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RESEARCH ARTICLE

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Water balance of municipal wastewater irrigation in a coastal forested watershed

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Abstract

In the southeastern United States, coastal communities face challenges for water resources and wastewater treatment capacity. In North Carolina, 51 municipalities irrigate forests with municipal wastewater to absorb nutrients, reduce direct effluent discharge to surface waters, and recharge groundwater. Most facilities have land-applied wastewater for decades, but there are no quantitative studies on the hydrologic impacts of this practice. This study developed a simulated water balance for the largest forest land-application system in North Carolina which treats wastewater daily by irrigating 30 km^2 of a mixed hardwood-loblolly pine forest. A distributed hydrological model (MIKE SHE) was adapted to simulate 20 years of watershed evapotranspiration (ET) and water table depth (WTD) under irrigated and non-irrigated conditions. We found that irrigation impact to annual and monthly WTD was negligible in years with average and above average rainfall. For wet years, drainage increased with irrigation while ET and WTD remained similar to nonirrigated conditions. In dry years, ET was 31 to 39 mm higher in irrigated forest than nonirrigated forest though the change in groundwater storage remained close to zero annually. Our simulation study suggested that the drivers of on-site drainage were predominantly rainfall and irrigation, and the annual watershed drainage increased in volumes equal to 93%–100% of the added annual irrigation input. This study offers insights to water balance dynamics in irrigated forests and coastal forest resiliency to variable wastewater hydraulic loading.

KEYWORDS
irrigation, loblolly pine, MIKE SHE, wastewater, water balance

1 | INTRODUCTION

In southeastern U.S. coastal communities, the sustainable management of forests and water resources is complicated by increased human population growth and climate change (Klepzig, Shelfer, & Choice, 2014; Manda & Klein, 2014; Webster, Holland, & Curry, 2014). Another challenge to coastal North Carolina communities is growing demand for municipal wastewater treatment. North Carolina is unique among the southeastern U.S. states in its utilization of land treatment for municipal wastewater treatment. Fifty-one municipalities use forest systems to treat primary- and secondary-treated municipal wastewater (Nielsen, 2011). These permitted facilities irrigate municipal wastewater onto forests to absorb nutrients and recharge groundwater (Nichols, 2016). Most facilities have land-applied wastewater for several decades with irrigation amounts equivalent to average annual rainfall of 1,346 mm (Birch, Emanuel, James, & Nichols, 2016). There are no quantitative studies on the hydrologic impacts of these green infrastructure systems postinstallation nor any evaluation of their response to chronic hydraulic loading.

The practice of wastewater land treatment experienced a resurgence in the late 1980s and early 1990s in the United States (Nichols, 2016), particularly in the Southeast, where wastewater land treatment is used to irrigate agricultural fields or forests with 25 to 50 mm of wastewater each week. Over several decades of operation, land treatment systems have been found to operate at a lower cost of treatment than other conventional wastewater treatment facilities that discharge treated wastewater directly to surface waters (Muga & Mihelcic, 2008). Land treatment of wastewater has likely effects to forest communities and processes. For example, studies of forest land treatment systems in North Carolina have shown greater biomass productivity for several hardwood tree species than the same species in nonmanaged hardwood forests (Ghezehei, Shifflett, Hazel, & Nichols, 2015). A catchment study observed that the wastewater fraction in shallow groundwater and surface waters increased from 50% to 76% and 3% to 58%, respectively, as site conditions transitioned from wet to dry (Birch et al., 2016). A recent watershed analysis at the same site showed that forest land treatment did not contribute more chemicals of concern to groundwater and surface waters than comparative off-site water sources (Hedgespeth et al., 2019).

In spite of the importance of hydrology on potential impacts of wastewater application on water quality and forest ecosystems, no process-based studies have been conducted on the effects of coastal forest land treatment systems on watershed water balances, evapotranspiration (ET), drainage, and water table depth (WTD) dynamics. However, prior studies on southern coastal forests provide important context to how irrigation might impact coastal forest hydrology. Harder, Amatya, Callahan, Trettin, and Hakkila (2007) observed that southern coastal-plain forest hydrology has dynamic drainage in response to rainfall; runoff coefficients varied substantially from less than 10% in dry years to more than 50% in wet years. These findings suggest that added water inputs to a forest land treatment system would increase drainage when rainfall is normal or above normal. Amatya, Chescheir, Williams, Skaggs, and Tian (2019) reported that shallow water tables, a common feature to coastal plain geography, responded similarly to drought and wet periods. Guinn Garrett, Vulava, Callahan, and Jones (2012) also found surface water to be highly connected with groundwater contributing up to 37% of stream water in the dry, dormant season. Sun et al. (2010) found that ET (>70% precipitation) exceeded drainage and represented the major water loss from coastal forest systems. Even under extreme drought, ET remained relatively stable due to trees accessing shallow groundwater (Liu et al., 2018). Amatya and Tian (2016) estimated that water surplus above potential ET is a long-term trend for coastal forest areas. Because the goal of forest land treatment systems is to treat wastewater without surface ponding of water to avoid hydric soil formation, these systems are irrigated at a slow and constant rate. Modelling the annual and seasonal water balances of these systems across variable rainfall would provide insight to their hydrological response and contribute to a better understanding of coastal forest hydrological dynamics.

This study was conducted at the largest forest land treatment facility in North Carolina and the third largest forest land treatment facility in the United States of America. We used MIKE SHE, a process-based hydrological model, as a tool to estimate hydrologic responses to forest irrigation for the last 20 years. The model has been validated for applications in the Atlantic Coastal Plain (Dai et al., 2011; Lu, Sun, McNulty, & Comerford, 2009; Zhang & Ross, 2015) and was shown to predict better results than DRAINMOD for a coastal forested watershed in South Carolina (Dai et al., 2010). Although DRAINMOD has been used to evaluate drainage design parameters for wastewater irrigation in Ohio for a uniform field (Öztekin, Holdsworth, Brown, Kurunc, & Rector, 1999), we preferred a grid-based distributed model for this simulation study to account for spatial land heterogeneity, location, and amount of irrigation in irrigated areas.

The objectives of this study were to (1) construct annual and monthly water balances for the study site using daily measured and simulated hydrologic data, (2) test the MIKE SHE model for estimating annual and monthly ET and WTD in irrigated and nonirrigated conditions using water table data collected manually every 3 months for the last 17 years and daily for the most recent year, and (3) evaluate and characterize the impacts of wastewater irrigation to annual and monthly ET, WTD, and drainage across climate variability. ET and WTD were estimated for both irrigated and nonirrigated conditions in order to calculate drainage in the water balance model. Drainage was defined as the combination of surface runoff and lateral groundwater drainage to surface waters. The water balance was calculated monthly and annually for the study period using simulated ET and WTD and measured precipitation and irrigation. Based on prior literature for coastal forests, we hypothesized that irrigation would increase ET, elevate the water table, and increase drainage. We expected that the irrigated site would become more saturated over time.

2 | MATERIALS AND METHODS

2.1 | Site description

The study site is located 12 km outside the City of Jacksonville in Onslow County, North Carolina (latitude 34.758760, longitude −77.433002). The 106-km^2 watershed is delineated around the outlet of Southwest Creek, an ungauged third-order headwater stream to the New River with 13 tributaries (Figure 1, Figure S1). The topography is relatively flat with a mean

surface slope of 1.65% and ground surface elevations ranging from 3 to 8.5 m above mean sea level (Birch et al., 2016). The irrigated forest is primarily loblolly pine, *Pinus taeda* (L.), but contains mixed hardwood species. There are 26 soil types in the watershed but three dominant soils consisting of Baymeade fine sand (21%), Norfolk loamy fine sand (13%), and Foreston loamy fine sand (12%). The average low and high temperatures are 10°C to 23°C with an average 30-year annual rainfall of 1,379 mm (NOAA, Jacksonville EOC). Twenty-eight wastewater spray fields cover 30 km^2 of the watershed (Figure 1). The permitted maximum annual hydraulic loading of wastewater ranges 1,244 to 1,590 mm per year and is based on soil types within each spray field. Large rainfall events can occur during the Atlantic hurricane season from June to November.

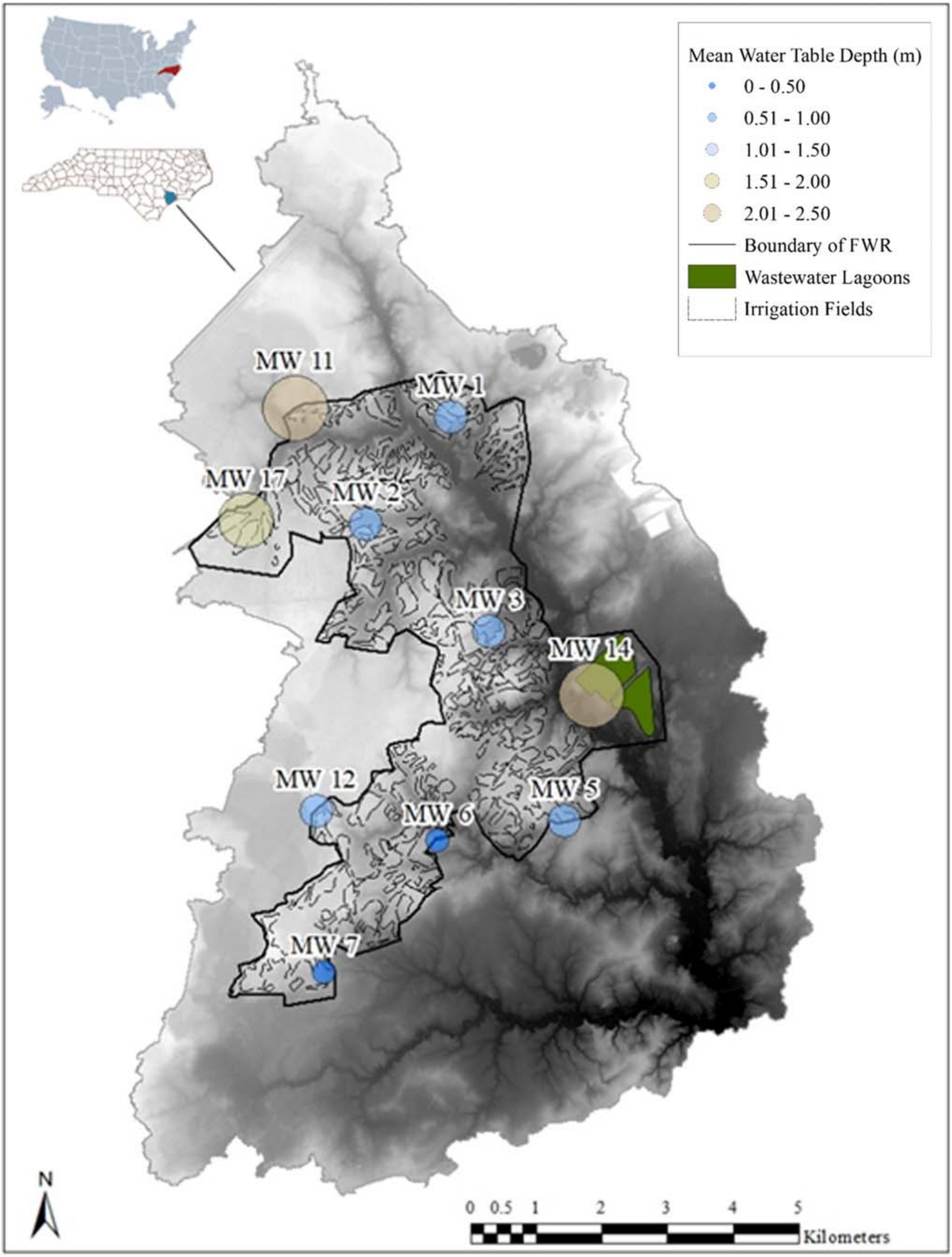

FIGURE 1 Map of 106 km^2 watershed and 30 km^2 monitoring wells (MW) with mean water table depth (metres) for 2000 to 2018. Well 14 is not in the irrigated forest system

The Forest Water Reuse (FWR) facility has records dating back to its establishment in July 1998 that detail daily wastewater irrigation at spray fields, rainfall at ten precipitation collectors across the site, and quarterly, every three months, WTD measurements for the 16 monitoring wells distributed throughout the forest, site boundaries, and near the wastewater storage lagoons. Ten monitoring wells were instrumented with pressure transducer data loggers (Hobo U20L 13-foot water level loggers) beginning in August 2017. Daily potential evapotranspiration (PET) was estimated using standard FAO Penman-Monteith equation (Allen, Pereira, Raes, & Smith, 1998) for a grass reference for the study period (1998–2018) using observations from nearby weather stations at Albert Ellis Airport (latitude 34.829, longitude −77.612) and New River MCAS (latitude 34.707, longitude −77.445). The leaf area index (LAI) is a parameter used in calculating actual ET in the model. For forested areas, LAI was estimated based on general monthly LAI for *P. taeda* plantations in the southeast (Liu et al., 2018) while monthly grass LAI in nonforested areas was based on ground layer LAI as found in Hoffmann et al. (2005).

2.2 | MIKE SHE model descriptions

ET is a major loss of water in forest ecosystems (Sun et al., 2010; Sun, Alstad, et al., 2011) and was not measured on site. Accurate measurement of ET would have been extremely costly and required the site to be instrumented from the beginning of wastewater treatment operations. Runoff (or drainage) was also not measured on site in the large ungauged watershed. Thus, we relied on a hydrological model to estimate ET so that drainage could be calculated in the absence of measurements and examined scenario conditions such as nonirrigation at the water reuse forest.

We adopted the widely used MIKE SHE, a modelling tool (Ma, He, Bian, & Sheng, 2016; MIKE, 2017) for coastal systems where the hydrology is controlled by a shallow water table (Lu et al., 2009). We chose the MIKE SHE model because the model has been used in lower coastal plains and can handle the spatial heterogeneity of topography, soil, vegetation, and precipitation inputs (Zheng, Hao, Huang, Sun, & Sun, 2020). MIKE SHE simulates the full hydrological cycle and soil water movement of both saturated and unsaturated zones in a distributed fashion with various spatial resolutions (Lu et al., 2009; Ma et al., 2016; Zheng et al., 2020). The model was parameterized according to the guidelines established in the MIKE SHE Manual (MIKE, 2017), including spatial data on topography (DEM, NC Spatial Data), soil type (Onslow County GIS Data), forested and nonforested areas (National Land Cover Database), LAI (Hoffmann et al., 2005; Liu et al., 2018), temporal data on measured daily precipitation using 10 locations within the irrigated forest, and uniformly distributed grass reference ET (ET_0) or PET (State Climate Office of North Carolina). The 2-layer water balance method, which does not represent variation in soil properties with depth, was used to model unsaturated flow. This approach is recommended for use in coastal areas with shallow water tables (MIKE, 2017). Saturated flow in MIKE SHE was modelled using the 3D finite difference method. Drainage to the streams is calculated by tracking the groundwater table levels in the soil cells adjacent to surface water in the stream network (MIKE, 2017) and is eventually routed to the outlet of the watershed.

2.3 | MIKE SHE model calibration and validation

The MIKE SHE model was calibrated using groundwater table depth, and the soil hydrologic properties parameters within the 2-layer water balance method were adjusted to achieve the best fit of measurements on WTD. Five soil groups were identified to represent the diversity of soils within the watershed (Table 1). The soil group parameters were calibrated manually using the soil hydrologic properties parameters. These parameters included the water content at saturation, water content at field capacity, water content at wilting point, and saturated hydraulic conductivity (Table 1). The parameters were adjusted one at a time, and the performance statistic was observed for every well. The wells were treated equally, and the scenario with the most wells calibrated above 0.6 *R* correlation was used for the final model run. The measured WTD was used as the calibration parameter to compare with simulated values to evaluate model performance (Table 2). Daily measured WTD were available from August 2017–September 2018 and used for model calibration, whereas quarterly (four times per year) measurements were used from 2000 to 2017. Multiple grid cell resolutions were used during calibration but did not improve results, therefore the grid cell size of 1 km^2 was used to balance computational time and data availability. The model was calibrated to recent years (2014–2018) which included daily measurements from 2017 to 2018 and was validated to prior years (2010–2013) which used quarterly measurements to assess model performance. The model performance statistics (Table 2) include mean error, mean absolute error (MAE), root-mean-square error (RMSE), standard deviation (STD), correlation coefficient (*R*), and Nash–Sutcliffe efficiency (*E*) as described in the MIKE SHE manual (MIKE, 2017). We focus on the model performance as evaluated using Pearson's *R* coefficient and RMSE. There are many parameters that are factored into the calculation of ET in the MIKE SHE model. These include LAI, root depth, ET depth, and canopy interception capacity. Because these were not measured in our study and were based on assumption from the literature, there are uncertainties in the estimation of ET in our model. Similarly, the drainage parameters like surface detention storage, depth of shallow aquifer for the 2-layer water balance, and drainage depth were not measured in our study and were based on assumptions. These uncertainties influence the calculation of drainage and must be acknowledged as a part of our model. Other critical MIKE SHE parameters include C1 to C3, which impact the calculation of ET, along with the Manning coefficient for calculating overland flow and others presented in Table 3. These values were not measured and increase the uncertainty of the model.

TABLE 1 Calibrated 2-layer unsaturated zone soil property parameters

	Parameter values		
	Minimum	Maximum	Final
Baymeade fine sand			
Water content at saturation	0.339	0.451	0.395
Water content at field capacity	0.1	0.2	0.132
W0ater content at wilting point	0.02	0.09	0.05
Saturated hydraulic conductivity (m/s)	3.2×10^{-5}	9.2×10^{-5}	8.3×10^{-5}
Foreston loamy fine sand			
Water content at saturation	0.342	0.478	0.41
Water content at field capacity	0.15	0.22	0.181
Water content at wilting point	0.06	0.15	0.09
Saturated hydraulic conductivity (m/s)	2.5×10^{-6}	4.6×10^{-5}	3.3×10^{-6}
Norfolk loamy fine sand			
Water content at saturation	0.342	0.478	0.41
Water content at field capacity	0.15	0.35	0.227
Water content at wilting point	0.1	0.2	0.15
Saturated hydraulic conductivity (m/s)	1×10^{-5}	2×10^{-5}	1.6×10^{-5}
Fine sandy loam			
Water content at saturation	0.334	0.506	0.42
Water content at field capacity	0.15	0.35	0.196
Water content at wilting point	0.1	0.25	0.107
Saturated hydraulic conductivity (m/s)	1×10^{-5}	9×10^{-5}	5.4×10^{-5}
Muckalee loam			
Water content at saturation	0.373	0.529	0.451
Water content at field capacity	0.15	0.35	0.198
Water content at wilting point	0.05	0.3	0.093
Saturated hydraulic conductivity (m/s)	3×10^{-6}	2×10^{-5}	9×10^{-6}

TABLE 2 Calibration and validation metrics

	Calibration 2014–2018						Validation 2010–2013					
Monitoring Well	ME	MAE	RMSE (m)	STD	R	E	ME	MAE	RMSE (m)	STD	R	E
17	−0.68	0.68	0.69	0.14	0.81	−9.11	−1.94	1.94	2.67	1.83	0.21	−1.07
11	−1.19	1.19	1.26	0.44	0.75	−4.18	−2.26	2.26	2.66	1.42	0.58	−2.13
1	−0.34	0.37	0.44	0.29	0.62	−0.66	−0.40	0.46	0.68	0.55	0.67	−0.06
2	−0.50	0.50	0.54	0.21	0.69	−2.48	−0.68	0.68	0.85	0.51	0.92	−0.49
3	−0.37	0.38	0.48	0.30	0.71	−0.28	−0.70	0.74	1.27	1.05	0.78	−0.19
14	−0.43	0.43	0.50	0.26	0.70	−1.72	−0.72	0.73	1.05	0.77	0.11	−0.84
12	−0.39	0.39	0.48	0.29	0.70	−0.49	−0.47	0.53	0.70	0.52	0.53	−0.36
6	−0.06	0.13	0.15	0.13	0.76	0.32	0.06	0.23	0.37	0.36	0.32	−0.13
5	−0.32	0.33	0.39	0.23	0.69	−0.51	−0.89	0.89	1.09	0.63	0.53	−1.43
7	−0.28	0.31	0.41	0.30	0.75	0.01	0.20	0.42	0.48	0.43	0.68	0.26

Abbreviations: E, Nash Sutcliffe correlation coefficient; MAE, mean absolute error; ME, mean; R, correlation coefficient; RMSE, root mean square error; STDRes, standard deviation of the residuals.

TABLE 3 Critical parameters controlling evapotranspiration, overland flow, unsaturated flow, and saturated flow

Water movement	Parameter	Final value	Unit
Evapotranspiration	C1	0.3	
	C2	0.2	
	C3	20	mm/day
	C_{int}	0.225	mm
Overland flow	Surface flow Manning coefficient (m)	10	$m^{(1/3)}$ /s
	Detention storage	10	mm
	Initial water depth	0	mm
Unsaturated flow	ET surface depth	0.5	m
Saturated flow	Horizontal hydraulic conductivity	0.0001	m/s
	Vertical hydraulic conductivity	1 e-05	m/s
	Storage coefficient	0.0001	1/m

Abbreviation: ET, evapotranspiration.

2.4 | Comparison of water balances

The water balance was estimated under both nonirrigated and irrigated conditions to evaluate the impacts of irrigation management on ET, WTD, and drainage. The equation used for annual and monthly water balances in this study was

$$Q = P + I - ET - \Delta S, \tag{1}$$

where Q is drainage, P is precipitation, I is irrigation, ET is evapotranspiration, and ΔS is change in groundwater storage. Water balances were calculated for the 10 MIKE SHE modelling cells containing monitoring wells, and the averages of these wells were assumed to represent the site as a whole. The water balance equation does not include deep seepage as a drainage factor because it is assumed to be negligible (Harder et al., 2007). Daily precipitation and irrigation rates are provided by the FWR. Precipitation and irrigation data were spatially distributed in the MIKE SHE model, and the measured values were also used separately in the calculation of the water balance. Shapefile polygons outlining the irrigation fields were used to spatially distribute the irrigation data. The precipitation data was spatially distributed within the watershed using 10 shapefile polygons. The precipitation data from the 10 precipitation collectors on site was matched with the polygon in its respective location. ET is the actual ET output from MIKE SHE, modelled on a daily time step. Change in groundwater storage is calculated as the change in head of the water table multiplied by specific yield. In this study, the change in head was the depth to the phreatic surface output from the MIKE SHE model and the specific yield was the average value (0.225) manually calibrated in the MIKE SHE model soil layers. The calibrated and validated MIKE SHE model was used to simulate irrigated ET and WTD with daily irrigation as a model input, after which irrigation was removed from the MIKE SHE model inputs and a second model simulated for nonirrigated ET and WTD. The simulated daily results for ET and ΔS from the two models were used to create two water balances from the water balance equation to assess the long-term impact of irrigation on ET, WTD, and drainage in the forested land treatment system. Equation 1 was used with MIKE SHE outputs from irrigated model. For the nonirrigated model, I in Equation 1 was set to zero. Annual and monthly water balances were created for each of the 10 groundwater wells that provided WTD data, after which the average of the water balances (annual and monthly) was used to represent the site as a whole.

3 | RESULTS

3.1 | Climate, irrigation, and WTD dynamics

Daily precipitation and irrigation from 2000 to 2018 are shown in Figure 2a. Figure 2a shows rainfall from two major events, most notably, Hurricane Florence in September 2018. The NOAA 30-year annual normal (1981–2010) at Jacksonville Emergency Operations Center (latitude 34.7965, longitude −77.401) 12 km from the study site is 1,379 mm. The mean annual rainfall measured on site during the study period (1998–2018) is 1,458 mm (STD = 308 mm). For this study, "above average" years are defined as having one STD above the mean annual rainfall (1,458 mm + 308 mm = 1,766 mm) in addition to having above average mean annual irrigation (>826 mm). Below average years are defined as years with one

STD below mean annual rainfall (1,458 mm—308 mm = 1,151 mm) in addition to having below average irrigation (<826 mm). Mean rainfall trends higher from May to September and lower from October to April. Variability for precipitation, irrigation, and PET are shown for annual and seasonal measurements in Figures S2 and S3, respectively.

The FWR facility irrigated approximately 826 (±205)-mm treated water per year with annual mean irrigation exceeding 826 mm for 2003, 2004, 2013, and 2015–2018 (Figure 1a). The average annual irrigation is 58% of the NOAA 30-year mean and 55% of the mean rainfall for the study period. The mean monthly PET exceeds mean monthly irrigation from March to September, with exceedances ranging from 22 to 84 mm, whereas monthly irrigation depths are greater from May until October (83–90 mm) and are lower from November through April (57–73 mm). Annual PET ranges from 27 to 187 mm less than the NOAA 30-year annual normal for rainfall but exceeds annual rainfall in 2001, 2002, 2007, 2011, and 2013. Although the PET was greater than precipitation at times, combined precipitation plus irrigation was 393 to 2,058 mm greater than PET. On average, months with higher PET than precipitation occur from March to June, although the total monthly input to the site is always greater than the PET because of irrigation.

WTD for the site monitoring wells showed seasonal variation from 2000 to 2018 (Figure 2a) although levels among wells were not uniform (Figure 2b). WTD varied across the watershed and greater depths were observed for periods of drought and during high ET summer months (Figure 2b). Recent daily WTD measurements (Figure 2c;

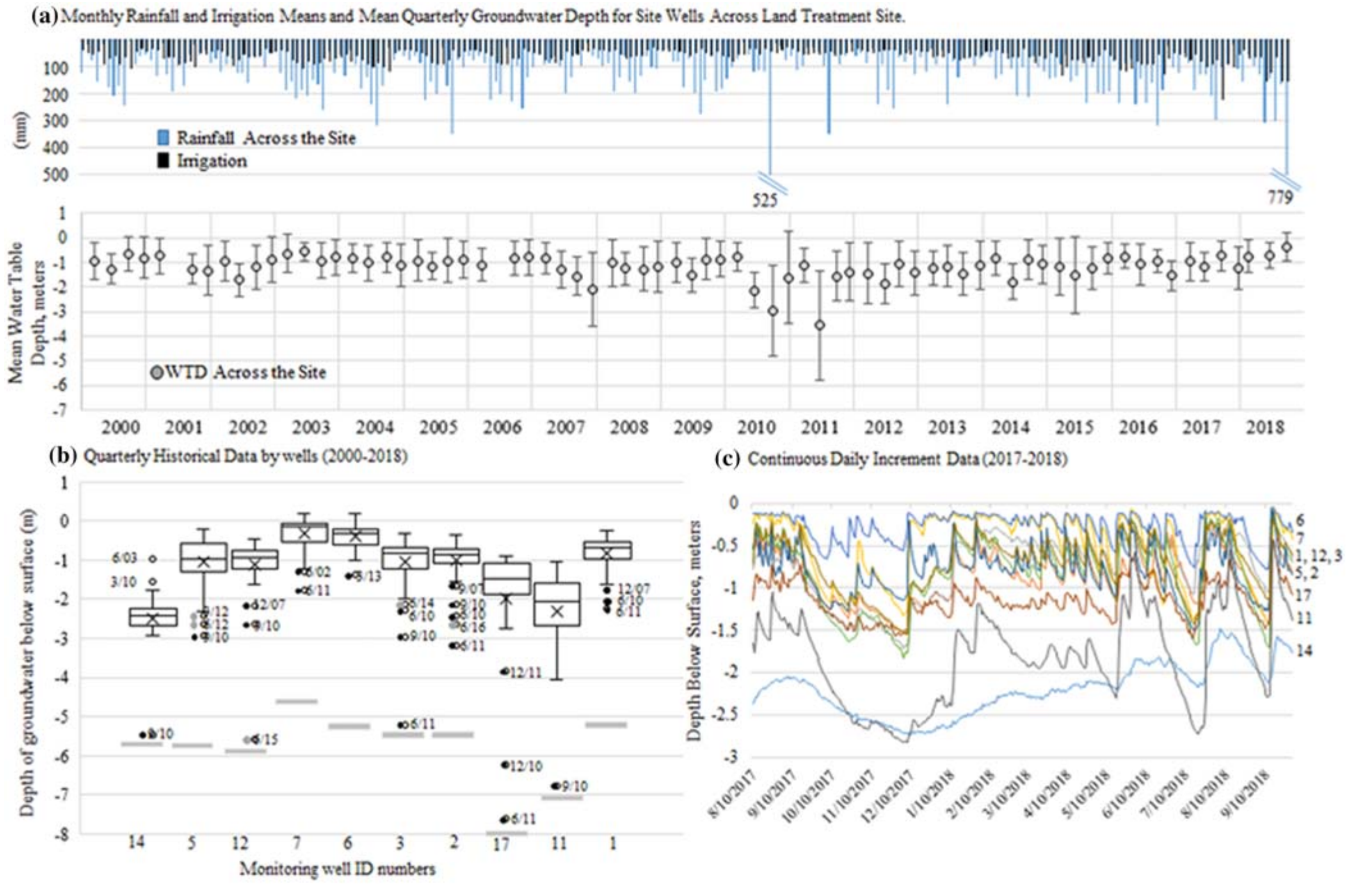

FIGURE 2 (a) Mean depth of groundwater below surface (m) based on quarterly measurements from 2000 to 2018 and daily measurements from 2017 to 2018. Box plot of groundwater depth below surface (m) for (b) quarterly measurements for individual wells. Total depths of each well are noted with a grey bar and month/year are provided for outlier groundwater depths for drought (black circles) and high evapotranspiration (grey circles). (c) daily average of 30-min measurements by data loggers for monitoring wells on site. Well 14 is not irrigated

August 2017–September 2018) fell within the range of historic quarterly levels (Mar 2000–June 2017), between 0 and 3 m below the ground surface. Figure 1b shows mean WTD derived from quarterly measurements from 2000 to 2018 wherein depth trends upward toward the ground surface from the site perimeter to wells located in the centre of the site. Mean depths are similar for wells except Well 14 and Wells 6, 7, 11, and 17. The WTD in Well 14 likely represents the height of surface water in the main channel, due to its location close to the main stream channel, on a nonirrigated, nonforested slope with compacted soil. The relative mean depths of Wells 6, 7, 11, and 17 reflect the topographical elevation of their respective locations. Wells 11 and 17 are adjacent to agricultural fields and may be influenced by groundwater extraction during water-short periods when crops are irrigated.

3.2 | MIKE SHE model performance

The model was calibrated with recent data (2014–2018) that included measurements on daily time steps. Correlation coefficients ranged from 0.62 to 0.81 during the calibration period (Table 2). The RMSE values ranged from 0.15 to 1.26 m, with the highest error seen in Well 11. *E* values ranged from −9.11 to 0.32. The MAE values ranged from 0.13 to 1.19 m (Table 2). Well 6 had the lowest RMSE for the calibration period, whereas Well 11 had the highest (Figure 3, Table 1, Figure S4). Simulated WTDs for Wells 11 and 17

were consistently higher than the measured data. The model was validated with 2010–2013 quarterly measurement data. R correlations ranged from 0.21 to 0.92, RMSE values ranged from 0.37 to 2.67, E values ranged from −2.13 to 0.26, and MAE values ranged from 0.23 to 2.26 m (Table 2).

The simulated water table in irrigated and nonirrigated conditions was often closer to the soil surface than measured WTD in the monitoring wells (Figure 3). The model performed better in wet conditions, for example, the period of continuous precipitation from 8/3/17–8/15/17 and 7/20/18–8/12/18. In these times, the simulated water table is more similar to measured WTD than during dry conditions, where the simulated WTD does not drop as rapidly as measured WTD. In fact, with the exception of nonirrigated Well 14, none of the simulated WTDs increased below a depth of 1 m even

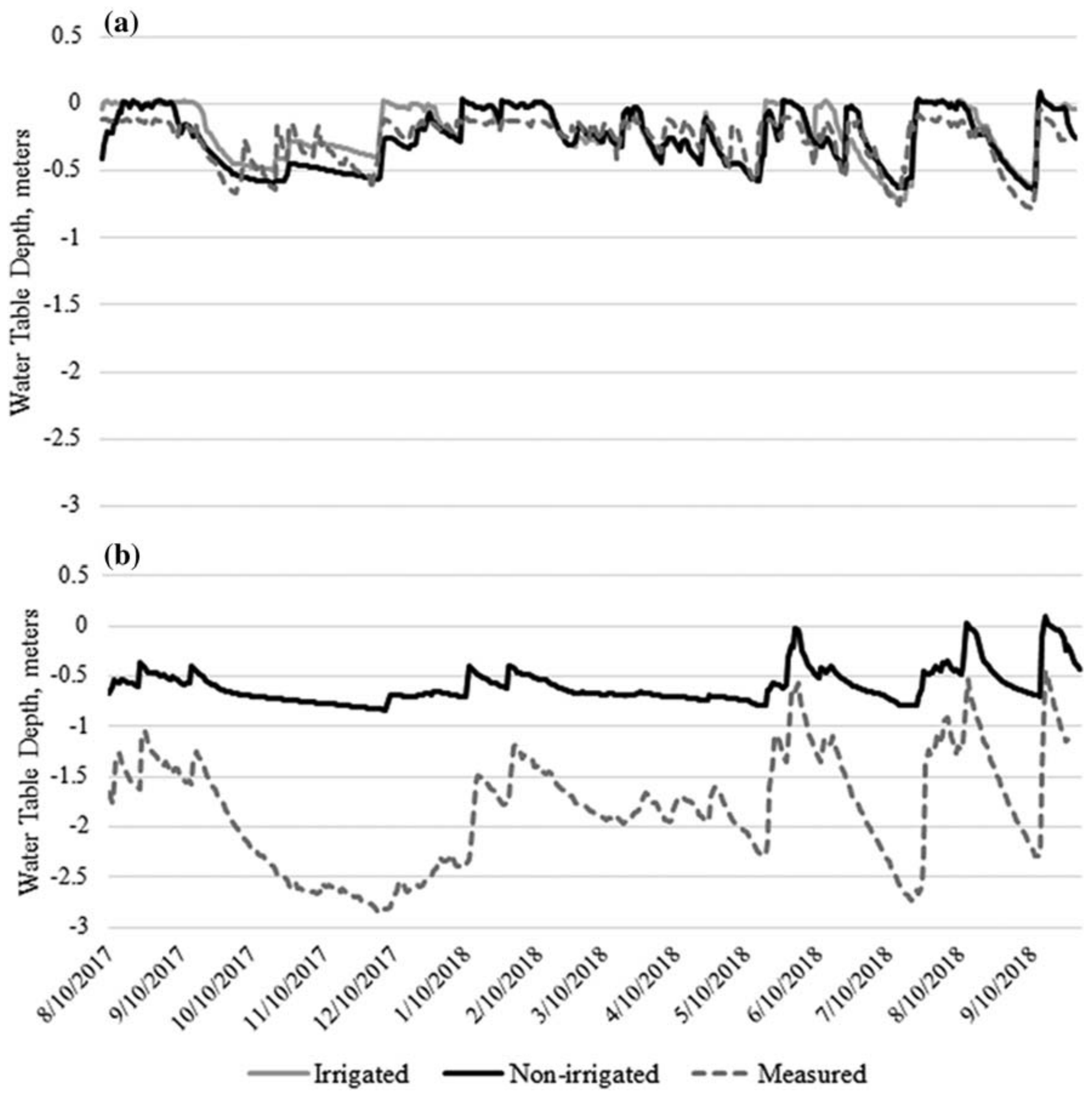

FIGURE 3 Irrigation and nonirrigated daily simulated water table depth, along with daily measured water table depth for (a) Well 6 and (b) Well 11

though measured WTDs across the site did increase below a depth of 1 m in the fall 2017 season, except for Well 6 (Figure 3).

3.3 | Change in simulated water balance due to irrigation

The annual total inputs (precipitation plus irrigation) exceeded annual precipitation by 45% to 86% and irrigation increased the total input to the system by 662 mm to 1227 mm (Figure 4). The difference in groundwater storage due to irrigation ranged from −11 to 14 mm

annually. The annual change in groundwater storage remained close to zero in both irrigated and nonirrigated conditions during all dry, average, and wet years (Figures 4 and 5). The plot differences in change in groundwater storage are difficult to decipher because both were close to zero and overlapping. Irrigation had little impact on annual patterns of groundwater storage, and the average water table across the site was slightly higher under irrigated conditions (6.2 ± 4.1 cm) than nonirrigated conditions. Although changes in groundwater storage (Figure 4) across the site were small over time, the water table did increase in some areas due to irrigation. Higher water table levels were observed for wells surrounded by irrigation (Well 3) or in low topographic areas (Well 6). In 2010, four new irrigation fields were installed near Wells 2 and 17, where water table levels varied little between irrigation and nonirrigated conditions. After irrigation began in the new fields, water table levels were higher for Wells 2 and 17 under irrigated conditions (Figure S5). Irrigation did influence monthly water table dynamics (Figure 5), and differences between monthly WTDs for irrigated and nonirrigated areas were more pronounced in years with lower annual precipitation.

Modelled ET was independent of annual precipitation patterns but dependent on soil water availability and climate conditions

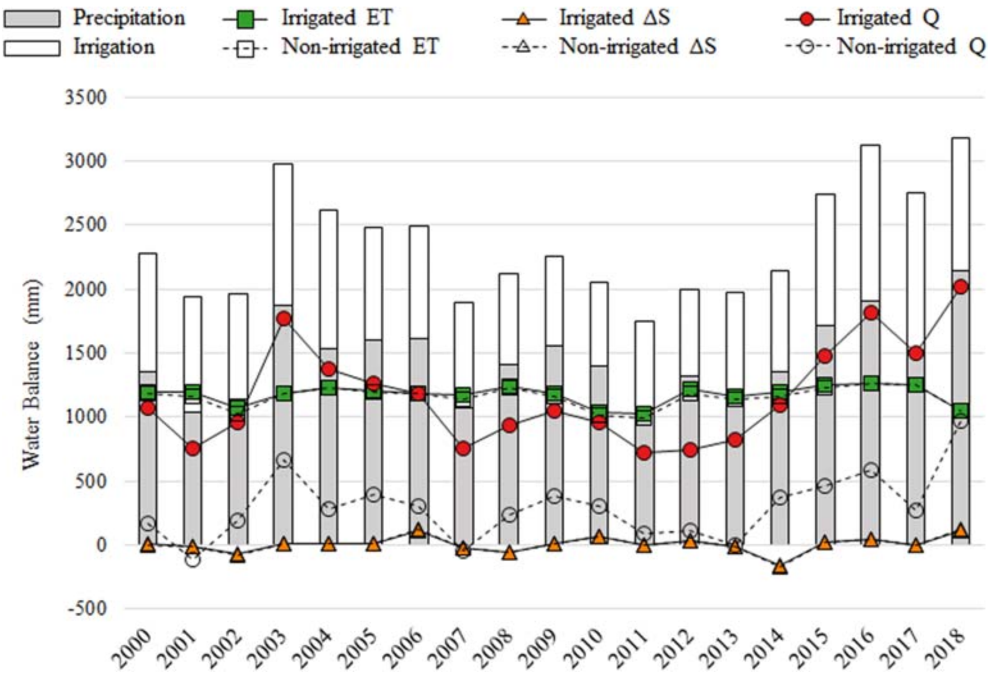

FIGURE 4 Mean annual water balance under irrigated and nonirrigated conditions across the site. It is difficult to see the overlapping values close to zero in the change in groundwater storage for irrigated and nonirrigated conditions

(i.e., PET). The stability of ET is consistent in both irrigated and nonirrigation conditions (Figure 4) when considering the site as a whole, with slight increases under the irrigated conditions. Irrigation increased overall ET by an average of 18 mm per year (Table 4). The increases in ET ranged from 0.25 to 40 mm annually. Annual ET to total input (ET/P + I) ratios ranged from 0.40 to 0.62 for irrigated annual ET (Table 4); ET to precipitation (ET/P) ratios ranged from 0.63 to 1.12 for nonirrigated annual ET (Table 4). The average ET across the site followed expected seasonal trends (Figure 5) with slight reduction of ET during dry summer months when water availability was limited. The monthly ET/P ratio was higher under nonirrigated conditions both in dry (0.56–2.12) and wet years (0.25–1.57) than the ET/(P + I) ratio in dry (0.37–1) and wet years (0.20–0.64).

Calculated drainage included runoff and lateral flow of groundwater, which ultimately discharges to surface water. Variation in the calculated drainage was influenced primarily by precipitation. Calculated annual drainage and monthly drainage in wet years followed trends in precipitation (Figures 4 and 5), whereas monthly drainage in dry years diverges from precipitation patterns when influenced by ET (Figure 5). Simulated ET had a greater influence on the amount of monthly drainage in dry years through its consistent use of available water. The annual increase in calculated drainage due to irrigation was equal to 93% to 100% of the annual irrigation volume (Figure S6). Irrigation increased monthly drainage in dry years by 6 to 105 mm and in wet years by 59 mm to 117 mm (Figure 5). The increase in drainage relative to irrigation volumes ranged from 88% to 119% of monthly irrigation, whereas in dry years, the increase in monthly drainage relative to irrigation varied from 12% to 158%. Not only were the monthly precipitation patterns different, but the irrigation volumes also differed due to facility management needs to irrigate in order to avoid excessive wastewater volumes in reservoirs (Figure 5). These findings suggest that relative impact on drainage is caused by irrigation because ET and groundwater storage changed very little at the annual scale in both dry and wet years. The MIKE SHE model contains uncertainties due to the poor calibration results which must be considered when evaluating the change in drainage due to irrigation. The model did not capture large WTD decreases during dry years; hence, drainage is likely overestimated during these times.

4 | DISCUSSION

4.1 | Wastewater land treatment forests and excess water

To our knowledge, this study is the first effort to evaluate municipal wastewater irrigation impacts on the water balance of a temperate, coastal forest system using a hydrologic simulation model. Non-forested areas have been evaluated for similar drainage design dynamics and parameterization (Öztekin et al., 1999), but this study specifically focused on a forest land treatment system of primarily *P. taeda*, or loblolly pine. A unique aspect of this study is the duration of irrigation, 20 years, the area of forest irrigation, 890 hectares, and the frequency of irrigation which was weekly throughout the year. Evaluations of other irrigated, coastal forest systems are limited for the southeastern United States of America and utilized irrigation only during the growing season.

The use of forests to treat municipal wastewater was researched and implemented in the United States from the 1950s–1990s although wastewater land application has existed for several centuries (Nichols, 2016). Research from the mid-1900s demonstrated that land treatment forests removed regulated chemicals and nutrients to meet permitted requirements (Pennypacker, Sopper, & Kardos, 1967) and increased nutrient levels in trees and soils (Stewart, Hopmans, Flinn, &

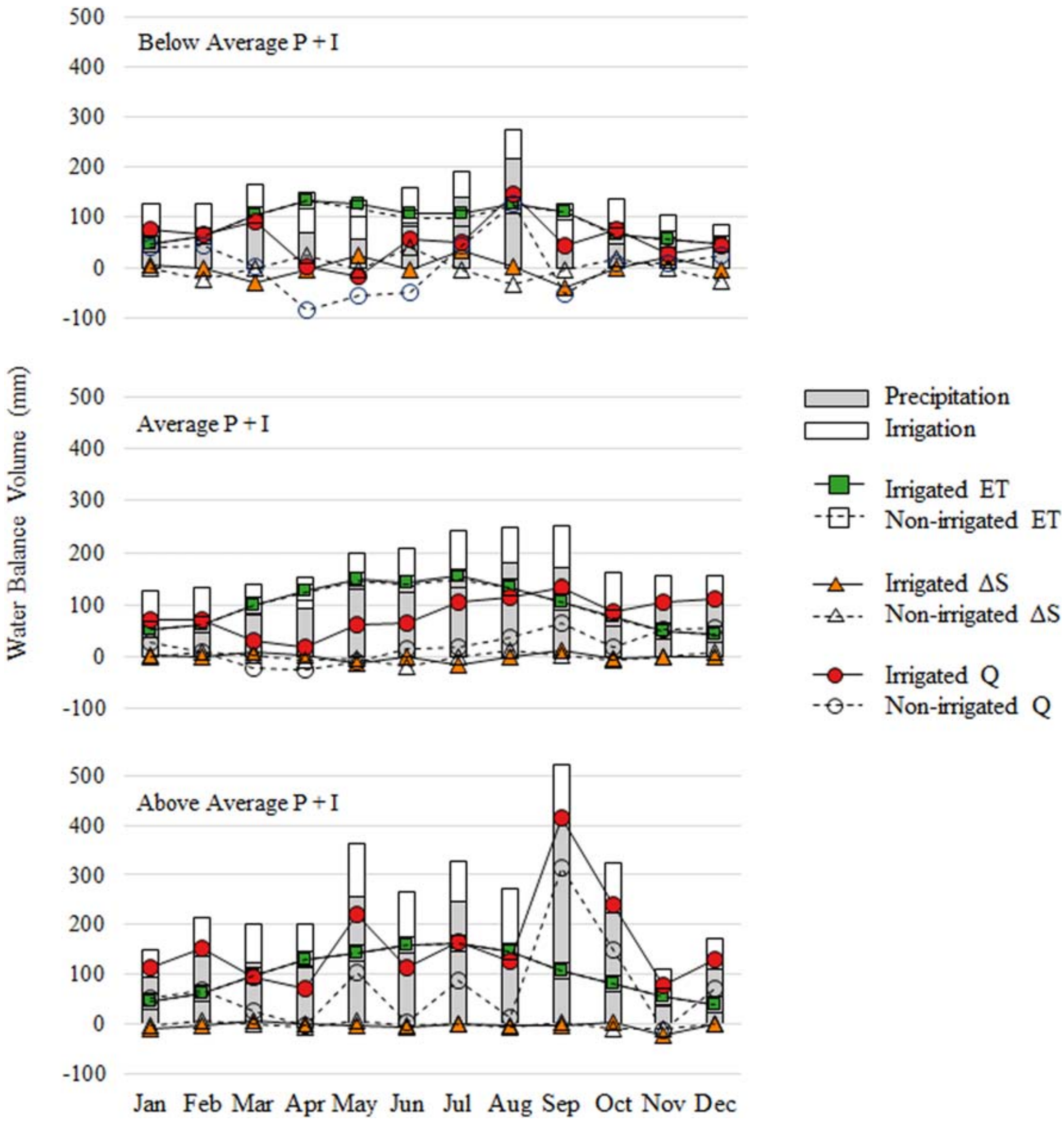

FIGURE 5 Monthly water balance for below average (one standard deviation below mean precipitation and below mean irrigation, 2002, 2007, 2011), average (2000, 2001, 2004–2006, 2008–2010, 2012–2015, 2017), and above average years (one standard deviation above the mean precipitation and above mean irrigation, 2003, 2016, 2018)

Hillman, 1990). A recent study at our study site used hydrometric data and stable isotopes to model wastewater fractions in groundwater and surface waters across the watershed during variable periods of drought and storms (Birch et al., 2016). The isotope study observed two pulses of pharmaceutical compounds, common to wastewater, in surface waters leaving the site that coincided with two major discharge peaks generated by a major tropical storm. The isotope study findings support current water balance results that drainage is driven by precipitation events. In a broader context, Abrahamson, Dougherty, and Zarnoch (1998) observed that irrigation of *P. taeda* had limited impact on forest water use but did increase water drainage below 1 m of the soil surface. These results agree with the water balance model that irrigation did not impact forest ET, except in drought periods, and that rainfall impacted storage and drainage at the watershed scale, more so than irrigation.

A primary interest of the water balance model was to understand how 20 years of constant and, at times, chronic irrigation influenced water balance dynamics, ecosystem services, and forest productivity. As previously noted, prior studies at the site have addressed water quality and related aspects of water regulation for ecosystem services (Hedgespeth et al., 2019; McEachran et al., 2018; McEachran, Shea, & Nichols, 2017) although water balance quantification is a necessary element toward a comprehensive assessment of forest land treatment practices. Water balance results are insightful to forest productivity in response to chronic and extreme hydraulic loading and supportive of results from a recent forest inventory of *P. taeda* productivity at the

TABLE 4 Mean, standard deviation, and coefficient of variance for annual precipitation, potential evapotranspiration, simulated evapotranspiration, change in groundwater storage, and drainage for the study period (2000–2018) along with ET/PET, ET/P, and Q/P ratios

Nonirrigated								
	P (mm)	**PET (mm)**	**ET (mm)**	**ΔS (mm)**	**Q (mm)**	**ET/PET**	**ET/P**	**Q/P**
Mean	1,458.50	1,270.91	1,156.75	5.85	295.90	0.91	0.82	0.18
STD	307.20	58.70	80.45	66.76	258.60	0.08	0.15	0.14
CV (%)	0.21	0.05	0.07	11.40	0.87	0.08	0.19	0.77
Irrigated								
	P + I (mm)	**PET (mm)**	**ET (mm)**	**ΔS (mm)**	**Q (mm)**	**ET/PET**	**ET/P + I**	**Q/P + I**
Mean	2,353.21	1,270.91	1,175.16	5.81	1172.24	0.93	0.51	0.49
STD	438.87	58.70	74.54	61.56	392.25	0.07	0.08	0.07
CV (%)	0.19	0.05	0.06	10.60	0.33	0.07	0.16	0.15

Abbreviations: ET, evapotranspiration; I, irrigation; P, precipitation; PET, potential evapotranspiration; Q, drainage; ΔS, change in groundwater storage.

site. The water balance model showed that irrigation did not increase ET substantially more than ET for nonirrigated conditions except during times of drought. Hence, even though the irrigated forest receives weekly hydraulic loading of wastewater and precipitation throughout the year, ET did not change in relation to ET for nonirrigated conditions. A recently completed 2019 forest inventory found that irrigation did not impact *P. taeda* productivity across the five major soil types of the forest land treatment site. Mean tree height, stem diameter, stem volume, understory biomass, and mean annual ring widths were not statistically different (ANOVA GLM, $p < 0.05$) between irrigated and nonirrigated *P. taeda* for three age classes. The main driver for significant growth differences was soil type (Goeke Dee, 2019). Previous studies suggested that ET and productivity of forest ecosystems was closely coupled (Sun, Caldwell, et al., 2011). Therefore, if irrigation impacted ET significantly, one would anticipate significant changes to *P. taeda* productivity, particularly for trees receiving irrigation during early stages of growth. Water balance and forest productivity metrics suggest that irrigation, in normal and above-normal rainfall years, does not impact *P. taeda* growth. One would anticipate growth differences between irrigated and nonirrigated *P. taeda* for below-normal rainfall based on the water balance model; however, significant differences in mean annual ring widths were not observed (ANOVA GLM, $p < 0.05$) when stratified by rainfall for drought and nondrought conditions (Goeke Dee, 2019). Drought conditions were less frequent and of shorter duration than normal and above-normal rainfall at the site; perhaps greater frequency and drought duration are necessary to observe increases of ET and productivity for irrigated *P. taeda*.

Extreme precipitation is expected to increase for the southeastern United States, particularly during warm months (Kunkel et al., 2020; Webster et al., 2014). Studying water balance dynamics for land treatment forests provides opportunities to gain insights for how coastal temperate forests may respond to greater rainfall regimes (Dai et al., 2011; Lu et al., 2009). As shown in this study, forest land treatment sites provide outlier conditions for hydraulic loading due to their inherent design and mandatory need to irrigate under above-normal rainfall conditions. At our study site, continuous emergency spraying has been necessary to maintain reservoir integrity due to above-normal rainfall and extreme storm events since 2015. Results of the water balance model and forest inventory suggest that the irrigation volume is not exceeding the drainage capacity of the site to the point of negatively impacting the ET and *P. taeda* productivity. As coastal communities consider infrastructure to manage future extreme storms and excess water, forest land treatment systems provide relevant context to utility and green infrastructure design.

4.2 | Irrigation effect to ET, groundwater, and drainage in the coastal plain

The effect of irrigation on the ET and groundwater storage was negligible, but irrigation increased the volume of drainage. There is a need to understand the impact that wastewater treatment has on the forest system hydrology, specifically how irrigation is distributed within the water balance. There is a need to understand where within the water balance the irrigation is being distributed in order to assess the impact that wastewater treatment has on the forest system. Irrigation reduced the impact simulated ET had on WTD by meeting ET demand. This impact was apparent when compared with the nonirrigated water balance, where high ET rates had a great influence on WTD when precipitation was low. In general, coastal plain forest systems are energy-limited and have stable ET due to high water availability (Amatya & Tian, 2016; Liu et al., 2018; Sun et al., 2010). Drought has limited impact on the ET of young and mature *P. taeda* in the coastal plain (Sun et al., 2010) due to shallow groundwater availability to trees. In this study, simulated ET rates were stable and interactions between ET, precipitation, and infiltration within the model determined simulated water table dynamics and, subsequently, calculated drainage. The addition of irrigation in this water-abundant system did not impede the ability of the forested land treatment site to drain water out of its boundaries. This can be seen in the change in storage remaining close to zero. Instead, this system remained a resilient coastal forest, with most of the added input to the system leaving through drainage. Excess drainage also suggests that the land treatment site can manage large rainfall events at current irrigation volumes.

Callahan, Vulava, Passarello, and Garrett (2012) observed that annual change in storage appeared negligible for coastal plain surficial aquifers where recharge to shallow aquifers nearly equals baseflow to surface waters. Irrigated plots of *P. taeda* showed less change in storage than nonirrigated plots for sandy soils in North Carolina (Abrahamson et al., 1998). This connectivity between groundwater and surface water keeps WTDs consistent with shallow water table characteristics of coastal forests and support forest productivity, particularly for *P. taeda* productivity. However, the function of the land treatment of wastewater depends on the slow rate of movement of wastewater through the soil profile and surficial aquifer. Increases in the water table due to irrigation could reduce the function of the land treatment system. The forest land treatment system is not permitted to allow surface ponding of irrigation water, and continuous water table monitoring has demonstrated rapid water table decline even after extreme storm water events due to drainage. More convincing is the observation that irrigated forest productivity is not significantly different than nonirrigated forest productivity regardless of stand age (Goeke Dee, 2019).

This study observed that wastewater irrigation influence is localized to the areas with uniform irrigation sprinklers (Figure S7). In these areas, the water table was higher, but, for most wells, the WTD remained unchanged under irrigation (Figure S5). The few differences between annual change in storage for irrigated and nonirrigation conditions indicate that the watershed is not becoming saturated over time with irrigation. These findings are similar to those of (Amatya, Skaggs, & Gregory, 1996), where controlled drainage had little impact on annual groundwater storage or ET. Within the high water table areas, there was increased responsiveness of the WTD to storm events. La Torre Torres, Amatya, Sun, and Callahan (2011) observed that rainfall runoff ratios corresponded to precipitation amounts when the antecedent moisture conditions were high. This study and the work by Amatya et al. (1996) indicate that irrigation and high water tables could increase the amount of runoff during precipitation events, as seen in our model results, though potential errors from our assumption that specific yield is constant with depth must be considered. Uniform specific yield leads to an overestimation of drainage under extremely wet, shallow WTD conditions which is observed in water balance results where change in groundwater storage is negative. However, the recent completed forest inventory of this site supports this finding that irrigated areas with higher water tables do not have lower tree growth due to drainage (Goeke Dee, 2019).

ET is calculated in MIKE SHE based on root depth and water availability in the unsaturated and saturated zones. ET differences between irrigated and nonirrigated conditions were minimal, indicating that the system is energy, not water, limited. This observation is similar to that of (Amatya et al., 1996), who found that their coastal North Carolina forest system was limited by reference PET, not soil moisture. Our study simulated annual ET/P ratios ranging from 0.63 (*wet year*) to 1.12 (*dry year*) in nonirrigated forest conditions (Table 4), similar to the range of ratios in wet and dry years (*0.55* to *1.07*) found in the literature for coastal forests (Amatya & Tian, 2016). The high ratios of ET to potential ET in our study were similar to other studies (Amatya & Tian, 2016) and ranged from 0.73 to 1.0 (Table 4) with the lowest ratio occurring in the drought year 2011. The ET simulated in this study (993 to 1,267 mm) was on the high end of annual ET in the coastal plain (Liu et al., 2018; Sun et al., 2010). Long-term mean annual potential ET in the coastal plain is 1,140 mm (Amatya & Tian, 2016), which is around 100 mm less than the mean potential ET (1,271 mm) used for both irrigated and nonirrigated conditions in this study.

Results of the MIKE SHE model suggest that monthly ET and change in groundwater storage in an irrigated forest are similar to nonirrigated forests under average rainfall and above-average rainfall years (Figure 5). ET and groundwater storage variability are greater in dry years wherein the irrigated forest responds to drought with increased groundwater consumption and decreased drainage volume (Figure 5). The increase in plant available water from irrigation increases the resilience of the forest to drought. The forest manages dry weather extremes by utilizing available water provisioned by elevated water tables and by ET reduction if available water is limited (Figure 5). Consequently, irrigated forest has high ET and low changes in groundwater storage during wet periods on a monthly basis. Simulated wet periods have high drainage but stable groundwater storage because the site is well-drained (Figure 5). These dynamics show that the saturated forest land treatment system manages intense rainfall events primarily though drainage. During wet periods when ET is equal to PET, the forest has reached its maximum capacity of ET and thus drainage is driven by the amount of irrigation applied to the site.

The stability of ET and groundwater storage in coastal plain systems makes drainage the most variable component of the water balance for irrigated forest systems. Precipitation drives drainage and potential outflow in forested coastal plains (Harder et al., 2007). In this study, both irrigated and nonirrigated drainage followed the pattern of rainfall volume on an annual scale (Figure 4); however, the pattern of drainage was more complex on a monthly scale (Figure 5). Monthly magnitude and timing of drainage followed rainfall patterns in wet years (Figure 5), but drainage patterns were more varied during average and dry years (Figure 5) due to interactions between rainfall, irrigation, ET, and changes in groundwater storage. The responsiveness of drainage means that the site is well-drained, and groundwater and surface water are highly connected in this system. High precipitation events cause increased runoff in this system and downstream discharge. Though small, increases in ET due to available water decrease the amount of drainage leaving the downgradient system. The absence of a stream gage in the main channel of this watershed adds to the uncertainty of the discharge estimates, but the high variation in discharge corresponding to precipitation events was observed visually during our work at the site and has been reported in the literature (Epps et al., 2013; La Torre Torres et al., 2011).

4.3 | Study limitations

This study represents the first study to assess the hydrologic impacts of applying waste water in forests. As in any modelling study, uncertainty exists in field data for model parameterization and calibration, and in the model structure and algorithms that describe the hydrological processes. Parameterization of MIKE SHE is difficult, especially for the complex soil properties that control WTDs and lateral groundwater flow and the dominant flow path in the study region. The general soil property information reported by soil databases may not be sufficient to characterize the top 10-m soils. Limited by long-term monitoring instrumentation to measure ET and runoff for a large heterogeneous watershed, this study relied on a simulation model that was calibrated and validated with groundwater measurements. When compared with the literature, estimated ET by the MIKE SHE values were reasonable for mature pine plantations; however, errors may exist for estimated differences between irrigated and nonirrigated stands given that the MIKE SHE was not designed to simulate ground irrigation. The processes of soil/litter evaporation under irrigated conditions are not modelled in MIKE SHE, which could lead to overestimated ET from canopy-intercepted input. Model validation is limited to water table level only, and in some cases such as extreme drought, the model simulations did not match elevations and overestimated WTD. We suspect that the extreme drawdown of WTD reflects groundwater extraction for irrigation on agricultural operations adjacent to the land treatment site. Nevertheless, we believe that the ET estimates were reasonable when compared with the literature for southeastern coastal plain forests. In addition, the historic measurements of the WTDs were not continuous, preventing validation of extreme dry events. Future studies also should measure streamflow using standard gaging flumes at selected subwatersheds to constrain model calibration for aquifer and soil parameters.

The MIKE SHE is best suited to simulate watershed water cycle in a landscape with a shallow groundwater table such as our study site. However, there are limitations. First, the procedure to model ET as a function of PET, WTD, and rooting depth may not be adequate for saturated conditions where atmospheric demand may play a more dominated role in tree transpiration. Second, assumptions for depth-invariant specific yield and hydraulic conductivity could create potentially large errors in groundwater flow and estimated drainage. Finally, full coupling of surface runoff in channels and groundwater flow is needed.

Some modelling issues are reflected in the fact that the Nash–Sutcliffe efficiency (*E*) values for evaluating WTD simulations were low. A few reasons may contribute to this result. First, although *E* has been used in the literature for quantifying the match of modelled streamflow measurements, its use to assess model performance for modelling groundwater table depth has only shown limited success. For example, Lu et al. (2009) used groundwater measurements as their calibration parameter and produced negative *E* values similar to our study. Second, *E* has performed better in normal conditions than drought conditions when the water table falls below well depths. The MIKE SHE model over predicted water level for extreme dry conditions. *E* values in this study also reflect difficulties in model calibration across multiple wells and a large spatial area. Lu et al. (2009) calibrated their model to nine individual wells and noted that calibration to one particular well can improve that well's performance while simultaneously deteriorating a different well's performance. Finally, the water table at our study site was highly variable which resulted in many negative *E* values. The model performance on a daily time step was poor, as is expected with smaller time steps, versus monthly or annual (Engel, Storm, White, Arnold, & Arabi, 2007). The 2011 drought had low WTD outliers for nearly every well. The model performed poorly during the dry years of the validation period. Poor model performance could be due to errors in ET estimation or due to lateral drainage causing unexplained dramatic decreases in groundwater levels during the drought. Water table decreases were up to 8 m, which are much deeper than expected in the NC coastal plain under natural conditions. Additionally, WTD returned to normal depths rapidly in the fall and winter, but the model did not capture that recovery response. As discussed before, Well 14 is uniquely located on the site. Consequently, in the validation period, Well 14 has a much lower *R* correlation due to differences between modelled and measured WTD in response to rainfall events.

5 | CONCLUSIONS

We applied the process-based MIKE SHE model to examine the effects of wastewater irrigation on annual and monthly water balances in a temperate, wet, coastal forest dominated by loblolly pine. Although there is uncertainty in our modelling approach, our study suggests that irrigation slightly increased ET and elevated the water table and largely increased drainage during 20 years of wastewater treatment at a forest land treatment site. The model validation was limited due to the number of wells, their large spatial distribution, and the absence of river discharge data. The model results should be considered as conservative estimates of impacts of irrigation on drainage. Our study suggests that irrigation only slightly increased forest ET and elevated the water table, but drainage increased dramatically due to wastewater addition.

The impacts of irrigation on ET and WTD were greatest in areas surrounded by irrigation, with minor impacts to the forest at the perimeter of irrigation fields. Irrigation slightly increased ET by forests across the site, particularly in dry years. The fluctuation of the water table varied more with decreasing annual precipitation; few differences were observed between irrigated and nonirrigated conditions in wet and average years. As a result of small changes in ET as a whole (0–48 mm), the increase in annual drainage due to irrigation was close to the volume of irrigation applied. Irrigation was a consistent input to the forest; thus, precipitation was the key driver of the variability of drainage.

Coastal forests are groundwater dependent systems that can be vulnerable to drought events in dry years. Wastewater irrigation reduced this vulnerability by increasing water table elevation and water availability for ET. In wet years, ET and groundwater levels remained constant while drainage increased in response to rainfall and irrigation. Improved understanding of water balance dynamics can help improve wastewater treatment efficiency at the forested land treatment site. Our simulation model suggests that seasonal irrigation volumes can begin earlier than currently practiced to take advantage of higher ET rates in early spring. This study highlights a research opportunity to improve our understanding of forest resiliency to excess water and the role of forests in coastal hydrologic systems.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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