

COMPARISON OF HYDROLOGY OF TWO ATLANTIC COASTAL PLAIN FORESTS

D. M. Amatya, T. M. Williams, J. E. Nettles, R. W. Skaggs, C. C. Trettin



HIGHLIGHTS

- Outflow, driven by water table position on these forest systems, is highly variable, depending on its soil water storage.
- The hydrologic responses of both forest sites were similar during extreme climatic events or disturbances.
- Effect of forestry drainage on runoff was obscured by its large interannual differences.
- Long-term monitoring provides better insights on climate and vegetation management effects on flow regime and model validation.

ABSTRACT. *This article compares the short-term and long-term hydrology of two typical forests in the humid Atlantic Coastal Plain, including a relatively undisturbed forest with natural drainage in South Carolina (SC) and a drained pine plantation in North Carolina (NC), using monitoring and modeling approaches. Highly dynamic outflow (O) from both of these systems is driven by the water table (WT) position, as influenced by rainfall (R) and evapotranspiration (ET). The annual runoff coefficient (ROC) varied from 5% in dry years to 56% in wet years, depending on the soil water storage (SWS), with a significantly higher average value for the NC site despite its deeper WT, on average, than the SC site. Although both sites behaved similarly in extreme climate conditions, the change in SWS above the WT influenced the annual RO, ROC, and ET. The 17-year average annual ET of 1114 mm (R – O, assuming annual balanced SWS) for the SC site was significantly higher ($p = 0.014$) than the ET of the drained NC site (997 mm) despite the SC site's lower mean annual R of 1370 mm, compared to 1520 mm for the NC site. This may be due to both the higher potential ET (PET) and soil water-holding capacity of the SC site. The SC site had higher frequency and duration of WT near the surface during winter, deeper summer WT, and higher correlation of annual ET to annual R ($r^2 = 0.90$ vs. 0.15), suggesting that the SC site was often moisture-limited, particularly during the growing season. Most of the streamflow in these systems occurred during winter, with low ET demands. However, summer periods with tropical storms also resulted in large RO events, generally with higher frequency and longer durations at the drained NC site. These results are similar to an earlier short-term comparison with an unstable behavior period at the SC site after Hurricane Hugo (1989). This study highlighted (1) the differences in hydrology between coastal forests drained for silvicultural production and undrained natural forests managed only for restoration, (2) the importance of long-term monitoring and the effects of regeneration as well as vegetation management on flow regime, and (3) the application and limitations of two widely used models (MIKESHE and DRAINMOD) in describing the hydrology of these forests. Long-term studies can be a basis for testing new hypotheses on water yield, stormwater management, wetland hydrology, vegetation restoration, bioenergy production, and climate change, in addition to applications of proper models for assessing the eco-hydrologic impacts of land use and climate change on freshwater coastal forests linked with downstream riparian rivers and estuaries affected by tidal fluxes and sea level rise.*

Keywords. *Drainage, Evapotranspiration, Hydrologic models, Pine forest, Poorly drained soils, Runoff coefficient, Water table.*

Submitted for review in March 2019 as manuscript number NRES 13387; approved for publication as a Research Article by the Natural Resources & Environmental Community of ASABE in July 2019.

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Forests control about 60% of the regional hydrology in the southeastern U.S. (Sun et al., 2002). The southeastern coastal plain, a 160 to 320 km wide belt that extends along the Atlantic and Gulf coasts from Virginia to East Texas, is covered mostly by pine forests (Henderson and Grisino-Mayer, 2009). Most of those forest lands in the lower coastal plain are drained by small streams, which are typically headwaters of blackwater rivers. These streams are characterized by low-gradient stream beds and side slopes, and relatively broad stream bottoms and riparian buffers with slow-moving water. Soils within the region often have poorly drained clayey subsurface layers that restrict internal drainage, resulting in a high water table (WT)

on a seasonal or more frequent basis. Because of these physical features, headwater catchments in the lower coastal plain often contain forested wetlands (Harms et al., 1998).

Rapid rise of the WT with rainfall is common in southeastern low-gradient forested watersheds (Trousdel and Hoover, 1955; Young and Klaiwiter, 1968; Williams, 1979; Riekerk, 1989; Amatya et al., 1996; Sun et al., 2000; Amatya and Skaggs, 2011). Because the long-term annual rainfall in this region is generally higher than the long-term annual ET (Skaggs et al., 1994, 2011; Sun et al., 2002, 2010; Amatya and Skaggs, 2011; Dai et al., 2013), these poorly drained sites are often wet, especially in the winter and early spring. The main hydrologic characteristics of these watersheds are: (1) near-surface or shallow WT, which drives most stream outflows (as shallow surface runoff and drainage) (fig. 1a); (2) surface water detention (via wetland storage), which reduces flash flooding; and (3) delayed discharge of surface and subsurface water, which provides improved water quality downstream. These hydrologic mechanisms control biogeochemical processes (Beltran et al., 2010; Dai et al., 2011; Tian et al., 2012); support high biomass production; sustain diverse terrestrial, riparian, and aquatic communities; and provide recreational opportunities (Sun et al., 2004; Vose et al., 2011).

Historically, in the late 19th through mid-20th century, vast areas of forested wetlands on poorly drained soils in the southeastern coastal plain were artificially drained with networks of main canals, roadside collector ditches, and field lateral or third-stage ditches (Skaggs et al., 1994; Amatya and Skaggs, 2011; Hughes, 2014) to lower the WT (fig. 1b),

thereby improving soil trafficability during silvicultural operations and removing excess soil moisture to improve the productivity of the intensively managed plantation forests (Amatya and Skaggs, 2001; Lohmus et al., 2015; Fox, 2000; Beltran et al., 2010). However, this artificial drainage may have altered the flow paths and hydrology of the region by increasing runoff and making the soil drier than it would have been under unaltered conditions (USEPA, 2017).

Silvicultural practices such as harvesting, site preparation, bedding, planting, fertilization, regeneration, and thinning (Beltran et al., 2010; Amatya et al., 2011) and water management, including controlled drainage in some cases, may further complicate site hydrology and downstream water quality. For example, timber harvesting reduces ET, thus elevating the groundwater level (Trousdel and Hoover, 1955; Sun et al., 2000; Xu et al., 2002; Amatya et al., 2006a) and increasing the water yield, stormflow volume, and peak flow rate from a forested site until the canopy is regenerated (Riekerk, 1989; Ursic, 1991; Amatya et al., 1997a, 2006a; Lebo and Herrmann, 1998; Sun et al., 2000, 2004; Xu et al., 2002; Webb et al., 2012). Skaggs et al. (2019) reported that silvicultural operations such as site preparation and bedding for planting or regeneration dramatically reduced the saturated conductivity and transmissivity of the soil profile in a drained pine forest. However, the saturated conductivity and transmissivity increased to preharvest levels as the plantation matured, potentially increasing the drainage rates and removing wetland hydrology from the drained sites.

In recent years, plantation forests for timber production,

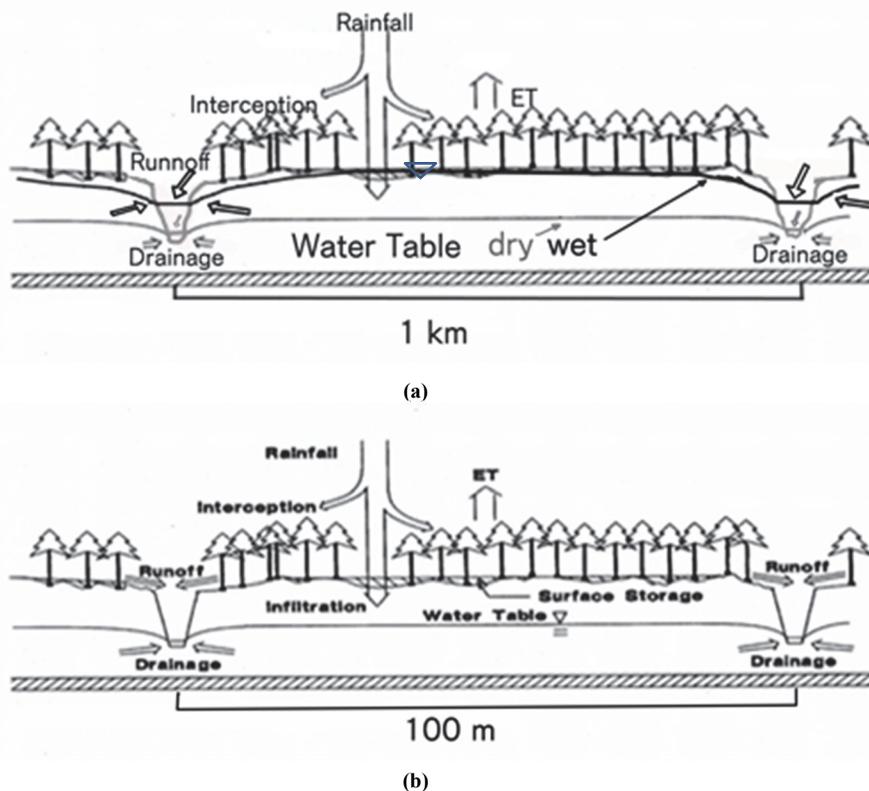


Figure 1. Schematic of water balance components in the lower coastal plain for (a) a naturally drained forest where natural drainages are at least 1 km apart and (b) a managed pine forest drained by field lateral ditches that are 100 m or more apart (after Amatya et al., 2019).

which are highly productive if managed well, have come under pressure from row crop agriculture (Skaggs et al., 2011) and biofuel production (King et al., 2013; Ssegane et al., 2017; Chescheir et al., 2018). Naturally drained forests on the coastal plain, on the other hand, are coming under pressure from rapid urban development (Sun and Lockaby, 2012; Driscoll et al., 2010). As a result, both of these systems are becoming vulnerable to altered hydrology, degraded water quality, and potential loss of other ecological functions (Callahan et al., 2012; Sun and Lockaby, 2012; Dai et al., 2013; Marion et al., 2013; Hitchcock et al., 2014; Lohmus et al., 2015), including conversion to other land uses, such as restoration of longleaf pine (*Pinus palustris*) in the south (Viner et al., 2016). At the same time, increasing extreme events, such as hurricanes, that are influential disruptors of ecosystem processes (Williams et al., 2013), as well as tropical storms, high-intensity storms, and droughts, have further complicated the hydrologic effects of land use conversion and development on the Atlantic and Gulf coasts. Therefore, understanding the interactions of climate, drainage, soils, and vegetation is critical for sustainably managing these coastal forests and reliably predicting the effects of land use and climate variability on hydrology and water quality, and in turn the potential impacts on water supply and other critical ecosystem functions (Sun et al., 2002; Keppeler et al., 2008; Amatya et al., 2011; Ford et al., 2011; Marion et al., 2013; McLaughlin et al., 2013). Evaluations using long-term data, where available, provide better insights into the baseline hydrology conditions of these forests for reliable assessment of the effects on forest ecosystem processes that otherwise would be difficult to capture or be missed by short-term experimental studies (Amatya et al., 2016a; Vose et al., 2014), as was recently shown by Jayakaran et al. (2014) for the effects of a hurricane on the vegetation and streamflow dynamics of a paired forested watershed in coastal South Carolina (SC). Furthermore, Tetzlaff et al. (2017) reported that observations and data from long-term experimental watersheds are the foundation of hydrology as a geoscience.

Recently, in a long-term (1988-2008) field-measured water balance study using a drained watershed (fig. 1b) as a reference, among two other treatments at a managed pine forest site in Carteret County in coastal North Carolina (NC), Amatya and Skaggs (2011) found that the mean annual runoff varied from 5% in a dry year (2001) to 56% in the wettest year (2003), with an average of 32%, of the annual rainfall. Similarly, in a multi-model approach for describing the long-term (1946 to 2008) water balance for a relatively undisturbed natural forest at Santee Experimental Forest on the SC coastal plain (fig. 1a), Dai et al. (2013) also found that runoff varied from 5.5% to 44.4%, with an average of 24%. Coincidentally, these values were found to be similar to a two-year field study by Harder et al. (2007), who found measured runoff as low as 8% for a relatively dry year (2004) to as high as 47% for a wet year (2003). In a short-term (1996 to 2001) comparative hydrologic analysis between the drained Carteret (NC) site and the undrained Santee (SC) site, Amatya et al. (2003a) reported that although the average annual rainfall was lower, the undrained watershed in SC had much shallower WT depths with

somewhat frequent outflows compared to the drained NC watershed, but limited data prevented for runoff comparison. The study emphasized the need for long-term rainfall and ET in comparative assessments of the hydrology of these poorly drained coastal watersheds. Multiple hydrology and water quality studies using long-term data from USDA-ARS experimental watersheds with predominantly agricultural land use on the coastal plain of Georgia were reported by Bosch et al. (2007). McLaughlin et al. (2013) synthesized information on precipitation and ET and their potential influence on water yield using data from multiple short-term studies conducted on various pine forests of varying stand age and density in Florida and two in coastal NC, including the Carteret site, to test the hypothesis that management of forested uplands for lower basal area, currently a priority for habitat improvement on public lands, may also increase water yield through decreased evapotranspiration (ET), without differentiating such effects due to drained forests. Most recently, Amatya et al. (2016a) synthesized hydrologic processes, including runoff magnitude, duration, and frequency, for ten long-term reference watersheds at USDA Forest Service Experimental Forests across the nation.

Studies exclusively comparing the hydrology of a drained forest watershed with that of an undrained natural forest on the coastal plain are limited, except for Sun et al. (2002), who compared two coastal pine forests (one undrained site in pine flatwoods in Florida and the drained Carteret site, as in this study) with a more inland and upland NC site. Our comparison for two coastal sites differs from that of Sun et al. (2002) in that (1) we used longer-term data from the Carteret site covering two extreme climatic events and analyzed the WT at both sites, (2) our 160 ha undrained Santee site has different vegetation (mixed pine and hardwood) from the slash pine and cypress wetlands of the 60 ha Florida site and has more poorly drained soils than the well-drained sandy soils of the Florida site, and (3) we briefly review some of the earlier modeling studies using the complex process-based MIKESHE (DHI, 2005) and the relatively less complex DRAINMOD (Skaggs, 1978; Skaggs et al., 2012), which were used to compare and contrast the hydrology of the two coastal forest types at the Santee and Carteret sites using the same data. Another motivation of this study was to determine whether the data from a short-term study can adequately describe the hydrology of these two distinct coastal forest watersheds with long-term monitoring covering periods of various natural and anthropogenic disturbances.

Results from an earlier short-term study by Amatya et al. (2003a) that the undrained Santee site yielded higher and more frequent outflows during wet years, despite its lower mean annual rainfall than the drained Carteret site were brought into question by a recent flow regime analysis, presented by Jayakaran et al. (2014), showing the impacts of Hurricane Hugo in 1989 on streamflow during the 1996-2001 period. This article revisits the earlier short-term annual and seasonal streamflow, ROC, ET, and depth to WT results and compares them against an additional five years of data (2004-2008), when the watersheds at Santee and Carteret were both undisturbed. The second objective evaluates the same hydrologic parameters using longer-term data

through 2008 from both sites that exclude the hurricane-influenced years of 1989-2003 at the WS80-SC site, followed by a discussion on the assessment of ROC, streamflow, flow duration, WT, and ET while comparing Hugo's immediate (1990-1994) disturbance effects on streamflow at WS80-SC with harvest effects on adjacent similar watersheds at the D1-NC site. The third objective was to provide a brief review of past modeling studies that tested two widely used DRAINMOD (Skaggs, 1978; Skaggs et al., 2012) and MIKESHE (DHI, 2005) models to describe the hydrology of these two distinct coastal forests, while identifying the strengths and limitations of each of these two models using the simulated results for streamflow, WT, and ET for the 2003-2008 period.

SITE DESCRIPTION

WS80-SC: NATURALLY DRAINED WATERSHED

The USDA Forest Service Santee Experimental Forest was established in 1937 (<http://www.srs.fs.usda.gov/charleston/santee/index.php>) in eastern SC to support long-term scientific studies of coastal forest ecosystems and their management (Amatya and Trettin, 2007). The study site is located at 33.15° N and 79.8° W within the Santee Experimental Forest near the town of Huger (fig. 2).

Two headwater watersheds (WS77 and WS80) drain first-order streams to Turkey Creek in the south, a tributary of Huger Creek, draining to the East Cooper River, a major tributary of the Cooper River, flowing into the Charleston Harbor System. These low-gradient watersheds, with elevations from

4 to 10 m and 0% to 3% slopes, were instrumented in the mid-1960s to study water budgets, rainfall-runoff processes, flooding patterns, and effects of rainfall on WT depth and soil moisture (Amatya and Trettin, 2007). The 155 ha WS77 watershed was established in November 1963, followed by the 206 ha WS80 watershed in October 1968. Both watersheds contained stands of greater than 80-year-old loblolly pine (*Pinus taeda* L.). The treatment (WS77) and control (WS80) watersheds were originally used to study hydrologic and water quality effects of prescribed burning to restore red-cockaded woodpecker (*Picoides borealis*) habitat. In November 2001, a small part of the watershed in the northeastern corner of WS80 was allowed to drain separately through a culvert, reducing the watershed size to 160 ha. Neither watershed was monitored from May 1982 until November 1989. The Santee Experimental Forest and surrounding area experienced the full force of Hurricane Hugo (Category IV) on 21 September 1989. More than 80% of the trees were destroyed, and nine long-term studies were prematurely terminated due to storm damage (Hook et al., 1991; Williams et al., 2013). Soon after Hugo, WS77 was salvage harvested, while WS80 was allowed to regenerate with debris in place. The watersheds now contain naturally regenerated vegetation, with loblolly pine and hardwoods predominating. The leaf area index (LAI) of the mixed pine and hardwood stands of the control watershed (WS80) was measured monthly during 2008-2009 using a LI-COR meter. It varied from 1.7 to 4.0 m² m⁻², with an average of 2.90 m² m⁻² (Dai et al., 2010a).

Common soils in the area are somewhat poorly to very poorly drained Aquic Alfisols and Ultisols of clay and fine

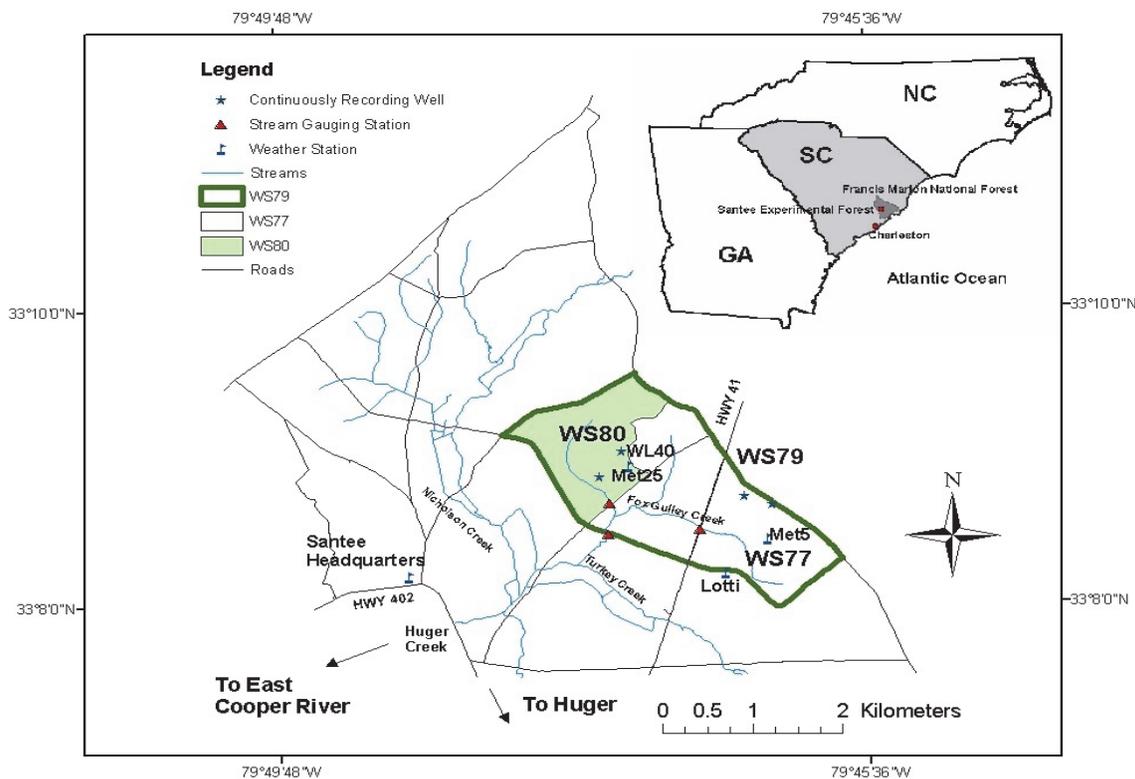


Figure 2. Location map of two first-order (WS77 and WS80) and one second-order (WS79) experimental watersheds within the Santee Experimental Forest near Huger, South Carolina. Locations of monitoring stations are also shown (after Harder et al., 2007).

sediments, which typically contain argillic horizons at 1.5 to 2.0 m depth (fig. 1a) and are influenced by seasonally high WT (SCS, 1980). These topographic and soil characteristics indicate a high surface water detention capacity and slow surface water drainage. The climate is mild and wet, with an average temperature of 18.3°C and average annual precipitation of 1370 mm (Dai et al., 2013).

For more than three decades, this paired system has been used for collaborative studies on watershed ecohydrology, biogeochemistry, and forest management (including prescribed fire and thinning), as well as the effects of hurricanes, climate variability, and changes to forest ecosystems (Richter, 1980; Richter et al., 1983; Amatya et al., 2006b; Amatya and Radecki-Pawlik, 2007; Harder et al., 2007; Dai et al., 2010a, 2013; Vose et al., 2011). A chronology of activities that occurred during the period of this review is presented in table 1. Detailed descriptions of this site and past studies of it are given elsewhere (Amatya et al., 2006b; Amatya and Trettin, 2007).

D1-NC: ARTIFICIALLY DRAINED WATERSHED

The NC watersheds were instrumented in 1987 by Weyerhaeuser Company to examine the long-term impacts of water management and silvicultural treatments on the tree growth, hydrology, and water quality of a managed pine forest (McCarthy et al., 1991; Amatya and Skaggs, 2011). The study site is located on a loblolly pine plantation in Carteret County (34° 48' N, 76° 42' W) (fig. 3). Research was initiated in 1986 by North Carolina State University, with field data collection beginning in 1987 and continuous hydrologic data collection starting in 1988. The site consists of three artificially drained

experimental watersheds (D1, D2, and D3), which are 24.7, 23.6, and 26.8 ha, respectively (fig. 3). The soils are flat and poorly drained under natural conditions. Each of the three watersheds is drained by four 1.4 to 1.8 m deep lateral ditches spaced 100 m apart. The soil is a hydric series, Deloss fine sandy loam (fine-loamy mixed, Thermic Typic Umbraquilt) with a shallow WT. The depth to the restricting soil layer is about 3 m (fig. 1b). McCarthy et al. (1991) and Amatya et al. (1996, 2006a) provide detailed descriptions of the site, including the history of the control, a loblolly pine stand planted in 1974 and commercially thinned in late 1988. LI-COR-based LAI of the pine stand on the control watershed (D1) measured between 1996 and 2004 varied from 1.35 to 4.84 m² m⁻². Details of the hydro-meteorological measurements and site descriptions are given by Amatya et al. (1996), Beltran et al. (2010), and Amatya and Skaggs (2011).

Hydrologic calibration occurred from 1988 to 1990, followed by paired watershed studies to evaluate the effects of controlled drainage treatments using both raised weirs and orifice weirs at the outlet (Amatya et al., 1996, 2000, 2003b). Effects of silvicultural operations were also studied, such as harvesting of D2 (Amatya et al., 2006a; Skaggs et al., 2006), thinning of D3 (Amatya and Skaggs, 2008), and fertilization of both D2 and D3 (Beltran et al., 2010). Watershed D1 was maintained in conventional drainage and served as the control watershed for these comparative studies. All treatments applied on these watersheds are shown in table 1. In early 2009, D1 was harvested. Another watershed (D0) was added (not shown), and treatments were modified to evaluate the hydrologic and water quality impacts of an intensive biofuel

Table 1. Chronology of activities at the South Carolina (SC) and North Carolina (NC) watershed sites.

Year	SC Watersheds		NC Watersheds			Activities and Disturbances
	WS80	WS77	D1	D2	D3	
1963	-	Y	-	-	-	Weir installed
1968	Y	-	-	-	-	Weir installed; calibration starts
1974	-	-	Y	Y	Y	Pine trees planted at 2100 t ha ⁻¹
1976	Y	Y	-	-	-	Water quality monitoring and calibration end
1977-1981	-	Y	-	-	-	Prescribed burn of 20% of WS77 each year
1980	-	-	Y	Y	Y	Thinned to approximately 100 t ha ⁻¹
1981	Y	Y	-	-	-	Data collection discontinued
1981	-	-	Y	Y	Y	Fertilized at 169 kg N ha ⁻¹
1987	-	-	Y	Y	Y	Instrumentation installed
1988	-	-	Y	Y	Y	Commercial thinning to 370 t ha ⁻¹ ; calibration starts
1989	-	-	Y	Y	Y	Fertilized at 225 kg N ha ⁻¹
Sept. 1989	Y	Y	-	-	-	Hurricane damage; 80% of mature trees killed
1990	-	-	Y	Y	Y	Calibration ends; downstream pump installed to minimize submergence
1990	-	Y	-	-	-	Salvage and large dead stems removed
1990-1994	-	-	-	Y	Y	Raised weir levels; controlled drainage
1993	-	Y	-	-	-	Regeneration on WS77 decreases flow
1995	-	-	-	Y	-	Whole-tree harvest
1995-1999	-	-	-	-	Y	Controlled drainage with an orifice weir
1996	-	-	-	Y	-	Site preparation and bedding
1997	-	-	-	Y	-	Trees planted at 2100 t ha ⁻¹
1999	-	Y	-	-	-	NDVI of WS77 = pre-hurricane conditions
2001	Y	-	-	-	-	Reduced watershed size from 206 to 160 ha
2001	-	Y	-	-	-	Mastication; understory vegetation mowed
2002	-	-	-	-	Y	Thinned to 285 t ha ⁻¹
2003	-	Y	-	-	-	Growing season prescribed burn of >80% area
2003	Y	-	-	-	-	Pine basal area = pre-hurricane area
2005	-	-	-	Y	Y	Fertilization: D2 at 115 kg N ha ⁻¹ and D3 at 172 kg N ha ⁻¹
2006	-	Y	-	-	-	Biomass thinning of understory
2007	-	Y	-	-	-	Prescribed burn
2009	-	-	Y	-	Y	Whole-tree harvest; site preparation for replanting pine and establishing switchgrass

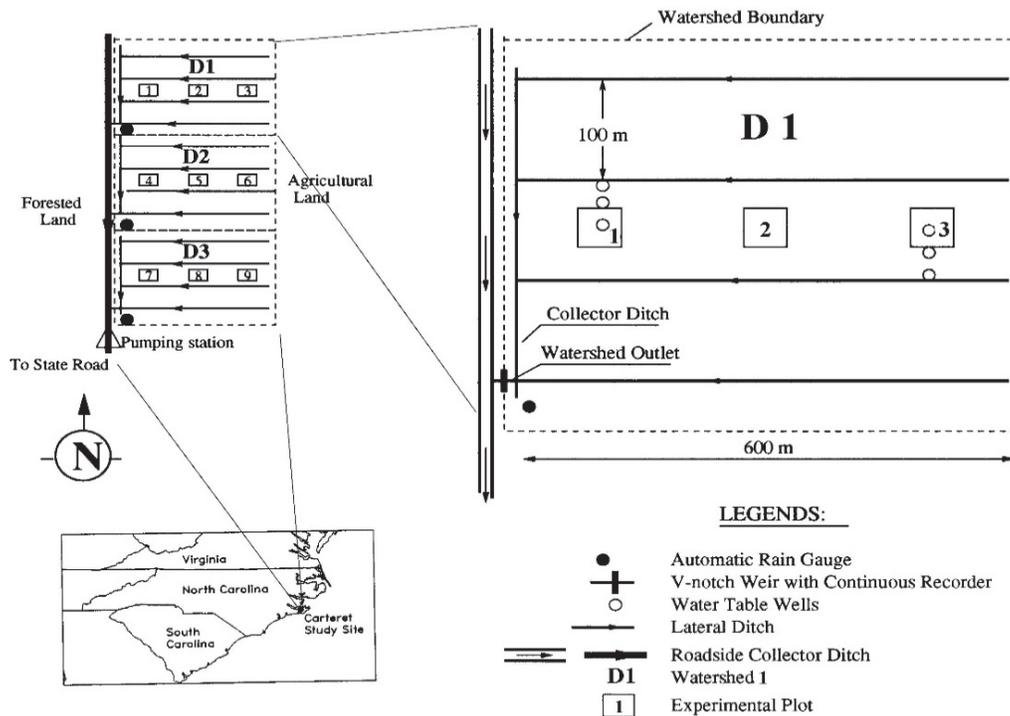


Figure 3. Location and layout of three experimental watersheds (D1, D2, and D3) in Carteret County, North Carolina.

treatment. The treatments focused on planting and intercropping switchgrass, a cellulosic biofuel crop, and are described by Ssegane et al. (2017).

HYDRO-METEOROLOGICAL MEASUREMENTS

WS80-SC Site

Daily precipitation and daily maximum and minimum temperatures were measured manually at the weather station at the Santee Experimental Forest headquarters from 1946 to 1995 and thereafter have been measured automatically using a Campbell Scientific datalogger. Since 2003, climate parameters, including air temperature, relative humidity, wind speed and direction, vapor pressure, and solar and net radiation, have been measured automatically at 30 min intervals using sensors. Continuous data measured at the weather station were used to estimate the Penman-Monteith PM-forest PET (Amatya and Harrison, 2016). Streamflow rates at the WS77 and WS80 watershed outlets were measured based on stage-discharge relationships of compound V-notch weirs, where stages have been measured at 10 min intervals at the flow gauging stations using data loggers since 1963. The streamflow rate was calculated using standard rating curve methods developed for these weirs, and the 10 min values were integrated into daily, monthly, and annual flows normalized to mm per day by dividing the watershed area. Hourly WT measurements were continuously recorded in upland and lowland groundwater wells of about 3 m depth (fig. 2). Details of the hydro-meteorological measurements and data processing for the Santee paired watersheds are reported elsewhere (Amatya et al., 2006b; Amatya and Trettin, 2007; Dai et al., 2013). Current and historical data and metadata can be accessed at <http://www.srs.fs.usda.gov/charleston/santee/index.php>.

D1-NC Site

Rainfall was measured with a tipping-bucket rain gauge backed up by a manual rain gauge on the western side of the study watershed. Since 1988, air temperature, relative humidity, wind speed, and solar and net radiation have been continuously measured with an automatic weather station, as described by Amatya et al. (1996) and Amatya and Skaggs (2011), to estimate daily PET using the P-M method. A 120° V-notch weir with an automatic stage recorder, located in a water-level control structure at a depth of about 0.3 m from the bottom of the outlet ditch, was used to continuously measure drainage outflow using the stage measured at 12 min intervals with standard weir equations. The flow rates were integrated to obtain daily, monthly, and annual flows. In 1990, a pump was installed downstream in the roadside collector ditch to prevent weir submergence during larger events. An additional recorder was placed downstream from the weirs in May 2005 to determine if weir submergence occurred and to correct flows in that event. Hourly WT measurements have been continuously recorded in groundwater wells located at the front and back sides of the watershed (fig. 3). Details of the measurement are given by Amatya and Skaggs (2011), Beltran et al. (2010), and McCarthy et al. (1991).

DIFFERENCES IN CHARACTERISTICS BETWEEN THE TWO SITES

The characteristics of the control watersheds, WS80-SC at Santee and D1-NC at Carteret, are summarized in table 2. The area of WS80-SC is more than six times that of D1-NC, with a slightly greater elevation range and slope. Both watersheds are on poorly drained, high WT soils. WS80-SC is 1° farther south, with an average temperature that is 2.3°C higher than

Table 2. Comparison of characteristics for WS80-SC and D1-NC.

Parameter	WS80-SC (Santee)	D1-NC (Carteret)
Location	33° N, 80° W	34° N, 76° W
Elevation (a.m.s.l.)	7.0 m	3 m
Watershed size	160 ha	25 ha
Long-term annual precipitation	1370 mm ^[a]	1517 mm ^[b]
Long-term mean annual temperature	18.5°C ^[a]	16.2°C ^[b]
Long-term mean annual PET	1136 mm ^[a]	1010 mm ^[b]
Slope	<4%	<0.2%
Drainage type	Natural, first-order stream	Artificially drained, pattern drainage
Dominant soil type	Wahee series, sandy clay loam, somewhat poorly drained	Deloss series, fine sandy loam, very poorly drained
Saturated conductivity	0.86 to 6.9 m d ⁻¹ ^[c]	3.9 m d ⁻¹ ^[d]
Drainable porosity	0.05 m m ⁻¹	0.05 m m ⁻¹
Deep seepage	Negligible	Negligible
Mean annual water table depth	0.85 m	0.95 m ^[b]
Vegetation	Mixed pine and hardwood	Loblolly pine with natural understorey

^[a] Source: Dai et al. (2013).

^[b] Source: Amatya and Skaggs (2011).

^[c] Source: Dai et al. (2010a).

^[d] Source: Amatya and Skaggs (2001).

D1-NC. The long-term average annual precipitation of 1370 mm at WS80 (Dai et al., 2013) is about 10% lower than the average of 1517 mm at D1-NC (Amatya and Skaggs, 2011). However, the average annual Hargreaves-Samani (1985) potential evapotranspiration (PET), adjusted with Penman-Monteith (Monteith, 1965) PET-based factors, for WS80-SC (Dai et al., 2013) was 1136 mm, which is 11% higher than the PET of 1010 mm at D1-NC (Amatya and Skaggs, 2011). Amatya et al. (2003a) presented an early comparison of the hydrology of these two forested watersheds for 1996-2001, with post-hurricane effects on WS80-SC.

After Hurricane Hugo in 1989, the forest stands on WS80-SC were regenerated (Hook et al., 1991). Only 15 years (2004) after regeneration, the runoff characteristics had returned to pre-Hugo levels (Amatya et al., 2006b; Jayakaran et al., 2014). At Carteret, post-harvest (1995) disturbance effects in a treatment watershed (D2), adjacent to the D1 watershed (Amatya et al., 2006a; Tian et al., 2012) shown in figure 3, were compared with post-Hugo effects on WS80-SC (table 1). Using twelve years (1969-1980) of pre-Hugo data and five years (2004-2008) of post-Hugo data from WS80-SC, Santee was analyzed to exclude hurricane influence. Seventeen years (1988-2004) of data from D1-NC were used. These longer-term data were also analyzed to compare streamflow (using daily flow duration), runoff coefficient (ROC = total flow / total precipitation), and ET with the results of the shorter-term study above using means, standard deviations, F-tests for equal variances, and t-tests for significance in differences in Microsoft Excel 2013.

Additional analyses were conducted to examine the effects of soil water storage as a result of seasonal antecedent conditions using the daily WT depths on annual drainage, ROC, and ET for 2004 to 2008 for the NC and SC sites. ET was estimated as a residual in the water balance (rainfall – drainage – ΔS), where ΔS is the change in storage between

the end and beginning of the year. Measured daily WT depths at the beginning and end of the year were used together with the average drainable porosity of the soil for each site (table 2), which was obtained in earlier studies as 0.05 m m⁻¹ for both Santee (Harder et al., 2007) and Carteret (Amatya and Skaggs, 2001), to compute the change in soil water storage. A unique example of storage effects on hydrology was also demonstrated using 2001-2003 data for the D1-NC site. We also calculated the PET/P (dryness index) and ET/P (evaporative index) ratios defined elsewhere (Creed et al., 2014; Evaristo and McDonnell, 2019) to partition potential effects of drainage from climate on water yield using a theoretical Budyko curve for these two sites, both with forest land use.

Standard t-tests were conducted using Microsoft Excel 2010 for examining the significance ($\alpha = 0.05$) between the mean hydrologic parameters (rainfall, outflow, ROC, ET, and WT) from the two study sites for periods with equal samples. An F-test was computed first for each measured parameter to examine sample variance for using the appropriate t-test. A Z-score test for rainfall that uses its long-term mean and standard deviation to obtain a normalized measure of precipitation anomaly between the two sites was also calculated to characterize a year or a period as wet, dry, or moderate for each site. Other statistical parameters used in various analyses were mean, standard deviation, coefficient of determination (R^2), Nash-Sutcliffe efficiency (NSE), and mean absolute error (MAE).

Earlier results from two different hydrologic models, DRAINMOD (Skaggs, 1978) and MIKESHE (DHI, 2005), with varying levels of complexity that had been tested and applied earlier on these coastal forest watersheds with poorly drained high WT soils for the 2003-2008 period (Amatya et al., 2003b; Harder et al., 2006; Dai et al., 2010a, 2010b) were briefly reviewed to identify the strengths and limitations of each model in simulating the hydrologic processes for these two different forest types. DRAINMOD is a process-based, field-scale model developed to describe hydrologic processes such as infiltration, surface runoff, subsurface drainage, and ET for poorly drained and artificially drained soils (Skaggs et al., 2012). The model, modified further for simulating the hydrology of pine forests, is based on water balances at a mid-point WT in the soil profile and on the field surface between parallel ditches or tiles draining the field. MIKESHE (DHI, 2005) is a modularized and spatially distributed watershed hydrological model with a friendly graphical user interface (GUI) for simulating the complete water cycle at each grid cell and in the total watershed. This watershed hydrological model simulates the water balance of any area with a complete terrestrial water cycle, including 3-D saturated water movement in soils, 2-D water movement of overland flow, 1-D water movement in river and streamflow, unsaturated water movement, and ET. A comparison of the general strengths and limitations of these two models, including some other widely used models, is provided elsewhere (Golden et al., 2014; Amatya et al., 2013). Model performance evaluation statistics, including Nash-Sutcliffe efficiency (NSE), mean absolute error (MAE), and normalized

Table 3. Measured annual average temperature (Temp.), total rainfall (P), average water table (WT) depth, total flow, flow as a percentage of rainfall (ROC), PET/P, and evapotranspiration (ET = rainfall – flow – ΔS) on watersheds WS80-SC at Santee and D1-NC at Carteret for 2004–2008. Average WT depth for 2004 may be biased due to data gaps when WT fell below 100 cm, the limit of the shallow well recorder.

Watershed and Year	Temp. (°C)	P (mm)	WT (cm)	Flow (mm)	ROC (%)	PET/P	ET (mm)
WS80-SC							
2004	17.4	962	64	73	7.6	1.21	905
2005	17.1	1540	65	307	19.9	0.79	1156
2006	17.4	1255	76	139	11.1	0.93	1112
2007	17.7	923	144	58	6.3	1.30	910
2008	17.2	1562	54	318	20.4	0.74	1206
Mean	17.4	1248	80.6	179	13	0.99	1058
D1-NC							
2004	16.6	1313	97	389	29.6	0.81	924
2005	16.4	1777	72	798	44.9	0.59	979
2006	16.7	1329	83	247	18.6	0.90	1082
2007	16.9	1201	110	257	21.4	1.04	944
2008	16.7	1360	116	247	18.2	0.89	1113
Mean	16.7	1396	95.6	387.6	26.5	0.85	1015
Statistical Tests^[a]							
	Temp.	P	WT	ROC	ET/P	PET/P	
F-test	0.33	0.28	0.16	0.17	0.43	0.22	
t-test	0.0007	0.41	0.66	0.050	0.09	0.31	

^[a] F-test probability used for testing if variances of data from separate watersheds were not significantly different; t-test probability used for testing if separate watershed mean values were not significantly different. All t-tests assumed equal variance among samples, as all F-test p-values were >0.05. The WT in 2004 for WS80-SC was sporadic, with several missing days, and was not included in the analysis.

percent error (NPE) (Tian et al., 2012), were used in the discussion of the application of these two models.

RESULTS AND DISCUSSION

SHORT-TERM (2004–2008) ANALYSIS

Annual hydrologic parameters (rainfall, flow, and ET) for the 2004–2008 period, when neither control was disturbed, are compared in table 3. The five-year (2004–2008) average rainfall (P) for WS80-SC was 1248 mm, which was lower than the 1396 mm for D1-NC (table 3), consistent with the previous five-year interval (1996–2001) reported by Amatya et al. (2003a). The year 2007 had the lowest rainfall at both sites, and the lowest ROC was at WS80-SC. Annual runoff and ROC were generally higher on drained watershed D1-NC (mean ROC = 26.5%) than WS80-SC (mean ROC = 13.0%) (table 3), with the largest difference in 2005 (fig. 5a) and the smallest in 2008 (fig. 5b). This is consistent in pattern with the longer-term results of 21% for WS80 (Amatya et al., 2006b; Dai et al., 2013) and 31% for D1 (Amatya and Skaggs, 2011).

These results are also similar to the values of 13% for an undrained pine flatwood site in northern Florida and 30% from a ten-year (1988–1997) mean for the same D1-NC watershed reported by Sun et al. (2002). The authors speculated that the PET as partially controlled by the climate, in addition to precipitation, topography, and vegetation, all might have contributed to these differences, although the climate was a key factor. Effects of topography were ruled out, as both sites were low-gradient systems. Interestingly, the five-year mean ET of 1069 at the undrained WS80-SC forested wetland site was lower than the long-term PET of 1135 mm

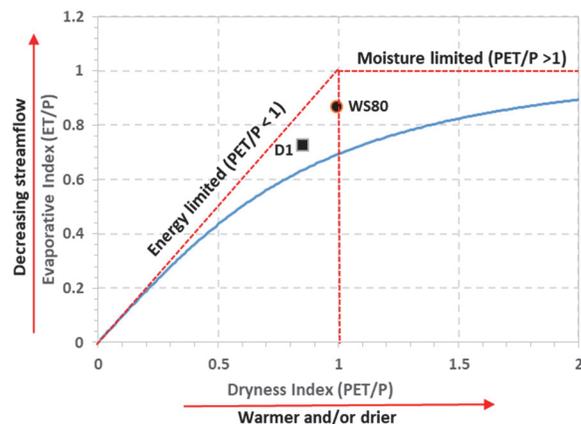


Figure 4. Original Budyko (1974) curve (dryness index vs. evaporative index) shown in blue. Dotted red lines represent the division between moisture limited ($PET/P > 1$) and energy limited ($PET/P < 1$).

(Dai et al., 2013), indicating that its ET limited by soil moisture, in contrast with D1-NC, where the ET of 1015 mm was very close to the long-term PET of 1010 mm (Amatya and Skaggs, 2011), indicating no soil moisture limitation. We attributed this to the higher PET/P ratio at the undrained WS80-SC site (0.99) compared to the drained D1-NC site (0.85). Sun et al. (2002) observed the same phenomenon between the Florida pine flatwood wetland with an even higher PET/P ratio of 1.13 (more moisture limited) and the D1-NC site and attributed the difference to the same reason. The fact that both the mean PET/P of 0.99 and ET/P of 0.87 for WS80-SC and the PET/P of 0.85 and ET/P of 0.73 for D1-NC lie above the Budyko curve (fig. 4) with a positive offset indicate the effects of forest vegetation, yielding lower than expected streamflow for both watersheds (Zhang et al., 2001, 2004; Donohue et al., 2006; Creed et al., 2014; Evaristo and McDonnell, 2019). Interestingly, both sites' PET/P and ET/P values plot between the tropical coniferous and montane grasslands above the curve recently published by Evaristo and McDonnell (2019), with WS80-SC closer to the tropical coniferous site than D1-NC. This also indicates that less runoff is likely from WS80-SC than from D1-NC. The vertical offset of ET/P between the two sites (fig. 4) indicates that climate has more effect on the increased water yield from D1-NC than drainage and/or soil factors. The further left location of D1-NC compared to WS80-SC indicates that this site is more energy limited. We further discuss the effects of rainfall on the ROC of D1-NC using longer-term data (table 4) in the next section.

We examined the role of hurricanes and tropical storms on the hydrology (particularly daily flow) of these two different Atlantic Coastal Plain sites using two specific years (2005 and 2008) of data (fig. 5). During both years, D1-NC produced slightly more runoff during the winter than the summer. However, in the relatively wet year of 2005, back-to-back storm events on May 5 (142 mm of rain), June 2 (86 mm), and July 13 (80 mm) and three much larger storms (Hurricane Ophelia on September 13–14 with 208 mm of rain in 38 h, on October 7–8 with 197 mm in 48 h, and on October 24 with 46 mm) resulted in considerably higher runoff and ROC at D1-NC (fig. 5a) (Beltran et al., 2010) than WS80 (table 3). However, in January through April 2008,

Table 4. Comparison of long-term annual hydrologic parameters for control watersheds WS80-SC at Santee and D1-NC at Carteret. Annual ET estimated as the difference between rainfall (P) and flow.

Watershed and Year	P (mm)	Flow (mm)	ET (mm)	ET/P	ROC (%)	
WS80-SC	1969	1408	260	1148	0.82	19
	1970	1362	251	1111	0.82	18
	1971	1776	495	1281	0.72	28
	1972	1094	174	920	0.84	16
	1973	1429	315	1114	0.78	22
	1974	1413	229	1184	0.84	16
	1975	1440	283	1157	0.80	20
	1976	1449	292	1157	0.80	20
	1977	1274	141	1133	0.89	11
	1978	1225	146	1079	0.88	12
	1979	1546	356	1190	0.77	23
	1980	1414	297	1117	0.79	21
	2004	962	73	889	0.92	8
	2005	1540	307	1233	0.80	20
	2006	1255	139	1116	0.89	11
	2007	923	58	865	0.94	6
	2008	1562	318	1244	0.80	20
Mean	1357	243	1114	0.83	17.1	
SD	218.6	111.0	107.8	0.06	5.8	
CV (%)	15.24	39.62	10.27	7.00	33.3	
D1-NC	1988	1406	209	1197	0.85	15
	1989	1876	659	1218	0.65	35
	1990	1236	240	996	0.81	19
	1991	1575	519	1056	0.67	33
	1992	1619	584	1035	0.64	36
	1993	1514	585	929	0.61	39
	1994	1528	437	1081	0.71	29
	1995	1404	457	947	0.67	33
	1996	1646	705	941	0.57	43
	1997	1382	385	987	0.71	28
	1998	1658	770	888	0.54	46
	1999	1377	614	763	0.55	45
	2000	1718	857	861	0.50	50
	2001	852	45	807	0.95	5
	2002	1718	426	1292	0.75	25
	2003	2331	1308	1023	0.44	56
	2004	1313	389	924	0.70	30
Mean	1538	541	997	0.67	33.3	
SD	310	286	143	0.13	12.9	
CV (%)	20.16	52.87	14.34	19.38	38.9	
Statistical Tests ^[a]						
F-test	0.086	0.0002	0.238	0.001	0.001	
t-test	0.058	0.0007	0.014	0.0001	0.0001	

^[a] F-test probability used for testing if variances of data from separate watersheds were not significantly different; t-test probability used for testing if separate watershed mean values were not significantly different. Rainfall (P) and ET tests assume equal variance, while runoff (flow), Est_ET/P, and ROC tests indicate unequal variance.

the cumulative daily flow pattern was similar (>10 mm) to 2005 at both sites (fig. 5b), primarily due to WT below 70 cm or lower. Later, a long summer drought occurred on both watersheds, followed by a somewhat opposite pattern thereafter, with WS80 experiencing wetter events in September-October. For example, a large tropical depression at WS80-SC (164 mm in 24 h on October 23) caused the flow patterns to diverge, with daily flow exceeding 50 mm as a result of ponding of 8 cm at one of the upland wells (fig. 5). This is consistent with Harder et al. (2007), who found that stream outflow at the Santee Experimental Forest sites increases exponentially after the WT ponds more than 2 to 4 cm on average, potentially due to shallow surface runoff from large saturated areas with high conductivity. In contrast, the data in figure 5a show a rapid increase of outflow exceeding 25 mm d⁻¹ in 2005 when the WT rose to or above

the surface of D1-NC, possibly due to the large WT gradient from the midpoint of the field to the ditch outlet and high conductivity and transmissivity of the topsoil layer (Skaggs et al., 2011, 2019). These responses are also consistent with earlier observations from 1996-2001 data (Amatya et al., 2003a) and 1988-1997 data for D1-NC (Amatya and Skaggs, 2001). Notably, the outflow from D1-NC is primarily subsurface drainage to ditches, as the surface runoff is minimal due to raised pine beds (Skaggs et al., 2016).

Daily flow in response to WT depth for these undrained and drained sites for the same two years (fig. 5) showed that flow on both watersheds corresponded to the rise in WT depth, but the small drained watershed (D1-NC) flowed until the WT depth fell below about 70 cm, compared to only 40 cm for the larger undrained watershed. The daily WT was somewhat shallower in 2005 on both watersheds compared to 2008. The calculated Z-score values closer to unity (0.9997 for WS80-SC and 0.9999 for D1-NC) obtained using the normalized measure of rainfall for 2005 showed no significant difference in mean daily WT depths between the sites (table 3). However, that was not the case in 2008, with Z-score values of 0.9999 for WS80-SC and 0.0084 for D1-NC. Apparently, the lower than normal rainfall at D1-NC caused much deeper WT depth than at WS80-SC (table 3).

The mean annual WT depth of 85 cm at WS80 was shallower than at D1-NC (95 cm) (table 3), with longer a duration of near-surface levels during the winter and summer rainy periods (fig. 5) but deeper summer recessions during drier years such as 2004 and 2007 (much lower than average rainfall, table 3). Amatya et al. (2019) also reported the mean daily growing season WT depth of 105 cm on the undrained WS80-SC watershed for the 2004-2016 period as 7 cm shallower than the drained D1-NC site for 1988-2008. During some summer-fall periods (e.g., 2006 to 2008, fig. 5b) with high rainfall and ET demands, daily WT depths were deeper, creating larger storage at WS80-SC than at D1-NC, although the WT depths were generally deeper than 100 cm at both sites, with a larger variability (SD = 77.2 cm) at WS80-SC than at D1-NC (46.8 cm). The very small calculated Z-scores of only 0.0197 for WS80-SC and 0.0333 for D1-NC for the five-year (2004-2008) period indicate that both sites were relatively dry, with minimal effects of rainfall on WT.

In a North Carolina study, Skaggs et al (2011) reported that, due to a deeper WT in a drained pine forest than a nearby forested wetland at the end of a prior year, there was no outflow from the drained site until later in the following year, in contrast with the wetland, which had immediate flows due to the higher WT and saturated soils. This resulted in a slightly higher annual flow from the wetland than the drained site. Although it appears that runoff from lateral drainage may result in a deeper water table, on average, on drained forests, the authors reported otherwise for a drained forest on a young pine stand with a shallower WT than that of the forested wetland during the summer, likely due to lower ET from the shallow-rooted young pine trees. These results indicate the influence of WT (and antecedent soil water storage) on drainage and possibly ET, as shown below.

Analysis of the same five-year data (table 3) considering the annual change in soil water storage estimated using the

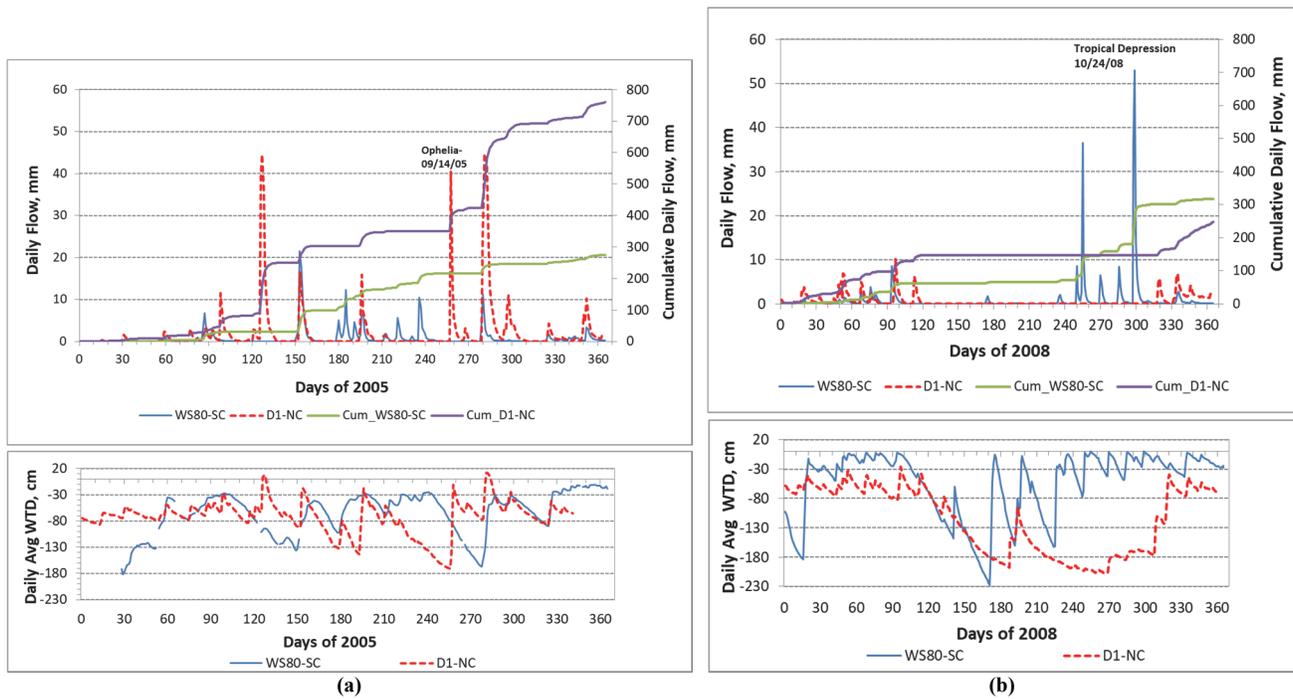


Figure 5. Comparison of measured daily flows and corresponding daily average water table depth (WTD) for the WS80-SC and D1-NC watersheds for (a) 2005 and (b) 2008.

measured daily WT at the beginning and end of the year with estimated drainable porosity for these poorly drained systems (Harder et al., 2007, for WS80-SC; Amatya et al., 1996, for D1-NC) (table 2) resulted in substantially different annual ET in some years compared to the estimates without soil water storage. For example, considering soil water storage, ET with excess moisture at the end of 2005 was only 1189 mm at WS80-SC (table 3), compared to 1266 mm without storage. On the contrary, ET in the relatively dry year 2007, with a soil water deficit, was 909 mm with storage at WS80-SC, compared to only 865 mm without storage. Similarly, the ET in 2007 was 972 mm with storage at D1-NC, compared to 944 mm without storage. This indicates the importance of soil moisture storage when quantifying the year-to-year variability in ET in these systems, as will be further illustrated below for a specific period at D1-NC for a period prior to 2004.

Due to the relatively short no-treatment period, a direct comparison (table 3) could be made only for temperature, which was significantly different between the sites ($p < 0.001$), while no significant difference ($p > 0.05$) was found between the sites for the rest of the hydrologic parameters. However, if we also include previously published data from these two watersheds (table 4), there are trends for higher (but barely significant at $p = 0.058$) rainfall, runoff, and ROC and lower estimated ET at D1 in contrast with WS80 that are highly significant ($p < 0.05$), as shown in table 4. However, the variances of annual rainfall and ET are not significantly different ($p = 0.058$), suggesting that these climatic parameters come from similar distributions.

LONG-TERM ANALYSIS

Runoff Coefficient, Streamflow (Runoff), Flow Duration, and Storm Events

Binstock (1978), Richter (1980), and Gilliam (1983) calculated the runoff and ROC for WS80-SC for some years between 1969 and 1980. The ROC varied from 11% to 28%, with an average of 16.7%, as shown in table 4, with no outflows during some months (May to October), indicating the absence of dependable baseflow, as reported by Young (1968). Data from the drained D1-NC site exhibit a similar wide range of runoff and ROC values, although the variance is significantly higher. The D1-NC data in table 4 (Amatya et al., 2006a) show three-year to five-year averages that had similar variations to those found for WS80-SC. Both watersheds showed wide variation in runoff between years with above-normal or below-normal rainfall. For both watersheds, the coefficient of variation (CV) of runoff is roughly double that of rainfall. In a comparative study examining the scaling effects on daily streamflow dynamics, Amatya and Radecki-Pawlik (2007) reported that the calculated pre-Hugo runoff-rainfall ratios for WS80 were consistent with those found on naturally drained forested watersheds in the NC coastal plain (Chescheir et al., 2003). Our long-term mean ROC values for the D1-NC site are also similar to those reported by Chescheir et al. (2003) with multi-site-year data for drained pine forests in eastern NC. These long-term ROC means for both sites are somewhat higher than the short five-year means shown in table 3, potentially with no significant difference between them.

Analysis of individual years with similar annual rainfall (within ~3% of each other) for both sites (table 4) showed that the mean ROC for the drained D1-NC site was consistently higher than that for the undrained site. For example, the years

1969-1970, 1974, and 1980 at WS80-SC had very similar rainfall to 1988, 1995, 1997, 1999, and 2008 at D1-NC. Despite the slightly higher average rainfall of 1399 mm at the undrained WS80-SC site than the 1386 mm at D1-NC, the mean ROC for the D1-NC site was 28%, compared to only 19% at WS80-SC. Analysis for the years 1979, 2005, and 2008 showed average rainfall of 1549 mm, which was again slightly higher than the mean of 1539 mm for 1991, 1993, and 1994 at D1-NC and also yielded a higher ROC of 33% for D1-NC, compared to only 21% for the undrained WS80-SC site. The same was true for the relatively drier years 1977, 1978, and 2006, with an average rainfall of 1251 mm yielding a lower mean ROC of only 11% for WS80-SC, compared to 20% for the drained D1-NC site but with a slightly lower mean rainfall of 1219 mm. Both sites had almost the same rainfall of 1776 mm in 1971 at WS80-SC and in 2005 at D1-NC, but the drained D1-NC site yielded 45% ROC, compared to only 28% for WS80-SC, potentially indicating the effects of drainage and soil on increased flow. Notably, for extreme conditions, both watersheds responded with similar ROC as well as WT depths. For example, the ROC of 5.8% for the driest year (2007, with 923 mm of rainfall) at WS80-SC was similar to 5.3% for 2001 at D1-NC when the WT depths at both watersheds were well below 2 m from the surface. The 2331 mm of rainfall at D1-NC was slightly lower than the highest recorded rainfall of 2171 mm in 2015 at WS80-SC (Amatya et al., 2016b). Both sites yielded the highest ROC of 56% at D1-NC and 46% at WS80-SC, respectively, with WT elevations at both watersheds ponding on the surface. These results support the hypothesis that the outflow from both of these poorly drained systems is dependent on the position of the WT and influenced by the balance of rainfall and ET (Amatya et al., 2019). This was supported by a study in coastal NC by Skaggs et al. (2011), who found WT frequently near the surface for an undrained wetland site, but outflow was similar to another adjacent drained forest for a limited wet year (1996), primarily because of higher outflow from the wetland site in the early part of the year due to wetter antecedent moisture at the wetland site with the WT near the surface at the end of 1995, in contrast with 1.5 m deep WT on the drained forest site.

A comparison of daily flow duration curves using ten years (1970-1979) of pre-Hugo data for WS80-SC and ten years

(1988-1997) of data for D1-NC (Amatya and Skaggs, 2001) is shown in figure 6. The flatter slope at the high flow range for D1-NC is an artifact caused by weir submergence; otherwise, the daily flow magnitudes for D1-NC were consistently higher than for the undrained WS80-SC site. The flow duration for flow ranges lower than 4 mm d⁻¹ is shown in the inset graph in figure 6, with higher-magnitude flows for D1-NC than for WS80-SC for the same frequency. During the low-flow period, WS80-SC flowed 58% of the time, consistent with Amatya and Radecki-Pawlik (2007), while the smaller D1-NC watershed flowed approximately 50% of the time, as expected. Again, this is consistent with Sun et al. (2002), who reported sustained mean monthly flow for the Florida pine flatwood site even during the summer months of June and July when flow on the D1-NC site was negligible, potentially due to higher ET of the managed pine forest. In both studies, the contribution of deep groundwater flow was assumed to be negligible due to the presence of restrictive clay horizons (Harder et al., 2007; Skaggs et al., 2016) (fig. 1).

Wilson et al. (2006) reported an average increase in stream outflow by 41% from WS80-SC for three years following Hurricane Hugo, consistent with the findings of Shelby et al. (2005) for NC coastal watersheds. Because the hurricane destroyed more than 80% of the mature forest canopy (Hook et al., 1991), primarily due to winds exceeding 200 km h⁻¹, these results can possibly be compared to studies of forest harvest in which almost 100% of the trees are removed. For example, after the 1995 harvest and logging of watershed D2 (table 1), which is adjacent to D1-NC at Carteret (fig. 3), flow increased by 35%, on average, for a three-year (1996-1998) post-harvest period compared to the control (D1-NC) (Amatya et al., 2006a), indicating effects similar to Hugo at WS80-SC but without salvage logging (Jayakaran et al., 2014). Although the increase in flow at both sites was likely due to a decrease in total ET, soil disturbance due to logging may also have played a role at the D2 site in NC.

Harder et al. (2007) examined WS80-SC daily event runoff for an extremely wet year (2003) and a relatively dry year (2004), finding a wide variability of daily outflows as affected by antecedent WT conditions. The largest daily out-

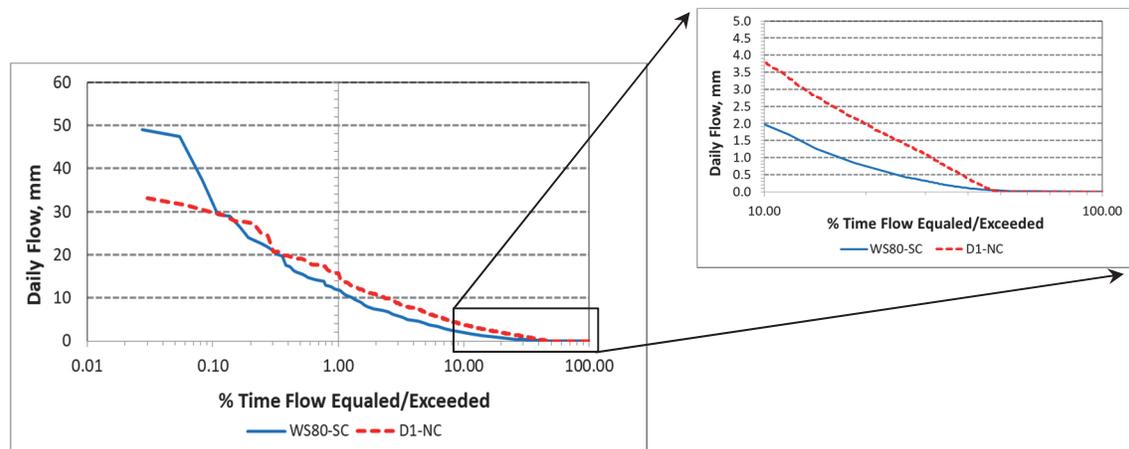


Figure 6. Daily flow duration curves using ten years of data for WS80-SC (1970-1979) and D1-NC (1988-1997).

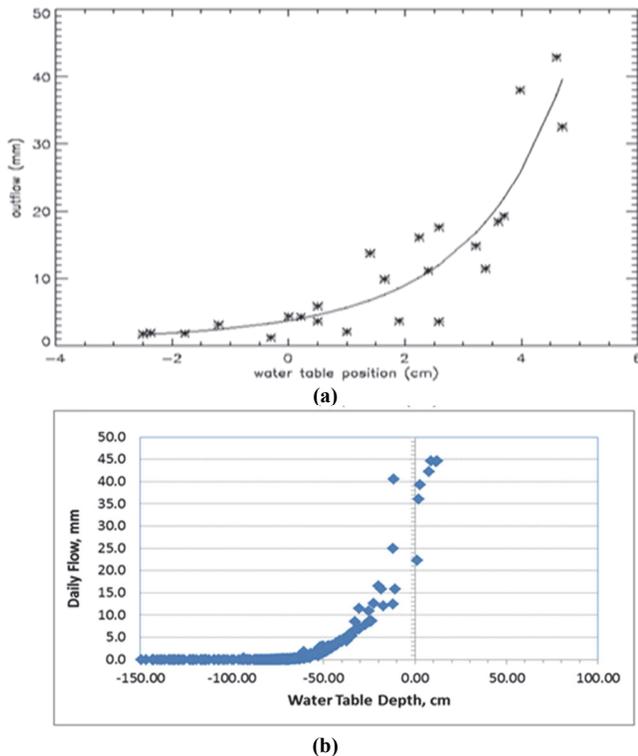


Figure 7. Daily stream outflow as a function of daily average water table position for (a) WS80-SC for events producing >1 mm in 2003 (after Harder et al., 2007) and (b) D1-NC for daily events in 2005.

flow events were generally associated with initial WT elevations already at or near the ground surface (fig. 7a), while many large summer rainfall events produced little or no outflow due to deeper WT positions, particularly at the drained NC site (fig. 7b). The large rain (130 mm) associated with Tropical Storm Gaston in late September 2004 produced total outflow of only 20 mm. Daily rainfall events greater than 30 mm generally resulted WT nearing the surface at the SC site, except when the WT was deeper than 1.5 m following a long dry period (fig. 5b). This is consistent with observations made by Amatya et al. (1998) for flatter watersheds in coastal NC, where outlets were frequently submerged for events greater than 25 mm d⁻¹ during wet winter and spring periods. Using storm event data from a three-year (2008-2011) period and an empirical hydrograph separation method, Epps et al. (2013) found that the total storm response as ROC varied from 0.01 to 0.74, with an average of 0.34, for WS80 and from 0.0 to 0.93, with an average of 0.39, for another forest site on a sandy soil near Georgetown, also in coastal SC. This is also consistent with Sun et al. (2002), who found that the ROC varied from near zero for small events with >50 mm of rainfall and dry antecedent conditions to as much as 0.57 for large events with >150 mm of rainfall and wet antecedent conditions, potentially as a result of the saturated overland flow characteristic of low-relief, poorly drained sites such as this one, as reported by Slattery et al. (2006). The WT elevation was shown to influence streamflow and runoff production (Epps et al., 2013), which cannot be predicted by precipitation alone for any given storm event, nor is antecedent precipitation a good indicator of the antecedent moisture condition (AMC), as was also

shown by La Torre Torres et al. (2011) for a nearby third-order forested watershed.

WT Depth, PET, and Outflow for Dry (2001), Wet (2002), and Wettest (2003) Years at D1-NC

The data in figure 8 show the daily cumulative rainfall, drainage outflow, and potential ET (PET) in figure 8a and the daily WT elevation in figure 8b for a three-year (2001-2003) period at D1-NC (Amatya and Skaggs, 2011). The WT dropped as low as 0.5 m (<1.5 m depth) in the summer of 2001, which was the year with the lowest rainfall (852 mm) and outflow (45 mm) in the 21-year recorded period (Amatya and Skaggs, 2011). With a large storage deficit created by the end of 2001, as shown by both the cumulative PET higher than the rainfall (fig. 8a) and the deeper WT in figure 8b, outflow in 2002 did not initiate until mid-March (day 72, fig. 8a). As a result, the above-average rainfall of 1718 mm yielded only 426 mm of total outflow in 2002. On the other hand, with wet antecedent conditions, as shown by the total cumulative rainfall much higher than the PET and the near-surface WT at the end of 2002 (fig. 8b), drainage in 2003 continued to occur throughout the winter and spring. Additional rainfall throughout the year totaling 2310 mm (the wettest year in the record; Amatya and Skaggs, 2011), which was much higher than the PET (fig. 8a), resulted in the outflow exceeding 50% of the precipitation.

The hydrologic patterns observed on WS80-SC in 2004-2005 and in 2007-2008 were similar (fig. 5a for 2005 and fig. 5b for 2008) to 2001-2002 at the D1-NC site. Both 2005 and 2008 had only small or no flow in the early part of the year due to drier conditions observed in the later parts of 2004 and 2007, respectively. This indicates that soil water storage in the later part of a year, as an antecedent condition, may play a significant role in the following year's seasonal outflow, and potentially the annual outflow in general in this shallow soil forested ecosystem. In some years, annual outflow also depends on the precipitation in the preceding months. For example, 2000 was a wet year at D1-NC, with above-normal rainfall (1718 mm) and an average WT depth of 79 cm. However, the following year (2001) had the lowest ROC (5%) of the study period (table 4) due to much drier conditions, with no flow after March for the rest of the year (fig. 8a).

Evapotranspiration

The recent five-year, short-term estimate of mean annual ET considering soil water storage (1070 mm) is close to the ET of 1047 mm reported earlier by Richter (1980) using pan evaporation data for a 15-year (1964-1979) historical period at the Santee site as well as that of the Florida wetland site reported by Sun et al. (2002). Recently, Dai et al. (2013) applied three hydrologic water balance models: Thornthwaite (Alley, 1984; Dingman, 2002), MIKESHE (DHI, 2005), and Forest-DNDC (Li et al., 2000) with 63 years (1946-2008) of long-term data on the second-order WS79 watershed (containing the WS80-SC watershed) and obtained similar ET estimates of 1043, 974, and 1030 mm, respectively. These estimates for the Santee site and for the Carteret site (1015 mm) are comparable (within $\pm 3\%$) to the five-year (2005-2009) mean of 1038 mm obtained for another drained pine forest at Parker Tract, NC, using eddy covariance (EC)

methods (Sun et al., 2010; Domec et al., 2012) and well within the recently reported ten-year (2006-2015) ET of 1076 ± 104 mm (Yang et al., 2017; Liu et al., 2018). The similar mean annual ET values from the two study sites potentially indicate no effects of drainage on mean annual ET (Amatya et al., 2019). Furthermore, for a dry year (2001), the ET/P ratio exceeded unity at D1-NC, which is also consistent with Sun et al. (2010), indicating that the ET from forests can exceed the rainfall when large soil water deficits occur. These results indicate that annual ET values estimated using the difference between rainfall and outflow without considering the change in soil water storage for sites with shallow WT depths should be cautiously interpreted, as they can be seriously misleading due to interannual variability in rainfall, particularly in years with wet or dry antecedent conditions. For example, our long-term mean annual ET estimate of 1114 mm without storage effects (table 4) that includes the analysis reported by Gilliam (1983) for the earlier 1969-1980 period prior to Hurricane Hugo for the WS80 site were higher than (1) the estimate of 1047 mm obtained by Richter (1980) using pan evaporation data for the same period; (2) the 1000 mm estimate obtained by Lu et al. (2003) using a multivariate regression model consisting of precipitation, area covered by forest, elevation, and latitude as explanatory variables; and (3) the five-year (2004-2008) estimate of 1069 mm obtained using storage effects for the post-Hugo regeneration period.

Using longer-term data from both sites without considering storage (table 4), both sites yielded similar ET values that were close to 94% of rainfall for the dry period and 72% to 75% of rainfall for the wet period, but the overall annual average ET as a percentage of rainfall was significantly ($p < 0.0001$) higher (83%) at WS80-SC than at D1-NC (67%) (table 4), with higher drainage outflow at D1-NC. These observations are in agreement with Sun et al. (2002), who re-

ported ET values of 87% of mean annual rainfall for the Florida pine flatwoods for 1979-1992 and 70% for Carteret for 1988-1997, respectively. The coefficients of determination (R^2) between long-term annual rainfall and ET in table 4 showed that rainfall only explained 16% of the variation ($R^2 = 0.16$) in ET at D1-NC, in contrast with WS80-SC, where rainfall explained 91% of the variation ($R^2 = 0.91$). Similar analysis using annual rainfall and ET, as the difference between rainfall and streamflow, for the adjacent large (5240 ha) Turkey Creek watershed (Amatya et al., 2015) also explained 60% of the variation ($R^2 = 0.60$) in ET, indicating that ET may often be limited by soil moisture in this region, while the poorer relationship at D1-NC suggests energy limitation.

MODELING CONSIDERATIONS

Modified forestry versions of DRAINMOD have been extensively validated for predictions of daily WT and drainage outflow and used for various applications for the drained Carteret site (McCarthy et al., 1992; McCarthy and Skaggs, 1992; Amatya and Skaggs, 2001). With the availability of 21 years (1988-2008) of data, as described above (tables 3 and 4), a process-based DRAINMOD-FOREST model, a recently modified version (Tian et al., 2012), successfully predicted the daily WT depths (NSE = 90; MAE = 0.10 m) and monthly drainage outflows (NSE = 0.87; MAE = 10.1 mm month⁻¹) for the 25 ha drained pine forest watershed (D1-NC) in this study (Tian et al., 2012). However, the authors reported poor model performance, with overpredicted outflow for the driest year (2001, with 852 mm of rainfall) with the deepest WT of 0.5 m a.m.s.l. and the following wet year (2002, with 1718 mm of rainfall) (fig. 8), possibly due to errors in antecedent soil water conditions and ET, consistent with those reported by other studies. The mean annual predicted ET was 65% of the rainfall, slightly lower than the measured ET/P ratio of 0.67 (table 4). In another study, DRAINMOD-FOREST predictions of monthly ET also agreed well with measured ET based on eddy covariance (NSE = 0.78) after validating with eight years of

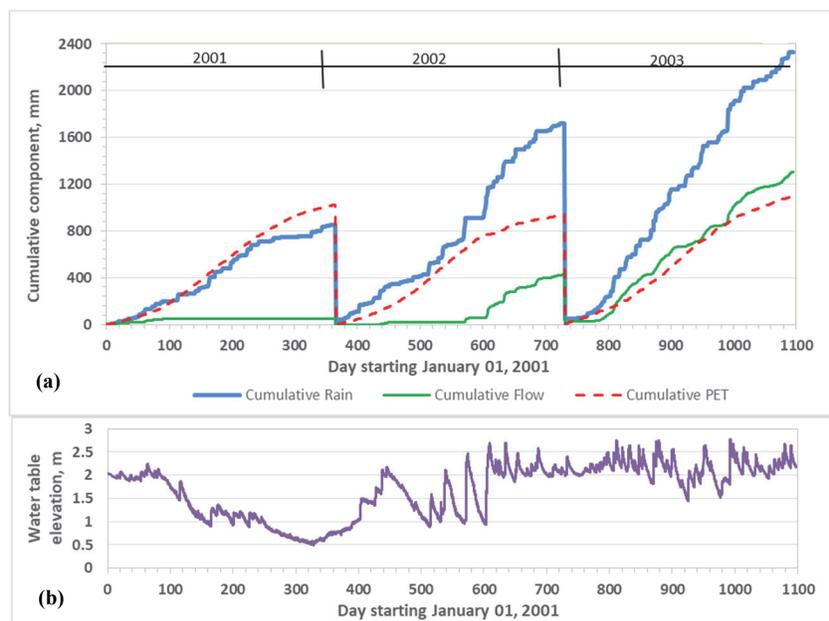


Figure 8. (a) Daily cumulative rainfall, flow, and potential ET (PET) and (b) hourly water table elevation at D1-NC for 2001-2003.

WT and monthly flow data for a 90 ha drained watershed at another pine forest in eastern NC (Tian et al., 2015). However, those DRAINMOD-based models developed for drained systems have not yet been fully tested for simulating watershed outflow for naturally drained (without ditches) forests, although some work has been done to simulate field or plot-based WT depths only (Caldwell et al., 2007; He et al., 2002; Vepraskas et al., 2004).

A first attempt to test the capability of DRAINMOD to model streamflow from the undrained WS80 watershed was made by Amatya et al. (2003c) using short-term (1997-1999) data with minimal field calibration. Although the model's daily flow predictions responded well to all rain events, including a no-flow period, the results were not satisfactory, with as much as 32% underprediction as well as discrepancies in peak flow rate and time to event peak. The authors attributed the discrepancies to this field-scale model's assumption of instantaneous arrival of excess flow (surface and subsurface) at the outlet, without considering the effects of flow routing, which seems to be important even in this first-order watershed with a clear concentrated flow path of the 1.2 km stream draining a 206 ha area at the outlet (until 2001 and 160 ha thereafter) (Golden et al., 2014). Furthermore, surface water moves slowly through longer paths on these low-gradient natural sites, allowing a longer recession of flows (Amatya et al., 2003a). By using field calibration of some drainage, surface storage capacity, and soil hydraulic parameters, including PET estimates, better predictions of monthly outflows were obtained by Harder et al. (2006), with NSE of 0.92 and MAE of 11 mm month⁻¹, comparable to the results obtained with DRAINMOD-FOREST for the Carteret site (Tian et al., 2012). In all of the above studies,

PET estimated by the process-based Penman-Monteith (P-M) method, requiring complete weather variables, was used as a model input variable. Flow prediction by DRAINMOD has been shown to be very sensitive to PET inputs, in addition to other parameters (Kim et al., 2012), particularly during the growing season with high ET demands.

Dai et al. (2010b) reported successful calibration and validation of a complex, process-based, distributed hydrologic model (MIKESHE) for the same undrained WS80 watershed, with NSE of 0.93 for monthly flow and NSE varying from 0.56 to 0.90 for daily WT depth for the 2003-2008 study period, similar to our 2004-2008 period, indicating that the complex process-based model performed only slightly better in predicting flow, although the study covered varying time periods. The predicted mean annual ROC for 2004-2007 was 16%, which was slightly higher than the 13.0% for the measured data (table 3), as with DRAINMOD for the D1-NC site. Later, Dai et al. (2012) compared the daily WT and monthly flow predicted by the distributed watershed-scale MIKESHE model and the field-scale DRAINMOD model for the 2003-2007 period for the same undrained WS80 site. Although both models performed reasonably well in predicting monthly and annual average WT depths and streamflow, with acceptable NSE values (0.55 to 0.99) for the five-year period (2003-2007), MIKESHE predicted better results than DRAINMOD for daily hydrologic dynamics. This was attributed to the use of finer spatial characteristics and surface flow routing in the MIKESHE, in contrast with the watershed-average parameters without any flow routing used in DRAINMOD. Although both models used a bi-criterion approach to validation using both WT and flow, both models showed relatively large uncertainties in simulating streamflow

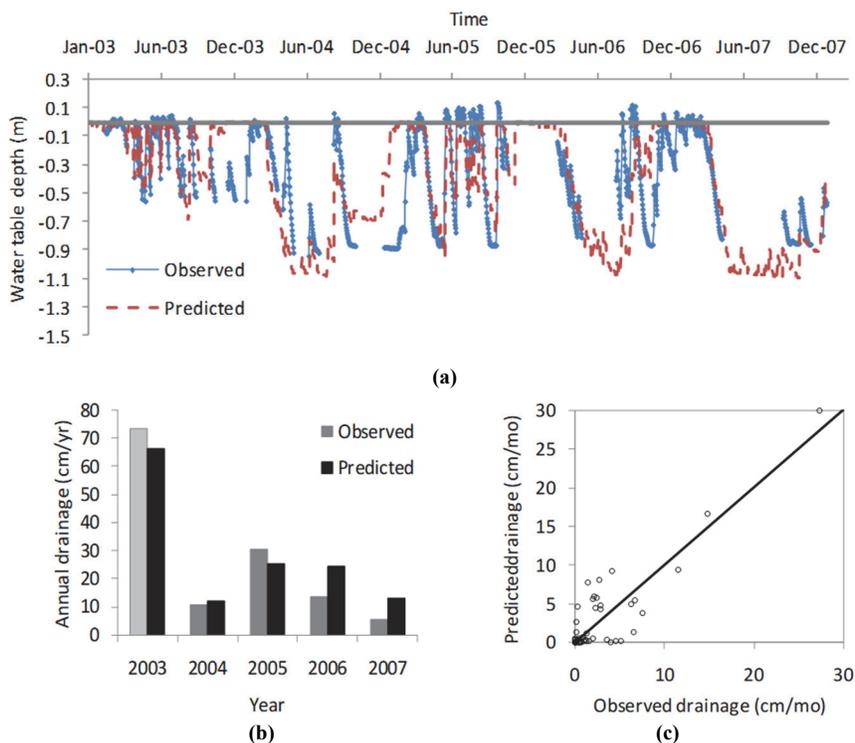


Figure 9. Observed and DRAINMOD-FOREST predicted (a) daily water table depths, (b) annual drainage outflow, and (c) monthly drainage outflow for the undrained WS80 watershed in South Carolina.

for dry years and somewhat overpredicted flow. While the overprediction of flow during dry periods by MIKESHE is likely an artifact of the model, which does not allow a river or stream to dry out (Lu et al., 2006), neglecting the canopy interception for rainfall soon after dry periods may be a cause in DRAINMOD, but not in DRAINMOD-FOREST, as discussed below. Lower subsurface drainage was predicted by MIKESHE than by DRAINMOD for dry years, higher for extremely wet years, and similar for normal climate years; the differences were likely due to the somewhat lower surface detention storage used by the MIKESHE. The simulated proportion of subsurface drainage varied from 33% (MIKESHE) to 38% (DRAINMOD) of mean annual streamflow and was somewhat lower, as expected, than the mean event-based baseflow (subsurface drainage) of 45% reported by Epps et al. (2013) for 20 storm events measured from 2008 to 2011 on this watershed.

A preliminary effort in testing the process-based DRAINMOD-FOREST model without any calibration on the same undrained WS80 site for the same 2003-2007 period (unpublished data) was, however, poor and unsatisfactory based on comparison of the observed and predicted daily WT depths and annual and monthly drainage outflows, as shown in figure 9. The model overpredicted WT depths during the relatively dry years of 2004 and 2006 (fig. 9a) and overpredicted annual drainage outflows in 2004, substantially in 2006 and 2007 (fig. 9b), compared to the measured data (table 3). Accordingly, the calculated average NSE and MAE were 0.27 and 0.19 m, respectively, for daily WT predictions and 0.76 and 13 mm, respectively, for monthly flow predictions.

As a result, the predicted mean annual runoff coefficient of 0.15 for 2004-2007 was slightly higher than the ROC value of 0.11 for the measured data for the same period (excluding 2008) of this study. However, the proportion of simulated subsurface drainage of nearly 50% of total streamflow, on average, was slightly higher than the value of 45% reported by Epps et al. (2013) for this watershed. This might have been due to the somewhat higher drainage density, with 500 m ditch spacing and 0.65 m depth assumed for this undrained forest in the initial run without any calibration. Interestingly, the predicted mean annual actual ET of 843 mm without canopy evaporation was only <6% lower than the ET of 894 mm predicted by MIKESHE, and the simulated average annual canopy interception of 12.9% of rainfall was slightly higher than the 11% interception measured by Harder et al. (2007) for this watershed. However, the total ET of 1007 mm was less than the ET of 1035 mm in table 3 for the same 2004-2007 period and the long-term simulated ET of 1043 mm reported for this site by Amatya et al. (2016c). Similarly, the simulated mean ET/P (0.69 ± 0.12) reported by the authors was very close to the measured mean (0.67 ± 0.13) for the D1-NC site (table 4) for the 1988-2008 period. However, the simulated mean ET/P (0.78 ± 0.12) for the longer period (1946-2008) was about 6% lower than the measured mean (0.83 ± 0.06) obtained for the more recent 17-year period (table 4). These results indicate that more accurate predictions of both flow and ET, as well as the WT depth, on these forests may be possible with better calibration of both models.

Amoah (2008) found satisfactory predictions of daily

streamflow for 2003-2004 calibration (NSE = 0.97) and 2005-2006 validation (NSE = 0.81) periods for the adjacent WS77 forest watershed (fig. 2) when using DRAINWAT, the DRAINMOD model with a flow routing component. However, the WT predictions were in poor agreement. Amoah et al. (2012) found that DRAINMOD achieved better prediction of flow when using a distributed depressional storage capacity parameter based on the topography, as opposed to a lumped parameter, indicating the potential influence of surface topography on seasonal antecedent moisture conditions and therefore peak flow rates. Unfortunately, a watershed-scale version of the more recent process-based DRAINMOD-FOREST model is not yet available.

Based on these studies, we argue that DRAINMOD-based models, calibrated with field input parameters including PET, should be adequate to describe the monthly and annual streamflow of uniform canopy forests at the scale of this watershed (WS80) but perhaps not the daily flow dynamics or spatial soil moisture characteristics, including WT depth. In such a case, a spatially distributed model with a flow routing component, such as MIKESHE/MIKE11 or DRAINWAT, a watershed-scale version of DRAINMOD (Amatya et al., 1997b), or another similar model (Golden et al., 2014; Amatya et al., 2013), should perhaps be applied. Using DRAINMOD calibrated with the minimum drainage intensity for the undrained WS80-SC site, we can possibly test the hypothesis that drainage intensity and soils perhaps have less effect than climate on the lower observed outflow (or ROC) of WS80-SC, as compared to the D1-NC site (tables 3 and 4).

SUMMARY AND CONCLUSIONS

This article compared and contrasted both short-term and long-term results from studies conducted on two first-order Atlantic Coastal Plain forested watersheds. The first site (WS80-SC), a naturally drained watershed in the USDA Forest Service Santee Experimental Forest in South Carolina, was compared to the second site (D1-NC), a managed pine forest watershed on artificially drained land at Weyerhaeuser Company's Carteret site in North Carolina. Artificial drainage resulted in lowering the WT, with a deeper average WT on the normally more poorly drained soil at the site that received the greatest average rainfall. Long-term data showed that runoff was very dynamic, with variations in runoff and the runoff coefficient (ROC) that were 2.5 times greater than the variations in either rainfall or ET. Daily and storm event runoff were closely but non-linearly related to WT position, with extreme runoff (>70% of rainfall) occurring at both sites when the WT approached or exceeded the surface, consistent with a recent finding by Guerin et al. (2019), who investigated groundwater's contribution to floods using isotopic measurements on a small tropical forest catchment. Despite a clear difference in average WT position, the influence of forestry drainage on runoff was obscured by large interannual and spatial differences in runoff. Variation of the paths and intensity of hurricanes and tropical storms, in relation to the location of the two sites, produced large differences in storm rainfall, generally occurring during the late summer and fall, and causing large and unpredictable differences in runoff.

Although the drained D1-NC site had a deeper WT and produced greater runoff, it also had more rainfall and a slightly lower ET limited by energy. By contrast, the WS80-SC site, with lower rainfall but higher ET, was more often limited by soil moisture. Wide variability in the observed runoff response indicated that results obtained to assess the effects of forest disturbances, management, and restoration activities should be interpreted cautiously. Unfortunately, there were only limited estimates of ET and its components for these coastal forests using reliable measurements and/or water balance on both the spatial and temporal scales. This is particularly unfortunate for wetland forests, where surface evaporation and transpiration are highly related to WT elevation and surface inundation.

Validated hydrologic models are often used to understand the interaction mechanisms of various processes and their associated impacts on hydrology. In this study, a deeper WT was generally observed with lower ET, due to either drainage or water limitations during dry periods. A brief review of the modeling approach for these two forest types showed similar results using DRAINMOD for the drained D1-NC site, generally with deeper WT yielding lower long-term ET, compared to the MIKESHE predictions of higher ET for the undrained WS80-SC site, with WT frequently near the surface. The modeling results also provided insights into partitioning of stream runoff into surface flow and drainage for these two forest types. Future work should also consider exploring the partitioning of overstory and understory ET (a capability of the current version of DRAINMOD-FOREST), which is critical but poorly understood, as well as the potential effects of deep groundwater, which have so far been assumed to be negligible in these poorly drained coastal forests. Both of these factors may affect the hydrologic pathways and transport mechanisms that are critical for accurate assessment of water resources.

Another highlight of this long-term assessment is the hydrologic resiliency of coastal forest watersheds. The extreme hurricane event (Hugo) in 1989 on WS80 produced increased runoff after destruction of the forest overstory, but runoff was similar to pre-hurricane data after the forest regeneration by 2008 (Jayakaran et al., 2014). The immediate short-term impact of Hugo on streamflow increase was comparable to the harvest of a drained managed pine forest (adjacent to the D1-NC site), the hydrology of which was also reported to return to baseline level by 2005 (Amatya et al., 2006a). However, some effects on flow of the harvested watershed may also have been due to soil disturbance during logging, which was not the case with WS80-SC. These results from freshwater systems reveal the importance of collecting long-term data in addition to the short-term hypothesis testing of runoff and ET that is often needed to corroborate hydrologic model predictions.

In relation to the generalized climatic model of runoff and evapotranspiration obtained in the Budyko curve (fig. 4), both watersheds produced less runoff than predicted by the curve. Because both watersheds are characterized by relatively young and vigorously growing loblolly pine forests, the rates of carbon assimilation and transpiration could be relatively high (Sun et al., 2007). Zhang et al. (2004) observed that forested catchments tend to show higher ET than grassed catchments, and their evaporative index (ET/P) is most sensitive to changes

in catchment characteristics for regions with an index of dryness (PET/P) of about 1.0, which applies to WS80-SC rather than D1-NC in our study. Accordingly, the relative positions of these two watersheds in relation to the water and energy limitations in the original Budyko curve suggest that the greater runoff at D1-NC was more likely related to climatic differences than to drainage and/or soil. However, the large WT fluctuations at WS80-SC (fig. 5), as discussed above, combined with the higher clay content of the subsoil, may have led to greater unsaturated moisture storage and greater ET losses than at D1-NC during dry periods. These results highlight the difficulty in accurately partitioning evaporative losses as well as surface runoff and baseflow for poorly drained wetland forests with highly dynamic WT depths.

BROADER APPLICATIONS OF STUDY AND FUTURE STUDIES

The results and findings from studies on long-term reference forest watersheds may also serve as a baseline for assessing the impacts of natural (e.g., extreme events) and anthropogenic (e.g., land use change) disturbances, including the effects of urbanization on hydrology and local climate (Hao et al., 2015, 2018), the effects of vegetation structure and species composition (e.g., bioenergy crops) on rapidly developing humid coastal regions, and for cross-site comparisons, as conducted by Amatya et al. (2016a) for flow regimes from multiple reference forests. Recently, Tian et al. (2019) used long-term historical high-resolution rainfall data from the WS80-SC watershed with two other similar forest sites in North Carolina and Arkansas to assess the effects of extreme precipitation events on peak discharges, as frequently used in the design of forest road culverts, gauging stations, and stream crossings. Similarly, the long-term data from WS80-SC as a reference in the paired system is being used to assess the short-term and long-term watershed-scale effects of restoring longleaf pine (*Pinus palustris*) as a replacement for loblolly pine (*Pinus taeda*) on the water yield of the treatment watershed (WS77) using a paired watershed approach backed up by hydrologic modeling (Trettin et al., 2019). On the other hand, historical long-term data from the drained D1-NC site were used as the basis for assessing the impacts of land cover change using switchgrass, as a cellulosic biofuel, intercropped in a managed pine forest on water quantity and quality (Muwamba et al., 2015, 2017; Amatya et al., 2016d; Ssegane et al., 2017) and verifying estimates of ET using high-resolution satellite images based on remote sensing approaches (Panda et al., 2019). Storm event outflow from the D1-NC site was also used in calibrating a modified version of the SCS curve number (CN) method for a drained forest watershed (Walega and Amatya, 2019). Furthermore, studies at the drained D1-NC site were used to compare drainage across Europe (Hirt et al., 2011) as well as water management practice on drained peatland forests in Finland (Koivusalo et al., 2008; Haahiti et al., 2016, 2018), indicating that our work on drained forests may also be applicable for broad synthesis studies (Holden et al., 2004, 2006; Skaggs et al., 2016). Long-term rainfall and WT data from both of the study sites, along with other sites with similar soils on the Atlantic Coastal Plain, were recently used in analyzing drivers and their effects on wetland hydrology (Amatya et al., 2019; Skaggs et al., 2019). Long-term data from both sites

were also a basis for developing and applying the various eco-hydrological models discussed above, including a recent model for intercropped forest (Tian et al., 2016).

It seems clear that there is great potential for novel research and management applications using the long-term data from these and similar forest sites (Amatya et al., 2016a). However, additional studies, using high-frequency isotopic and eddy flux measurements backed up by high-resolution satellite images, for more detailed understanding of the hydrological (particularly ET and streamflow generation and partitioning) and ecological process interactions on these types of reference watersheds are needed for accurately quantifying the impacts of management actions at the catchment scale as well as for restoration practices. This is critical because of the large variability in runoff that we observed in these two complex systems, which may be exacerbated by the increasing frequency of climatic extremes. Without understanding the variability in runoff mechanisms, there will likely be an over-attribution to treatment on these watersheds. Such detailed studies, accompanied by innovative technology, will allow better model parameterization and calibration for reliable model applications on poorly drained forests. In addition, short-term studies, such as the study by Skaggs et al. (2011) comparing the three-year hydrology of a drained forest and an adjacent natural forested wetland, that avoid the effects of climate, soils, and topography should be conducted on a longer-term basis to potentially provide more accurate assessment of drainage impacts on hydrology and WT dynamics (as a surrogate for wetland hydrology and other groundwater-dependent ecosystem processes). Forest managers need to be able to extrapolate the principles learned in these experimental watersheds to their own forests. Furthermore, additional research is suggested to examine forest systems drained for timber production as an ecosystem modifier, as argued by Lohmus et al. (2015), who revealed a complex of feedback-regulated, largely indirect, and wide-ranging impacts of the changed hydrology on the biodiversity in such systems.

ACKNOWLEDGEMENTS

We would like to thank Dr. Zhaohua Dai at Michigan Tech for some earlier data analysis; Andy Harrison, Hydrologic Technician at the USDA Forest Service Center for Forested Wetlands Research, for some data processing; and Cliff Tyson, Sandra McCandless, Joe Hughes, and Joe Bergman, the latter three formerly with Weyerhaeuser Company, for field data support. We also extend our thanks to Dr. Shiyong Tian at North Carolina State University and Dr. Herbert Ssegane at Climate Corporation for their useful feedback on the manuscript. The findings and conclusions in this article are those of the authors and should not be construed to represent any official USDA or U.S. Government determination or policy.

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