Forest restoration increases isolated wetland hydroperiod: a long-term case study

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Abstract. Geographically isolated wetlands (GIWs) are well known as “hotspots” for biodiversity and other ecosystem services, making their value on landscapes disproportionate to the area they occupy. GIWs are dependent on regular cycles of inundation and drying, which makes hydrology a primary controlling variable for sustaining functions and associated ecosystem services. Although human activity has degraded GIWs in many regions, relatively little work has focused on upland management as a way of sustaining, or even improving, GIW structure and function. We present a case study of longleaf pine forest restoration, by hardwood removal, on the characteristics of a wetland hydroperiod over a 10-year study. Our study wetland, W-51, is 0.89 ha with a catchment area of 31.2 ha located on a ~11,400-ha private preserve in Baker County, Georgia (31.25°N, 84.49°W). Beginning in 2006, continuous water level and climate data were recorded in the wetland and adjacent well transects across the wetland catchment. In autumn 2009, hardwoods were removed or deadened in the catchment resulting in a 37% reduction in tree cover. The effects on the hydrologic system were measured through 2016 by examining pre- and post-removal water levels, water yield ecosystem (WYe), and standardized recession rates (RRstd). The study included periods of above and below normal rainfall. Generally, wetland hydroperiods began in December and ended in May, but varied with rainfall pattern and amount. Hardwood removal increased WYe and decreased RRstd resulting in greater catchment water availability as reflected in water levels. Hardwood removal affected both the ascending and recessing limbs of wetland hydroperiods, substantially increasing the availability of ponded water in the wetland. Our results quantify changes in wetland hydrologic characteristics associated with forest management activities, which appear to have reduced forest water demand. Our study was a management case study, limited in scope but conducted in a realistic setting. More extensive studies (paired, replicated designs) are needed to better understand the implications at both the local scale, that is, managing critical aquatic habitat for wildlife populations, and at the regional scale, that is, providing support for landscape-scale connectivity and water yields.

Key words: evapotranspiration; forest restoration; hydrology; hydroperiod; isolated wetland; recession rate.

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INTRODUCTION

Geographically isolated wetlands (GIWs) are well known as hotspots for biodiversity and other ecosystem services, making their value on landscapes where they occur highly disproportionate to the area they occupy (Leibowitz 2003, McClain et al. 2003, Cohen et al. 2016, Smith et al. 2017, Evenson et al. 2018). They are known for high levels of both plant diversity and animal diversity (Smith et al. 2017), provision of critical habitat for diverse wildlife taxa (Smith et al. 2017), and high rates of productivity and carbon...
retention (Watt and Golladay 1999, Marton et al. 2015), and they can help regulate watershed function (Evenson et al. 2018, Lee et al. 2018). Thus, GIWs have been the subject of wide interest for research and conservation in recent decades. By their nature, isolated wetlands are dependent on regular cycles of inundation and drying making hydrology a primary controlling variable for sustaining these functions and associated ecosystem services. Current conservation priorities for GIWs focus on minimizing degradation from landscape development. Research priorities have largely focused on providing evidence of landscape hydrologic connectivity due to the policy implications, at least in the United States (e.g., McLaughlin et al. 2014).

Relatively little work has focused on the management of forested uplands surrounding GIWs as a way of sustaining, or even improving, wetland function and resilience. Jones et al. (2018) reported substantial effects of simulated upland forest management on wetland inundation and amphibian habitat in the southeastern United States. GIWs are especially common landscape features in the southeastern U.S. Coastal Plain and peninsular Florida (Kirkman et al. 1999). This region also coincides with the historic range of longleaf pine (*Pinus palustris* Mill., Pinaceae). Like GIWs, longleaf pine woodlands are noted for high biodiversity and serve as critical habitat for endemic species. Natural longleaf pine savanna is characterized by low basal area and is known for its diverse flora of warm-season grasses and forbs (Kirkman and Jack 2017). When managed with prescribed fire and maintained at relatively low basal area, longleaf savanna is gaining recognition for its potential value to improve water yield (McLaughlin et al. 2013, Brantley et al. 2017). The southeastern Coastal Plain also has a long, complex history of land use and disturbance (Golladay et al. 2016). This has caused landscape fragmentation, reduction, or elimination of frequent fire, and the near elimination of longleaf pine savanna (Martin et al. 2013, Golladay et al. 2016). The elimination of frequent surface fire throughout the eastern United States has led to the expansion of many fire-sensitive shade-tolerant species, particularly hardwoods, with cascading ecological effects (Nowacki and Abrams 2008, Alexander and Arthur 2010). These changes in forest composition and structure have affected water yields at stand and landscape levels (Caldwell et al. 2016).

The proliferation of hardwood species is particularly prevalent and problematic in and around GIWs where higher soil moisture in wetland margins favors hardwood species over pines and native groundcover. Hardwoods displace herbaceous species through shading, reduce longleaf pine regeneration, and alter fire regimes, reinforcing further hardwood encroachment (Nowacki and Abrams 2008, Martin and Kirkman 2009). Dominate by hardwood trees and shrubs is hypothesized to change the hydrologic function of wetlands. In many isolated wetlands, direct interception of precipitation inputs appears to be augmented through discharge from shallow groundwater (e.g., Winter 1988, Crownover et al. 1995, Winter 1999). This assertion is based, in part, on observations that the increase in wetland water volume following rainfall often greatly exceeds the volume attributable to direct precipitation and overland flow as well as direct measurements of hydraulic gradients around wetlands (e.g., Winter 1999). Gentle topography and relatively porous surface soils make lateral movement of shallow groundwater water, a major contributor to sustaining ponded water in isolated wetlands (Winter 1988). However, this groundwater water source is also susceptible to uptake and consumption by plants and therefore sensitive to vegetation change (Winter 1999). Higher leaf area and resulting higher water demands of woody vegetation relative to grass-dominated communities make higher streamflow and/or less groundwater recharge (Briggs et al. 2005, Elliott et al. 2017). Thus, trees and shrubs shorten wetland hydroperiods by removing more water from the soil and shallow water table than herbaceous vegetation (i.e., act as hydrologic pumps), which to varying degrees is hydraulically connected to the wetland (Winter 1988, Crownover et al. 1995, Winter 1999).

Forest management activities that change stand conditions through alterations in biomass or species composition may affect ecological processes such as carbon storage, evapotranspiration (ET), biogeochemical transformations, and water yields at both local and landscape scales (Cohen...
et al. 2016, Jones et al. 2018). Using linked process models, Jones et al. (2018) simulated the effects of upland forest biomass change in wetland hydrologic regimes and functions. Their simulations showed exponential declines in duration and volume of wetland hydroperiod associated with increasing forest biomass. Based on their modeling results, they suggested that forest management has a cascade of biogeochemical and ecological consequences through direct alteration of landscape hydrology and wetland hydroperiod. Similarly, Jones et al. (2012) noted that for unit watersheds, forest vegetation and associated ecosystem processes can mediate the effects of climate change and watershed development. Changes in hydrology related to vegetation management can also alter greenhouse gas emissions and modify rates of nutrient/sediment sequestration and transformation (Cohen et al. 2016, Jones et al. 2018). Biological consequences of shortening or altering the timing of hydroperiods mean that fauna dependent upon wetland hydroperiod for reproduction have insufficient development time and are unable to complete their life cycles. This is a major concern for wetland breeding amphibians, a group of special conservation concerns in the southeastern United States (Chandler et al. 2017, Jones et al. 2018).

The relationship between forest management and hydrology is well established in headwater catchments drained by small streams (Douglas 1983, Ford et al. 2011). However, insufficient quantitative data have been collected on upland vegetation management and its effects on GIWs. As a part of a larger landscape-scale forest restoration effort (Holland et al. 2019), encroaching hardwoods were removed from an upland catchment surrounding a grass-sedge marsh-type, isolated wetland in an actively managed longleaf pine savanna within the Gulf Coastal Plain of southwestern Georgia. This gave us the opportunity to quantify the effects on the wetland hydroperiod prior to and following hardwood removal. We hypothesized that the removal of fire-sensitive shade-tolerant hardwoods would reduce vegetative water demand and increase wetland hydroperiod. We focused our analysis on wetland recession rates because they represent a period when vegetation has a strong effect on wetland drying; that is, early growing season. Here, we present the results of a 10-year study demonstrating how upland forest management alters the hydroperiod of an isolated wetland. The length of the study permitted the evaluation to include years with above and below normal rainfall both before and after the hardwood removal treatment. The study provides data that support projections of hydrologic models by demonstrating alterations of water storage/movement at a local wetland catchment scale.

**Methods**

**Study site**

Our study was located at the Jones Center at Ichauway, a ~11,400-ha private preserve and research center in Baker County Georgia (31.250° N, 84.495° W). Climate at the site is classified as humid subtropical with long, hot summers and short, cool winters. Mean annual temperature is 19°C and ranges between −10°C and 39°C. Mean annual rainfall is 1310 mm and is distributed relatively evenly throughout the year. Ichauway is dominated by second-growth, fire-maintained longleaf pine forest, much of it with diverse, native groundcover. Over 10,000 geographically isolated wetlands dot the surrounding karst landscape of the Dougherty Plain, part of the southeastern U.S. Coastal Plain (Martin et al. 2012; Fig. 1a), with ~100 GIWs occurring on Ichauway. On-site wetlands range in size from 0.25 to >60 ha (Kirkman et al. 2000).

Our study wetland, W-51, is 0.89 ha and occupies a shallow bowl-like catchment with an area of 31.2 ha and an elevation change of 6.7 m from outer catchment margin to lowest point in the wetland. W-51 is classified as a marsh (Kirkman et al. 2000), with dominant vegetation being emergent grasses/sedges, scattered shrubs, and an open canopy. The typical length of inundation, that is, hydroperiod, is from late winter to early summer with maximum inundation depth of less than 1.5 m. Soil samples taken while inserting shallow wells (see Hydrologic measurements and Appendix S1: Fig. S1) indicated that surface soil thickness ranged from 1 to 3 m and was composed of fine sands and sandy loams. Clay content tended to increase with depth. Surface soils sat atop a low permeability clay layer that roughly followed the contours of the soil surface. The clay perching layer tended to be...
Fig. 1. (a) Location of study site at Ichauway in southwestern Georgia; (b) catchment boundary, land cover,
well-defined in surrounding uplands, but was generally more diffuse in wells adjacent to the wetland margin. Water chemistry data indicate that there is no dynamic linkage between seasonal wetlands at Ichauway and underlying aquifers (Battle and Golladay 2001).

Changes in upland vegetation
The forest surrounding our study wetland has undergone long-term vegetation change associated with human land use and changes in fire regime. Historically, hardwoods were only found on the north end of the wetland and in small clusters. By 2006, hardwoods, primarily Quercus species, encircled the wetland ecotone and caused a fire shadow, that is, an area of low flammability, thus reducing the chance of prescribed fire burning through the wetland. As trees matured, a dense, nearly continuous canopy reduced sunlight to grasses and forbs essentially eliminating the groundcover and fire carrying capacity while decreasing floral diversity in the ecotone (Clayton and Hicks 2007). To protect and restore groundcover diversity, W-51 and its catchment were included in high-priority areas for longleaf/groundcover restoration efforts in the early 2000s. hardwoods and shrubs were removed or deadened on the catchment during October 2009 by Jones Center land management staff. Smaller hardwoods, including Quercus nigra and Prunus serotina, were removed mechanically, while larger hardwoods, mostly Quercus virginiana, were girdled.

Because this study was a retrospective analysis of a land management action, our options were limited in quantifying and validating vegetation change from those actions. Thus, to estimate the impact of hardwood removal on catchment vegetation structure, we focused only on changes in tree cover, which was the primary treatment goal, and estimating changes in other parameters (i.e., leaf area index or biomass) would introduce unnecessary error and could not be validated. We used high resolution (~0.30 m) pre- and post-treatment color infrared aerial photographs to determine the change in catchment tree cover from hardwood removal treatments. These photographs were collected in 2006 and 2011 as part of a comprehensive land cover monitoring program using contracted aerial observation platforms (Fig. 1b). Imagery in 2006 was collected using color infrared film that generated a digital orthomosaic, while in 2011 digital cameras were available. All analyses were completed in ArcGIS 10.3. Photographs were clipped using an approximation of the wetland contributing area, or catchment using the digital terrain model (DTM) generated by the National Ecological Observatory Network LiDAR Point Clouds collected in September 2018. The DTM was processed using ArcGIS Hydrology Spatial Analyst Tools (ArcMap 10.3; ESRI 2018). Small imperfections in the DTM were removed using the Fill tool, and the Flow Direction tool was used to determine the direction of flow out of each cell. Finally, the Basin tool was used to delineate the drainage basin based on the flow direction raster.

We estimated catchment tree cover before and after hardwood removal treatments on the part of the catchment that received the treatment. We first clipped known agricultural areas from the image using land cover maps maintained by the Jones Center. We then used an unsupervised maximum-likelihood classification with 100 iterations and 25 classes. This number of classes was the minimum number that consistently distinguished between woody cover and nonwoody cover and 100 iterations ensured the accuracy of pixel classification without excessive processing time. Classes were grouped by visually comparing the classified image to the aerial photograph and combining classes until two cover classes, tree and non-tree, were developed (Fig. 1b). The high resolution of the images made visual validation of tree cover change a practical solution since field data could not be gathered, and individual trees and even small shrubs could be easily distinguished from herbaceous vegetation,
water, or roads. Pixel counts of each class, tree or non-tree, were compared for each photograph to calculate tree cover pre- and post-treatment, and calculate catchment tree cover change after treatment for both the forested land cover and the whole catchment.

Hydrologic measurements

We installed hydrologic and climatic monitoring equipment in the catchment in 2006, anticipating hardwood removal in late 2009. A platform was installed in the deepest part of the ponded area, and a stilling well was constructed to house a float-operated automated water-level data logger (Ott, Thalimedes). Water levels in the wetland were recorded at one-hour intervals. Shallow groundwater wells were established through the W-51 catchment on approximately north/south and east/west transects. Locations and elevations of all wells and level recorders were established using a total station and standard survey techniques. A USGS road benchmark was used as a datum, and all elevations are expressed with respect to average mean sea level (AMSL). For simplicity, we present examples from the eastern location (E1–E3) to show patterns of water level in wells and the wetland during selected hydroperiods. Data from all transects are presented in Appendix S1. On each transect arm (N, S, E, W), soil borings were collected at three locations including outer catchment (1), mid-catchment (2), and wetland ecotone (3). Each bore was converted to a monitoring well with 5.1 cm diameter PVC well casing. Sand was used to fill the well annulus proximate to the screened interval, and a bentonite seal was installed above the well to prevent vertical leakage of rainwater. Well transects extended laterally approximately 80–100 m from the wetland ecotone, and well depth ranged from 1.5 to 3.7 m. Elevation of the soil surface increased approximately 2–3 m over the length of transects (Fig. 2). Soil borings were used to define soil characteristics at each site, as mentioned above. Data loggers (Ott, Orphimedes) were installed in wells to record water levels at

Fig. 2. Cumulative rainfall vs. study average rainfall for water years at W-51. The date of hydroperiod inundation, that is, the presence of ponded water, is indicated by a triangle for each water year. Note that Yr_w begins on 1 October; therefore, Yr_w 2007 begins on 1 October 2006.
one-hour intervals. Data collection began during spring 2006 and ceased on 31 December 2015.

Data analysis

Daily water levels in wells and the wetland were calculated from hourly values. Daily values served as the basis for all subsequent calculations and analyses. For each year of the study, wetland hydroperiod was defined as the beginning of inundation, that is, visible ponded water at the deepest part of the wetland basin, through seasonal drying. Determinations were based on visual inspection of water levels in the wetland as recorded by the depth logger. While hydroperiod initiation and duration varied with antecedent rainfall, a period of ponded water occurred during all years of the study. Wetland hydroperiod did not correspond well with calendar year. It was not unusual for wetlands to begin to inundate in late fall or early winter (November–December). With this in mind, we used the convention of water year (Yrw, 1 October–30 September) to group our hydrologic data. For example, the 2010 Yrw begins on 1 October 2009 and ends on 30 September 2010. The Palmer Drought Severity Index (PDSI) was used to classify water years as wet and dry based on whether they occurred during periods of PDSI surplus (positive) or PDSI deficit (negative; Appendix S1: Fig. S2, data obtained from the National Centers for Environmental Information [NCEI], www.ncei.noaa.gov).

Daily recession rates (RRd) were calculated on the drying limb of wetland hydroperiods for each year of the study. Daily recession was calculated as the difference in water levels across each successive day. When rainfall occurred during the drying limb, we excluded recession rates for a minimum of 2 d following the succession of rainfall or until the pond level stabilized, in subsequent calculations (e.g., Chandler et al. 2017). To compare recession rates across years and climatic conditions, we calculated standardized recession rate (RRstd), which is equal to RRd/PET (Chandler et al. 2017). Potential evapotranspiration (PET) was estimated using the FAO Penman-Monteith approach (Zotarelli et al. 2010). Data to populate the PET calculations came from a weather station at the study site (rainfall, temperature) or a nearby climate station (wind, insolation, relative humidity, etc.; http://www.georgiaweather.net/, Newton, Georgia, USA).

Water yield for an ecosystem, WYe, is derived from the equation (e.g., Royhatyn et al., 2018): \[ P = ET + R + G + \Delta S, \]
where \( P \) is rainfall, \( ET \) is evapotranspiration, \( R \) is the surface run-off, \( G \) is the groundwater recharge, \( \Delta S \) is the soil water storage, where
\[ WYe = R + (G + \Delta S). \]
and therefore,
\[ WYe = P - ET. \]

For this study, \( G + \Delta S \) was unmeasured, and \( R \) can be assumed to be 0, that is, no surface run-off. We did not measure ET of the wetland catchment, that is, actual evapotranspiration (AET). However, our calculations of potential evapotranspiration (PET) are a useful surrogate for AET and should provide a close approximation during periods of water surplus and low transpiration, for example, winter dormant season. Therefore, we can infer changes in \( G + \Delta S \) associated with hardwood removal by examining the threshold of P-PET required within a water year to initiate wetland hydroperiod. Thus, our estimate is:

\[ WYe = P - PET. \]

This allows for the comparison of hydroperiods for water years across our forest management treatment. Note that we are not attempting to partition water between soil storage, shallow groundwater, and loss to deeper aquifers, but instead look at the potential for water availability to change in the local hydrologic system across water years based on the threshold for hydroperiod initiation in relation to WYe.

Comparisons of RRstd and WYe were performed using SigmaPlot for Windows V14 (Systat Software). Groups were compared using either t-test (P-PET, met assumptions of normality) or the Mann-Whitney rank-sum test (RRstd), did not meet assumptions of normality). We used the estimated WYe threshold for post-hardwood removal to project changes in the timing of inundation for pre-hardwood removal hydroperiod (Table 1). We also used post-hardwood removal RRstd to project how slower recession might have affected the length of wetland hydrology under the pre-removal conditions (see Results).
RESULTS

Study period climatology

The study encompassed two major regional droughts (PDSI < −3.0) during 2006 through mid-2008 and from mid-2010 until the end of 2012. Late 2008 through 2009 and 2013 through mid-2014 were periods of water surplus (Appendix S1: Fig. S2). This pattern of climate variability allowed us to compare the hydrology of W-51 during wet and dry periods both pre-hardwood removal and post-hardwood removal.

Forest change

Catchment area was 31.2 ha, of which 37% was open field (i.e., wildlife food plots) and 63% was either pine woodland or wetland. Pre-treatment tree cover and post-treatment tree cover for the forested sections were 58% and 37%, respectively. For the entire catchment, pre-treatment tree cover and post-treatment tree cover were 37% and 23%, respectively, for an overall change of −37% from the hardwood removal treatment. While hardwoods were removed from the entire catchment, efforts were concentrated in the area adjacent to the wetland (Fig. 1).

Water infiltration and inundation

Water year (Yrw) cumulative rainfall ranged from 936 mm (2011) to 1611 mm (2013) over the course of the study. Cumulative rainfall during dry years (Yrw 2007, 2008, 2011, and 2012) ranged from −900 to 1300 mm and in wet years (2009, 2010, 2013, 2014, 2015) from 1400 to 1600 mm. Differences between wet and dry years were most apparent in seasonal distribution of rainfall with wet years having above normal cumulative growing season rainfall (Fig. 2). With the exception of Yrw 2014, wetlands were not inundated at the beginning of water years (October 1). In wet years, the period of no standing water tended to be brief (<2 weeks), while during dry years, the wetlands were typically dry for most of the late growing season (August–November).

Wetland inundation followed late fall and winter precipitation and began in late November through February depending on rainfall distribution and cumulative total. Inundation (hydroperiod initiation) began following 10–360 mm of cumulative water-year rainfall depending on antecedent soil moisture/groundwater conditions. Hydroperiod initiation was responsive to the distribution and volume of rainfall during the preceding year. Years with above-average rainfall, particularly during the growing season, resulted in earlier wetland inundation requiring less cumulative rainfall during the subsequent Yrw hydroperiod (Fig. 2).

Water levels in the upland wells responded similarly to cumulative rainfall (Fig. 3). With the onset of late fall/early winter rainfall, water levels in the ecotone well and the wetland began to increase. Water levels in mid- and outer catchment wells appeared to be sensitive to rainfall showing rapid rises and declines in response to individual rainfall events. Water levels in the wetland and the ecotone well showed a general increase in water table elevation. This pattern is consistent with the presence of a subsurface perching layer that created a hydraulic gradient toward the wetland in the center of the catchment (e.g., Appendix S1: Fig. S1). Two years (Yrw−2008, pre; Yrw−2011, post) are shown for simplicity; however, the wetland/well transect behaved similarly across the water years of the study. No treatment effect, that is, hardwood removal, was apparent at this relatively coarse scale. Yrw−2008 and Yrw−2011 were similar in drought severity (Appendix S1: Fig. S2), with Yrw−2011 having slightly less rainfall during the wetland hydroperiod but following a relatively wet Yrw. During Yrw−2011, water levels in the upland wells remained at or above the elevation of the wetland inundation level (Appendix S1: Fig. S1).

| Water year | Day of Yrw hydroperiod begins (calendar date) |
|------------|---------------------------------------------|
|            | Actual | Projected | Difference (d) |
| 2007       | 82 (12/22/06) | 19 (10/21/06) | 63 |
| 2008       | 90 (12/30/07) | 17 (10/18/07) | 73 |
| 2009       | 71 (12/10/08) | 23 (10/23/08) | 48 |
| 2010       | 62 (12/02/09) | 0 (10/01/09) | 62 |
Water yield ecosystem (WYe)

WYe (P-PET) had a lower annual range for wetland hydroperiod (Fig. 4) compared with cumulative rainfall (Fig. 2). Prior to hardwood removal, the threshold for hydroperiod initiation was a WYe of 108.8 mm accumulated since the beginning of the water year (1 October). Following removal, the threshold was significantly reduced to −35.6 mm (t-test, P < 0.001). Using the new threshold for years prior to hardwood removal resulted in substantially earlier estimates for hydroperiod initiation. Prior to treatment, inundation generally began during month 3 of the water year (December), while the projected inundation based on hardwood absence began during month 1 (October; Table 1).

Standardized recession rates

Hardwood removal resulted in significant declines in wetland recession rate in both dry and wet years (Fig. 5). We used the average maximum water depth of our wetland (0.6 m) and substituted the post-hardwood recession rate to project the effect of hardwood removal in the pretreatment period. This projection suggested an increase in the length of wetland inundation from 218 to 271 d in dry years, and 238 to 278 d in wet years.

Fig. 3. Examples of water levels in catchment wells and in W-51 before (2008) and after (2011) hardwood removal.
Vegetation controls on wetland hydroperiod and the effects of forest restoration

The hydroperiod of many isolated wetlands, that is, those without direct connections to surface waters, is controlled by the balance of precipitation (P) and evapotranspiration (ET) over the contributing area of the wetland. The relatively small volume of water stored in many of these wetlands relative to the size of the contributing area may make them particularly sensitive to small changes in the balance between P and ET in the catchment. Isolated wetlands fill during periods when $P > ET$ and dry when $P < ET$ (Crownover et al. 1995, Kirkman et al. 1999). Thus, inundation occurs following pulses of precipitation that infiltrate soils raising the local water table elevation, and when a hydraulic gradient is established, both soil water and shallow groundwater flow toward depressions in the land surface (Winter 1988, Kirkman et al. 1998). Our study wetland responded to dormant season rainfall and was influenced by antecedent conditions. During or following years with above-average rainfall, wetland inundation began earlier or at a lower total rainfall accumulation (Figs. 2 and 4). Above-average rainfall increased the length of wetland hydroperiod during years of water surplus as indicated by PDSI (Appendix S1: Fig. S2) and cumulative water-year rainfall (Fig. 4).

Our study design incorporating continuous water-level data in both the wetland and adjacent upland wells provided valuable descriptive information about the wetland hydroperiod. During most years, ponded water was apparent in the wetland once 200–400 mm of accumulated rainfall was recorded. The wetland showed the presence of ponded water before upland wells responded indicating that direct interception of water over the wetland basin accounted for early hydrologic response. As more rainfall accumulated, water levels in the two outermost catchment wells showed pulsed responses, while the wetland showed steadily increasing water levels. This, along with the presence of a perching layer beneath uplands soils and higher elevation of the

![Figure 4](image-url)

Fig. 4. Annual water-year ($Y_{w}$) ecosystem water yield ($W_{ye} = P - PET$) calculated for W-51 in the main figure. The inset shows the threshold ($P - PET$) required to initiate wetland hydroperiod.
water table in upland wells, provides evidence for a hydraulic gradient and lateral flow of groundwater in our wetland catchment. The presence of complex hydraulic gradients between uplands and surface waters has been previously noted (Heimberg 1984, Crownover et al. 1995, Bailey et al. 2014). The delayed response of ecotone wells may have resulted from several factors. First, the perching layer in ecotonal wells was more diffuse and the delay in filling may reflect slower water infiltration due to greater soil complexity (Bailey et al. 2014). The presence of an upland perching layer may have permitted the development of a transient water table resulting in rapid lateral movement of water from outer wells to lower elevation areas closer to the wetland boundary (Bailey et al. 2014). The abundance of plants in ecotonal areas may result in bands of greater ET and soil moisture depletion directly adjacent to the wetland (Meyboom 1966, Kirkman et al. 1998). Clearly, the patterns of local groundwater movement into GIWs are complex and worthy of greater study.

Within forests, ET is a major source of water loss and return to the atmosphere and is mediated by physiological and structural characteristics of forest vegetation including species water requirements, leaf area, stand density, and growing season length (Hwang et al. 2018). ET is strongly influenced by vegetation water demand at a catchment scale, which influences water balance of aquatic systems by lowering the water table and reducing the hydraulic gradient causing groundwater discharge (Jones et al. 2012). The small area and relatively shallow depth of many isolated wetlands result in a relatively small inundated volume, which, in turn, may be highly sensitive to small changes in catchment ET. Thus, wetland recession rates and associated drying will be strongly influenced by adjacent forest structure and species composition interacting with climatic factors such as rainfall and temperature (Jones et al. 2018). We observed wetland recession commencing during the beginning of the growing season as rainfall patterns changed from frontal systems to scattered showers (Golladay et al. 2016), daily temperature and vegetation water demand increased, and cumulative PET exceeded by cumulative P (Fig. 4). However, during wet years timing of recession could be delayed through the summer (Fig. 4). The influence of rainfall was also apparent in comparisons of recession rate, which showed significantly slower recession during wet years compared with dry years.

The importance of catchment vegetation on wetland hydroperiod was apparent by its strong effect on wetland recession rate with significant reductions occurring post-treatment during both wet and dry years (Fig. 5). Modeling studies have postulated that reductions in forest biomass could extend wetland hydroperiods (Jones et al. 2018) and our study is consistent with model predictions. Alterations in forest biomass or structure that reduce ET leave greater shallow groundwater within catchments, which, in turn, provides for greater groundwater flow, greater catchment yields (Hwang et al. 2018, Royhatyn et al. 2018), and/or longer wetland hydroperiods (this study, Jones et al. 2018). Reduced ET and greater groundwater also explain the lower cumulative rainfall threshold for hydroperiod we observed post-hardwood removal (Fig. 4). This carryover water or lag effect from one water year...
to the next has also been observed in forested watersheds (Hwang et al. 2018). These results have implications for wetland management through forest restoration in that the removal of live oaks, whose presence reflects the reduction in surface fire occurrence, provides substantial hydrologic benefits at local scales and if implemented at a landscape scale might improve regional water yields (e.g., Jones et al. 2018).

As with inundation, groundwater data obtained from monitoring wells provided descriptive insights on hydrologic processes during wetland recession. Prior to hardwood removal, during mid- to late recession the elevations of water in catchment wells receded faster than wetland levels and was sometimes lower. This indicates that under some circumstances the hydraulic gradient in the wetland catchment reversed, and potential flow gradient was radially outward rather than inward toward the wetland. Similar multidirectional flow of water has been reported in other isolated wetlands (e.g., Meyboom 1966, Crowenover et al. 1995), but this possibility has been overlooked in more recent modeling studies and is likely a source of uncertainty in partitioning vegetation use and change in groundwater storage, and recharge of underlying aquifers, if present (e.g., Chandler et al. 2017).

**Broader implications, the significance of wetlands in landscapes, and model shortcomings**

While the value of wetlands has been long recognized, isolated wetlands have presented a challenge to both the policy and management communities. In the southeastern United States, they are recognized as sites of concentrated biological diversity and/or breeding sites for rare fauna such as amphibian and invertebrate species (e.g., Kirkman et al. 2012, Chandler et al. 2017, Smith et al. 2017). However, supporting regional biodiversity has not been successful in justifying wetland conservation and many isolated wetlands have been lost or degraded by landscape development (Martin et al. 2012, Stuber et al. 2016). In landscapes with GIWs, their apparent lack of connection to surface waters means that they are not considered waters of the United States (WOTUS), and thus, they lack recognition under the Clean Water Act (Golden et al. 2014, Cohen et al. 2016). We contend that the failure to appreciate GIWs is caused by lack of appreciation for hydrologic controls, such as vegetation and the complexity of water movement, at a landscape scale.

Recent studies using hydrologic models strongly suggest that GIWs provide support for water yields at a landscape scale. Generally, the approach has been used to develop a landscape-scale hydrologic model, calibrate the model for streamflow or water yield, and then run scenarios involving GIW alteration/elimination (Feng et al. 2013, Evenson et al. 2015, 2016, Lee et al. 2018). In the Coastal Plain landscape of the Chesapeake Bay region, model simulations suggested that GIWs were more important than riparian wetlands in reducing surface run-off during storms and promoting groundwater recharge (Lee et al. 2018). In the prairie pothole region of North Dakota, simulated loss of wetlands affected water storage (wetlands > 3 ha) and water residence time (wetlands < 3 ha) in a 1700 km² watershed (Evenson et al. 2018). Using linked process models (forest structure/ET, and wetland hydrology), Jones et al. (2018) showed an inverse relationship between forest basal area and wetland hydroperiod in pine flatwoods of the southeastern U.S. Coastal Plain. Collectively, modeling efforts suggest that GIWs can be viewed as nodes within a landscape-scale hydrologic systems that act to modify imbalances between hydrologic inputs (rainfall) and outputs (run-off, watershed yield) at both local and landscape scales (Cohen et al. 2016).

Modeling studies, while useful for projecting hydrologic connectivity, do not quantify rates of water transfer into and out of isolated wetlands with varying climate and forest management conditions (Hwang et al. 2018). Determination of those rates and processes is a critical research priority recognized in modeling studies with calls for direct measurements of water balance (Lee et al. 2018), shallow groundwater measurements in wetland catchments (Chandler et al. 2017), rates of individual wetland run-off generation (Thorslund et al. 2018), and changes in hydrologic processes associated with land management (Lee et al. 2018, Jones et al. 2018). Recognition of critical research priorities is not a recent phenomenon as Winter (1988) noted that most field studies of wetland hydrologic processes have generally been short term and not included entire wetland catchments. Our results...
provide direct measures of changes in wetland hydrologic characteristics associated with forest management and subsequent changes in forest water demand. They have important implications at both the local scale, that is, managing critical breeding habitat for amphibian populations, and the landscape scale, that is, providing support for increasing water yields over broader spatial scales.

CONCLUSIONS

Our study shows that isolated wetland hydrology can be strongly controlled by forest structure and composition within a wetland catchment. This regulating role is evident across precipitation cycles and is substantial during both wet and dry years. In our study, the reduction in forest water demand through hardwood removal was reflected in an increase in the length of wetland hydroperiod. Our study wetland filled earlier in response to cumulative rainfall and dried more slowly. Ultimately, this has important implications for water storage and yield, at both wetland catchment and landscape scales. We acknowledge the limited geographic scope of our study, but suggest that it quantifies upland/isolated wetland connection over an extended period in a realistic forest management setting. Perhaps more importantly, our study area is representative of many landholdings in the southeastern United States. It is situated on a private land where local management is the primary focus (Golladay et al. 2016). As such, it provides valuable insight into the potential contributions to and controls on isolated wetland hydrology in a local setting. We suggest that manipulation of upland forest composition should be considered in areas where water conservation and wetland hydroperiod are priority concerns.

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