

Decadal shifts in the population growth, regeneration, and health of *Taxodium distichum* in swamps of the Cache River Watershed, Illinois

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ABSTRACT

Population projection models based on long-term trends in regeneration and tree survival can be used to predict the future stability of swamp forest species using water management. Population growth and regeneration of a foundational tree species in North American cypress swamps (*Taxodium distichum*) were compared in the Cache River watershed of southern Illinois USA over several decades. This study examined *T. distichum* population growth in several regional swamps within the Cache River watershed along a moisture gradient to model growth, stability, or decline based on data of life history stage transitions, using data from the following time periods: the 1990s, 2002–2011, and 2012–2022. Using data from the 1990s, *T. distichum* populations in Crawford Tract in Buttonland Swamp were projected to increase over time because of the high frequency of juvenile stages at elevations with growing season drawdown and winter flooding. The situation in Crawford Tract had changed by 2019–2021 because *T. distichum* seedlings had not transitioned (i.e., saplings and young trees were absent) although no old-growth trees had died during or after the 1990s. In other regional swamps, projection models were constructed over two decades (2006–2021) along a dry-to-wet flood gradient including Section 8 Woods, Deer Pond, Wildcat Bluff, Snake Hole, and Heron Pond. Populations in swamps on the drier end of the gradient had more support from juvenile stages. In a tree health survey in 2022 within the impounded interior of Buttonland Swamp (not Crawford Tract), seventy percent of the old-growth *T. distichum* trees were stressed or declining. Coupled with the fact that no seedlings or saplings were observed, the populations in the interior of the swamp were not stable. Many factors could further stress these ancient *T. distichum* including interactions of impoundment with chemicals, disease, and changes in air temperature and precipitation. Overall, the seedling abundance in these swamps increased in these environments from 2012 to 2021, except in the most flooded swamps. From a management perspective, drier conditions were more conducive to the success of the earlier life history stages of *T. distichum* in these old-growth forests.

1. Introduction

Population matrix modeling can be used to identify the demographic transitions and the environmental variables influencing life stages dynamics, which are important in determining population stability and/or growth. Stable populations have a balance of fertility and mortality rates of adults, that in conjunction with early life stage transition, support constant populations over time (Artzrouni, 2005; Silvertown et al., 1996). Study approaches in this field are a key tool in conservation management (Caswell, 2002; Crone et al., 2011), and are becoming important for the status assessment of species under consideration for protection by the U.S. Fish and Wildlife Service under the U.S. Endangered Species Act (McGowan et al., 2017).

It is not necessary for populations of ancient individuals such as old-growth trees to regenerate each year (Caswell, 2002). *Taxus floridana* Nutt. has periods of episodic recruitment, but these early life stages do not seem to contribute much to population stability (Kwit et al., 2004). Some riparian forest species have normal recruitment episodes only once per decade, e.g., *Populus deltoides* along free-flowing rivers of the western United States (Lytle & Merritt, 2004). A better understanding of the long-term influence of life stage dynamics on population stability and the health of long-lived forest species is essential for understanding the potential impacts of changing environments (Middleton, 1999).

The health of sensitive species is an important metric for assessing any threat to the survival of ancient populations and can be projected using transition matrix models (Bierzychudek, 1999). Additional

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assessment of the future stability of populations can be explored by using standard approaches to assess the health of trees using estimates of tree crown occupancy and other metrics (Schomaker et al., 2007). A change in population health in response to the environment can give insight into the likely outcomes of environmental change as well as management to offset undesirable changes in future natural systems (Jones & Driscoll, 2022; Campbell et al., 2022; Zinnert et al., 2021).

Wetland species are threatened by agricultural waste such as herbicides and pesticides (Knuffman et al., 2020; Erndt-Pitcher & Kemper, 2022), landscape-level land drainage for agriculture, water storage projects on floodplains for irrigation or municipal water consumption, extreme climatic events (Middleton, 1999), subsidence, erosion, deposition, and geophysical processes (Middleton & Boudell, 2023). Floodplain water environments are changing in the midwestern United States, and stream gages show an increase in water height in recent decades (Mallakpour & Villarini, 2015), along with increasing precipitation levels and decreasing elevation along the Cache River in southern Illinois (Middleton, 2023a). From 1991 to 2000 to 2011–2020, the trend of several normal temperature variables increased including normal decadal minimum annual air temperature, mean July air temperature, maximum July air temperature, and minimum July air temperature (Middleton, 2023a). Normal annual precipitation increased in 2011–2020 in comparison to 1991–2000 and 2001–2010 (119.3 to 135.2 cm year⁻¹, respectively) (Middleton, 2023a).

Landscape-level land drainage is an especially important wetland change along the Cache River in Illinois where the floodplains along the lower part of the river were cut off from the flow of the river by a regional water drainage system after ~ 1916 (Post Creek Cutoff) for the purpose of agricultural development (Muir et al., 1995). Engineered drainage systems often drain entire regions, and make wetland restoration and management challenging (Middleton, 1999). In the case of the Cache River, the downcut main channel has limited hydrologic interaction with the surrounding floodplain along the Cache River watershed, in Illinois USA (Bennett et al., 2001). Channel incision greatly reduces the opportunity for the interplay of water, processes, and biota between the channel and floodplain (Middleton, 1999; The Nature Conservancy, 2002).

These changes have the potential to hinder the life history stages of *T. distichum* that support populations along the Cache River in southern Illinois. Successful *T. distichum* regeneration requires seasonal inundation to disperse short-lived seeds away from the shade of mother trees (Middleton, 2000). In one experiment, seeds placed in mesh bags in the field had less viability over time (Day = 0, 100, 359: 67, 39, and 4 % viability, respectively) (Middleton, 2000). Seeds will not germinate until a growing season drawdown and moist but unflooded conditions support seedling growth (Middleton, 1999, 2000, 2003). *Taxodium distichum* seedlings die if overtopped by water for even a day or two during the summer growing season (Demaree, 1932; DeShield et al., 1994; Middleton, 1999, 2000). *Taxodium distichum* saplings become successively more tolerant of flooding as they age (Middleton, 1999). Infrequent transitions between the early life history stages of a long-lived species can indicate the potential for population decline if old trees die (Caswell, 2002).

For adult *T. distichum*, production levels as measured by leaf litter fall declined by the 1990s in permanently flooded in contrast to seasonally flooded conditions in Buttonland Swamp (Middleton & McKee, 2005). Recently, there is new concern that trees in the Midwest and elsewhere are stressed by both impoundment and/or herbicide drift; the specific chemicals of concern are dicamba, 2–4 D, glyphosate, and atrazine (Knuffman et al., 2020; Erndt-Pitcher & Kemper, 2022).

The floodplain forests along the Cache River Illinois lie at the junction of four ecological provinces and represent the northern terminus of the Mississippi Embayment (Ramsar, 2023). These ecosystems are a critical part of the Mississippi Flyway, and support some of the rarest natural species in the highly agricultural state of Illinois, and so portions of the system were designated as a Ramsar site in 1994 (Ramsar, 2023).

While drainage projects from the early 1900s dried the floodplain for agriculture (Muir et al., 1995), these floodplains still support large remaining stands of ancient *T. distichum* forests along the Cache River, particularly in Buttonland Swamp. As a conservation effort, the Diehl Dam was constructed in the 1980s to keep water levels higher in Buttonland Swamp (Muir et al., 1995). However, the Diehl Dam created an impoundment, which is unsuitable for the regeneration of *T. distichum* (Middleton, 2002).

Moisture levels on the floodplains of the Cache River have likely changed due to long-term shifts in climate and environment (Middleton, 2023a). The surface height of floodplains surrounding the Cache River decreased by as much as –0.9 cm per year in the last twenty years (Middleton, 2023a), noting that floodplains can scour following major rain events (Wyrrick & Pasternack, 2016), especially when adjacent to downcutting channels (Simon, 1989). When floodplain elevation loss along the Cache River is combined with hydrologic changes related to increased precipitation and/or evapotranspiration as air temperatures increase, the current Cache floodplain may either be more or less flooded than several decades ago (Middleton, 2023a). Therefore, a study of changes in the life stages of *T. distichum* over decades is highly relevant to the conservation status of this foundational species.

The objective of this research was to create baseline information on regeneration and population growth of *T. distichum* to aid adaptive management decisions in support of conservation. This study examined *T. distichum* population growth within several regional swamps along a moisture gradient to model growth, stability, or decline based on data of life history stage transitions from the 1990s, 2002–2011, and 2012–2022.

Hypotheses tested by this research included: 1) early life history stage responses will be higher in drier environments including reproduction (biomass seeds, cones, male strobili), abundance of seedlings, and height of saplings as related to site type along a moisture gradient, or time frame; 2) early life history stages response will be higher in drier years as represented by the El Niño-Southern Oscillation (ENSO) events such as a La Niña event within six months prior to the growing season (i. e., summer field collection time); and 3) stochastic growth models of *T. distichum* population numbers (Caswell, 2002) will project earlier population stability in drier environments.

2. Materials and methods

2.1. Description of the study area

Buttonland Swamp is located along the Cache River IL USA (37°17'50" N, 89°03'10" E; Fig. 1) and has been hydrologically altered by agricultural drainage projects with some subsequent restoration activities. The original hydrology of Buttonland Swamp included seasonal overbank flooding for up to 6–8 months each year leading to the creation of a ridge and swale floodplain (Cache River Drainage Commissioners of Illinois, 1905). In 1915, a cutoff was created to divert the main flow of water upstream of Buttonland Swamp directly to the Ohio River with the purpose of drying the floodplain for agriculture. The swamp maintained an old-growth forest after the construction of the Post Creek Cutoff, but Buttonland Swamp was drier and cut off from water inputs from the main channel after this diversion was created (Bennett et al., 2001). Riverine *T. distichum* swamps in the northern parts of the southeastern United States typically have high water in the winter and low water in the summer (Voigt & Mohlenbrock, 1964; Wharton et al., 1982).

The construction of the Diehl Dam in the Cache River channel in 1982 was designed to reflood Buttonland Swamp (Muir et al., 1995) but created impounded conditions (Middleton & McKee, 2005). After impoundment, the production of the trees was lower in flooded portions of Buttonland Swamp (Middleton & McKee, 2005). At the same time, regional flood magnitude and frequency increased in southern Illinois from 1962 to 2011 (Mallakpour & Villarini, 2015). In this study, note that the “dry-to-wet gradient” is based on relative water depths in these

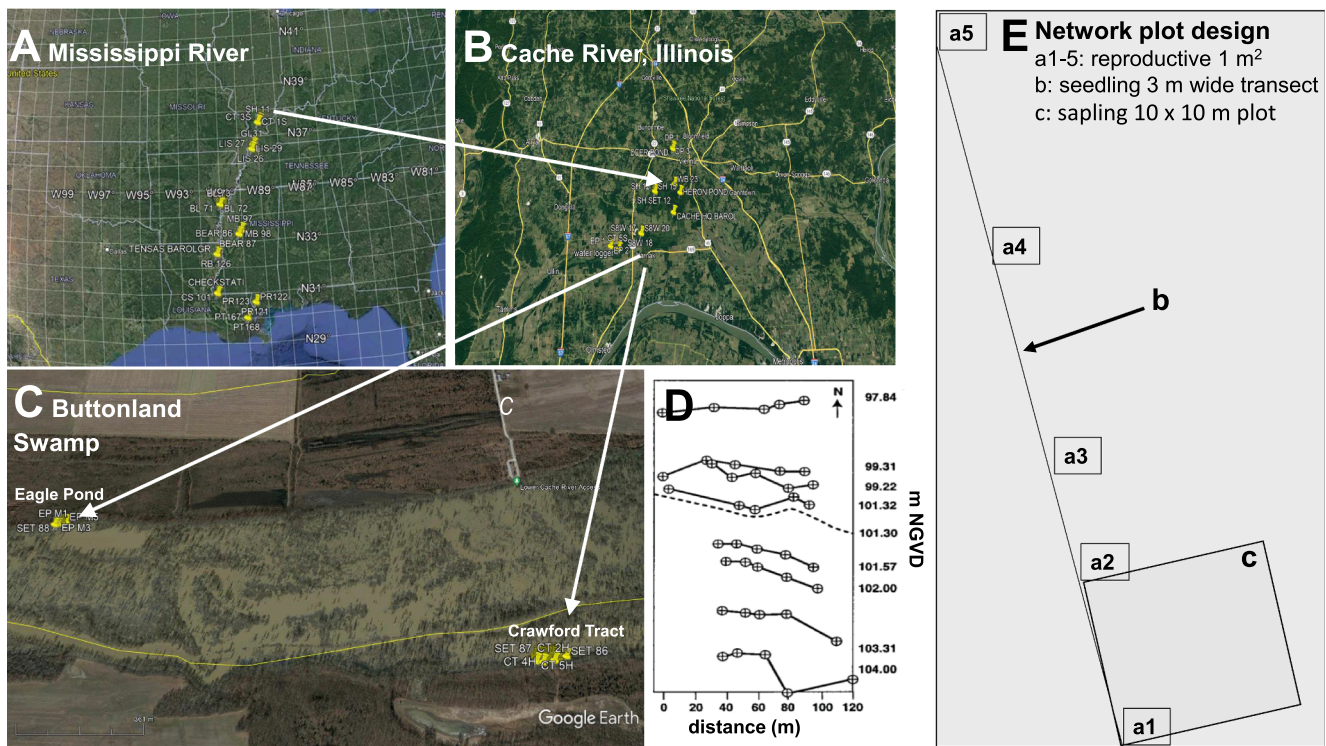


Fig. 1. Map of the region, river, and study site locations for the decadal comparison of tree health and regeneration of *Taxodium distichum* including **A** Mississippi River floodplain study network each with 4–7 replicates per location, noting that the Illinois site was the only network site used in this study; **B** Cache River Illinois study sites including Crawford Tract, Eagle Pond, Deer Pond, Wildcat Bluff, Snake Hole, Heron Pond, and Section 8 Woods; **C** IDNR/USGS study sites along Lower Cache River including Crawford Tract (CT), which was first studied in 1991–2002 and then re-established in 2017; **D** Plot design in Crawford Tract in 1991–2002 re-established in 2017 with five plots along each of three transects (elevation: ~101.32, 101.57, 102.00 m NBVD) with the interior of the swamp northward and an old field southward. Note that point bar transects from the 1991–2002 study in the former Cache River channel could not be located (~99.31, 97.84) and may have been removed by dredging. The plot layout at each site **E** is set up along a 125 m long transect with plots as follows: reproductive biomass in five, one m² plots (a1–a5), seedling along a 3 m wide swath of the transect (b), and saplings within a 10 × 10 m or 100 m² plot between Plots a1–a2 (c).

regional sites during day-of-site-visit measurements within a similar time frame along the Cache River since the 1990s.

2.2. Description of the study species

Taxodium distichum (L.) Rich. is a gymnosperm in the family Cupressaceae with deciduous needles (USDA, 2023). The species occurs in wet forests in the southeastern United States including the Atlantic Coastal Plain, U.S. Gulf Coast to southeastern Texas (Little, 1971; USDA 2023), and the Mississippi Embayment (Little, 1971; Middleton, 2020). It is long-lived and grows to 35 or more meters tall and 1.8 m in diameter (USDA, 2023), noting that much larger individuals also occur in study sites along the Mississippi River Alluvial Valley (Middleton, USGS, personal observation). *Taxodium distichum* has knees that protrude above the ground, which are a part of the root system (Middleton, 2020). The root system includes a tap root, and horizontal roots that extend 6 to 9 m and then bend downward (USDA, 2023). Geographically, the knees are tallest in the center of the species' distributional range in wet conditions (i.e., Arkansas), with tree and knee height generally smaller northward and southward from the center of the range within the Mississippi Embayment (Middleton, 2020). For regeneration, *T. distichum* has specific environmental requirements (Middleton, 1999). To germinate, the seeds need flooded conditions (USDA, 2023) after seed fall from the trees (after October in Illinois), followed by moist drawdown during the growing season (Middleton, 2000). Seedlings need relatively unshaded conditions to grow quickly (USDA, 2023).

2.3. Study design

The current study was designed to reestablish plots that were created

in Crawford Tract during the 1990s (as described in Middleton & McKee, 2005, and Middleton, 2000). At the Crawford Tract site, five vegetation plots were placed in stratified random positions within each 25 m section of a 125-m transect in each study swamp. In Crawford Tract in 2016 (Fig. 1D; Appendix A), three 125-m transects followed elevations roughly equivalent to those measured originally in the 1990s with a theodolite (99.22, 101.32, and 101.57 m NGVD29) (Fig. 1D, Appendix A). In Eagle Pond in 2019, a single transect was laid out at a similar elevation and fashion as Crawford Tract (~101.32 m NGVD29; Fig. 1E). In Crawford Tract and Eagle Pond, there were 15 plots and 5 plots, respectively. The plots in Crawford Tract were used in the 1990s to document litter production and seed dispersal (as described in Middleton, 2000; Middleton & McKee, 2005). From 2006 to 2022, one transect each was monitored in Deer Pond, Snake Hole, Heron Pond, Section 8 Woods, and Wildcat Bluff (Fig. 1E; Appendix A) (Middleton, 2020). The studies in these floodplain forests were set up to match those conducted elsewhere in the southeastern United States within the North American Baldcypress Swamp Network, which is a system of study sites used to study geographic shifts in ecosystem processes in *T. distichum* swamps of the southeastern US (NABCSN; Middleton, 2020).

2.4. Field and lab assessment of tree status

The following studies were conducted to observe changes in *T. distichum* seedling and sapling transition, and tree mortality in the 1990s vs. ~ 2018–2022 in Buttonland Swamp (Crawford Tract and Eagle Pond) vs. other regional sites in 2006–2022 (Deer Pond, Snake Hole, Heron Pond, Section 8 Woods, and Wildcat Bluff).

2.4.1. Regeneration: seeds, seedlings, and sapling

During the summer survey of each year (Appendix A), seedlings, saplings, and trees were recorded to provide information for the *T. distichum* transition models for each site. Floating litter traps were set up to capture reproductive litterfall (biomass) of *T. distichum* (seeds, cones, male strobili) and numbers of fallen *T. distichum* cones in plots during the summer survey (Fig. 1E; Middleton, 1995) and retrieved during the winter survey after seasonal peak litterfall (Appendix A). Floating litter traps (circular bucket bottoms: 0.27 m in diameter) were tied to a wooden post at stratified random positions along 25 m intervals along a 125 m transect with 5 plots per site (Fig. 1E). The weight of reproductive biomass was recorded after heating in a drying oven at 70 °C and the presence of *T. distichum* reproductive biomass was noted for each trap, from the peak fall period during the winter survey each year (Appendix A). Note that over the course of the year, 95 % of the annual total reproductive biomass fell from October to late November (Middleton et al., 2021). All reproductive and seed fall data are available in Middleton (2023b).

During each year of the study during the summer survey (Appendix A), *T. distichum* seedlings were counted along one 125 m transect and 3 m wide between Plot 1 and 5 at each site (Fig. 1E). Note that there were more transects during the intensive spatial distribution of seed deposition conducted during the 1990s in Crawford Tract, noting that in 2016–2019, the lower transect (L) and middle transect (M) were at similar elevations as in the 1990s surveys. Also, the height of saplings was recorded within a 10 m² plot between Plot 1 and 2 during the summer field survey (Fig. 1E; Appendix A). An ancient *T. distichum* tree was associated with each of the plots, and none of these died during the years of the study from the 1990s–2021. All reproductive data related to vegetation transition and regeneration are available in Middleton (2023c).

Presence/absence data for reproductive biomass, seedlings, saplings, and trees from the 1990s in Crawford Tract were assessed from published data (Middleton & McKee 2005). Day-of-visit water depth was measured at each plot and recorded during the summer and winter surveys (Appendix A).

2.4.2. ENSO categorization of summer season status

The ENSO event categorization for each site and year was based on the type of event (El Niño, La Niña, a sequential El Niño to La Niña, or no event) determined to be present six months prior to the summer field survey (Appendix A). The time and type of ENSO event were determined from information in NCAR (2023).

2.4.3. *Taxodium distichum* tree health in the interior of Buttonland Swamp

A health assessment was conducted in 2022 on *T. distichum* trees located south of the northern canoe landing of Buttonland Swamp toward the Cache River channel within an accessible area cleared of *Cephalanthus occidentalis*. *Taxodium distichum* trees were selected for health assessment along a north-to-south canoe route at least 50 m from the last assessed tree. Each successive study tree was chosen by following the direction of a random throw along the canoe path until a total of ten trees were selected. The GPS coordinates of each study tree were recorded.

Tree health metrics were assessed and recorded for the selected trees by two, two-person teams. The live crown ratio was based on an assessment of the percentage of live crown material filling the actual percentage of canopy filling the crown of the tree divided by the potential percentage, i.e., 100 % (Schomaker et al., 2007). The tree stress level for each tree was visually assessed as healthy, stressed, in major decline, or dead. A healthy tree had a vigorous canopy with no dieback, epicormic branching, or significant disease. A stressed tree had canopy dieback of 50 % or less, and no epicormic branching, defoliation, or significant vine coverage in the canopy. A tree with significant decline had canopy dieback of 50 % or more, significant epicormic branching, significant defoliation, broken top, and/or major vine infestation. A

dead tree had a standing stem with no live foliage and sloughing or absent bark.

Also recorded for each sampled tree was information on tree class (sapling, young, old), and height in meters estimated with a Nikon range finder. Suspected herbicide damage to leaves was recorded as % foliage affected, deformed/curled leaves, chlorotic leaves, and other damage. Leaf color was determined using a Munsell color chart. The presence/absence of *T. distichum* seedlings and saplings were recorded within 25 m of the sampled tree. Percentage (%) cover of duckweed (*Lemna minor*; a small floating plant) within 5 m of a tree was recorded. Note that *Lemna* spp. were once very common in this swamp (Middleton, USGS, personal observation, 1990–2002). Each sampled tree was photographed with a cell phone or waterproof camera. All tree health data are available in Middleton (2023d).

2.4.4. Model parameters for population matrix models in R

For the population projection models, population numbers in Year “0” for each life history stage (seeds, seedlings, saplings, and trees) were set up using the following approach. The initial number of seeds per tree was equal to the average number of seeds in the annual number of cones falling in the reproductive litter per tree (plot) as a mean of all sites and all years during the years of the study (mean cones site⁻¹ year⁻¹ = 0.028). The mean number of seeds per cone from cones collected along the Mississippi River Alluvial Valley is 13.9158 ± 0.1454 seeds cone⁻¹, which varies with geographical location and annual precipitation level (Middleton et al., 2021). The average number of seeds captured per litter trap (0.27 m diameter bucket bottom) was 0.0268.

The number of seed produced per tree was based on its canopy spread projected to the ground surface. The ground coverage area of the canopy of mature cypress trees (~60 years old) was measured for five trees growing next to a pond in Louisiana by measuring the aerial extent of the widest part of the canopy in two directions (roughly ellipsoidal), with two widths (a and b) ranging from a × b lengths of 6.9×6.9 m and 9.3×10.9 m (Middleton, 2023b); the area of an ellipse is: $3.14 \times a \times b$ (Selby, 1967). The mean area of these ellipsoids representing the ground cover of these trees was 2.36 ± 27.79 m² and the mean area of the litter trap was 0.23 m².

Following this procedure, the mean number of seeds produced per tree at Time ‘0’ was 303 seeds tree⁻¹. The number of seedlings at Time ‘0’ = 3.8, saplings = 2.1, and trees = 1. While the actual numbers were not known for seeds, seedlings, saplings, and trees, these mean annual values provided reasonable values for population numbers at the initial time step of the model.

To project the population number over fifty years for each site, for each year (step), the population number for each life stage was determined using the annual estimates presence/absence of each life history stage at the site as the mean annual value of the proportion of presence/absence in the sample plots at each site (Table 1) divided by the number of years of the study, which varied in size for each life history type. All population projection data are available in Middleton (2023e).

2.5. Data analysis

2.5.1. Water category of sites

Sites were grouped along a water gradient based on day-of-visit water depths measured at plots (Appendix A). Water depths of plots during each survey were used to determine the mean water depth in the summer (MeanSummerWaterDepth) and winter (MeanWinterWaterDepth) for each year and site. The pseudo T criterion was using PROC CLUSTER to determine the water grouping along a water gradient of sites (1 = driest to 4 = wettest; dry, medium-dry, medium-wet, wet) using day-of-site-visit water depths measured in these swamps during summer and winter surveys (see Appendix A).

The results showed four water groupings including dry (Crawford Tract & Eagle Pond), medium-dry (Section 8, Heron Pond & Deer Pond), medium-wet (Snake Hole), and wet (Wildcat Bluff), which were used in

Table 1

For each site, the frequency of life history stages was assessed for seeds in 1-m² plots (fecundity), seedlings along transect (s_0), and saplings (s_1) in 1–10 m² plot (i.e., life stage frequency). Tree frequency (s_2) was “1.0” because no trees died during the study. In the popgen model, transition values were assigned as: seed = $f/50$, seedling = $s_0/500$, sapling = $s_1/10,000$, with the chance of tree mortality = $1/50,000$. Population growth is described by “ λ ”, which is the dominant eigenvalue based on a change in population size between population counts (Caswell, 2002). The interior of Buttonland Swamp was only surveyed in 2022 for the tree health survey and not in other years.

Site	Water category	Decade or year	Seed frequency	Seedling frequency	Sapling frequency	Tree frequency	Notes
Crawford Tract - L	Wet	1990s	0.0000	0.0000	0.0000	1.0000	No sapling-tree transition
Crawford Tract - L	Wet	2016–2021	0.0000	0.0000	0.0000	1.0000	No sapling-tree transition
Crawford Tract - M	Dry	1990s	1.0000	0.0000	1.0000	1.0000	
Crawford Tract - M	Dry	2016–2021	0.0670	0.0667	0.0067	1.0000	All transitions present
Eagle Pond	Dry	2019–2021	0.9000	0.4375	0.0875	1.0000	All transitions present
Heron Pond	Medium-dry	2006–2021	0.8000	0.5000	0.0875	1.0000	All transitions present
Section 8 Woods	Medium-dry	2006–2021	0.7567	0.3750	0.0500	1.0000	All transitions present
Deer Pond	Medium-dry	2006–2021	0.7895	0.4375	0.0875	1.0000	All transitions present
Snake Hole	Medium-wet	2006–2021	0.5500	0.4375	0.00625	1.0000	All transitions present
Wildcat Bluff	Wet	2006–2021	0.6938	0.3750	0.0125	1.0000	All transitions present
Buttonland Swamp Interior	Permanently flooded	2022	No data	0.0000	0.0000	1.0000	No transitions present

the zero-inflated Poisson and negative binomial ANCOVAs. All statistical analyses were done using SAS 15.1 (2018) unless otherwise specified.

2.5.2. *Taxodium* reproductive biomass, seedling abundance, and sapling height

Annual total reproductive biomass of *T. distichum* (i.e., fallen seeds, cones, and male strobili) from peak reproductive biomass was determined as the percentage of reproductive biomass collected during this peak season of reproductive biomass fall between early October and late November (in southern Illinois), noting that a very small percentage of material fell during other times of year (<5%) (Appendix C). Mean seedling abundance and sapling height were assessed for each site and year as the total number of individuals along the transect or the total height of saplings within the 100 m² plot, respectively.

Reproductive biomass (g), seedling abundance transect⁻¹, and mean sapling height per 100 m² plot⁻¹ were analyzed using zero-inflated Poisson and negative binomial ANCOVAs using PROC GENMOD for the five predictors of SummerWaterDepth, WinterWaterDepth, water category (dry, medium-dry, medium-wet, wet), decade (2006–2011 vs. 2012–2021) and ENSO type (El Niño, La Niña, a sequential El Niño to La Niña event, or no event). Three cases were analyzed: including (1) five main effects [watercategory, decade & ENSO event type], (2) five main effects with decade replaced with year, and (3) five main effects with watercategory*decade interaction with the dry category = 1 dropped because studies were not conducted in Crawford Tract and Eagle Pond from 2006 to 2016.

A Poisson test was applied to see whether a Poisson and/or Negative binomial distribution should be fit to the responses. All possible models were run to see which models converged. All possible AICc models were run for each of the three response variables to determine convergence, and the outcome was used to select best-fit models for ANCOVA comparison (i.e., three model cases with two data distribution assumptions including negative binomial and Poisson). The residuals and Pearson residuals were checked in the final model to meet assumptions. The final model predicted equations for each response is ‘predicted response’ = $(1 - [1 + \exp(-ZM)]^{-1}) * \exp(AM)$, where AM is the predicted abundance model(log-transformed) and the abundance of the response is greater than zero. ZM is the predicted zero model (logistic transformed), with $ZM = \text{logit}(p) = \log(p/(1-p))$, where $p = P(\text{abundance of the response} > 0)$. Alpha = 0.05 was used for all analyses.

The Delta Method was used to determine mean $\pm$ S.E. reproductive biomass by year and also with respect to ENSO event category with $i = 1, 2, 3, 4$, i.e., correlation to El Niño, La Niña, sequential El Niño to La Niña,

or no event. The mean was expressed, following the methodology in Ver Hoef (2012), as follows: mean = $\exp(\text{estimate of ESNO} = i) = \exp(\text{est})$; stder r = $\exp(\text{estimate of ESNO} = i) * \text{Stderr}(\text{estimate of ESNO} = i) = \text{stderr}(\text{est})$.

2.5.2.1. Final reproductive biomass model. The best model (AICc = 247.4596, Pseudo R-square = 36.02 % (UCLA, 2022) follows.

Reproductive Biomass Model when reproductive biomass > 0: AM = $\log \text{mean}(\text{ReproductiveBiomass}) = 0.6646 - 0.5785 * I(\text{Year} = 2006) - 0.6629 * I(\text{Year} = 2007) - 0.8091 * I(\text{Year} = 2008 \text{ \& ENSO} = \text{“La Niña”}) + 0.1041 * I(\text{Year} = 2009) + 0.6722 * I(\text{Year} = 2010 \text{ \& ENSO} = \text{“El Niño La Niña”}) + 0.4897 * I(\text{Year} = 2011) + 2.7631 * I(\text{Year} = 2012) + 0.0311 * I(\text{Year} = 2013) + 1.5121 * I(\text{Year} = 2014) - 1.1612 * I(\text{Year} = 2015) + 1.3529 * I(\text{Year} = 2016) + 0.2427 * I(\text{Year} = 2017) + 2.0804 * I(\text{Year} = 2018) + 0.4399 * I(\text{Year} = 2019) + 2.6906 * I(\text{Year} = 2020) - 0.0109 * \text{MeanSummerWaterDepth} + 0.0018 * \text{MeanWinterWaterDepth} - 1.7302 * I(\text{ENSO} = \text{“None”})$.

Zero model to predict presence/absence of reproductive biomass: $ZM = \text{logit}(p) = \log[p/(1-p)] = -27.2838$, where $p = P(\text{reproductive biomass} > 0)$, $1-p = P(\text{reproductive biomass} = 0)$, $I(\text{“TRUE”}) = 1$ & $I(\text{“FALSE”}) = 0$.

2.5.2.2. Final seedling abundance model. The best model (AICc = 254.4194, Pseudo R-square = 13.64 %) follows.

Seedling Abundance Model when seedling > 0: AM = $\log \text{mean}(\text{SeedlingNumber}) = 1.9671 - 2.8221 * I(\text{Decade} = 2006_2011) - 0.0381 * \text{MeanSummerWaterDepth} - 0.0203 * \text{MeanWinterWaterDepth}$.

Zero model to predict presence/absence of seedlings: $ZM = \text{logit}(p) = \log[p/(1-p)] = -24.9479$, where $p = P(\text{MeanSeedlingNumber} > 0)$, $1-p = P(\text{MeanSeedlingNumber} = 0)$, $I(\text{“TRUE”}) = 1$ & $I(\text{“FALSE”}) = 0$.

2.5.2.3. Final mean sapling height model. The best model (AICc = 154.8883, Pseudo R-square = 24.35) follows.

Height Model when sapling height > 0: AM = $\log(\text{MeanSaplingHeight}) = 1.6126 - 3.8676 * I(\text{SiteWaterCategory} = 1) + 0.5017 * I(\text{SiteWaterCategory} = 2) - 2.2268 * I(\text{SiteWaterCategory} = 3) - 0.1122 * \text{MeanSummerWaterDepth}$.

Zero Model to predict height of sapling: $ZM = \text{logit}(p) = \log(p/(1-p)) = -0.0803 + 23.6334 * I(\text{Decade} = 2006_2001) - 0.8536 * \text{MeanSummerWaterDepth} - 1.8392 * \text{MeanWinterWaterDepth}$, where $p = P(\text{MeanSaplingHeight} > 0)$, $1-p = P(\text{MeanSaplingHeight} = 0)$, $I(\text{“TRUE”}) = 1$ & $I(\text{“FALSE”}) = 0$.

2.5.3. Age class transitions for *Taxodium distichum* life stages

Mean annual seed weight, seedling and sapling regeneration, and tree mortality were noted in plots and along transects in each site, noting that not all sites were sampled each year (Appendix A). Mean percent frequencies (presence/absence) of each life stage for *T. distichum* (seeds, seedlings, saplings, and trees) were assessed for each site, plot, and year as the total number of plots of occurrence divided by the number of plots along each transect within a site. Also, the early life stages were compared along a moisture gradient (drier to wetter sites: Section 8 Woods, Crawford Tract mid-elevation, Crawford Tract low-elevation, Eagle Pond, Heron Pond, Section 8 Woods, Deer Pond, Snake Hole, and Wildcat Bluff).

2.5.4. Stochastic population modeling of *T. distichum* by site

Life history transition values were assessed as the frequency or mean presence/absence of a life stage in plots at each site during each year for each size class including seeds (fecundity), seedlings, saplings, and trees (Table 2). The frequencies of stage classes per site were assessed as the plot average per site during the entire study. Tree frequency (ss_3) was given a value of “1.0” because one tree was associated with each plot, and no trees died during the study.

In the popgen model (R Core Team, 2022), transition values were assigned as: seed to seedling = $f/50$, seedling to sapling = $s_0/500$, sapling to tree = $s_1/10,000$, with the chance of tree mortality = $1/50,000$. Note that ‘ f ’ is the probability that a seed is present whereas $1/50$ is the probability that the seed transitions to a seedling next year. Likewise, s_1 is the probability that a seedling is present and $1/500$ is the probability that it transitions to a sapling during the next year. Finally, s_2 is the probability that a sapling is present and $1/10,000$ is the probability it transitions to a tree during the next year. Trees survive with a probability of $49,999/50,000$. Each year, it is assumed that a tree produces 303 seeds and that the seeds and seedlings the tree produces survive one year (Middleton, 2000). Hence, the popbio projections were run one year at a time to adjust for these yearly events to get the final projections (1, 50, and 500 years). For each site, the life stage transition probability was multiplied by the life stage frequency to give the number of individuals likely to transition to the next life history stage. The interior of Buttonland Swamp did not have any life stage transitions (seed, seedling, sapling) so no population projection was possible.

The stage transitions were handled this way under the assumption that stage transitions had occurred at some time in the past even though these were seldom observed in these old-growth forests. Despite attempts to tag individuals to capture seedling-to-sapling transition and sapling-to-tree transitions, such events were not observed in these populations of *T. distichum*. All one-year-old saplings tagged in the 1990s in an old field at Crawford Tract had died by 2016 (Middleton, personal observation).

The transition and frequency information was entered into R with the popbio library (R Core, 2022) to project population growth for 50 years. Population growth is the projected change in population number over time or “ λ ”, which is the dominant eigenvalue based on the change in population size between population counts (Caswell, 2002). Stochastic modeling was used to make projections of future population changes of *T. distichum* for Crawford Tract at low and medium elevations based on data from current and previous decades (i.e., 1990s vs. 2016–2022). Population projections were also constructed for other swamps including Eagle Pond (2019–2022 only), Deer Pond, Snake Hole, Heron Pond, Section 8 Woods, and Wildcat Bluff with life stage transition data collected from 2006 to 2021.

2.5.5. *Taxodium distichum* tree health in the impounded interior of Buttonland Swamp

Means of each variable (e.g., mean live crown ratio) were constructed by adding the total value of the variable for each tree and dividing by the total number of trees assessed in the interior of Buttonland Swamp.

3. Results

3.1. *Taxodium distichum* reproductive biomass plot⁻¹ (litterfall)

Mean annual *T. distichum* reproductive biomass ($\text{g m}^{-2} \text{y}^{-1} \pm \text{S.E.}$) (seeds, cones, and male strobili) varied by year and ENSO event type ($p < 0.0001$, and 0.0229 , respectively; Table 2a; Fig. 2 A and B, respectively). Year 2012 had a higher mean annual reproductive biomass than 2008, 2017, or 2021 (mean reproductive biomass in 2012 vs. 2008, 2017, and 2021: 30.806 ± 25.35 vs. $1.76 \pm 1.67 \text{ g m}^{-2} \text{y}^{-1}$, respectively; Table 2a & Fig. 2A; Bonferroni adjusted comparison: $\alpha = 0.05/6 = 0.0083$). Note that 2008 was the only year with a La Niña event with no preceding El Niño event (Table 2a). With respect to ENSO events, the estimated mean annual reproductive biomass $\pm \text{S.E.}$ was highest in a year with a shift from El Niño to La Niña (2010) vs. years with other ENSO event types (3.65 ± 1.36 vs. $1.04 \pm 0.5 \text{ g m}^{-2} \text{y}^{-1}$, respectively; Table 2a & Fig. 2B; Bonferroni adjusted comparisons: $\alpha = 0.05/6 = 0.0083$). Note that mean annual *T. distichum* reproductive biomass did not vary by site, site water category (i.e., dry to wet), decade, and summer and winter day-of-visit water depth ($p > 0.05$; Table 2a).

3.2. *Taxodium distichum* seedling abundance transect⁻¹

The mean abundance of *T. distichum* seedlings transect⁻¹ varied by decade ($p < 0.0001$, but not by site, site water category (i.e., dry to wet), year, decade, ENSO event, and summer and winter day-of-visit water depth ($p > 0.05$; Table 2b). The mean total abundance of seedlings transect⁻¹ was lower during the decade 2006–2011 than during 2012–2021 (2.21 ± 1.48 vs. 1.75 ± 1.29 seedlings transect⁻¹, respectively; $p < 0.0001$, Table 2b; Fig. 3).

3.3. Sapling height 100 m² plot⁻¹

Saplings were present in these swamps except in Section 8 Woods in 2006–2011, Crawford Tract in 2016–2022, and the interior of Buttonland Swamp in 2022 (Table 1). The mean height of saplings 100 m² did not vary by site, site water category (i.e., dry to wet), year, decade, or ENSO event ($p > 0.05$; Table 2c). Mean height of saplings 100 m² decreased with increasing summer and winter day-of-visit water depths ($p = 0.0003$ vs. 0.0019 ; Fig. 4 A and B, respectively) following a non-linear relationship related to the expected values of the Zero and Expected Models: expected value (SaplingHeight) = $1/[1 + \text{expected}(-\text{ZeroModel})] * \text{expected}(\text{HeightModel})$ (Table 1c; see Section 2.5.3.3 (above)).

No adult *T. distichum* trees died in these study sites i.e., no dead standing or fallen *T. distichum* trees were observed from the 1990s through 2021.

3.4. Stochastic projection of *T. distichum* in future decades

Population sizes (N) of the life stages of *T. distichum* were predicted for each swamp over one, fifty, and five hundred years into the future based on a starting number of fifty trees using popbio in R (R Core, 2022) (Tables 1 and 3). The eigenvalues of these populations were all very near “1”. Adult tree numbers increased very little during the 1–500 years of model runs (Table 3). Eagle Pond had slightly higher numbers of saplings and adult trees in the model runs, especially by Year 500 (Table 3).

3.5. *Taxodium distichum* tree health in the interior of Buttonland Swamp

Based on a survey via canoe in the interior of Buttonland Swamp, the mean live crown ratio for *T. distichum* trees in the survey was 43.3 %, the mean height was 16.3 m, and 70 % of the trees showed signs of stress or decline (Appendix E). The % cover of lemnids (e.g., *Lemna minor*) was less than 1 % m².

Table 2a

Outcome of a zero-inflated negative binomial model using PROC GENMOD to test responses of log *T. distichum* reproductive biomass (cones, seeds, and male strobili; ReproductiveBiomass) based on the model comparison of the interactions of year (2006–2021), and ENSO event status, with the covariables of mean summer day-of-visit water depth, mean winter day-of-visit water depth for cypress swamps along the Cache River, Illinois. ENSO event types included El Niño, La Niña, sequential El Niño to La Niña, or no event six months before the summer field survey (Appendix B). This model comparison was chosen as the best model fit of a set of all possible models based on AICc model outcomes (AICc = 247.4596). See Section 2.5.3.1 for details of the model equations.

Variable	df	Chi-square Likelihood Ratio	Maximum likelihood estimate or Pseudo r^2	p	Significance	Contrast Lsmeans	ENSO event type	p
Overall model	16	106.55		<0.0001	***			
Intercept	1		0.6646	0.5225				
Year	13	55.07		<0.0001	***			
2006	1		−0.5785	−0.5785			None	
2007	1		−0.6629	−0.6629			None	
2008	1		−0.8091	−0.8091		a	La Niña	
2009	1		0.1041	0.1041			None	
2010	1		0.6722	0.6722			El Niño to La Niña > other ENSO type	
2011	1		0.4897	0.4897			None	
2012	1		2.7631	2.7631		b	None	
2013	1		0.0311	0.0311	*		None	
2014	1		1.5121	1.5121			None	
2015	1		−1.1612	−1.1612			El Niño	
2016	1		1.3529	1.3529			None	
2017	1		0.2427	0.2427		a	None	
2018	1		2.0804	2.0804			None	
2019	1		0.4399	0.4399			None	
2020	1		2.6906	2.6906			None	
2021	1		0.0000	0.0000		a	None	
ENSO Event	1	5.17		0.0229				
Mean water depth summer	1		0.61	0.4337				
Mean water depth winter	1		0.02	0.9001				
Dispersion	1		0.1697					

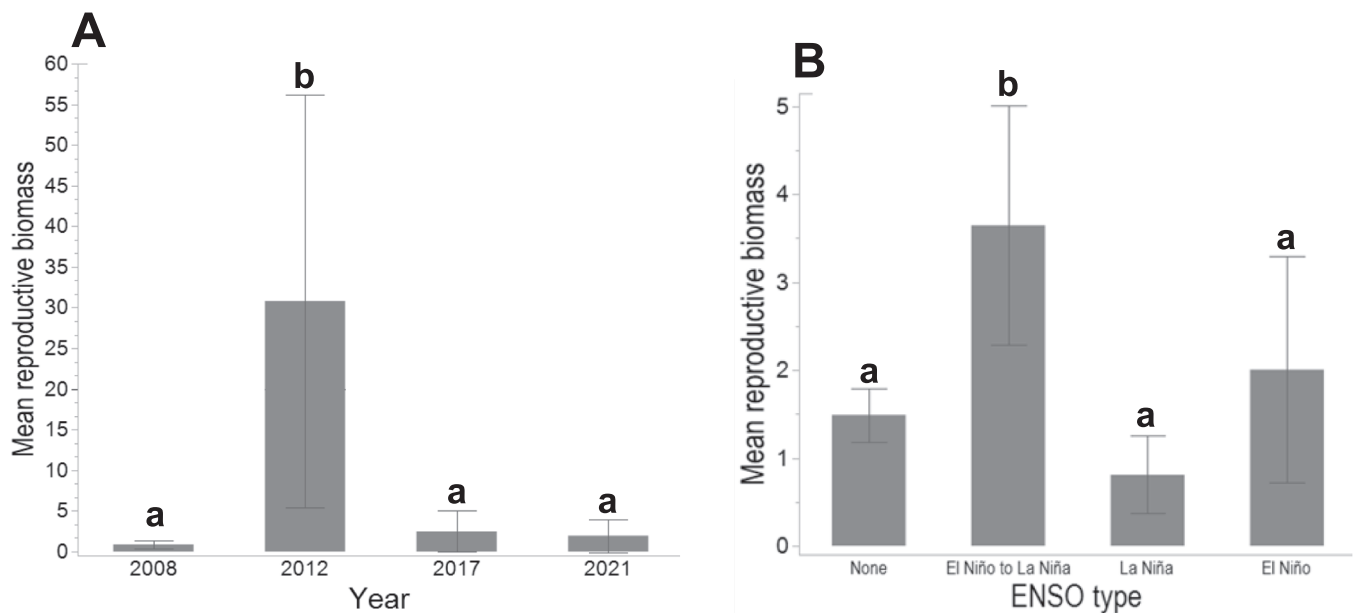


Fig. 2. Comparison of mean $\pm$ S.E. for reproductive biomass for A) Year 2012 vs. 2008, 2017, and 2021 and B) ENSO (Events: None, El Niño, La Niña vs. a sequential El Niño to La Niña) using the Delta Method based on the outcome of a zero-inflated negative binomial model using PROC GENMOD (Table 1a). Different letters indicated significant differences in means based on Bonferroni-adjusted comparisons.

4. Discussion

An understanding of environments that support early life history stages can contribute to management approaches that support the population stability of long-lived species (Hanley et al., 2019). The findings of this study supported Hypothesis 1, i.e., “Early life history stage responses will be higher in drier environments including

reproductive biomass (seeds, cones, male strobili), abundance of seedlings, and height of saplings as related to site type along a moisture gradient, or time frame.”.

In this study, all life history stages responded positively to drier conditions in drier swamps on the water gradient, or year, including regenerative biomass, seedling abundance, and sapling height (Table 1a-c; Fig. 2A and B, Fig. 3). Also note that adult tree production is also

Table 2b

Outcome of a zero-inflated negative binomial model using PROC GENMOD to test responses of log *T. distichum* seedling abundance (log meanSeedlingNumber) based on the model comparison of the interactions of decade (2006–2011 vs. 2012–2021) with the covariables of mean summer day-of-visit water depth, mean winter day-of-visit water depth for cypress swamps along the Cache River, Illinois. This model comparison was chosen as the best model fit of a set of all possible models based on AICc model outcomes (AICc = 254.4194) below. See Section 2.5.3.2 for details of the model equations.

Variable	df	Chi-square Likelihood Ratio	Maximum likelihood estimate or Pseudo r^2	p	Mean $\pm$ S.E.	Contrast Lsmeans	Water depth $\pm$ S.E.
Overall model	3	38.113		<0.0001	***		
Intercept	1		1.9671	0.3701			
Decade	1	16.570		<0.0001		2006_2011 < 2012_2021	
2006–2011	1		–2.8221	<0.0001			
2012–2021	0		0.0000				
Mean water depth summer	1	2.50		0.1140			
Mean water depth winter	1	2.08		0.1495			
Dispersion	1	3.2841					

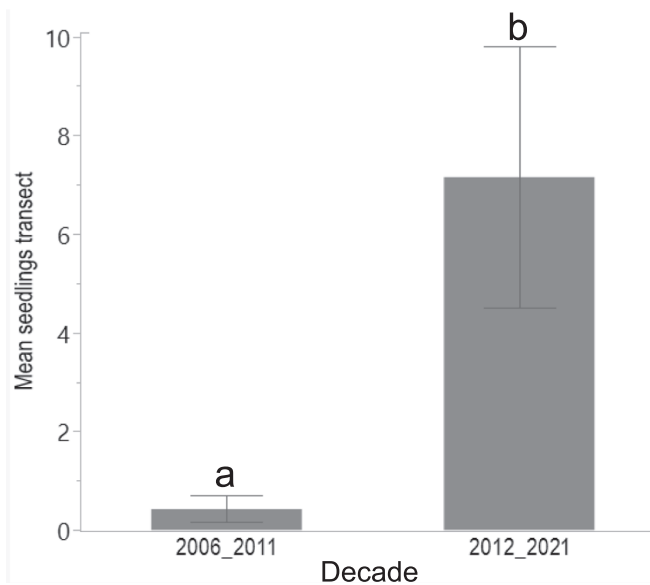


Fig. 3. Comparison of mean $\pm$ S.E. for total seedlings transect⁻¹ (Table 1b) using the Delta Method based on the outcome of a zero-inflated negative binomial model using PROC GENMOD. Different letters indicated significant differences in means based on Bonferroni-adjusted comparisons.

highest in drier conditions in the swamp (Middleton & McKee, 2005). Thus, the hypothesis that the responses of early life history stages would respond positively to drier conditions was supported by the findings of this study.

4.1. Water environment and early life stage response

Hypothesis 2 tested the idea that the early life history response would be higher in drier years as represented such as La Niña events six months event within six months prior to the growing season.

Generally, *T. distichum* had higher reproductive biomass during the drier years during the last two decades. For example, reproductive biomass was higher in 2012 than in some other years (2008, 2017, and 2021; Fig. 2A), noting that 2012 was one of the hottest and driest years to that date in Illinois (Ford, 2022).

For masting species such as *T. distichum*, the role of ENSO climate events that lead to large synchronous production of reproductive material and seedling recruitment is fascinating (Ascoli et al., 2017; Penisi, 2021) but not completely understood for *T. distichum*. Reproductive biomass production was 3 times higher in 2010 following a sequence of wet winter El Niño with a dry spring during a La Niña (Fig. 3; Table 1a). This sequence of events may have provided ideal conditions for the support of the early development of *T. distichum* seeds, cones, and male strobili during the winter El Niño followed by high levels of production in low water during the summer La Niña in southern Illinois. Admittedly, this twenty-year study did not provide enough information on this unusual ENSO sequence to fully test the idea.

Seedling abundance was highest during the second decade of the study (2012–2021) and that period was likely wetter than 2002–2011

Table 2c

Outcome of a zero-inflated negative binomial model using PROC GENMOD to test responses of the log *T. distichum* sapling height based on the model comparison of the interactions of site water category (dry, medium-dry, medium-wet, and wet), with the covariables of mean winter day-of-visit water depth for cypress swamps along the Cache River, Illinois. This model comparison was chosen as the best model fit of a set of all possible models based on AICc model outcomes (AICc = 154.8883). See Section 2.5.3.3 for details of the model equations.

Variable	df	Chi-square Likelihood Ratio	Maximum likelihood estimate or Pseudo r^2	p	Mean $\pm$ S.E.	Contrast Lsmeans	Water depth $\pm$ S.E.
Model	7	42.42		<0.0001	***		
Intercept	1		1.6126	0.0525			
Site water category	3	7.95		0.1000		Not significant. 1 < 2, alpha = 0.10	
Mean water depth summer	1	13.34(Wald)		0.6048			
Decade	1	0.09		0.7609			
Mean water depth summer	1	0.27(Wald)		0.0003	***		
Mean water depth winter	1	9.63		0.0019	**		
Dispersion	1		5.72				

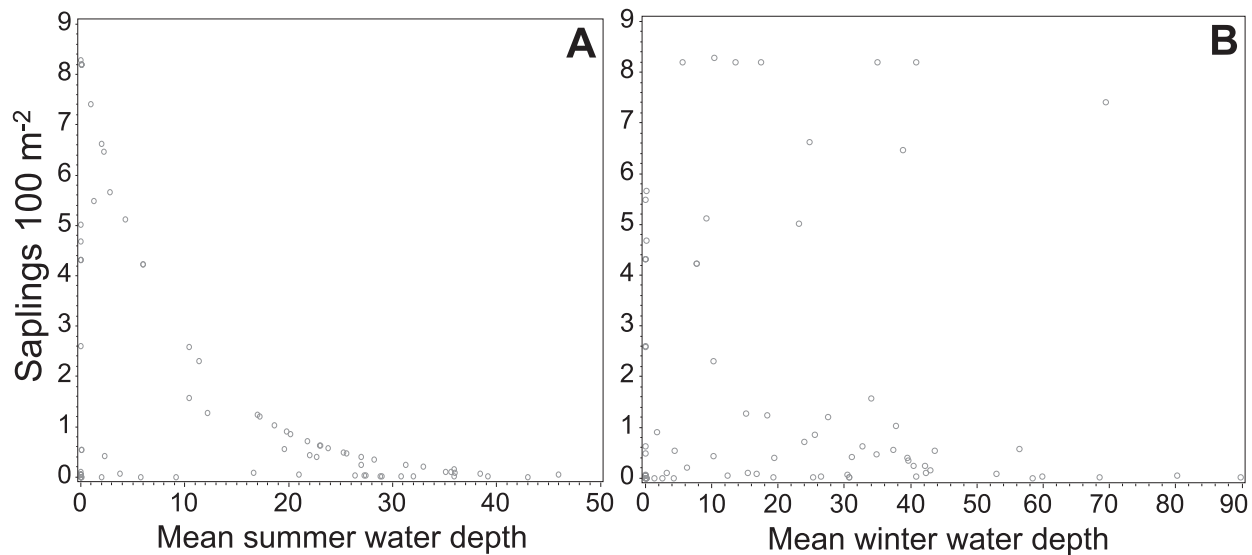


Fig. 4. Comparison of the relationship of predicted mean sapling height 100 m^{-2} for A) summer and B) winter day-of-visit water depths (cm) (Chi-square: 0.27 (Wald) vs. 9.63, respectively; $p = 0.0003$ and 0.0019 , respectively). The non-linear relationship is based on the outcome of a zero-inflated negative binomial model using PROC GENMOD (Table 2c) that predicts the relationship of sapling height with summer and winter day-of-visit water depths using the equation: expected value (SaplingHeight) = $1/[1 + \text{expected}(-\text{ZeroModel})] * \text{expected}(\text{HeightModel})$ (see Table 1c).

(Table 2b). While it is known that the precipitation increased during the second decade of (Table 1 and 2b) the study (Middleton, 2023a), there was no difference in the water categorization of the sites over that time as determined by the cubic clustering (CCC) analysis. Admittedly, the method to determine water environments at the sites was not refined enough to fully answer these questions, particularly given that the elevations of the floodplains in the region were dropping during those decades (Middleton, 2023a).

The masting of other tree species has been linked to ENSO events driven by the North Atlantic Oscillation (NAO) in certain European tree species such as *Fagus sylvatica* L. (European beech) and *Picea abies* (L.) H. Karst (Norway spruce) as based on long-term data sets (Ascoli et al., 2017; Pennisi, 2021). In North America, it is thought that masting and seedling growth in white spruce are driven by El Niño-Southern Oscillation (ENSO) events with a warm wet winter followed by a dry spring (Ascoli et al., 2017). This study did not have higher *T. distichum* reproductive biomass following all El Niño events (Table 1a).

As long-term studies delve into the role of the climate oscillation inherent in ENSO shifts, a better understanding is emerging of the relationship of the environment to the life history stage responses, which ultimately may be of value in assessing population stability and management. In Death Valley desert shrubs such as *Encelia farinosa*, *E. frutescens*, and *Ambrosia salsola*, germination, establishment, and mortality are related to ENSO events with the early establishment and adult mortality linked to precipitation and cold phases, respectively, and the longer-term changes in climate-related to decreasing populations (Ehleringer & Sandquist, 2018). Clearly, this topic deserves much more detailed study.

4.2. Population stability across a water gradient

Hypotheses 3 stated that stochastic growth models of *T. distichum* population numbers (Caswell, 2002) will project earlier population stability in drier environments and were assessed using population projection models compared across a dry-to-wet gradient in the Cache River region. This hypothesis was supported to some extent because many swamps on the drier end of the gradient had higher values of biomass, abundance, and height of life history stages than those at the wetter end of the gradient. However, Crawford Tract which was the driest of the swamp had higher levels of seedling abundance after

2006–2011 (Table 2b). The site was once on the edge of an old field (Middleton, 2002), and perhaps the site was less stressed by farmland edge effects by 2016–2021. The wetter but un-impounded elevation of Crawford Tract-M had projections of stability at higher population levels by 2016–2021 rather than during the 1990s (Table 3). Population growth comparisons were not supported by statistical analysis, so only general trends can be stated here.

The relative contribution to population growth contributed by early life history stages may be less critical to long-lived species than the transition of medium-sized trees and saplings to the tree stage, which is the case for *Fagus grandifolia* in the hurricane-affected forests of northern Florida (Batista et al., 1998). In situations where no trees die over time steps, populations are very stable (Enright & Watson, 1991; Van Mantgam & Stephenson, 2005; this study). However, infrequent regeneration episodes may be key to the indefinite persistence of populations for long-lived species (Silvertown et al., 1996), even if regeneration may be relatively unimportant compared to adult survival for long-term population stability (Hunt, 2001). Regardless of this, it is questionable if sites such as the interior of Buttonland Swamp can maintain populations following catastrophe (e.g., tornado, ice storm, disease, herbicide drift) because of the limited opportunity for recruitment in impounded conditions (Middleton, 2002). There is little question that the interior of Buttonland Swamp has limited population stability because of the failure of some early life history stages (seedlings and saplings), while the adult trees are stressed and declining (Appendix E).

Management to improve the survival of all life history stages does not necessarily increase the population numbers of a species (Hanley et al., 2019). While sensitivity analysis to determine management strategies to increase numbers of all life stages can be useful in support of populations (Hanley et al., 2019), such an analysis was beyond the scope of this research. Stable population growth rates (λ) are close to “1” (Smith & Ludwig, 2011), so the projection modeling in this study showed that the study populations of *T. distichum* along the Cache River are not predicted to either increase or decrease in the future, except for the interior of Buttonland Swamp (Table 3).

An environment where the trees are stressed and there is no regeneration could risk a reduction in future population numbers. This situation poses the question of best management practices if trees in settings with little or no regeneration. From a management perspective, swamps in dry and medium-dry environments (e.g., Eagle Pond) projected the

Table 3

Stochastic model to project *T. distichum* populations numbers for life stages (seeds, seedlings, saplings, trees) after 1, 50, and 500 years listed by site with the lowest to highest tree number in the projected population. This model population of 50 trees was tracked from Years 1, 50, and 500. Life stage frequency values were assessed for each site including for seeds (fecundity), seedling, saplings, and trees as a mean of values in plots over the years of study and assessed using the popbio package in R (R Core Team, 2022). Transition values were assigned as: seed = $f/50$, seedling = $s_0/500$, sapling = $s_1/10,000$, with the chance of tree mortality = $1/50,000$. These transition probabilities were multiplied by the stage frequency to give the number of individuals likely to transition to the next life history stage.

Year 1 Site	Water category	Years of study	Seed	Seedling	Sapling	Tree	Tree number after 1 year	Eigenvalue
Crawford Tract - L	Dry	1990s	296.9400	6.056200	2.107600	0.99998	49.99900	1.00000
Crawford Tract - L	Dry	2016–2021	303.0000	0.000000	2.100000	0.99998	49.99900	1.00000
Crawford Tract - M	Dry	1990s	296.9400	6.056200	2.107600	0.99998	49.99900	1.00000
Wildcat Bluff	Wet	2006–2021	298.7959	4.202700	2.102847	0.999983	49.99913	0.99993
Section 8 Woods	Medium-dry	2006–2021	298.4146	4.583975	2.102839	0.999991	49.99953	0.99990
Snake Hole	Medium-wet	2006–2021	299.6670	3.331338	2.103312	0.999993	49.99966	0.99991
Crawford Tract - M	Dry	2016–2021	298.9398	4.057667	2.105053	0.999994	49.99970	0.99995
Deer Pond	Medium-dry	2006–2021	298.2151	4.783212	2.103307	0.999998	49.99992	0.99991
Heron Pond	Medium-dry	2006–2021	298.1520	4.846100	2.103782	0.999998	49.99992	0.99994
Eagle Pond	Dry	2019–2021	297.5460	5.450200	2.107579	1.000001	50.00005	0.99999
Year 50								
Site	Water category	Years of study	Seed	Seedling	Sapling	Tree	Tree number after 50 years	
Crawford Tract - L	Dry	1990s	296.6491	6.048016	2.700607	0.999001	49.95003	
Crawford Tract - L	Dry	2016–2021	302.7032	0.000000	2.100000	0.999001	49.95003	
Crawford Tract - M	Dry	1990s	296.6491	6.048016	2.700607	0.999001	49.95003	
Wildcat Bluff	Wet	2006–2021	298.5430	4.198992	2.257094	0.999137	49.95683	
Section 8 Woods	Medium-dry	2006–2021	298.2817	4.581640	2.270717	0.999546	49.97731	
Snake Hole	Medium-wet	2006–2021	299.5725	3.330492	2.245464	0.999679	49.98394	
Crawford Tract - M	Dry	2016–2021	298.8646	4.056474	2.369378	0.999744	49.98721	
Deer Pond	Medium-dry	2006–2021	298.2041	4.782605	2.307419	0.999963	49.99816	
Heron Pond	Medium-dry	2006–2021	298.1429	4.84543	2.340258	0.999970	49.99851	
Eagle Pond	Dry	2019–2021	297.5981	5.449506	2.640414	1.000182	50.00908	
Year 500								
Site	Water category	Years of study	Seed	Seedling	Sapling	Tree	Tree number after 500 years	
Crawford Tract - L	Dry	1990s	293.9912	5.993827	8.119454	0.990050	49.502485	
Crawford Tract - L	Dry	2016–2021	299.9911	0.000000	2.100000	0.990050	49.502485	
Crawford Tract - M	Dry	1990s	293.9912	5.993827	8.119454	0.990050	49.502485	
Wildcat Bluff	Wet	2006–2021	296.3633	4.168335	3.667324	0.991844	49.592175	
Section 8 Woods	Medium-dry	2006–2021	297.6385	4.571761	3.808073	0.997399	49.869925	
Snake Hole	Medium-wet	2006–2021	299.3163	3.327644	3.547740	0.998832	49.941595	
Crawford Tract - M	Dry	2016–2021	299.3788	4.063454	4.793143	1.001481	50.074025	
Deer Pond	Medium-dry	2006–2021	299.3185	4.800479	4.180165	1.003716	50.185820	
Heron Pond	Medium-dry	2006–2021	299.4683	4.866970	4.510515	1.004434	50.221715	
Eagle Pond	Dry	2019–2021	301.6994	5.524608	7.546842	1.014014	50.700700	

most change in tree number after one, fifty, and five hundred years (Table 3). With no changes in water management, climatic fluctuation still might support periodic recruitment (Lertzman, 1995) in the interior of Buttonland Swamp.

In the context of future regeneration success in populations such as *T. distichum*, flooding status may be the most important factor (Middleton, 1999, 2003), noting that parts of the Midwest may be getting wetter as precipitation and flooding increase (Mallakpur & Villarini, 2015). Local precipitation data from a nearby weather station suggest a similar trend for the Cache River wetlands. Normal annual precipitation increased in 2011–2020 in comparison to 1991–2000 and 2001–2010 (119.3 to 135.2 cm year⁻¹, respectively; Table 2), making it likely that these swamps may be more flooded in the near-term future, particularly because their elevations are dropping (Middleton, 2023a).

5. Conclusions

Maintaining drier swamp environments for *Taxodium distichum* populations may provide higher levels of support from early life history stages (seeds, seedlings, saplings) for this foundational species of southeastern swamps. Projection models of *T. distichum* populations in southern Illinois reached stable population levels faster and at slightly higher population numbers in swamps at the drier end of the moisture gradient. This finding suggests that from a management perspective, periodic drought or drawdown may be helpful in supporting this foundational species. Precipitation has increased in southern Illinois during recent decades and elevation has declined on floodplains, so management might benefit from careful observations of water environments in the future. Management to maintain environments that foster seed-to-seedling-sapling-tree transitions at critical times (Hunt, 2001), could

also support *T. distichum* population growth and stability in these vital southern Illinois wetlands. The high and prolonged water levels in the interior of Buttonland Swamp may contribute to the declining health and stress of *T. distichum* trees in the future. This study is of broad value in understanding dominant tree responses to environments in forests where multiple stressors impact forests (e.g., historical hydrological alteration, logging, development, nutrient deprivation, and saltwater intrusion (Shaffer et al., 2009, Keim et al., 2013) because these factors also may interfere with the success of earlier life history stages that support forest population stability. This multi-decadal study of Cache River swamps in a range of moisture environments provides information on the best water management to support all life history stages of *T. distichum* and support population stability to aid in the conservation of these vital wetlands.

CRedit authorship contribution statement

Beth A. Middleton: Conceptualization, Data curation, Funding acquisition, Investigation, Methodology, Resources, Formal analysis, Validation, Visualization, Writing – original draft. **Darren J. Johnson:** .

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.

Data availability

Data will be released in ScienceBase at the time of publication.

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Appendix A. Mean water depths at site plots during summer and winter surveys. Surveys were conducted in Deer Pond, Eagle Pond, Wildcat Bluff, Snake Hole, and Section 8 Woods from 2006 to 2022. Surveys of Crawford Tract and Eagle Pond on the edge of Buttonland Swamp were conducted from 2016 to 2022 and 2019 to 2022, respectively. Winter water depths were measured on the day of litter harvest. Summer and winter water depths were measured on the day of vegetation sampling vs. End-of-year leaf-litter pickup, respectively. ENSO event by site was determined from information from NCAR (2023). A site was designated as influenced or not influenced (yes/no) by an ENSO event if an El Niño, La Niña, or a sequential El Niño to La Niña event occurred six months prior to the summer survey

Site	Year	Summer	Winter	Summer depth	Winter depth	El Niño	La Niña	El Niño to La Niña
Crawford Tract	2016	7/27/2016	12/08/2016	0	0	No	No	No
	2017	8/24/2017	11/21/2017	0	0	No	No	No
	2018	7/05/2018	12/19/2018	0	0	No	No	No
	2019	9/22/2019	11/26/2019	0	0	No	No	No
	2020	6/24/2020	12/1/2020	0	0	No	No	No
	2021	6/28/2021	11/30/2021	0	0	No	No	No
	2022	5/24/2022	11/29/2022	6.4	0	No	No	No
Eagle Pond	2020	6/24/2020	12/7/2020	3.8	Not measured	No	No	No
	2021	6/28/2021	11/30/2021	0	3.2	No	No	No
	2022	5/24/2022	11/29/2022	34.2	0	No	No	No
Deer Pond	2006	8/8/2006	12/14/2006	2.2	38.9	No	Yes	Yes
	2007	9/16/2007	12/3/2007	<0.1	40.9	No	No	No
	2008	9/6/2008	12/2/2008	0.1	0	No	No	No
	2009	8/25/2009	11/29/2009	35.9	43	No	No	No
	2010	8/24/2010	11/27/2010	0	0.1	No	No	No
	2011	8/17/2011	11/30/2011	23.8	56.4	Yes	No	No
	2012	7/31/2012	11/14/2012	0	0	Yes	No	No
	2013	8/12/2013	11/1/2013	27.0	19.4	No	No	No
	2014	7/9/2014	11/20/2014	23.0	0	No	No	No
	2015	8/28/2015	11/20/2015	28.2	39.6	No	No	No
	2016	7/25/2016	12/7/2016	33.0	6.2	No	No	No
	2017	8/25/2017	11/20/2017	0	0	No	No	No
	2018	7/5/2018	12/8/2018	17.6	Not measured	No	No	No
	2019	9/23/2019	11/26/2019	0	10.4	No	No	No
	2020	6/25/2020	12/7/2020	21.8	23.9	No	No	No
	2021	6/26/2021	11/29/2021	10.4	0	No	No	No
	2022	5/27/2022	11/29/2022	17.0	0	No	No	No
Snake Hole	2006	8/9/2006	12/14/2006	26.4	59.9	No	No	No
	2007	9/16/2007	12/3/2007	0.1	43.6	No	No	No
	2008	9/6/2008	12/2/2008	27.3	26.5	No	No	No

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Site	Year	Summer	Winter	Summer depth	Winter depth	El Niño	La Niña	El Niño to La Niña
Wildcat Bluff	2009	8/26/2009	11/29/2009	43.0	58.4	No	No	No
	2010	8/24/2010	11/28/2010	21.0	12.4	No	Yes	Yes
	2011	8/16/2011	11/30/2011	30.8	89.8	No	No	No
	2012	8/1/2012	11/15/2012	<0.1	4.9	No	No	No
	2013	8/12/2013	11/2/2013	16.6	16.8	No	No	No
	2014	7/9/2014	11/21/2014	32.0	30.7	No	No	No
	2015	8/28/2015	11/20/2015	36.0	71.5	Yes	No	No
	2016	7/26/2016	12/7/2016	28.8	19.3	Yes	No	No
	2017	8/25/2017	11/21/2017	2.3	31.2	No	No	No
	2018	7/5/2018	12/12/2018	33.0	Not sampled	No	No	No
	2019	9/24/2019	11/26/2019	27.4	40.8	No	No	No
	2020	6/25/2020	12/2/2020	39.2	30.8	No	No	No
	2021	6/29/2021	12/1/2021	29.0	25.3	No	No	No
	2022	5/24/2022	11/29/2022	35.5	<0.1	No	No	No
	2006	8/11/2006	12/14/2006	22.7	39.5	No	No	No
	2007	9/16/2007	12/3/2007	0	23.1	No	No	No
	2008	9/7/2008	12/2/2008	2.0	0.1	No	No	No
	2009	8/26/2009	11/29/2009	27.0	42.2	No	No	No
	2010	8/25/2010	11/28/2010	0	0.1	No	Yes	Yes
	2011	8/17/2011	11/30/2011	36.0	53	No	No	No
	2012	8/1/2012	11/15/2012	0	0	No	No	No
	2013	8/12/2013	11/2/2013	22.0	10.3	No	No	No
	2014	7/9/2014	11/20/2014	35.6	15.4	No	No	No
	2015	8/29/2015	11/20/2015	35.1	42.3	Yes	No	No
	2016	7/26/2016	12/7/2016	38.4	30.5	Yes	No	No
Heron Pond	2017	8/23/2017	11/21/2017	19.6	37.4	No	No	No
	2018	7/4/2018	12/18/2018	41.2	Not measured	No	No	No
	2019	9/23/2019	12/11/2019	10.4	34.1	No	No	No
	2020	Not sampled	12/2/2020	Not sampled	23.5	No	No	No
	2021	6/29/2021	12/1/2021	12.2	15.2	No	No	No
	2022	Not sampled	Not sampled			No	No	No
	2006	8/11/2006	12/15/2006	23.1	32.8	No	No	No
	2007	9/16/2007	12/3/2007	0.1	17.5	No	No	No
	2008	9/7/2008	12/2/2008	9.2	4.3	No	No	No
	2009	8/27/2009	11/29/2009	31.3	40.4	No	No	No
	2010	8/24/2010	11/28/2010	5.8	1.3	No	Yes	Yes
	2011	8/17/2011	11/30/2011	18.6	37.8	No	No	No
Section 8 Woods	2012	8/1/2012	11/14/2012	0	<0.1	No	No	No
	2013	8/11/2013	11/1/2013	17.0	18.4	No	No	No
	2014	7/8/2014	11/20/2014	19.8	1.8	No	No	No
	2015	8/29/2015	11/20/2015	6.0	7.7	Yes	No	No
	2016	7/25/2016	12/8/2016	2.8	0.1	Yes	No	No
	2017	8/24/2017	11/21/2017	<0.1	5.6	No	No	No
	2018	7/4/2018	12/18/2018	22.8	Not measured	No	No	No
	2019	9/23/2019	12/11/2019	2.0	24.8	No	No	No
	2020	6/25/2020	12/2/2020	17.2	27.6	No	No	No
	2021	6/26/2021	11/30/2020	12.8	0.1	No	No	No
	2022	Not sampled	Not sampled	Not sampled	Not sampled	No	No	No
	2006	9/9/2006	12/14/2006	1.0	69.5	No	No	No
	2007	9/16/2007	12/3/2007	0	2.6	No	No	No
	2008	9/6/2008	12/2/2008	0	0	No	No	No
	2009	8/26/2009	11/29/2009	25.6	34.9	No	No	No
	2010	8/23/2010	11/28/2010	0.1	0.1	No	Yes	Yes
	2011	8/17/2011	12/17/2011	46.0	80.2	No	No	No
	2012	8/1/2012	11/14/2012	0	0	No	No	No
	2013	8/11/2013	11/1/2013	1.3	0	No	No	No
	2014	7/9/2014	11/20/2014	4.3	9.2	No	No	No
	2015	8/29/2015	Not sampled	16.2	Not sampled	Yes	No	No
	2016	1/1/2016	12/7/2016	11.4	10.3	Yes	No	No
	2017	8/24/2017	11/21/2017	<0.1	13.6	No	No	No
	2018	7/4/2018	12/19/2018	25.3	Not sampled	No	No	No
	2019	9/22/2019	12/10/2019	<0.1	35.0	No	No	No
	2020	6/23/2020	12/7/2020	20.2	25.6	No	No	No
	2021	6/29/2021	11/30/2021	0.2	0.3	No	No	No
	2022	Not sampled	11/28/2022	Not sampled	0	No	No	No

Appendix B. Climate normal annual air temperatures (°C) and total annual precipitation (cm) by decade including 1991–2000, 2001–2010, 2011–2020. Annual air temperature and precipitation data are from the Carbondale Southern Illinois Airport, Illinois, and the Carbondale sewage plant (USW00093810 and USC00111265, respectively). Downloaded on June 23, 2022, from the NCEI Climate Normal tool (NCEI, 2022; from Middleton, 2023a)

Climate variable	Decade		
	1991–2000	2001–2010	2011–2020
Normal decadal mean annual air temperature	13.72	13.44	13.72
Normal decadal maximum annual air temperature	19.78	19.39	19.50
Normal decadal minimum annual air temperature	7.61	7.50	8.00
Normal decadal mean July air temperature	24.44	25.17	25.89
Normal decadal maximum July air temperature	29.56	30.89	31.61
Normal decadal minimum July air temperature	19.28	19.39	20.17
Normal decadal mean January air temperature	2.00	0.50	0.67
Normal decadal maximum January air temperature	6.50	5.44	5.50
Normal decadal minimum January temperature	−2.44	−4.39	−4.17
Normal decadal total annual precipitation	119.3	121.7	135.2

Appendix C. Definitions: Variable abbreviations and definitions for climate and environment variables (Adapted from Middleton, 2020)

Variable name (units)	Variable abbreviation	Variable type	Definition or equation
Normal decadal total annual precipitation (mm)	Normal annual precipitation	Climate Normal	The 10-year average of total annual precipitation by decade
Normal decadal mean annual air temperature (°C)	Normal annual temperature	Climate Normal	The 10-year average of mean annual temperature by decade
Normal decadal mean maximum annual air temperature (°C)	Normal maximum temperature	Climate Normal	The 10-year average of mean maximum annual temperature by decade
Normal decadal mean minimum annual air temperature (°C)	Normal minimum temperature	Climate Normal	The 10-year average of mean minimum annual temperature by decade
Normal decadal mean July air temperature (°C)	Normal July temperature	Climate Normal	The 10-year average of mean July temperature by decade
Normal decadal maximum July air temperature (°C)	Normal maximum July temperature	Climate Normal	The 10-year average of mean maximum July temperature by decade
Normal decadal minimum July air temperature (°C)	Normal minimum July temperature	Climate Normal	The 10-year average of mean minimum July temperature by decade
Normal decadal mean January air temperature (°C)	Normal January temperature	Climate Normal	The 10-year average of mean January temperature by decade
Normal decadal maximum January air temperature (°C)	Normal maximum January temperature	Climate Normal	The 10-year average of mean maximum January temperature by decade
Normal decadal minimum January temperature (°C)	Normal minimum January temperature	Climate Normal	The 10-year average of mean minimum January temperature by decade
Day-of-visit water depth (cm)	Water depth	Environmental	Plot water depth as measured on the day of a site visit

Appendix D. Mean annual litter weight of reproductive biomass ($\text{g m}^{-2} \text{y}^{-1} \pm \text{S.E.}$) in southern Illinois swamps including Crawford Tract and Eagle Pond (after 2016 to 2022) and other swamps along the Cache River (~2006–2022) in Decade 1 and Decade 2 (2006–2011 and 2012–2021, respectively). Note that frequency data was used in stochastic modeling

Site	Sample number	Collection years	Reproductive biomass $\pm$ S.E Decade 1	Reproductive biomass $\pm$ S.E Decade 2
Crawford Tract	90	2017–2021	n.a.	833.0 $\pm$ 555.6
Deer Pond	300	2006–2021	58.3 $\pm$ 27.0	58.3 $\pm$ 20.6
Eagle Pond	30	2020–2021	n.a.	330.3 $\pm$ 183.7
Heron Pond	300	2006–2021	120.0 $\pm$ 56.1	107.5 $\pm$ 54.1
Section 8 Woods	300	2006–2021	100.3 $\pm$ 58.8	64.6 $\pm$ 26.1
Snake Hole	300	2006–2021	141.9 $\pm$ 39.9	156.1 $\pm$ 75.0
Wildcat Bluff	270	2006–2021	38.9 $\pm$ 13.5	139.9 $\pm$ 63.1

Appendix E. Assessment of *T. distichum* tree health in the interior of Buttonland Swamp based on a survey of live crown ratio, leaf condition, tree stress, height, seedling/sapling presence using ten randomly selected adult trees in the interior of the swamp



Tree 1
Latitude/Longitude 37.29698 N / – 89.05297 W
Height (m): 17
Live crown ratio: 50 %
Herbicide % foliage: 100 %
Leaf curl: slight
Leaf epistasy: slight
Leaf yellow: slight
Leaf Munsell color:Hue: 5
(GY)
Value: 4
Chroma: 7
Epicormic sprouting: no
Tree health metric: stressed
Cypress seedlings/saplings: no/no
Lemna % cover: 0
Notes: Tree near the northern canoe launch



Tree 2
Latitude/Longitude: 37.29665 N / –89.05334 W
Height (m): 19
Live crown ratio: 25 %
Herbicide % foliage: 100 %
Leaf curl: slight
Leaf epistasy: slight
Leaf yellow: slight
Leaf Munsell color:Hue: 5
(GY)
Value: 4
Chroma: 6
Epicormic sprouting: no
Tree health metric: decline
Cypress seedlings/saplings: no/no
Lemna % cover: 0
Notes:

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Tree 3

Latitude/Longitude: 37.29615 N / -89.05188 W
Height (m): 16
Live crown ratio: 20 %
Herbicide % foliage: 100 %
Leaf curl: slight
Leaf epistasy: slight
Leaf yellow: slight
Leaf Munsell color:Hue: 5
(GY)
Value: 4.5
Chroma: 6
Epicormic sprouting: no
Tree health metric: decline
Cypress seedlings/saplings: no/no
Lemna % cover: < 1
Notes:

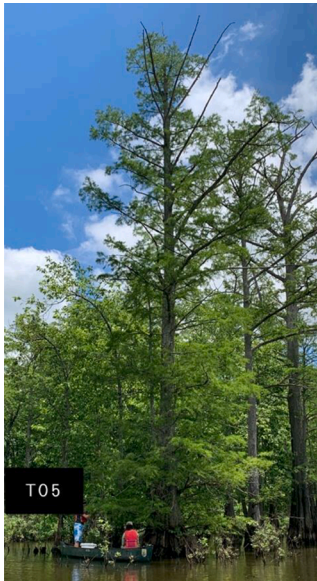


Tree 4

Latitude/Longitude: 37.29593 N / -89.05179 W
Height (m): 15
Live crown ratio: 30 %
Herbicide % foliage: 100 %
Leaf curl: slight
Leaf epistasy: slight
Leaf yellow: slight
Leaf Munsell color:Hue: 5
(GY)
Value: 4
Chroma: 7
Epicormic sprouting: no
Tree health metric: decline
Cypress seedlings/saplings: no/no
Lemna % cover: 0
Notes: No epicormic sprouting from trunk in the trees.

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Tree 5
Latitude/Longitude: 37.29568 N / -89.05152 W
Height (m): 19
Live crown ratio: 45 %
Herbicide % foliage: 100 %
Leaf curl: slight
Leaf epistasy: slight
Leaf yellow: slight
Leaf Munsell color:Hue: 5
(GY)
Value: 4.5
Chroma: 6
Epicormic sprouting: no
Tree health metric: decline
Cypress seedlings/saplings: no/no
Lemma % cover: < 1Notes: Small cypress tree
(4 m tall), 20 m from Tree 5.



Tree 6
Latitude/Longitude: 37.29508 N / -89.05147 W
Height (m): 15.5
Live crown ratio: 15 %
Herbicide % foliage: 100 %
Leaf curl: slight
Leaf epistasy: slight
Leaf yellow: slight
Leaf Munsell color:Hue: 5
(GY)
Value: 4.5
Chroma: 6
Epicormic sprouting: no
Tree health metric: decline
Cypress seedlings/saplings: no/no
Lemma % cover: 0Notes: Six small cypress trees
(10 m tall), 25 m from Tree 6.

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**Tree 7**

Latitude/Longitude: 37.29510 N / -89.05082 W

Height (m): 13

Live crown ratio: 60 %

Herbicide % foliage: 100 %

Leaf curl: slight

Leaf epistasy: slight

Leaf yellow: slight

Leaf Munsell color:Hue: 5

(GY)

Value: 4

Chroma: 5

Epicormic sprouting: no

Tree health metric: stressed

Cypress seedlings/saplings: no/no

Lemma % cover: 0Notes: Two small trees

(10 m tall), 20 and 75 m away from Tree 7; Nightjar in Tree 7.

**Tree 8**

Latitude/Longitude: 37.29443 N / -89.05030 W

Height (m): 18

Live crown ratio: 17 %

Herbicide % foliage: 100 %

Leaf curl: slight

Leaf epistasy: slight

Leaf yellow: slight

Leaf Munsell color:Hue: 5

(GY)

Value: 4.5

Chroma: 6

Epicormic sprouting: no

Tree health metric: relatively healthy

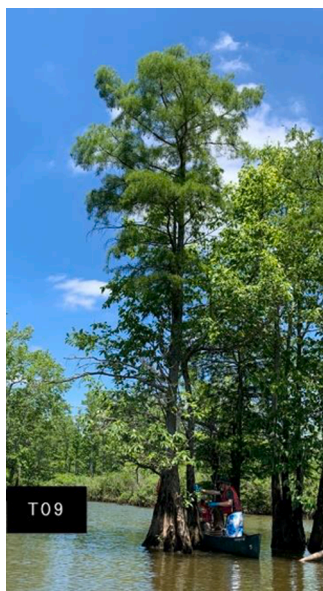
Cypress seedlings/saplings: no/no

Lemma % cover: 0Notes: Three small trees

(12 m tall), 12, 13, 15 m from Tree 7.

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**Tree 9**

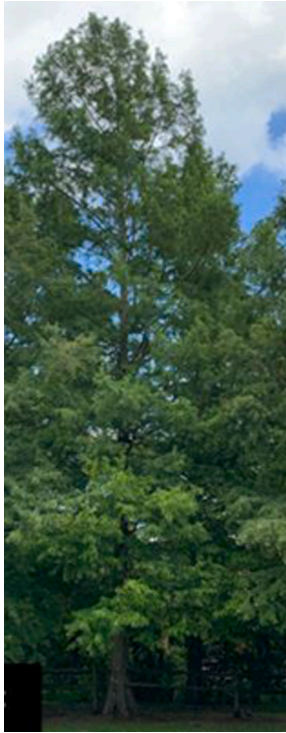
Latitude/Longitude: 37.29464 N / -89.05009 W
 Height (m): 11
 Live crown ratio: 80 %
 Herbicide % foliage: 100 %
 Leaf curl: slight
 Leaf epistasy: slight
 Leaf yellow: slight
 Leaf Munsell color:Hue: 5
 (GY)
 Value: 5
 Chroma: 6
 Epicormic sprouting: no
 Tree health metric: relatively healthy
 Cypress seedlings/saplings: no/no
Lemma % cover: 0
 Notes: Small trees
 (13 m tall), 40 m away from Tree 8. Near river channel.

**Tree 10 – 2nd cypress in foreground from left**

Latitude/Longitude: 37.29452 N / -89.04967 W
 Height (m): 19.5
 Live crown ratio: 90 %
 Herbicide % foliage: 100 %
 Leaf curl: slight
 Leaf epistasy: slight
 Leaf yellow: slight
 Leaf Munsell color:Hue: 5
 (GY)
 Value: 4
 Chroma: 7
 Epicormic sprouting: no
 Tree health metric: relatively healthy
 Cypress seedlings/saplings: no/no
Lemma % cover: 0
 Notes: Split trunk. Near river channel.

Buttonland Swamp south of northern canoe launch**Summary Trees 1–10**

Latitude/Longitude: 37.29 N / -89.05 W
 Mean Height (m): 16.3
 Mean Live crown ratio: 43.2 %
 Herbicide % foliage: 100 %
 Leaf curl: slight
 Leaf epistasy: slight
 Leaf yellow: slight
 Leaf Munsell color:Hue: 5
 (GY)
 Mean Minimum to Maximum Value: 4.1 to 4.5
 Mean Minimum to Maximum Chroma: 5.8 to 6.6
 Epicormic sprouting: no
 Tree health metric: decline = 5, stressed = 2, relatively healthy = 3
 Cypress seedlings/saplings: no/no
Lemma % cover: < 1

**WARC Tree – Lafayette, Louisiana**

Latitude/Longitude: 30.22547 N / -92.04590 W Height (m)

: 15 m

Live crown ratio: 90

Herbicide % foliage: 0 %

Leaf curl: no

Leaf epistasy: no

Leaf yellow: not yellow

Leaf Munsell color:Hue: 5
(GY)

Value: 4

Chroma: 4

Epicormic sprouting: no

Tree health metric: healthy

Cypress seedlings/saplings: no/no

Lemna % cover: 0

Notes: Planted tree on unflooded property of Wetland and Aquatic Research Center, Lafayette, LA.

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