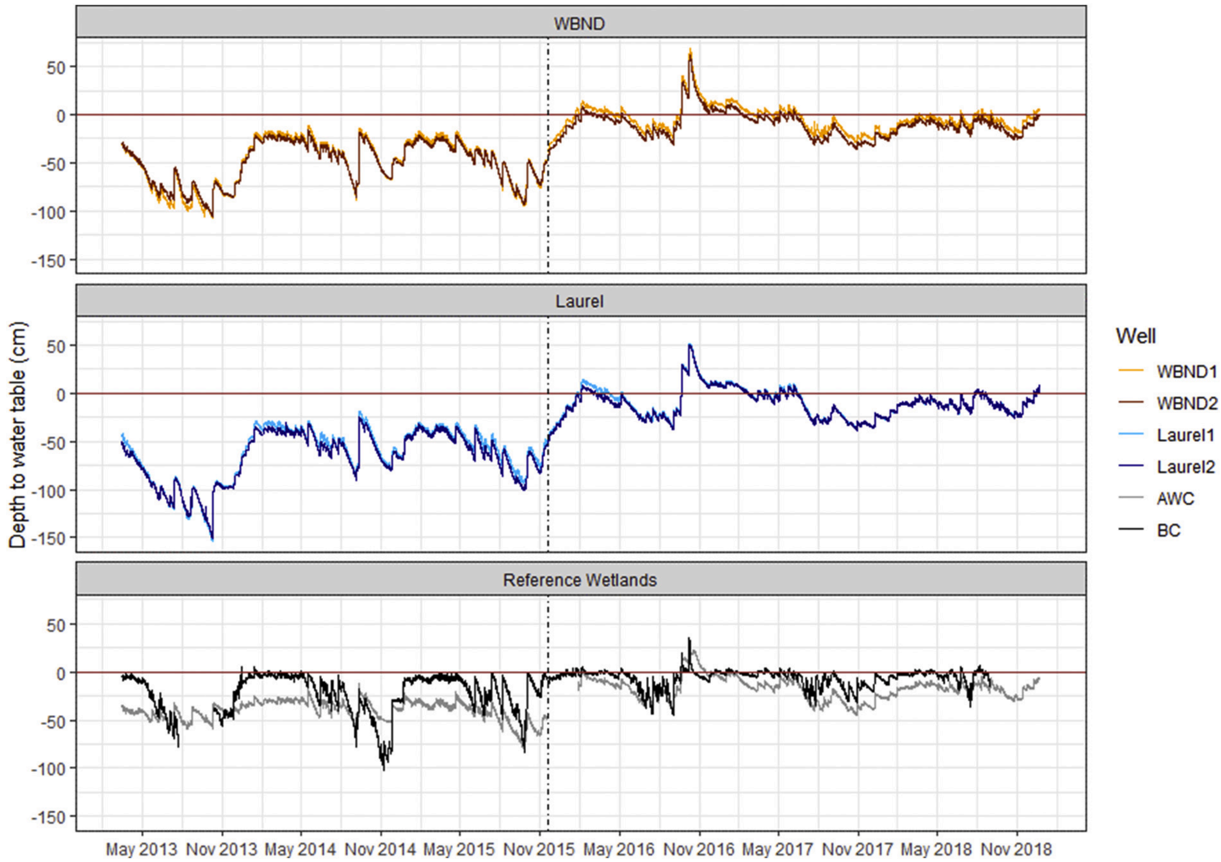

Fig. 3. Depth to water table for all six wells. Ground surface highlighted at 0. Vertical line represents the date of -WCS installation. WBND – Western Boundary restoration area; Laurel – Laurel restoration area; AWC – Atlantic white cedar reference; BC – Bald cypress reference.

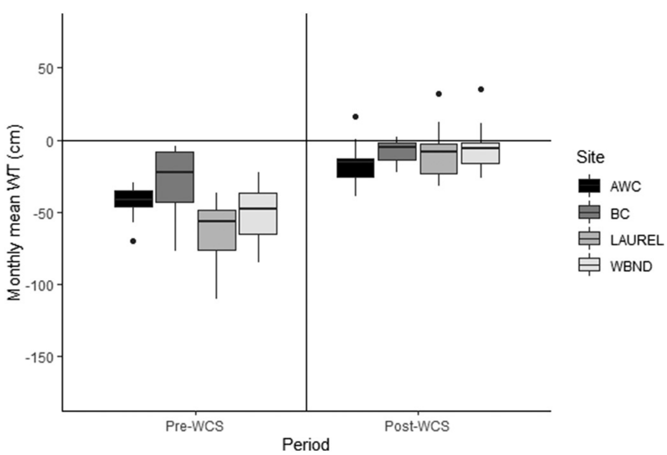

Fig. 4. Boxplots of the monthly mean WT levels for each restoration block and each reference wetlands. The bar indicates the median, the box spans from the 1st to 3rd quartile and the whiskers indicate 1.5* interquartile range (IQR). WBND – Western Boundary restoration area; LAUREL – Laurel restoration area; AWC – Atlantic white cedar reference; BC – Bald cypress reference.

Kim Saunders Ditch. Fig. 3 highlights the gradual increase in the WT immediately after WCS installation (installation date highlighted by the vertical line), the large increase in WT levels due to Hurricane Matthew in October 2016, and overall higher WTs in the post-WCS period for the two target restoration areas. Mean monthly values of the WT position for each period are shown in Fig. 4, which provides a coarse view of how the restoration areas increased in wetness and how they become more similar to the reference wetlands during the period following the installation of the WCSs. Further analysis described below provided a more accurate quantification of the changes in the hydropatterns of the restoration areas and comparisons of the restoration areas to the reference wetlands.

3.1. Jurisdictional wetland criteria

Laurel did not meet the duration of saturation requirement of the jurisdictional wetland hydrology criteria during any of the pre-WCS growing seasons (Table 2). Normal annual rainfall was observed for each of the pre-WCS year (Table 1). Therefore, it is likely that prior to WCS installation Laurel was not a jurisdictional wetland. Meanwhile, WBND met the saturation requirement in both 2014 and 2016. After installation of the WCS, both target restoration areas met and exceeded the duration of saturation threshold. Additionally, the mean WT during the growing season increased from deeper than 35 cm to within 10 cm below the soil surface in both restoration areas after WCS installation (Table 2).

Table 2
Maximum consecutive days the WT was at or above 30 cm below ground surface**.** Threshold value for GDS was 13 days. Mean WT levels for each growing season are also shown.

Period	Growing season	Consecutive days		Mean WT level (cm)	
		Laurel	WBND	Laurel	WBND
Pre-WCS	2013	0	0	−94.8	−72.2
	2014	9	53	−49.2	−36.4
	2015	0	16	−52.5	−42.9
Post-WCS	2016	159	257	−3.6	0.3
	2017	124	219	−13.1	−10.1
	2018	257	257	−11.5	−9.6

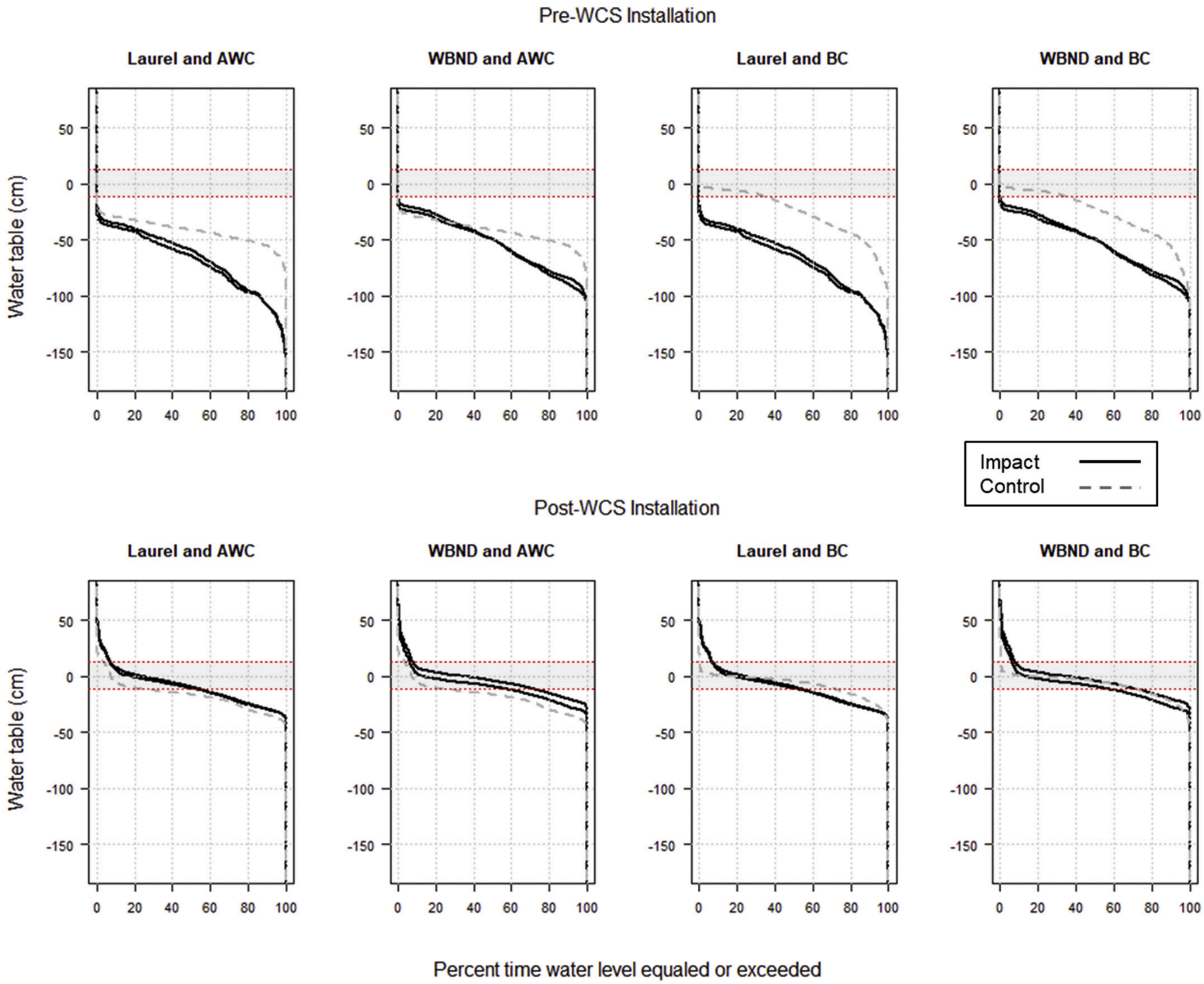

Fig. 5. ECDF plots for the growing seasons pre- and post-WCS installation. Impact represents either the Laurel or Western Boundary (WBND) restoration areas, while Control represents either the Atlantic white cedar (AWC) or Bald cypress (BC) reference areas, dependent on the comparison. The shaded area represents the mean annual water tables in six AWC swamps observed by Golet and Lowry (1987).

3.2. ECDF plots

Prior to WCS installation (pre-WCS, 2013–2015), the median WT levels measured in the Laurel site were − 59.1 and − 64.1 cm. During the same period, median WT levels measured in the WBND site were49.3 and − 49.9 cm. As expected, wetter conditions were observed in the reference wetlands; median WT levels were − 41.2 and − 22.0 cm for AWC and BC, respectively. Pre-WCS WTs were consistently closer to the soil surface in the BC reference wetland than in either of the target restoration areas. In the AWC reference wetland, WT levels were closer to the surface than the target restoration areas most of the time. The hydropatterns of the restoration areas were centered approximately 10 to 20 cm below the observed AWC hydropattern and 30 to 40 cm below the observed BC hydropattern (Fig. 5).

After WCS installation (post-WCS, 2016–2018) the WT in both restoration areas increased relative to the reference wetlands. In the restoration areas, the median WT at both wells were − 9.5 and − 10.8 cm in the Laurel site and − 4.2 and − 9.3 cm in the WBND site. During this period, wetter conditions, relative to the pre-WCS period, were also observed in the reference wetlands likely due to rainfall patterns during that period. Median WT levels were − 16.4 and − 4.5 cm for AWC and BC, respectively. The restoration area WTs were centered approximately 7 cm above the observed AWC hydropattern and approximately 5 cm below the observed BC hydropattern. These changes, along with overall changes in hydropatterns, were apparent in the ECDF plots (Fig. 5).

3.3. BACIP analysis

The interaction effect between location and period was significant (ANOVA Satterthwaite *t*-test, $p < 0.05$) for each restoration area and reference wetland combination (Table 3). The significant interaction effect indicated that the installation of the WCSs had a statistically significant impact on the WT within both target restoration areas. The installation of WCS had the effect of raising monthly mean WT levels in WBND by 24.7 cm and 19.8 cm relative to monthly mean WT levels in BC and AWC reference sites. Respectively. The effect was larger in the downstream Laurel site where monthly mean WTs increased 34.8 cm and 30.0 cm relative to the BC and AWC WT levels, respectively.

The statistical evaluation also indicated there were significant differences between restoration area and reference wetland monthly mean groundwater levels for all combinations pre-WCS installation. Post-WCS

Table 3
BACIP interaction effect for each combination of target restoration area and reference wetland.

Comparison (Impact/Control)	Effect (cm)	SE	T value	P value
Laurel/AWC	*30.00*	3.91	7.67	< 0.001
WBND/AWC	*19.84*	3.08	6.44	< 0.001
Laurel/BC	*34.84*	4.52	7.70	< 0.001
WBND/BC	*24.67*	4.01	6.14	< 0.001

installation there were no significant differences between the both restoration areas and the BC reference wetland (*t*-test, $p < 0.05$). Both restoration areas had monthly mean WT levels significantly closer to the soil surface than the AWC reference wetland post-WCS installation; however, this was likely due to the drier than expected conditions observed in the AWC reference wetland.

4. Discussion

Lichtler and Walker (1974) stated that it would be impossible to quantify the magnitude of change between the historical hydrology of the GDS and its current hydrology because the historical hydrology was unknown. However, changes in forest communities and susceptibility to peat fires clearly indicated that the GDS had become much drier than it was historically (Richardson, 1982; Laderman, 1989). The observation that one of the restoration areas did not meet the duration of saturation requirement for jurisdictional wetland criteria in any year Pre-WCS highlights the negative impact that drainage ditches have had on the hydrology in these areas.

For the goal of fire suppression, the results shown in Fig. 3 were encouraging. Studies have indicated that in tropical peatlands there is a critical WT level of 40 cm below the surface, below which dry peat becomes increasingly susceptible to fire (Jaenicke et al., 2010). Although Jaenicke et al. (2010) investigated peatlands in a tropical climate, the similar soil structure indicates that WT levels within 40 cm of the soil surface for substantial periods are likely also beneficial for fire suppression at DISW. Following the installation of the WCS, the WT levels in the restoration areas did not fall below −40 cm during either the 2017 or 2018 growing season, both of which had normal annual precipitation totals (Fig. 3 and Table 1).

For the goal of forest restoration, the results shown in Fig. 5 were again promising. While yearly WT levels in AWC forests vary considerably and an optimal AWC forest hydropattern has not been determined, a 7-year study by Golet and Lowry (1987) did find that the mean annual WT levels in six AWC swamps varied from 13 cm above ground to 11 cm below ground and the WT levels dropped 100 cm below the surface only during non-normal dry years. The average growing season WT levels in normal rainfall years post-WCS installation were 13.1 and 11.5 cm below the surface in the Laurel restoration area and 10.1 and 9.6 cm below the surface in the WBND restoration area indicating the hydropatterns in the target restoration areas were similar to those measured in other AWC swamps (Golet and Lowry, 1987; Laderman, 1989).

While the WCSs had a statistically significant positive effect on the WT levels within target restoration areas, a concern in these types of projects is that installation of WCSs may create wetter than optimum conditions. This would have detrimental effects on forest survival and establishment. There was an initial concern that the restoration areas had been over-saturated, WTs in both restoration sites were significantly closer to the soil surface than the AWC reference wetland WT post-WCS. However, within the AWC reference wetland, wind-downed trees and exposed root structures on existing AWC trees indicated that peat subsidence had occurred, likely because the area had become somewhat drier in recent years from unidentified changes in local drainage influences. The fact that the post-WCS hydropatterns in the target restoration areas were wetter than the AWC reference wetland but similar to both the BC reference wetland and the observations of Golet and Lowry (1987) should be viewed as a positive result. Relative to the restoration goals of the DISW, these results showed that the installation and management of WCSs produced a positive outcome.

As a rewetting technique, this study showed that properly maintained and operated WCSs increased the WT of the surrounding areas and produced more favorable conditions for future ecosystem restoration at the site. It should be noted that the magnitude of the changes in hydropatterns of this and other studies are likely dependent on several site-specific factors including soil type, vegetation type and cover, exiting drainage networks and regional climate. Therefore, direct comparison of data between sites in different regions should be made with caution. When the results of this study are considered alongside the results described in Landry and Rochefort (2012), predicted in Jaenicke et al. (2010), and observed in Menberu et al. (2016), it is apparent that WCSs are an important tool for large-scale hydrologic restoration of wetlands.

5. Conclusion

The determination of the success of the WCS installation and management was based on the observed changes in groundwater levels in the target restoration areas with respect to nearby reference wetlands during the growing seasons. These WT levels were analyzed using the jurisdictional wetland criteria, ECDF plots, and BACIP analysis. The jurisdictional wetland criteria analysis indicated that the Laurel site was likely not a jurisdictional wetland prior to the installation of the WCS. Following WCS installation, both restoration areas were saturated within 30 cm of the soil surface for at least 124 consecutive days per year during the growing season. ECDF plots indicated that the hydrologic regimes of the target restoration areas were substantially drier than those of the reference wetlands pre-WCS installation but were comparable post-WCS installation. The BACIP design and ANOVA analysis of mean monthly WT levels determined that the WCS had a significant positive impact on the hydropattern of the target restoration areas when external factors were controlled for, indicating that the shift in WT levels towards the soil surface was driven by the installation of WCS.

It is anticipated that the restoration of hydrology within the target areas will help reduce fire susceptibility, prevent peat oxidation, maintain carbon storage, and produce conditions that will favor the restoration of historical forest communities that once dominated the swamp. To maintain these successful results, the current operation and maintenance of these WCSs must be continued. This study demonstrated that proper design, installation, and management of the WCSs positively influenced the hydrologic restoration efforts within the DISW. Implementation of WCSs at other locations within this site should be used as a strategy to expand restoration efforts across the GDS. These results are also applicable to other large-scale degraded wetland sites and can be used as guidance for wetland managers faced with similar restoration opportunities.

Author's statement

All authors were active in Investigation. **Brock Kamath:** Formal analysis, Visualization, Data Curation, Writing-Original Draft. **Michael Burchell:** Conceptualization, Methodology, Supervision, Writing-Review & Editing. **J. Jack Kurki-Fox:** Field Maintenance, Visualization, Writing-Review & Editing. **Kris Bass:** Conceptualization, Resources, Writing-Review & Editing.

Author declaration

We wish to confirm that there are no known conflicts of interest associated with this publication and there has been no significant financial support for this work that could have influenced its outcome.

We confirm that the manuscript has been read and approved by all named authors and that there are no other persons who satisfied the criteria for authorship but are not listed. We further confirm that the order of authors listed in the manuscript has been approved by all of us.

We confirm that we have given due consideration to the protection of intellectual property associated with this work and that there are no impediments to publication, including the timing of publication, with respect to intellectual property. In so doing we confirm that we have followed the regulations of our institutions concerning intellectual property.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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