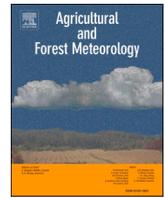


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Effects of land-use change and drought on decadal evapotranspiration and water balance of natural and managed forested wetlands along the southeastern US lower coastal plain

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ABSTRACT

Forested wetlands are important in regulating regional hydrology and climate. However, long-term studies on the hydrologic impacts of converting natural forested wetlands to pine plantations are rare for the southern US. From 2005–2018, we quantified water cycling in two post-harvest and newly-planted loblolly pine (*Pinus taeda*) plantations (YP₂₋₇, 2–7 yrs old; YP₂₋₈, 2–8 yrs old), a rotation-age loblolly pine plantation (MP, 15–28 yrs old), and a natural bottomland hardwood forest (BHF, > 100 yrs old) along the lower coastal plain of North Carolina. We quantified the differences in inter-annual and seasonal water balance and trends of evapotranspiration (ET) using eddy covariance over 37 site-years and assessed key climatic and biological drivers of ET. We found that the rotation-age plantation (MP) had higher annual ET (933 ± 63 mm) than the younger plantations (776 ± 74 mm for YP₂₋₇ and 638 ± 190 mm for YP₂₋₈), and the BHF (743 ± 172 mm), owing to differences in stand age, canopy cover, and micrometeorology. Chronosequence analysis of the pine sites showed that ET increased with stand age up to 10 years, then gradually stabilized for the remainder of the rotation of 28–30 years. YP₂₋₈ was sensitive to water availability, decreasing ET by 30–43% during the extreme 2007–2008 drought, but reductions in ET at MP were only 8–11%. Comparing to BHF, ditching with management enhanced drainage at YP₂₋₇ and YP₂₋₈, while drainage was lower at the mature pine site. This study provides insight into land use-hydrology-climate interactions that have important implications for forested wetland management in a time of rapidly changing environmental conditions of the LCP of the southern US.

1. Introduction

Wetland forests play a critical role in regulating regional ecohydrology (McNulty et al., 1997; Sun et al., 2001), water quality (Moreno-Mateos et al., 2012; White and Kaplan, 2017), and productivity of economically important crops, timber, and fishery resources (Bullock and Acreman, 2003) in the southeastern U.S. Compared to upland forests, our knowledge of wetland ecosystem hydrology is still inadequate to fully understand the regional roles of wetland ecological functions and services (Amatya and Skaggs, 2001; Moreno-Mateos et al., 2012;

Noormets et al., 2010; Sun et al., 2002; Tor-ngern et al., 2018). In particular, large areas of natural forested wetlands of the southeastern US lower coastal plain (LCP) were historically cleared and drained for timber production (Campbell and Hughes, 1991). Timber companies in the region lowered the water table and soil water content through construction of ditches and canals to facilitate industrial tree farming (Amatya et al., 1996; Kelting et al., 2000; McCarthy et al., 1991). These companies generally used bulldozers, bush hogs, back hoes and hydraulic dredgers to construct the ditches (Burke, 1996). However, we know less about the impact of forest management practices on the

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long-term impacts of this land use practice on energy, water, and carbon cycling at the ecosystem, watershed, and regional scales (e.g., a forest rotation period) (Sun et al., 2010; Tian et al., 2015).

Climate change projections suggest an increasing frequency of droughts in the southern US, and at the same time an intensification of the hydrologic cycle (IPCC, 2013; Trenberth, et al., 2007), bringing the risk of flooding. Therefore, a quantitative understanding of the effects of climatic extremes on coastal wetland hydrologic function is a pressing research need to inform LCP forest management.

Land use change, forest management practices, and climate fluctuations have the potential to affect the regional hydrologic cycle and wetland ecosystem function directly by altering evapotranspiration (ET) (Day et al., 2008; Ellison et al., 2017; Sun et al., 2010; White and Kaplan, 2017). ET is recognized as a key hydrologic control (Sposito, 2017), returning 50–90 % of the incident precipitation to the atmosphere in temperate forest ecosystems (Ford et al., 2007; Gholz and Clark, 2002; Sun et al., 2002). ET links hydrology to biology and is strongly associated with the primary productivity and biodiversity of terrestrial ecosystems (Jackson et al., 2005; Noormets et al., 2006; Wilson and Baldocchi, 2000). ET is a particularly important hydrologic flux under stresses caused by accelerating land-use and climate change (Jones et al., 2012; Oishi et al., 2018; Woodward et al., 2014). Hydrologic and water quality models developed for this coastal region are seldom validated with measured actual ET (Sun et al., 2010; Tian et al., 2015). Large uncertainties remain therefore in watershed scale ET estimates because it has been often derived as the residual of the water balance (Sun et al., 2002), and quantifying ET of vegetated surfaces at a fine spatial and temporal scales (e.g. watershed, daily, monthly) remains challenging. Improved estimation of ET, especially under extreme climate conditions such as drought (Vose et al., 2016), is needed to project potential effects of management and climate change on LCP forest water cycling. To advance this objective, researchers use eddy covariance methodology to directly measure ET, energy, and carbon fluxes at the ecosystem scale in recent decades (Wilson and Baldocchi, 2000; Baldocchi et al. 2001; Gholz and Clark, 2002; Sun et al., 2010).

To better understand the hydrology and C cycling of forested wetlands of the southeastern U.S., we maintain a cluster of eddy covariance flux towers as part of the AmeriFlux/FLUXNET network at four sites along the LCP of North Carolina (Aguilos et al., 2020; Domec et al., 2010; Miao et al., 2017; Minick et al., 2019; Mitra et al., 2019; Noormets et al., 2012; Sun et al., 2010; Liu et al., 2019). The four sites are comprised of two young pine plantations (2-7 and 2-8 years old), a rotation-age mature pine plantation (15–28 years old), and a natural, undisturbed forested wetland (>100 years old).

In the current study, we were interested in how the conversion of wetland forest to managed pine plantations, construction of ditches and extreme weather events (e.g. drought) impact the water balance of these lower coastal plain forests. Specifically, we aimed to (1) examine the seasonal and interannual variations in water balance; (2) determine the ratio of water leaving the system as drainage (D) and ET relative to the amount of precipitation inputs (P); and (3) evaluate the effect of drainage and land-use change, age, gross primary productivity (GPP) and climate variation (including a severe drought in 2007-2008) on ET and water balance. We hypothesized that (1) ET would be higher in rotation-age pine plantation than in young pine, owing mostly to differences in canopy cover, and also higher than in the natural BHF that had lower NPP and GPP (Aguilos et al., 2020); (2) ET increases with age commensurate with increases in GPP; (3) Ditched landscapes for intensive pine plantation management would decrease the fraction of P leaving the site as ET compared to a natural wetland forest; and (4) that a severe meteorological drought in 2007-2008 would affect young plantations more than rotation-aged plantations due to higher level of stress, shallower root systems and lower GPP, although ET and GPP would remain tightly coupled even during extreme conditions.

2. Materials and methods

2.1. Study sites

Under the FLUXNET database, our sites are coded as US-NC3, US-NC1, US-NC2, and US-NC4, herein referred to, respectively, as young loblolly pine YP₂₋₇ (2–7 years old) and YP₂₋₈ (2–8 years old), rotation-age mature loblolly pine MP (15–28 years old), and a natural (undrained) bottomland hardwood forest BHF (>100 years old; Fig. 1). The last numbers in the site denominations correspond to the site's actual age in 2018 except at YP₂₋₈ where it was 8 years old when measurements ended in 2011.

The three loblolly pine plantations (YP₂₋₇, YP₂₋₈, and MP) are managed by Weyerhaeuser NR Company located in Washington County, North Carolina, USA (Noormets et al., 2010; Sun et al., 2010). The MP stand is now in its 5th rotation, which was established in 1992. The MP carbon, energy, and water flux monitoring tower was established when the plantation was 15 years old in 2005. A thinning operation at the MP site occurred in 2009. Also in 2005, another nearby tower was built (YP₂₋₈) after a harvesting operation and replanting of 1-year old pine seedlings. Measurement at YP₂₋₈ ended in 2012, and it was replaced with YP₂₋₇ flux tower when another loblolly pine stand was harvested in 2012. The pine stands at the managed plantation sites are drained with a network of parallel ditches, 90-130 cm deep on 90-m spacing, and more widely spaced roadside canals. Watershed yield (outflow) is monitored on the downstream end of the drainage network using a V-notch weir. Parallel ditches and roadside canals divide the flat landscape into a mosaic of regularly shaped fields and blocks of fields (Diggs, 2004).

The BHF flux tower was constructed in 2009 in a 100+ year-old natural coastal bottomland hardwood forest at Alligator River National Wildlife Refuge in Dare County, NC, USA, to establish a frame of reference for the managed plantation sites. It has had minimal human disturbance since stand establishment. BHF is composed of mixed hardwood swamp forests where 'hummocks' around tree bases are usually above the water table, and non-vegetated low-lying 'hollows' are submerged for more than 70 % of the year (Miao et al., 2013; Minick et al., 2019). This wetland forest site is approximately 108 km away from the managed plantations sites (Fig. 1).

Pine plantation sites have elevations <5 m, while BHF was <1 m above sea level. At the plantation sites, the mean annual precipitation was 1366 ± 57 mm (1971–2018), and the mean annual temperature was 16.4 ± 0.4°C (2005–2018). The BHF site had mean average precipitation of 1239 ± 46 mm (1981–2018) and a mean annual temperature of 16.4 ± 0.6°C (2009–2018). A complete description of the sites is provided in Table 1 (Aguilos et al., 2020; Domec et al., 2015; Miao et al., 2017; Noormets et al., 2012, 2010; Sun et al., 2011; Liu et al., 2018).

2.2. Evapotranspiration measured by the eddy covariance system

Following AmeriFlux protocols, latent heat (LE) and sensible (H) flux measurements were made 9.0 m, 11.6 m, 28.7 m, and 33.3 m above the forest floor at YP₂₋₇, YP₂₋₈, MP, and BHF, respectively. These variables were quantified along with the measurements of CO₂ exchange using eddy covariance systems at all sites. An open-path analyzer (consisting of an LI-7500 infrared gas analyzer (LiCor, Lincoln, NE, USA), CSAT-3 sonic anemometer (Campbell Scientific (CSI), Logan, UT, USA), or in the case of BHF, a Gill Windmaster (R-350; Gill Instruments, Lymington, UK) was used to measure the ecosystem exchange of CO₂ and CR5000 dataloggers for the plantation sites and CR-1000 at the BHF site. The air temperature was derived from sound speed measured by a sonic anemometer and was also obtained using a thermocouple that was used for estimating H. A relative humidity and air temperature sensor (model HMP-45C; Vaisala, Helsinki, Finland) was also installed at the same height as the IRGA to quantify water vapor density.

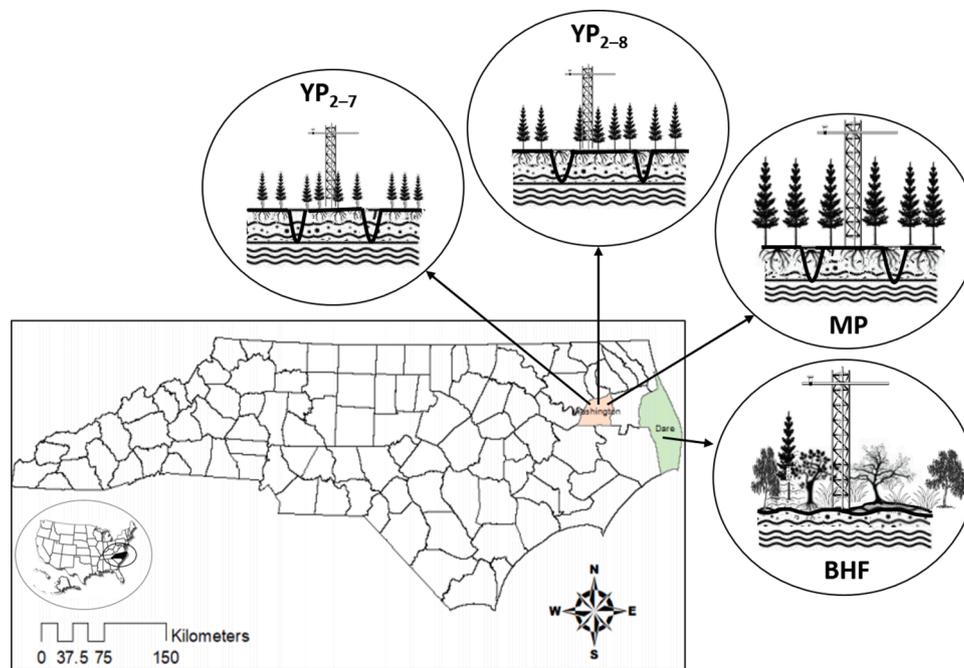


Fig. 1. Location of carbon and energy fluxes monitoring sites in the lower coastal plain of North Carolina.

Table 1

Study site description of eddy covariance flux towers located along the lower coastal plain of North Carolina to monitor long-term evapotranspiration and forest water cycling.

Location*/Parameters	YP ₂₋₇	YP ₂₋₈	MP	BHF
Latitude	35.7990	35.8118	35.8030	35.7879
Longitude	-76.6560	-76.7119	-76.6685	-75.9038
Eddy covariance flux tower sensor height (m)	6.5 (2013) 9.0 (2016)	6.0 (2005) 11.6 (2008)	22.5 (2005) 28.7 (2017)	30.0 (2008) 33.3 (2012)
Management activities	Ditched, conversion to commercial forest with subsequent production cycle (e.g. thinning, fertilization, etc.)	Ditched, conversion to commercial forest with subsequent production cycle (e.g. thinning, fertilization, etc.)	Ditched, subsequent production cycles	None
Changes in hydrology due to ditch construction	Decreasing water table	Decreasing water table	Water table dependent on precipitation	Slight fluctuations in water table depth
Successional stage	Plantation stage	Plantation stage	Plantation stage	Natural stage
Structural attributes				
Stand type	Pine plantation	Pine plantation	Pine plantation	Natural hardwood
Age (years, as of 2018)	7*	8*	28*	>100
Measurement period	2013 – 2018	2005 – 2011	2005 – 2018	2009 – 2018
Year of most recent harvest	2012	2004	1992	Undisturbed
Major species	<i>Pinus taeda</i>	<i>Pinus taeda</i>	<i>Pinus taeda</i>	<i>Nyssa sylvatica</i> , <i>Nyssa biflora</i> , <i>Taxodium distichum</i> , <i>Acer rubrum</i> , <i>Liquidambar styraciflua</i> , pond pine
Soil type	Belhaven series (loamy, mixed, with highly decomposed organic matter layer underlain by loamy marine sediments)	Cape fear series (slightly higher water holding capacity and a surface loam layer)	Belhaven series (loamy, with highly decomposed organic matter layer underlain by loamy marine sediments)	Pungo series (poorly drained with highly decomposed muck and less decomposed peat layers; underlain by highly reduced mineral sediments)

* YP₂₋₇ is young pine 2-7 years of age; YP₂₋₈ is young pine 2-8 years of age; MP is mature pine 15-28 years of age; and BHF is natural bottomland hardwood forest, >100 years of age.

2.3. Data quality control and gap-filling

Following the data processing procedure as in previous studies (e.g., Aguilos et al., 2020; Noormets et al., 2012, 2010; Sun et al., 2011), the 30-minute mean fluxes of H₂O were calculated as the covariance of vertical wind speed and the concentration of H₂O, representing the total water loss, including soil evaporation and canopy interception. Using

the Eddypro software (v. 6.1.0) (LICOR Inc, Lincoln, NE, USA), standard quality checks and corrections were applied for spike detection (Vickers and Mahrt, 1997), planar fit coordinate rotation of wind vectors (Wilczak et al., 2001), correction of the time lags between scalar concentrations and wind speed (covariance maximization), air density fluctuation (Webb et al., 1980), and high (Ibrom et al., 2007) and low pass filtering (Moncrieff et al., 2004). We flagged as 0 (high quality), 1

(medium quality), and 2 (low quality) flux outputs (Mauder and Foken, 2006). Fluxes with flags greater than 1 were discarded. Other sources of error and data losses were attributed to low friction velocity, power losses, and instrument malfunction. After all data quality checks and gap-filling procedures had been performed, the final data coverage of quality-controlled LE was 93 % for YP₂₋₇, 94 % for YP₂₋₈, 94 % for MP, and 97 % for BHF. Remaining gaps were filled using empirical correlations between observed ET and grass reference evapotranspiration calculated by the FAO Penman-Monteith method. Obvious outliers were removed on a 30-minute timescale. For example, data points with $LE > 800 \text{ W m}^{-2}$ or $LE < -200 \text{ W m}^{-2}$ were removed. For missing monthly precipitation data, we obtained the nearest US Weather Service station data or the automated or manual back-up rain gauge installed near the flux towers. The total 30-min ET was converted from latent heat flux, LE (W m^{-2}) by the formula: $ET = LE \times (0.01800/44000) \times 3600 \times 0.5$ (Sun et al., 2010). Daily ET was the sum of 30-min ET.

2.4. Other micrometeorological measurements

Additional micrometeorological parameters measured above the canopy at all sites included photosynthetically active radiation (PAR, LI-190, LiCor Inc.), net radiation (CNR-1 and CNR-4, Kipp & Zonen, Delft, the Netherlands), and precipitation (TE-525, Campbell Scientific Inc., Logan, UT, USA). Mean soil volumetric water content was measured for the top 30 cm using a vertically inserted CS616 time domain reflectometry probe (Campbell Scientific Inc.). The depth of the water table was monitored using ultrasonic water level datalogger (Infinites, Port Orange, FL, USA). To compare across sites and to remove the influence of soil texture on soil water content, drought intensity was quantified in the form of relative extractable water (REW) (Vicca et al., 2012; Domec et al., 2015):

$$REW = \frac{SWC - SWC_{wp}}{SWC_{fc} - SWC_{wp}} \quad (1)$$

where SWC is soil water content (SWC), and SWC_{wp} is the SWC at wilting point. The maximum extractable water over the entire rooting zone is equal to SWC at field capacity (SWC_{fc}) less SWC_{wp} . Soil texture was identified at the four sites, and SWC_{fc} and SWC_{wp} were obtained from soil water estimates by texture and organic matter content (Saxton and Rawls, 2006). Anomalous drought years were determined using an ecological drought indicator, the soil water stress index (SWSI) (Granier et al., 1999), computed as the difference between the daily REW and 0.4 (a threshold that induces stomatal closure in most trees), divided by 0.4.

2.5. The water balance equation

A simplified monthly and annual water balance equation for the study sites was developed as:

$$P = ET + D + \Delta S$$

where P is precipitation (mm), ET is ecosystem evapotranspiration (mm) that includes soil evaporation, canopy interception or wet canopy evaporation and plant transpiration (i.e., dry canopy transpiration); D is drainage (i.e., surface and shallow groundwater flowing out of the watershed) (mm); and ΔS represents the change in water storage (mm) in both the unsaturated and saturated soil zones.

The measured eddy fluxes are interpreted as representing the total ecosystem evapotranspiration (ET) that includes both plant transpiration and evaporation from soil and plant surfaces.

Change in storage (ΔS) was derived from change in water table depth at a monthly scale:

$$\Delta S = \Delta SWC \times WTD + \Delta WTD \times \theta_d$$

where SWC is the soil water content for the unsaturated zone measured

at a soil depth of 30 cm, WTD is the water table depth at the beginning of the month, ΔWTD is the change in the water table for the month, and θ_d is the drainable soil porosity (Sun et al., 2010). Drainage (D) was then calculated as residuals after deducting ET and change in storage term from precipitation.

$$D = P - ET - \Delta S$$

P and ET were summed 30-minute data and compiled as daily, monthly or annual sums. D and ΔS were derived monthly. It must be noted that systematic drainage system was constructed for the managed plantation sites whereas the natural forest depended mostly on natural drainage processes driven by local (shallow) topographic relief. Thus, the 'ditching effect' at the managed sites is partially reflected in D as derived from the water balance equation.

2.6. Grass reference evapotranspiration (ET_0)

Grass Reference ET (ET_0) represents actual ET for a standard grass with a fixed leaf area index and canopy conductance and without soil water stress (Allen et al., 1998). It was calculated by the FAO (Penman-Monteith) method. ET_0 is controlled by atmospheric factors and offers a convenient variable to compare actual forest ET across sites. In this study, ET_0 data were obtained from the North Carolina State University climate database system at research stations near our study sites. For the managed sites, we used the ET_0 data obtained from Tidewater Research Station at Plymouth, NC. ET_0 data from Pamlico Aquaculture Field Laboratory in Aurora, NC, was used for the BHF site since it was located in the same coastal setting. Considering the completeness of data, we used turfgrass ET_0 from both field stations. Daily ET_0 were summed to derive monthly and annual ET_0 .

2.7. Data analyses

We used generalized additive models (GAMs) to explore the relationships of ET to climatic variables such as net radiation (R_n), air temperature (T_{air}), relative extractable water (REW), albedo, precipitation (P), and gross primary production (GPP; Aguilos et al., 2020). Leaf area index (LAI) data were derived using two sources: 1) Landsat-scale LAI (Supplemental Fig. 1) retrieved from Landsat shortwave surface reflectance data using MODIS LAI products as reference using a regression tree analysis following the method developed in Gao et al. (2012); 2) A regression model developed from Albaugh et al. (2014) using LAI and basal area for pine stands across the US Southeast, ranging from 3 to 14 years of age. We then used the resulting regression equation to compute the LAI from YP₂₋₇, YP₂₋₈, and MP. We extended the equation up to age 28 by adding data from the MP site. The basal area in 2013 at YP₂₋₇ (seedlings) was considered negligible while missing LAI in 2014 and 2018 were filled using the stand age-LAI regression equation obtained from years with available annual LAI computed for this site.

To predict the best drivers among climatic and biological variables, we built GAM models with spline smoothers. To build the statistical models, we used the *mgcv* package, and used the *MuMin* package to obtain the best smoothing dimension (Aguilos et al., 2018; Leonardi et al., 2015; Shao et al., 2015). We used the Tukey HSD Test for inter-site comparison. Smoothed-curve fitting and linear relationships were carried out with a locally weighted linear or polynomial regression in the *ggplot2* package. All analyses were processed in R version 4.0.2 (R Core Team, 2020).

3. Results

3.1. Micrometeorology, soil water, and water table depth

Daily average micrometeorological data across the measurement periods (Fig. 2) revealed that MP received higher net radiation (R_n) with a daily mean of 130 W m^{-2} which was 10–14 % higher than YP₂₋₇ (118

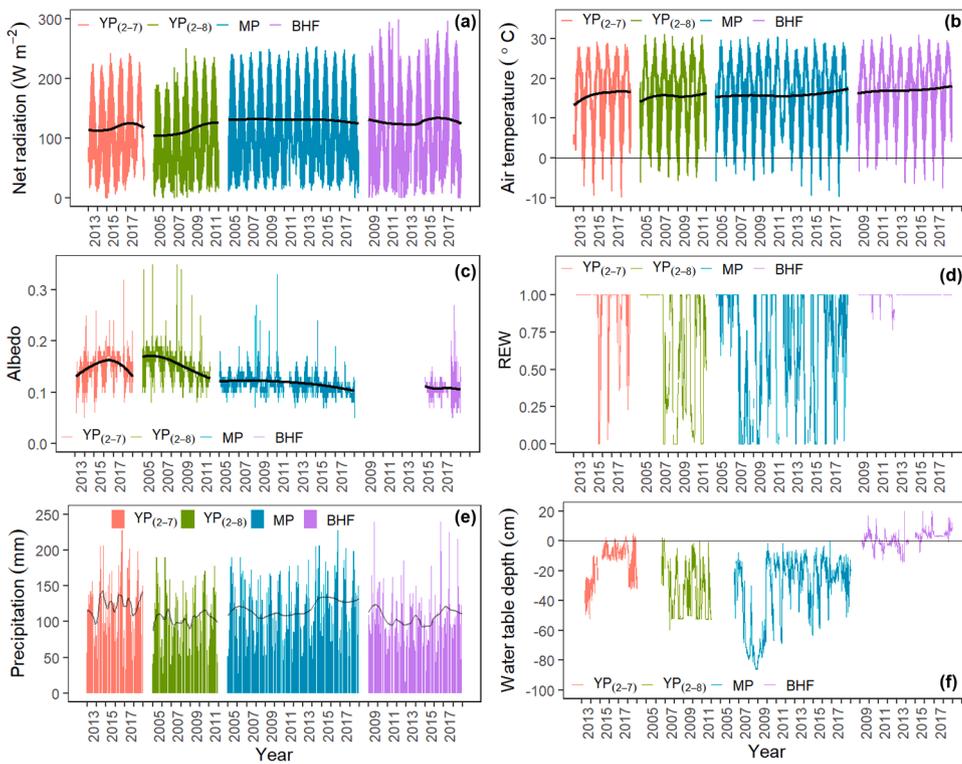


Fig. 2. Daily variation of climate variables: (a) Net radiation, (b) Air temperature, (c) Albedo, (d) Relative extractable water (REW), (e) Precipitation, and (f) Water table depth. Each site was assigned with different vertical line colors. Thick horizontal lines are the smoothed curves. Values are daily averages except for precipitation, where monthly sums were used in a bar plot. For interpretation of the references to color in the figure legend, the reader is referred to the web version of this article.

Wm^{-2}) and YP_{2-8} (114 Wm^{-2}), respectively. This pattern was opposite to that of albedo. MP had an albedo of 0.11, whereas YP_{2-7} and YP_{2-8} both had a higher value (0.15). Interestingly, both MP and BHF had similar albedo (0.11) and R_n (130 Wm^{-2}) values.

Daily variation in REW and WTD indicate that the younger plantations had slightly wetter soils than MP (Fig. 2 d, f). Annual precipitation (P) varied from 1282 – 1488 mm across all sites and years. Annual average air temperature ranged from $\sim 15 - 18^\circ\text{C}$ at all sites and

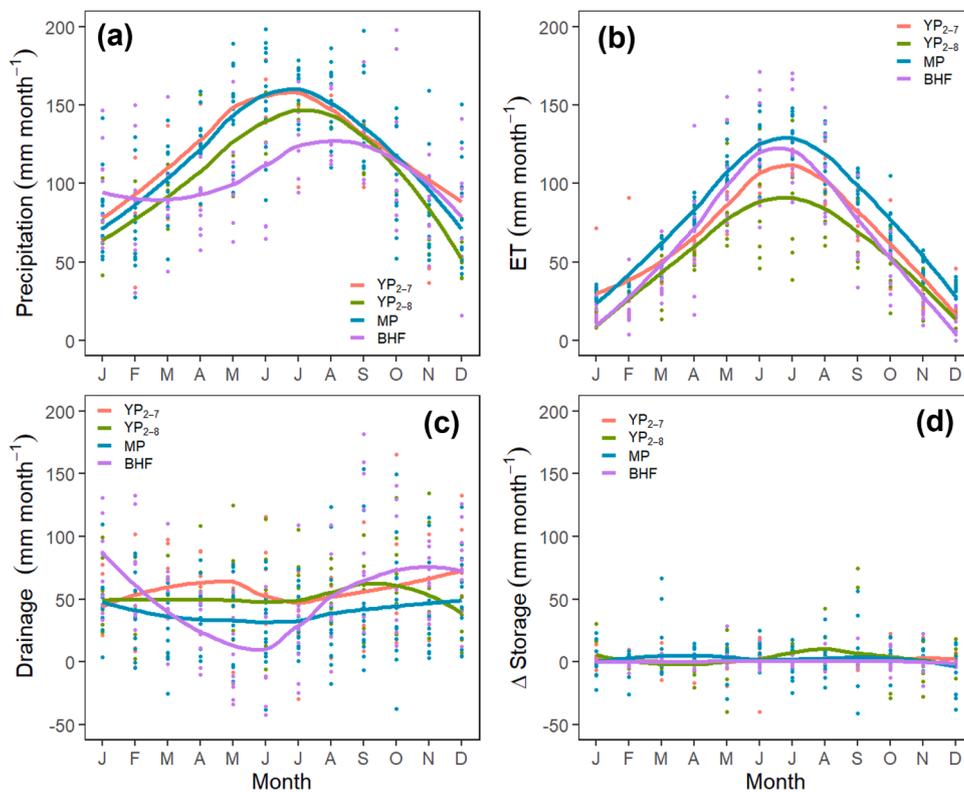


Fig. 3. Seasonal variability in the water balance parameters of precipitation (P), evapotranspiration (ET), drainage (D), and change in soil water storage (ΔS) at YP_{2-7} , YP_{2-8} , MP, and BHF sites across the measurement period. Monthly data each year is represented by different colored circles. A thick solid line denotes the smoothed curves of each site.

followed a similar pattern to Rn, where seasonal highs (growing season) and lows (non-growing season) were prominent throughout the study period (Fig. 2).

3.2. Water balances

3.2.1. Seasonal variation in ecosystem water balance

Precipitation (P) and ET followed the same seasonal trend across all sites, increasing from winter to spring, with a peak in summer and a gradual decline towards fall (Fig. 3 and Supplemental Fig. 2). High ET during dry seasons indicated that soil water was generally not limiting across sites. Among the managed sites, YP₂₋₈ had the lowest P and ET, while BHF had the lowest P of all sites, especially during the growing season (April to August). However, the capacity to release water through ET during the same period exceeded that of the young plantations (Fig. 3).

Drainage (D) in young managed plantations was higher than MP and was relatively stable year-round (Fig. 3). However, the seasonal fluctuations in D at BHF were pronounced, being very low during the growing season (April to July), compared to the plantation forests. Yet, D at the BHF site also exceeded D at the managed sites from September to December (Fig. 3), suggesting lower ET in the BHF when deciduous trees dropped their leaves in the Fall.

3.2.2. Interannual comparison of water balance

Annual P at managed plantation sites varied by 1 – 7 % within a site inter-annually (Table 2; Figs. 4 and 5). The rainfall amount at YP₂₋₇ and MP was statistically higher than YP₂₋₈ ($p < 0.05$). Precipitation at BHF was only statistically different from the younger forest YP₂₋₇ ($p < 0.05$),

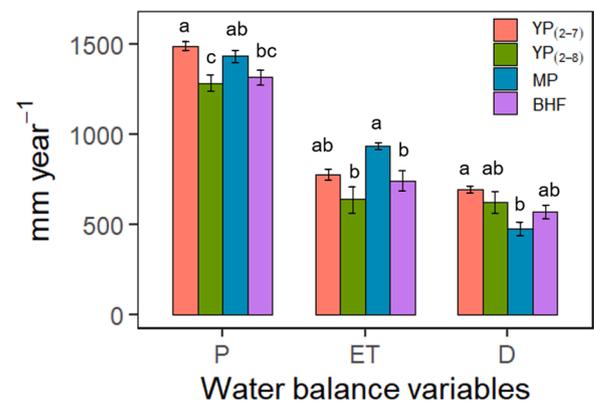


Fig. 4. Inter-site comparison of average annual precipitation (P), evapotranspiration (ET), and drainage flow (D) at YP₂₋₇, YP₂₋₈, MP, and BHF sites. Each site is represented by different colored bars. Different letters above each bar denote significant differences among sites ($p < 0.05$) using the Tukey HSD test. Error bars indicate \pm SE. Annual data across the measurement period for each site was used.

but not the other two plantation forests.

The MP site had the highest evapotranspiration with an overall annual average ET of 933 ± 63 mm compared to 776 ± 74 mm, 638 ± 190 mm, and 743 ± 172 mm for YP₂₋₇, YP₂₋₈ and BHF sites, respectively (Table 2; Fig. 4). Inter-annual variation in ET among younger managed sites was 1 – 44 % while the least variation occurred at the MP site (1 – 12 %; Fig. 5). Lower ET rates were observed in the first 3 – 4 years after harvest and subsequent plantation re-establishment at YP₂₋₇ and YP₂₋₈.

Table 2

Interannual variations in precipitation, grass reference evapotranspiration, actual evapotranspiration, drainage and change in storage at YP₂₋₇, YP₂₋₈, MP, and the BHF site.

Site	Year	Precipitation (P, mm)	Grass reference ET (ETo, mm)	Evapotranspiration (ET, mm)	Drainage (D, mm)	Δ Storage
YP ₂₋₇	2013	1380	1042	665	671	44
	2014	1508	1127	770	695	42
	2015	1518	1118	865	647	6
	2016	1533	1210	732	778	23
	2017	1437	1187	773	661	3
	2018	1552	1149	848	702	2
YP ₂₋₈	2005	1335	1059	568	767	0
	2006	1287	1156	513	749	25
	2007	1164	1219	490	627	47
	2008	1188	1194	444	795	-51
	2009	1234	1119	650	473	112
	2010	1505	879	897	570	38
MP	2011	1263	1037	903	385	-25
	2005	1486	1059	917	569	0
	2006	1398	1156	985	382	31
	2007	1305	1219	925	406	-26
	2008	1274	1194	859	382	33
	2009	1304	1119	852	469	-16
	2010	1525	879	949	538	38
	2011	1242	1037	1017	247	-22
	2012	1393	922	934	401	57
	2013	1349	1042	928	368	53
	2014	1585	1127	1049	426	110
	2015	1591	1118	885	730	-24
BHF	2016	1571	1210	1015	538	18
	2017	1464	1187	890	546	28
	2018	1552	1149	860	668	24
	2009	1490	954	720	770	0
	2010	1511	1089	982	516	13
	2011	1344	1107	930	420	-6
	2012	1174	979	606	581	-14
	2013	1173	950	597	556	20
	2014	1232	972	672	569	-9
	2015	1171	998	500	668	3
2016	1367	1043	824	540	3	
2017	1355	1057	971	388	-5	
2018	1345	1015	624	687	34	

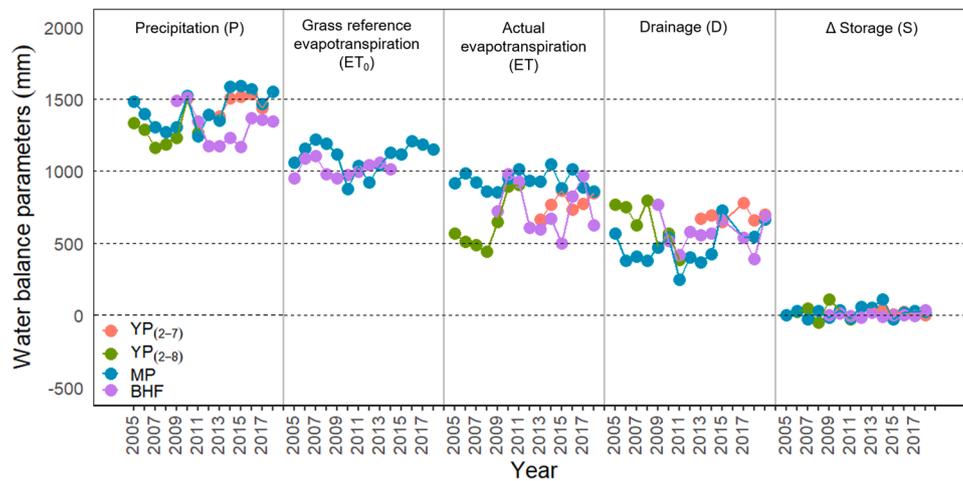


Fig. 5. Annual average water balance variables, including precipitation (P), reference grass evapotranspiration (ET₀), evapotranspiration (ET), drainage (D), and change in storage (ΔS) at YP₂₋₇, YP₂₋₈, MP and BHF sites. Each site is represented by different colored lines and circles across the measurement period.

Inter-annual variation in ET at MP was always higher than the younger plantations throughout the measurement period. Among the four sites, the BHF site had the most variation in inter-annual ET (3 – 49 %), compared to P variation at this site of only 2 – 15 %. (Fig. 5). The differences in ET between the managed MP and unmanaged BHF sites were statistically significant ($p < 0.05$).

Annual average drainage (D) in the managed sites ranged from 476 ± 129 mm to 692 ± 47 mm, although young plantations (YP₂₋₇ and YP₂₋₈) drained 148 – 216 mm more water annually than MP (Fig. 4). D at young plantation sites was 46 – 48 % of overall P. In comparison, only 33 % of P was drained from the MP site. Statistical differences were found between MP and YP₂₋₇ ($p < 0.05$), reflecting the ability of the older pine plantation to release more water through ET than from D. Meanwhile, D at BHF was 43 % of overall P, amounting to 570 ± 116 mm of D annually (Fig. 4). Although the natural forest drained 94 mm more than the older pine plantation (MP), no statistically significant differences in D were found between the BHF and the managed plantation sites ($P > 0.05$). Ecosystem ET was always higher than D, but it did appear that young plantations tended to drain comparable amounts of water as ET, with an annual difference of 14 to 84 mm at YP₂₋₇ and 14 mm at YP₂₋₈. The difference of 457 mm between ET and D at MP illustrates that much more water was released to the atmosphere than drained away. Similarly, BHF also had higher ET, 743 mm, compared to D, at 570 mm. The ΔS among sites did not vary ($P > 0.05$). Interannual variation in S was only -51 mm to 110 mm (Fig. 5).

3.3. ET₀/P, ET/P, D/P, and ET/ET₀ ratios

MP had the highest ET₀/P ratio (0.87) that was significantly higher than the young plantations and BHF, which had similar ET₀/P ratios (~0.77; Fig. 6). The annual ratio ranged from 0.58 to 1.02 at all sites (Fig. 8). ET₀ exceeded P only twice at YP₂₋₈ in 2007 and 2008. Overall mean ET/P ratio varied from 0.49 – 0.65 among all sites. (Fig. 6). The long-term ET/P ratio at the managed sites was ranked in the order MP > YP₂₋₇ > YP₂₋₈, with an average annual ET/P of 0.66, 0.52, and 0.49, respectively. At the BHF site, the annual average ET/P ratio was 0.56.

D was only about half of P (0.33 ~ 0.49) across all sites throughout the measurement period. The lowest D/P occurred at MP, with the amount of D only 1/3 of the amount of P (0.33). Interannually, D/P ranged from 0.26 ~ 0.67 (Fig. 7). Inter-site comparison of annual ET/ET₀ shows that only YP₂₋₈ and MP were statistically different (Fig. 6). The highest mean annual ET/ET₀ occurred at MP, averaging 0.86 (ranging from 0.72 – 1.08, Fig. 7). The lowest was observed at YP₂₋₈, at 0.60 but this site had high interannual variability, ranging from 0.37 to 1.02. Interannually, ET/ET₀ ratio at YP₂₋₇ ranged from 0.60 to 0.77, and

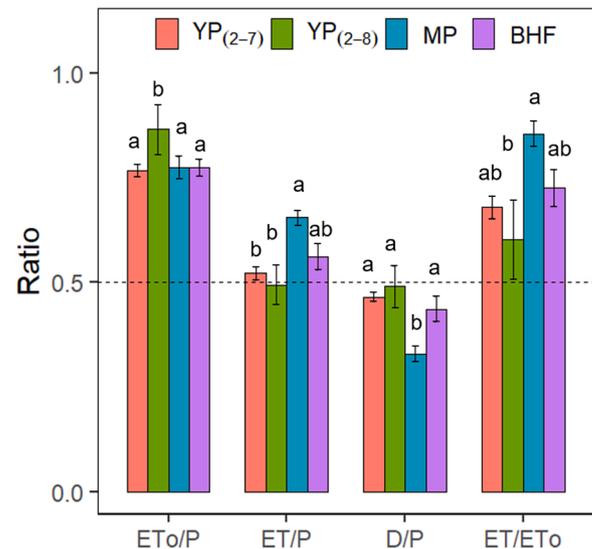


Fig. 6. Inter-site comparison of ET₀/P, ET/P, D/P, and ET/ET₀ at YP₂₋₇, YP₂₋₈, MP, and BHF sites. Each site is represented by different colored bars. Different letters above each bar denote significant differences among sites ($p < 0.05$) using the Tukey HSD test. Error bars indicate \pm SE. Based on annual data spanning the entire measurement period for each site.

ET and ET₀ were strongly correlated ($R^2 = 0.75$). The correlation of ET to the grass reference ET₀ at MP was $R^2 = 0.73$, but YP₂₋₈ had the weakest relationship ($R^2 = 0.34$). Almost 80 % of the variation in ET at BHF was explained by ET₀, with an annual average ratio of 0.73. These long-term observations demonstrate that ET on these coastal plain forests is almost always much lower than the potential water use of a reference grass.

3.4. Effect of drainage and land-use change on water balance

The present study found that 49 % – 65 % of P was lost to the atmosphere through ET, while 33 % – 48 % of P was removed through D in managed sites. The plantation sites had no apparent difference with BHF site ($p < 0.05$; Fig. 4) in this regard, where 56 % and 43 % of P was lost through ET and D, respectively. Although annual D in young managed plantations was comparable with that of the natural forest, D at the MP site was the lowest among all sites.

We found a direct linear relationship between P and ET across

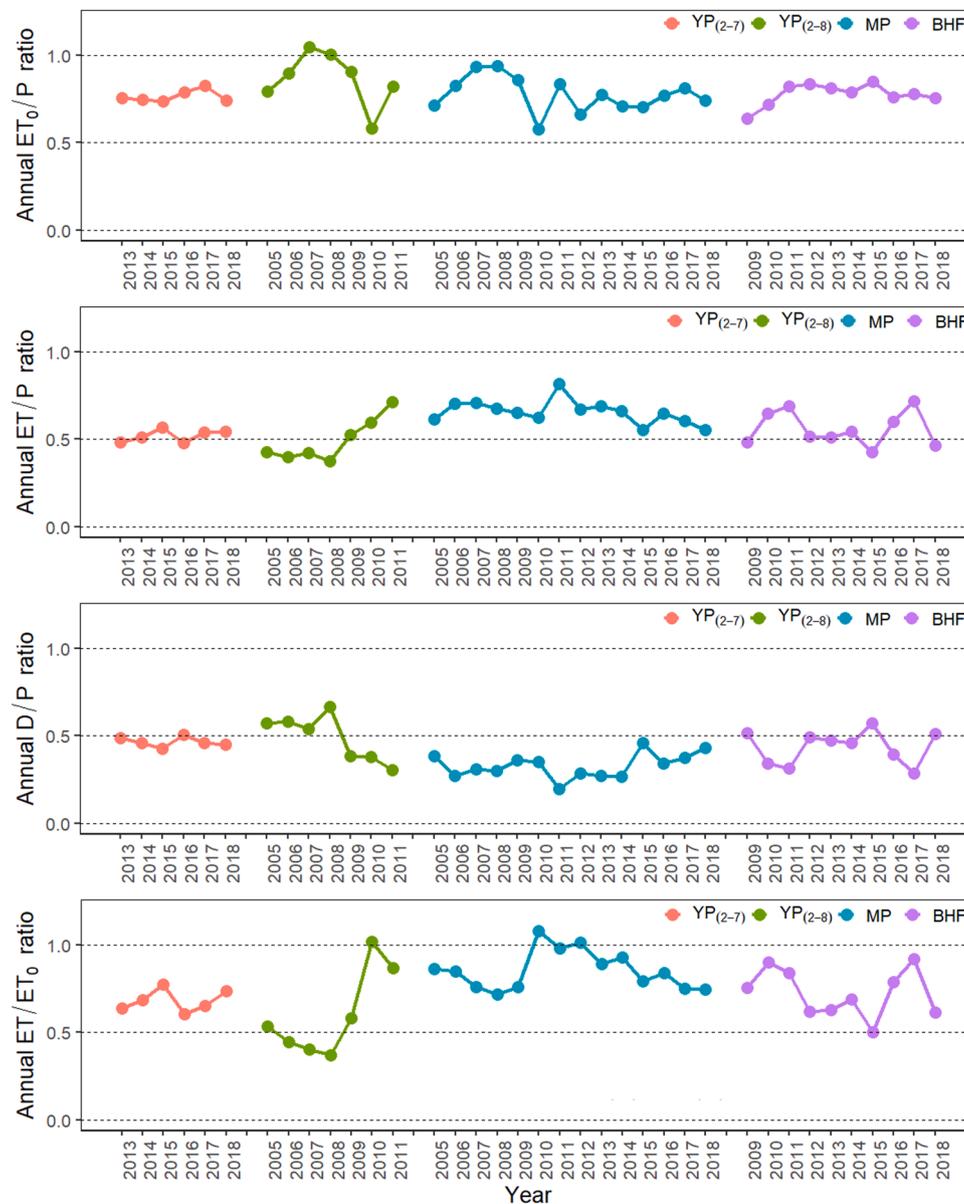


Fig. 7. Interannual variation of ET_0/P , ET/P , D/P , and ET/ET_0 at YP_{2-7} , YP_{2-8} , MP, and BHF sites across the measurement period. Thin color-coded lines per site connect monthly data.

managed sites ($R^2 = 0.40 - 0.55$) and a weaker relationship at the BHF site ($R^2 = 0.14$), suggesting a more pronounced coupling of ET and P at the managed sites than in natural forest (Fig. 8). We also found that P-D relationship in the YP_{2-7} and BHF sites were similar ($R^2 = 0.41$), and the relationship was stronger than at the MP site ($R^2 = 0.20$), suggesting higher drainage occurred at relatively wetter sites than in the mature plantation (Fig. 8).

3.5. Effect of stand age on ET

ET increased with stand development of plantation forests (Fig. 9). ET correlated well with stand age ($R^2 = 0.59$), increasing rapidly from stand initiation stage (YP_{2-7} and YP_{2-8}) to around 10 years of age and then more slowly afterward. The stand age-ET relationship followed a similar trend of stand age-LAI relationship, with $R^2 = 0.57$ using the remotely sensed LAI, and $R^2 = 0.40$ using a ground-based regression model (Fig. 10). Annual average LAI at YP_{2-7} , YP_{2-8} , and MP sites was 1.46, 2.63, and 3.00, respectively, using the remotely-sensed LAI and 2.02, 1.55, and 2.55, respectively, using the ground-based regression

equation (Supplemental Table 1). ET-LAI relationship is a little weaker with $R^2 = 0.27$ using the remotely sensed LAI and $R^2 = 0.46$ using the ground-based regression model. In both LAI derivation cases, a sudden decline in LAI at 20 years old occurred due to thinning operations done during this year. After thinning, LAI gradually increased and slowly stabilized 3 – 5 years later (Fig. 10). BHF was a mature stand over the observation period with little change in LAI, averaging 2.30, and ranging from 2.12 to 2.61 using remotely-sensed LAI data (Supplemental Table 1). Inter-annual variation in ET at the BHF site was, therefore, primarily driven by environmental factors.

3.6. Drought effects on ET

In general, the SWSI was well above 0 at all sites for most years of the study, indicating these lower coastal plain forests typically experienced very mesic conditions (Fig. 11). However, the years 2007 and 2008 were exceptionally dry, as indicated by the daily SWSI median values falling below 0 at YP_{2-8} and MP (the only sites with data for those years), which means REW levels were below 0.4, the threshold where plants suffer

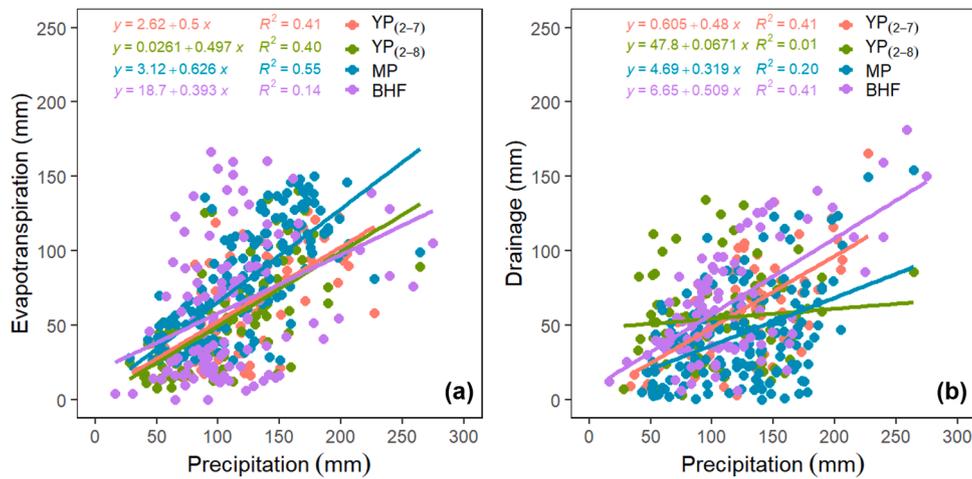


Fig. 8. Relationships between (a) P and ET and (b) P and D at YP₂₋₇, YP₂₋₈, MP, and BHF sites. Each circle indicates a monthly data colored according to the site. The colored lines indicate linear regression line.

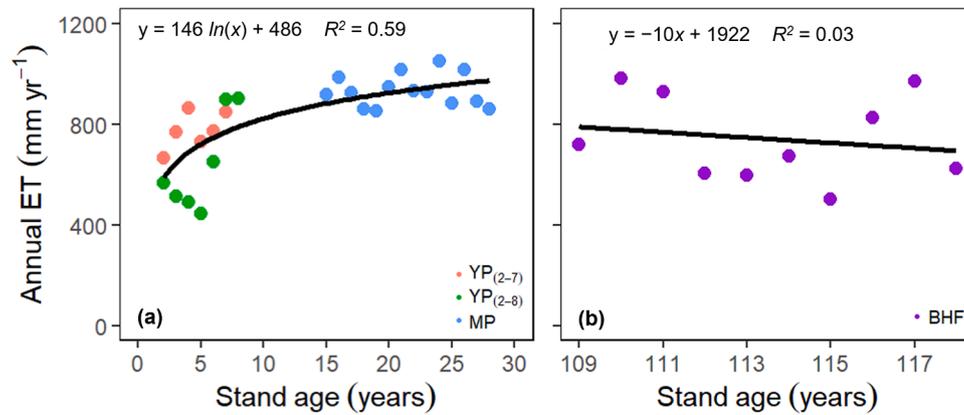


Fig. 9. Relationship between stand age and annual evapotranspiration (ET) at (a) managed sites and (b) BHF site. Each circle indicates an annual ET ratio colored according to the site. The black curve indicates the regression line.

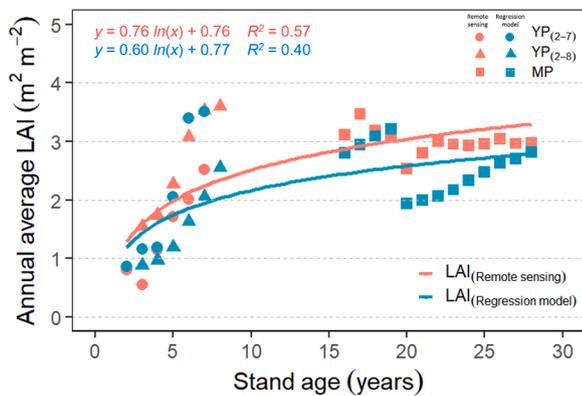


Fig. 10. Relationship between stand age and annual average leaf area index (LAI) using remote sensing and ground-based regression equation at YP₂₋₇, YP₂₋₈, and MP sites.

stress. During these years, the water table depth dropped to 50 to 80 cm below the soil surface (Fig. 2). During drought periods, we found an 8 – 11 % reduction in annual P at YP₂₋₈, with 1164 mm and 1188 mm in 2007 and 2008, respectively, compared to the long-term mean of 1282 mm. Annual ET declined by 30 – 43 % at YP₂₋₈ during this time, suggesting that ET responded proportionately greater than P (2-3 times the

decrease).

A similar magnitude in P reduction was observed at MP (10 – 12 %) during the severe drought of 2007/2008. Annual P was 1305 mm in 2007 and 1274 mm in 2008, which was 126 mm and 158 mm lower, respectively, than the long-term mean of 1431 mm. However, ET was much higher at MP compared to YP₂₋₈, and the reduction in ET due to the drought was less severe (1 – 9 %), possibly due to the older forest having greater stem capacitance and deeper roots than the younger forest.

The drought was even more intense when only considering the growing season (June – August) of 2007 and 2008, rather than the average growing season P over the entire measurement period. YP₂₋₈ experienced a 15 – 26 % reduction in growing season P, representing 39 mm – 73 mm difference from the overall mean growing season P, while it decreased 7 – 26 % at the MP (23 mm – 94 mm difference). We observed that the annual average available soil water at YP₂₋₈ was REW = 0.49 and at MP REW = 0.40, both at the drought threshold, whereby trees induce stomatal closure. At YP₂₋₈, the number of days where REW fell below the threshold of 0.40 was 179 in 2007 and 200 in 2008. MP had soil water content lower than the REW drought threshold for 188 and 258 days in 2007 and 2008, respectively. The water table depth fell below 50 cm and was generally much lower than in other years during these drought periods.

The Generalized Additive Modeling (GAM) with the climatic factors Rn, Tair, water table depth, albedo, and REW, and the biological control of gross primary production (GPP), revealed that the combined effect of these variables ranked Rn > albedo > GPP > Tair > REW > precipitation

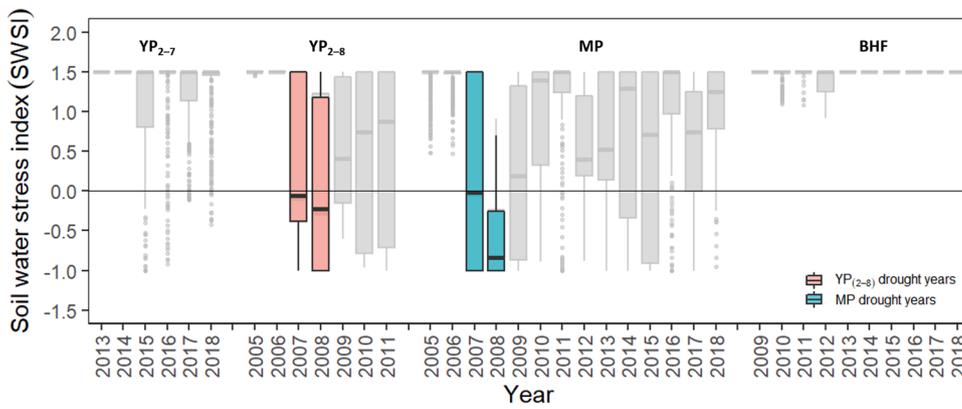


Fig. 11. Interannual variation in soil water stress index (SWSI). Daily SWSI values were used during the measurement period at YP₂₋₇, YP₂₋₈, MP, and BHF sites. Drought years were characterized by having an SWSI median falling below zero. These drought events occurring in 2007 and 2008 were highlighted with red color boxes for YP₂₋₈ and blue for the MP site. In the box plot, the thick horizontal line shows the median, and the box extends to the upper and lower quartiles, vertical dashed lines indicate the nominal range, open circles indicate points that lie outside of the nominal range. SWSI at 0 indicates that REW = 0.4 (threshold value). The greater SWSI above 0, the lower is the stress level.

explaining 91 % of the variation in ET at YP₂₋₈ during the drought period in 2007/2008. At MP, the predictors were ranked $R_n > GPP > T_{air} > REW > albedo > precipitation$, with their combined effects explaining 88 % of the variation in ET. Light availability was the best predictor of ET at both sites, with albedo the second key predictor at YP₂₋₈. YP₂₋₈ maintained the same amount of ET at albedo ranging from 0.10 – 0.20, whereas ET declined as the albedo increased at MP within the range of 0.10 – 0.15. Maintaining high ET at higher albedo during drought suggests that when the reflectance is high, the younger plantation still loses significant amounts of water to the atmosphere. In contrast, the older plantation was less responsive to an increase in albedo, suggesting that the mature forest better-conserved soil water as the reflectance was getting higher. Surprisingly, neither REW nor precipitation held the primary key to variation in ET during drought years when combined with other explanatory variables, indicating the significant effect of other climatic or biological factors.

We further analyzed the effect of the drought on ET using the Budyko framework, which is widely used in hydrological studies (Budyko, 1974; Tor-ngern et al., 2018). The Budyko curve is the relationship between the potential evapotranspiration (herein we use the grass reference evapotranspiration, ET_0) and actual evapotranspiration (ET), each normalized by precipitation (P). The Budyko curve is partitioned to conditions in which ET is limited by energy supply ($ET_0/P < 1$), regulated by net radiation and vapor pressure deficit, or by water supply ($ET_0/P > 1$), controlled by vegetation (canopy interception) and soil water storage. The larger the ET_0/P ratio, the less P contributes to water yield. This analysis showed a weak negative response of ET/P with ET_0/P ($R^2 = 0.12$) at YP₂₋₈ (Fig. 12), wherein the evaporative index tended to decrease with the severity of dryness. In the case of MP, the severe 2007/2008 drought did not cause the rotation-age plantation to

become water-limited.

3.7. The ET-GPP relationship during drought period

During the 2007/2008 drought period, we found a 6 – 9 % enhancement in GPP at the MP site while the YP₂₋₈ site had 20 – 53 % increase (Fig. 13). In contrast, we found a slight reduction in ET at YP₂₋₈ (30 – 43 %) and MP (8 – 11 %) thereby a higher water use efficiency (WUE) during drought period (Fig. 13). This inverse increase in GPP and reduction of ET during drought suggests a decoupling of ET and GPP during extreme dry conditions. The positive linear relationship between ET and GPP at MP and YP₂₋₈ sites during the drought period were relatively similar (Fig. 14). However, ET at YP₂₋₈ was sharply reduced, whereas GPP was not.

4. Discussion

4.1. Seasonal and inter-annual comparisons of site water balance

Overall, ET was lower than P during the late growing season (July – August) in all years and sites, suggesting there is sufficient plant-available water in the soil to meet atmospheric demand despite periods with no rain. This is in part due to adequate P throughout the year, but also, these lower coastal plain systems receive groundwater from adjacent uplands. These physiographic factors maintain the high water table and constrain D and ΔS components of the site water balance. With little soil water limitation, the high solar radiation during the growing season induces strong evaporative demand resulting in the high growing season ET. In dry periods during the growing season, pine forests rely on deep, extensive root systems to access soil water storage and

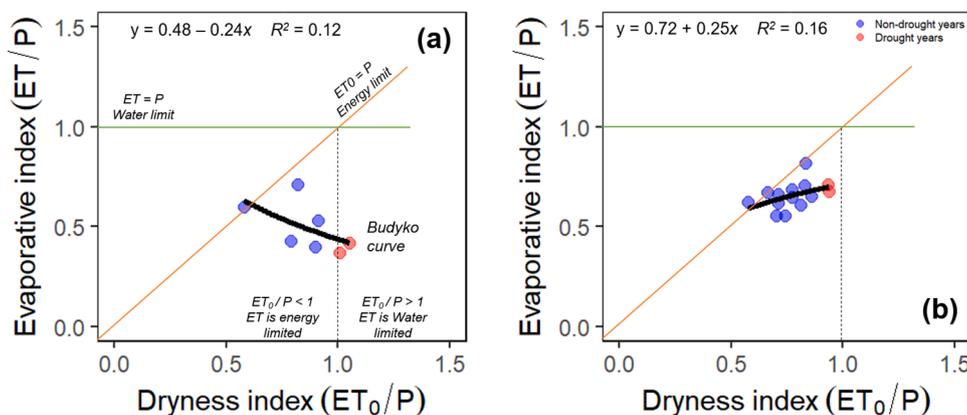


Fig. 12. The Budyko curves derived using annual data from YP₂₋₈ (a) and MP (b). Green and red lines show physical boundaries of the curve where dryness index (ET_0/P) increases with the evaporative index (ET/P) over the energy-limited region and where ET/P is independent of ET_0/P over the water-limited region. The blue circles correspond to non-drought years and red circles for drought years (2007 and 2008).

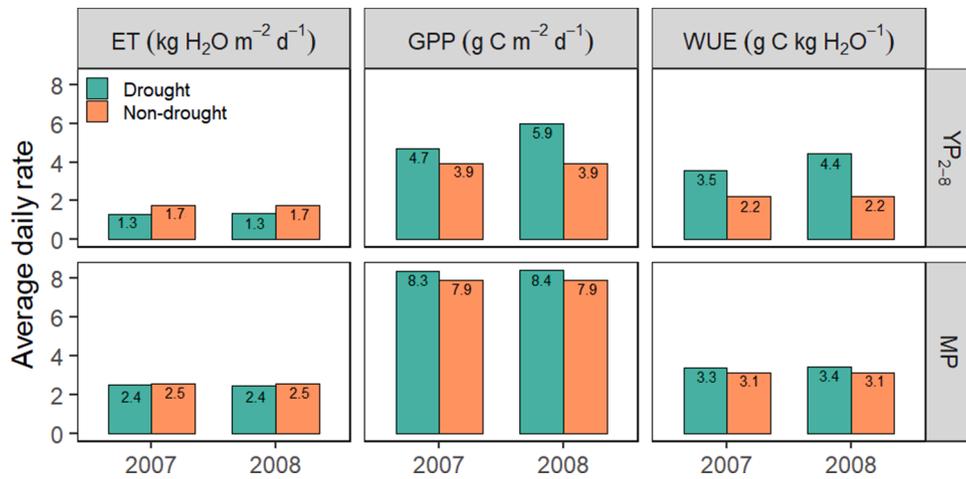


Fig. 13. The average daily rate of evapotranspiration (ET), gross primary productivity (GPP), and water use efficiency (WUE) at the YP₂₋₈ and MP sites during the drought and non-drought periods in 2007 and 2008.

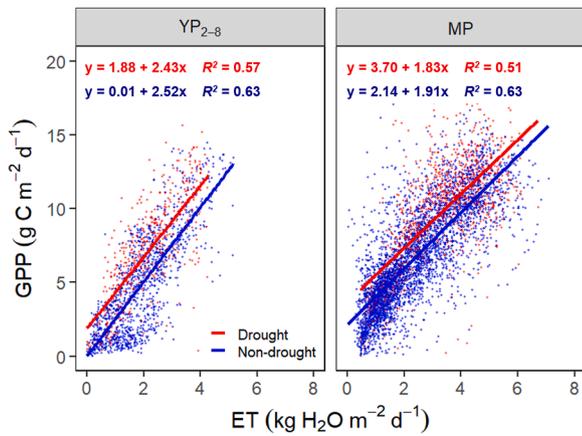


Fig. 14. Relationship between average daily rate of evapotranspiration (ET) and gross primary productivity (GPP) at the YP₂₋₈ and MP sites during the drought (red line and text) and non-drought (blue line and text) periods in 2007 and 2008.

groundwater (Domec et al., 2012a; Luis et al., 2005; Vincke and Thiry, 2008). Hydraulic redistribution and upward soil capillary water flux may also help sustain ET during the growing season (Allen et al., 2010; Domec et al. 2010). These root-based mechanisms of soil water access

are consistent with the higher rates of ET at MP compared to YP sites, where ET may be limited by smaller root systems (in addition to lower LAI, as shown in Supplemental Table 1), along with lower stem capacitance, in the early stages of stand development.

Annual ET ranged from 638 – 933 mm in the managed plantations and 606 – 982 mm at the natural wetland forest. ET for forest ecosystems in the southeastern US ranges from 486 to 1509 mm (Bracho et al., 2008; Ford et al., 2007; Oishi et al., 2010; Rao et al., 2011; Sun et al., 2010, 2002; Wilson and Baldocchi, 2000; Domec et al. 2012a) (Fig. 15). The ET rates found in this study are within the mid-range when compared to other hydrologic studies conducted in other forest ecosystems worldwide, reporting annual ET ranging from 280 to 1029 mm (Aguilos et al., 2019; Brümmer et al., 2012; Humphreys et al., 2003; Li et al., 2010; Tang et al., 2014; Vourlitis et al., 2014; Wilson and Baldocchi, 2000).

Our data support the hypothesis that the rotation-age pine plantation would have higher ET than the two young plantations. Climatic controls, especially net radiation and albedo, and higher leaf area, contributed to higher ET rates, consistent with a previous study at this site (Sun et al., 2010). This is also consistent with a broader comparison to other southeastern US studies (Figure 15), in which the primary drivers of ET were found to be Rn and albedo ((Rao et al., 2011; Wilson and Baldocchi, 2000), LAI, and productivity (Bracho et al., 2008; Sun et al., 2002), vapor pressure deficit (Oishi et al., 2010), and soil water availability (Bracho et al., 2008). The data also support the hypothesis that MP ET would be higher than that of BHF, and here we attribute that to

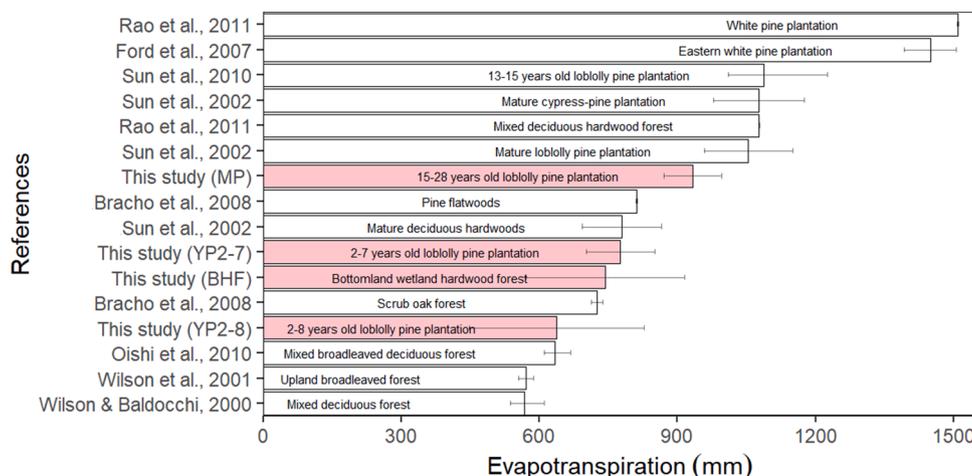


Fig. 15. Comparison of studies on annual evapotranspiration (ET) in major forest ecosystems in the southern United States.

biological and climatic effects.

The magnitude of inter-annual variation in ET at the rotation-age plantation, of 852 mm to 1049 mm, is within the range previously reported for the same site (Liu et al., 2018; Sun et al., 2010; Tian et al., 2015; Yang et al., 2020). The small variation in annual ET has been attributed to the shallow water table and deep root systems of this coastal plain setting (Domec et al., 2012). We also found that ET differed between the two young pine plantations, even though they were nearby. Differences between young plantations suggest that even in a single geographical location, there can be multiple trajectories in ET during stand development due to local differences in soil conditions, drainage, surrounding vegetation, inter-annual variation in climate, etc. (Amiro et al., 2010; Borders et al., 2004; Noormets et al., 2012).

4.2. ET , D , ET_0/P , ET/P , D/P , and ET/ET_0 ratios

Except for YP₂₋₈, where $ET/ET_0 = 0.34$, all other sites had high ET/ET_0 ratio of 0.73 ~ 0.81, closer to what was previously reported (0.92) across North Carolina watersheds (Sun et al., 2002, 2001), suggesting that our sites are generally not water-limited. Wetland-dominated watersheds do not necessarily have a higher ET/ET_0 ratio (Sun et al., 2002).

Our data support the hypothesis that ET is always lower than P on an annual basis at these wetland sites. However, the fact that ET can slightly exceed P in some months connotes important implications for ecosystem water balance and its response to climatic variability. Both growing season precipitation and soil water recharge in the non-growing season were necessary to meet evaporative water demand in the growing season (Sun et al., 2011a). Therefore, any shifts in seasonal precipitation patterns due to climate change (e.g., drought) could profoundly affect ecosystem water use patterns during the growing season, thus affecting the sustainability and productivity of lower coastal plain managed forest ecosystems.

Annual ET, averaged over 13 years at the managed sites, was lower than annual P by about half (ratio 0.40 ~ 0.55), indicating that the overall effects of ET on water table depth are not significant at lower coastal plain sites. This is consistent with another study that found lower ET than P under similar conditions (Almeida et al., 2007). An ET of about half of P indicates that the remainder of P contributes to watershed yield in this coastal plain area, typical of ecosystems with very high soil water availability (Liu et al., 2018).

4.3. Effects of ditching and land-use change on water balance

The lower coastal plain is flat and poorly drained, resulting in a shallow water table. Ditching was historically used in this region to drain excess water to improve site conditions for timber production (Amatya et al., 1996).

Our study found no statistical difference ($p > 0.05$) between managed sites and natural forest in terms of the amount of P being lost to the atmosphere through ET and removed through drainage. This result contrasted our hypothesis that draining the landscape with ditches during intensive forest management would decrease the fraction of P leaving the site as ET, as compared to the natural wetland forest. In addition, our data suggest that land-use change and ditching slightly enhances D at YP 2-7 and YP2-8, but ultimately decreases D when plantations reach maturity. Differences in precipitation regimes and topographical characteristics (i.e., proximity to streams) between the plantation and BHF sites may also contribute to differences in ET and D, making hydrologic comparisons difficult. However, the drainage pattern of BHF is quite different from the drained, managed plantations, with extremes in both growing and non-growing seasons (Fig. 2). This natural bottomland hardwood forest is occasionally submerged and often experiences prolonged inundation during the non-growing season (Miao et al., 2013). The water table at BHF is at full storage capacity during non-growing seasons, again due to the low topographic setting, which receives water from adjacent uplands yet has impeded drainage because

of proximity to the sea. This poor drainage is characteristic of very diffuse lower coastal plain watersheds with low topographic relief and poorly defined flow pathways for runoff, decreasing the rate of overland flow. However, the water table became deeper at BHF during the growing season during periods with no rain and high seasonal ET. Therefore, modeling studies would benefit by considering the seasonal change in site water balance in such hydrologically unique coastal plain settings.

Generally, studies suggest that forest harvesting results in increased drainage (Sun et al., 2001), for at least 8 years afterwards (Arthur et al., 1998). In our study, YP₂₋₇ and YP₂₋₈, which are recently clear-cut sites drained 122 mm – 216 mm more water (annual total = 624 mm – 692 mm) than MP (annual total = 476 mm), and 54 mm – 122 mm more than BHF (annual total = 570 mm). Thus, tree removal may raise the water table (Aust and Lea, 1992; Lockaby et al., 1997) due to the reduction in ET, consistent with observations in our two young plantation sites (Fig. 2). However, we caution that our study is not a paired watershed design, and thus inferences must be considered with care.

4.4. ET -stand age relationship

The positive relationship between ET and stand age observed in our study is also consistent with observations of others ((Jagodziński and Kalucka, 2008; Yang et al., 2020, 2017). Newly established plantations tend to have low ET due to low leaf area (Supplemental Table 1, Fig. 10) and developing root systems. However, ET increases with time commensurate with increases in productivity (Aguilos et al., 2020). In the current study, the ET-GPP coupling was moderate at the young pine plantations ($R^2 = 0.56$ at YP₂₋₇ and $R^2 = 0.73$ at YP₂₋₈). As the stands get older (>15 years old), ET gradually stabilizes until it approaches the rotation age of ~28 – 30 years old, the same period when GPP reaches a plateau. The ET-GPP relationship in our study was very strong ($R^2 = 0.86$ at MP). Others observed LAI recovery following a disturbance until 15 years of age, after which LAI gradually stabilized with increasing age (McMichael et al., 2004). LAI is therefore a key component of the dynamics in productivity and evapotranspiration. However, care must be taken in the use of allometric or remote sensing methods to obtain LAI. These methods may underestimate or overestimate LAI (Albaugh et al., 2014; Arias et al., 2007; Liu et al., 2015) due to overlapping or clumping between the leaves and light obstruction from branches, boles, and stems (Zheng and Moskal, 2009).

The positive ET-stand age relationship has implications for forest management in that the establishment of plantations of different age classes in parallel will result in reduced variation in water table levels, thus reducing the effects of extreme water anomalies. However, the ET-stand age relationship at BHF was insignificant despite a high ET-GPP correlation ($R^2 = 0.74$), suggesting greater climatic control of ET at the natural forest rather than biological constraints in this mature forest that does not undergo large changes in LAI from year to year.

4.5. Effect of drought on ET

A significant drop in precipitation in 2007 and 2008 resulted in increased depth to water table and decreased drainage at YP₂₋₈ and MP (Aguilos et al., 2020; Noormets et al., 2010; Domec et al., 2012b; Sun et al., 2010), and corresponding decline in ET. The long-term data presented here support our hypothesis that drought affects young plantations more severely than rotation-age plantations. The decrease in ET at MP (9 %) was much smaller than occurred at the young pine sites, which we attribute to the sharp increase in water table depth.

Typically, the high water table of these lower coastal plain sites provided abundant soil water to meet the atmospheric evaporative demand. Also, hydraulic redistribution by deep roots might have played a crucial role in replenishing soil water to the upper soil layers (Domec et al., 2010), thus sustaining tree transpiration and ET. A slight reduction of ET during drought at MP can be attributed to reductions in

canopy interception, tree transpiration, understory transpiration, and soil evaporation (Domec et al., 2012a; Liu et al., 2018; Oishi et al., 2010). It has been observed that a reduction in tree transpiration occurs with a decrease in soil water content, especially when REW drops below the threshold of 0.4 (Almeida et al., 2007; Granier et al., 1999; Vincke and Thiry, 2008). Prolonged soil water deficits in our study, with 177 – 188 days of REW falling below 0.4 in 2007, which got worse in 2008 (200 – 258 days), indicate that trees might be under constant water stress during these periods. However, trees at the MP site may have responded to severe stress by partially closing stomata (Domec et al., 2009), improving water use efficiency.

Our results support the hypothesis that drought would decrease ET at the managed sites. There was a higher proportional decline in ET, 30 – 43 %, at the young plantations compared to the rotation-age plantation (8 – 11 %). This higher reduction in ET at the younger stands is consistent with another remote sensing-based study, which reported that drought in this coastal region had a more significant effect on young stands (Yang et al., 2020). Drought-induced reduction in transpiration of young plantation forests might be due to low biomass/LAI, shallow root systems, low stem capacitance, and reduced plant hydraulic conductivity (Domec et al., 2012a).

In addition to differences in roots and biomass among the four sites, net radiation, which is influenced by canopy albedo, was also a key factor in explaining ET. Albedo significantly affects net radiation and is highly dependent on the dynamics of plant growth patterns, and is influenced by ‘greenness’ and ‘wetness’ of the land surface (Sun et al., 2010). Net radiation was the primary climatic control on ecosystem ET during the observed drought period, consistent with other studies (da Costa et al., 2018; Gaofeng et al., 2014; Stoy et al., 2006; Yu et al., 2008). The discrepancy in Rn between young (mean = 104 W m⁻²) and rotation-age (134 W m⁻²) plantations during the 2007/2008 drought was due to differences in surface albedo of 0.16 and 0.12 for the sites, respectively. Surface albedo has been shown to decrease with increasing canopy height (Kuusinen et al., 2016), increasing leaf area index (Bright et al., 2015; Forzieri et al., 2018), increasing biomass (Lukeš et al., 2014), and increasing age (Amiro et al., 2006; Bright et al., 2013; Kuusinen et al., 2014), which characterize differences between MP and younger plantations. Thus, MP absorbed more solar radiation, had lower reflectance, and thus lower surface albedo. Any alteration in forest structure may affect albedo and thus Rn (Gaofeng et al., 2014). Therefore, previous forest harvesting may have caused an increase in surface albedo at YP₂₋₈. Further, the understory at young plantations was mostly herbaceous and deciduous woody plants that lost leaves during the non-growing season, unlike with the evergreen canopy at MP, where needles were present all year-round. Lower canopy cover leads to higher surface reflectance and thus higher albedo at the younger pine forests.

Based on the Budyko’s curve (Fig. 12), the ET₀/P ratio during the 2007/2008 drought years was higher than 1.0, unlike at the MP site where ET₀/P remained less than 1.0. As drought conditions became more severe, the sensitivity of transpiration to water deficit increased, suggesting a down-regulation of stomatal conductance, which lowered tree transpiration under extreme drought. This deviation from the theoretical Budyko’s curve at the young plantation can be attributed to a management intervention (e.g., harvest), which changed the vegetation cover (Tor-ngern et al., 2018). Soil type (Wang et al., 2009), water storage capacity (Milly, 1994), the timing of soil water recharge (Potter et al., 2005), and other factors such as incident radiation, canopy leaf area, tree density, canopy conductance, and other micrometeorological conditions also affect evapotranspiration, (Budyko, 1974; Tor-ngern et al., 2018).

Overall, annual ET/P (0.49 ~ 0.66), mean annual evaporative index, and ET₀/P (0.60 ~ 0.86), mean annual dryness index, indicated that the young and mature plantations fell within the theoretical Budyko (1974) space, and within the 1:1 line breakpoint, meaning that all sites were more energy-limited rather than soil water deficient.

4.6. The role of GPP to changes in ET during drought period at managed sites

The positive ET-GPP relationship was more significant at MP compared to YP₂₋₈. Therefore, higher net radiation and LAI (Supplemental Table 2) maintained higher ET (Figs. 13 and 14) and higher GPP because there was little water stress due to more available groundwater and deeper root systems in MP. In contrast, at YP₂₋₈, ET rates were sharply reduced, but GPP was not. It is known that plants may delay bud break and leaf expansion with drought (Mijnsbrugge et al., 2016), or roots of understory vegetation may exploit a higher soil volume that potentially increases tolerance to drought (Warren et al., 2015). Further, ET response to drought at both young and mature plantations may have also been influenced by varying tolerance to a drought of different tree species or understory species that occurred at the sites, depending on drought intensity levels (Maréchaux et al., 2018; Stahl et al., 2013). Overall, our results confirm that in addition to available energy, biological controls (i.e., GPP, stomatal conductance, roots, LAI) play a critical role in ET processes. Therefore ET is a good integrator of many physiological and physical processes affecting forest water cycling (Ford et al., 2007; Stoy et al., 2006; Zhou et al., 2008) and the tightly coupled water-carbon cycling (Sun et al., 2011b).

Interestingly, a 6 – 9 % enhancement in GPP at the MP site and 20 – 53 % increase in GPP at YP₂₋₈ during the 2007/2008 drought period (Aguilos et al., 2020) were not accompanied by a similar rise in ET at both sites (Fig. 13), suggesting increased WUE (Domec et al., 2015). We found slight to moderate reductions in ET at YP₂₋₈ (30 – 43 %) and MP (8 – 11 %) during the drought. This result does not support our hypothesis that ET and GPP will remain tightly, positively coupled despite extreme climatic conditions (Fig. 14), and is inconsistent with the widely reported coupling of ET and GPP (Brienen et al., 2011; Fisher et al., 2007; Huang et al., 2015; Yu et al., 2008). Although we found that ET and GPP respond to the same primary climatic drivers, the magnitude of response differs for the two processes during anomalous soil water years. This asynchrony of the response represents a decoupling of carbon and water cycling under extreme conditions. GPP is usually affected less during severe water depletion since stomatal closure typically induces a stronger down-regulation of transpiration than photosynthesis (Maréchaux et al., 2018), and transpiration is linearly related to stomatal conductance. In contrast, photosynthesis may be limited by a variety of other factors and does not respond linearly to instantaneous changes in stomatal conductance (Chaves et al., 2003). Differences in sensitivity of ET and GPP to drought illustrate the challenges that must be overcome for ecosystem models to accurately simulate these processes.

4.7. Implications

To our knowledge, the present study represents the longest direct measurement of ET and forest water balance of lower coastal plain forested wetlands in the southeastern US, advancing our understanding of the hydrologic responses to land-use change (drained vs. natural hydrology), inter-annual variation in climate, and climate extremes (drought). Our results show that water use (ET) by young drained plantation forests increases over time, as trees grow and LAI stabilizes around age 10-15 years, the time of canopy closure. Mid-rotation plantation forests that have higher biomass and productivity had similar or higher ET compared to undisturbed natural forested wetlands. This has implications for drainage management in the coastal plain at the landscape scale. Forest land managers can sustain ecological functioning to extremes in water availability (drought or flooding) by using the relatively large ET of pine plantations, or ‘biological drainage’ (Hao et al., 2015), to moderate watershed yield (more from young plantations, less from mature), and water quality concerns from forest cutting and regeneration. The ditching practices of the past in coastal forested wetlands indeed increases flow and increases the risk of backflow and saltwater intrusion during storms and sea-level rise. Maintaining tree

stocks may help mitigate the hydrological and climatic regulation functions (i.e., ET capacity) from disturbances such as land-use change (i.e., ditching). Our data suggest that these drained wetlands are resilient to extreme episodic droughts due to the shallow water table. Therefore, maintaining the water table through controlled drainage (Amatya et al., 2006) that alters ditch water levels based on-site hydrologic conditions can become increasingly important to mitigate drought effects on forest productivity under a changing climate. However, these wetlands are vulnerable to long-term climate change and sea-level rise, which likely alters the hydrology of the physiographic region of the southeast lower US coastal plain (Zhang et al., 2018), with implications for forest productivity, mortality and carbon storage (Aguilos et al., 2020). Watershed management in coastal plain regions should consider the potential combined effects of climate change, sea-level rise, and land-use dynamics on water balance. This study provides a long-term benchmark of forest water balance for both natural and managed conditions.

Our study also suggests that wetland forest ET/ET₀ values are somewhat variable depending on the stage of stand development. Annual forest ET rates are generally lower than ET₀ in this region but can exceed ET₀ in the peak of the growing season. Forest productivity is tightly coupled with ET. Thus forest ecosystem services of carbon sequestration and water supply have a tradeoff relationship (Jackson et al., 2005). However, this relationship varies among forest stands of different ages. Hydrological or regional climate models must consider the vegetation dynamics and the biological controls on hydrology to fully assess the role of wetland forests in regulating local and regional hydrology and climate.

5. Conclusions

Based on 37 site-years of monitoring, we found that total annual ET in plantation forests was more than half of precipitation. Poorly drained pine plantation forests with high leaf area had comparable ET compared to undisturbed forested wetland with low productivity and higher inundation. ET was quite resilient to changes in precipitation variability (e.g., drought) due to the flat topography and shallow water table. Annual variability in precipitation explained most of the variation in drainage in both intensively managed pine plantations and natural bottomland hardwood forest. Pine plantation ET increased sharply with stand age and leaf area index during the early years (0-10), but then gradually stabilized thereafter. This study improves understanding of the effects of land-use change and climate on ET and water balance of coastal forested wetlands. Forest structure (age, leaf area) and drainage patterns should be considered when parameterizing ecosystem models to accurately project responses of southern coastal plains to rapidly changing environmental conditions.

Declaration of Competing Interest

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

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Supplementary materials

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