CASE STUDY

Swamp Reforestation in Coastal Louisiana, USA Exposes Landscape Scale Differences in Survival and Growth Across Two Hydrologically Restored Regions

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ABSTRACT

Coastal swamp forests fill ecological niches that provide valuable environmental services, yet their extent across coastal Louisiana has diminished from logging and saltwater intrusion. Previous reforestation attempts yielded mixed results, attributed to hydrological disturbances leaving regions of the coast vulnerable to environmental stress. Environmental conditions may be improving. In the Pontchartrain Estuary, the Caernarvon Freshwater Diversion (CFD) has pulsed Mississippi River water into lower estuary wetlands since 1991, and the Mississippi River Gulf Outlet Canal closure in 2009 lowered soil salinity further upstream in the estuary on the Maurepas Landbridge (MLB). However, how hydrological restoration impacts habitat restoration remains undocumented. The Pontchartrain Conservancy planted ~80,000 saplings from 2011–2021 in two regions (CFD, MLB) within the estuary. Species included *Taxodium distichum* (L.) Rich, *Nyssa aquatica* L., *Nyssa sylvatica* Marshal var. *biflora, Acer rubrum* L. var. *drummondii* (Hook. & Arn. ex Nutt.) Sarg., and *Fraxinus pennsylvanica,* though > 80% of the saplings planted were *T. distichum*. We assessed survival and growth for 7.3% of saplings. Survival differed between regions; survival was lower around the CFD (63%) and higher on the MLB (82%). Growth rates also differed; growth was higher in CFD (height: 0.18–0.69 m/yr; diameter 0.66–1.62 cm/yr) and lower in MLB (height: 0.00–0.44 m/yr; diameter 0.05–0.18 cm/yr). Growth varied temporally between areas, but trends were similar. Results indicate 1) hydrological restoration benefits coastal swamp reforestation, and 2) river water increased growth. As habitat restoration, including swamp reforestation, scales-up in Louisiana, these outcomes help resource managers and planners refine restoration goals and target areas to maximize restoration success.

Keywords: bald cypress, coastal forests, swamp restoration, tree monitoring, water tupelo

Coastal forested wetlands fill ecological niches that provide valuable environmental services. These forests thrive along the environmental continuum within estuaries and along coasts and create specific ecosystem types. Forested estuarine and coastal ecosystems provide wildlife habitat, improve water quality, sequester and store carbon, and offer economical and recreational opportunities (i.e., fishing, crawfishing, hunting, trapping, timber production, and ecotourism) (Engle 2011, Hansen and Nestlerode 2013, Holcomb et al. 2015). In southeast (SE) Louisiana, coastal swamp forests constitute the dominant coastal forest type, though their spatial extent has diminished since the turn of the twentieth century.

Historical logging reduced virgin *Taxodium distichum*–*Nyssa aquatica* (bald cypress–water tupelo) swamp forests by an estimated 526,091 ha in Louisiana (Conner and Toliver 1990a). Logging and other human activities during this time (the 1890s–1950s) created landscape-altering features (i.e., railroad ridges, spoil banks, shallow ruts, ditches, canals, etc.) that disrupted natural hydrological processes (Mancil 1980) and guided salt water further up the estuary during tropical storms (Keddy et al. 2007). Spoil banks along the canals also inhibited natural drainage, creating impoundments that held water longer and contributed to the gradual salinization of an area (Turner et al. 1994). River management (i.e., levees) prevented spring floods (oxygenated freshwater) from occurring, which further amplified regional salinization. Large shipping channels

Color version of this article is available online at: https://er.uwpress.org.

doi:10.3368/er.42.3.205
Ecological Restoration Vol. 42, No. 3, 2024
ISSN 1522-4740 E-ISSN 1543-4079

Restoration Recap

- Coastal swamp forest extent declined in coastal Louisiana due to past logging and systemic changes in surface water hydrology, leading to concomitant declines of numerous ecological services.
- Partial hydrological restoration may have improved reforestation success.
- About 80,000 saplings, including five different species of trees, were planted from 2011–2021 in two environmentally distinct areas of the Pontchartrain Estuary (Louisiana, USA) and were assessed for survival and growth rate differences.
- Saplings planted in the lower estuary exhibited lower survival than saplings planted in the upper estuary, and saplings planted near a freshwater source (i.e., river diversion) showed higher growth rates than trees planted without one.
- As habitat reforestation scales-up globally, these outcomes can help reforestation personnel refine project goals, predict outcomes, and target areas that maximize reforestation success.

like the Mississippi River Gulf Outlet (MRGO) became conduits for even more saltwater intrusion into the upper estuary (Shaffer et al. 2009). During the 1970s and 1980s, subsequent generations of *T. distichum* failed to thrive in these saltier environments, and longer hydroperiods (inundation) reduced seed germination and natural coastal swamp forest regeneration (Conner et al. 1986, Conner and Toliver 1990b). All of these alterations resulted in a coastal swamp forest ecosystem reduced in scale and ecosystem services provided.

The need for coastal swamp reforestation in Louisiana was known by the mid-twentieth century (Bull 1949) and began around the Lac des Allemands area of Barataria Estuary (Rathborne 1951, Conner et al. 2012) (Figure 1). The Joseph Rathborne Land Company (Harvey, LA) planted ~1,000,000 *T. distichum* saplings on company land between 1949–1950. Saplings survived a few years (~90%), and a second planting of 150,000 saplings in 1951 also had high survival (> 80%, Rathborne 1951, Conner et al. 2012). However, that effort may have been subject to nutria (*Myocastor coypu*) herbivory at a later date (Brown and Montz 1986), and since no other monitoring data are available, the outcomes and lessons learned from these early reforestation attempts are hard to determine.

T. distichum planting increased in the latter part of the twentieth century (1950s–1980s) as a component of research, not large-scale reforestation. Research focused on the effect of inundation and herbivory on growth, survival, and reproduction (Blair and Langlinias 1960, Krinard and Jonson 1976, Klimas 1987, Conner and Toliver 1987). During the mid-1990s, large-scale coastal swamp reforestation began again in SE Louisiana, but outcomes were mixed. Southeastern Louisiana University planted approximately 10,000 *T. distichum* saplings from 1993–95 on the Maurepas Landbridge (MLB) of the Pontchartrain Estuary (Figure 1). Sapling survival was 78% after two years (Myers et al. 1995), with higher survival and growth for protected and fertilized saplings. However, these saplings and an additional 72,000 planted in the same general area in the late 1990s experienced > 95% mortality, attributed to drought-induced high salinity and hurricane impacts (Shaffer et al. 2016).

Elevated porewater salinity and periodic pulses of even higher surface salinity have been consequences of altered hydrology, which limits the natural distribution of *T. distichum* and the potential for successful reforestation. Therefore, identifying and understanding *T. distichum* salt tolerance and intraspecific differences became a research priority in the 1990s to early 2000s. Outcomes from studies by Allen et al. (1994) and Conner (1994) indicated a salinity tolerance threshold between 8–10 ppt (100% mortality). However, Krauss et al. (2000) conducted a Louisiana field experiment using *T. distichum* seeds from a variety of sources (AL, FL, LA) and observed improved outcomes (higher growth) and greater salinity tolerance in genetically mixed *T. distichum* stands compared to monotypic ones. Conner and Inabinette (2005) expanded on this work and included a wider mix of seed sources. *T. distichum* seedlings from Louisiana were the best performers (63% survival) and withstood the highest salinity pulses (> 18 ppt).

In the Pontchartrain Estuary, the Caernarvon Freshwater Diversion (CFD) has intermittently pulsed Mississippi River water into the degraded contiguous wetlands of the lower estuary since 1991, and some research indicated that proximity to an oxygenated water source results in improved sapling survival and growth (Krauss et al. 2000). Further, the closure of the MRGO shipping channel in 2009 via a rock dam appeared to lower soil salinity in the upper estuary, including the MLB (CPRA 2022). However, whether these hydrological changes would result in better long-term reforestation outcomes remained unknown. The Pontchartrain Conservancy (PC) and its partners planted ~80,000 wetland tree saplings across two regions of the Pontchartrain Estuary of SE Louisiana from 2011–2021. The primary goal was to replant degraded coastal swamp forests; the monitoring objective was to determine whether there are differences in the survival and growth of planted saplings between the upper and lower regions of the estuary.

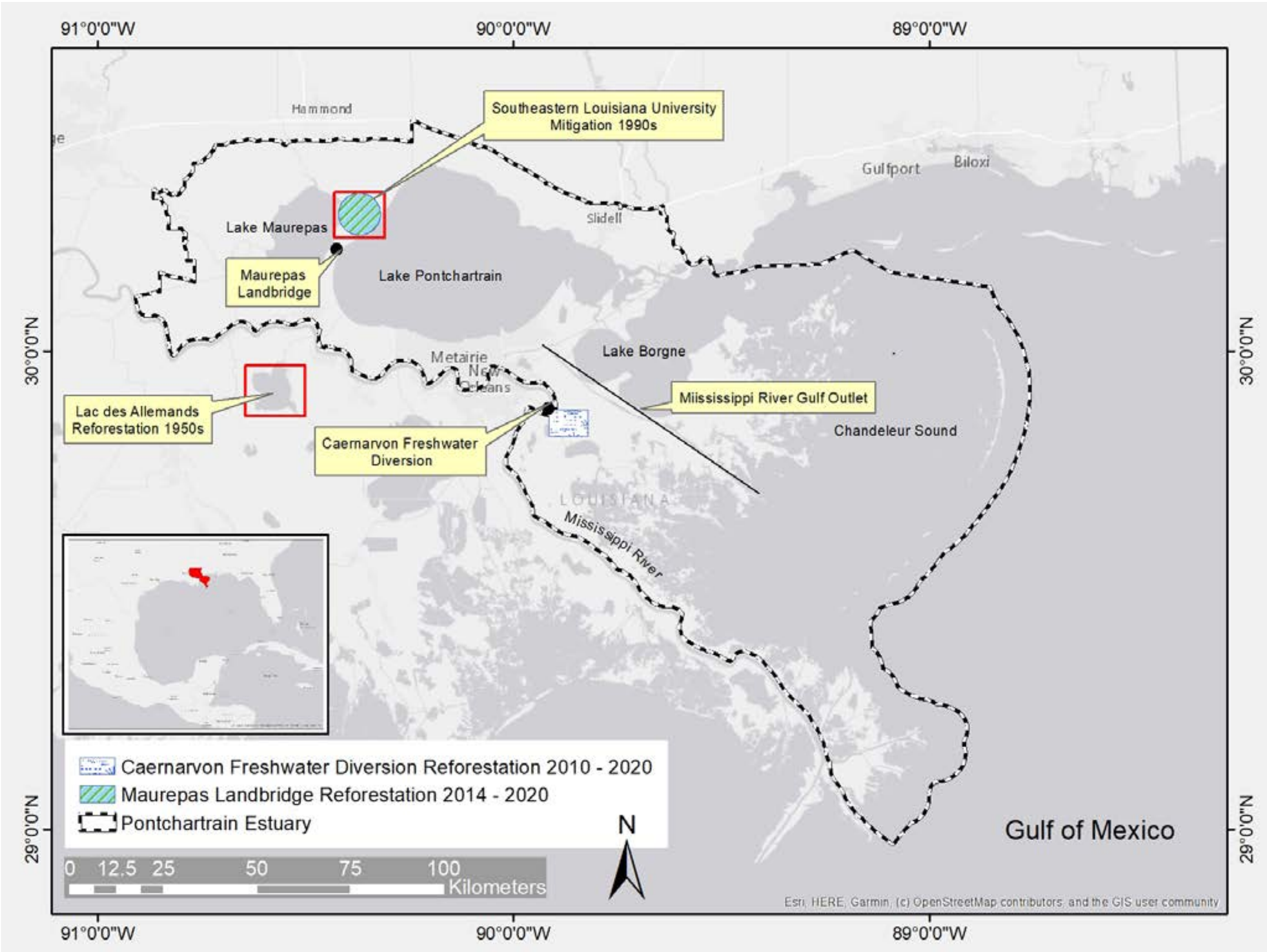

Figure 1. Pontchartrain Estuary and vicinity in Southeast Louisiana, USA, showing the areas of reforestation in this study. The Caernarvon Freshwater Diversion (CFD) is a freshwater diversion that opened in 1991 and releases Mississippi River water into lower estuary wetlands. At the Maurepas Landbridge (MLB), the 2009 Mississippi River Gulf Outlet Canal closure limited saltwater intrusion and lowered soil salinity.

Methods

Setting

The Pontchartrain Estuary is a 12,173 km² watershed and one of the largest estuarine systems along the Gulf of Mexico. It receives fresh water from rivers and streams in the upper estuary and saltwater from the Gulf (Figure 1). The watershed supports approximately 2.1 million people across a spectrum of rural, suburban, and urban land uses. Habitats across the estuary include pine upland, backwater swamps, tidal marshes, and barrier islands, with the 1,632 km² Lake Pontchartrain as the centerpiece.

The upper estuary contains several significant habitats, including longleaf pine upland forests, flatwood savannahs, and the Lake Maurepas region. The Lake Maurepas region and its adjacent wetlands contain alluvial river swamps. This region includes the MLB, an isthmus between Lakes Maurepas and Pontchartrain that is a critical landscape feature for reducing storm surge risks for regional communities and is completely cut off from the Mississippi River. The middle estuary, including Lake Pontchartrain and its surrounding wetlands, contains shoreline habitat, marshes, and submerged aquatic vegetation. The lower estuary includes a historic chenier, brackish and saline marshes, shallow bays, sounds, and the Chandeleur-Breton barrier island chain. The lower estuary also contains the CFD, a freshwater diversion that opened in 1991. It intermittently reconnects the waters of the leveed Mississippi River to its adjacent wetlands to reduce salinity further down the estuary (Das et al. 2012). The CFD does not impact wetlands in the upper estuary on or around the MLB.

These two regions represent two major conditions in coastal Louisiana and are geologically and hydrologically different. The CFD region (lower estuary) is connected to the Mississippi River delta and mimics historic riverine processes and current restoration initiatives through the operation of the diversion structure, while the MLB region

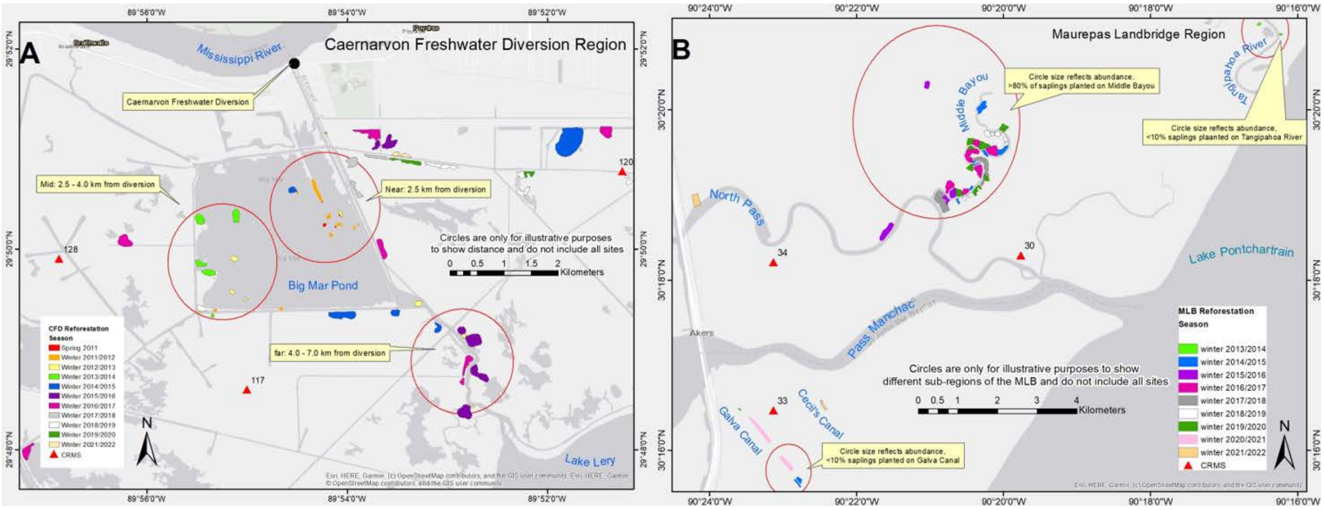

Figure 2. Regional maps of the Caernarvon Freshwater Diversion (CFD) reforestation area (A) and the Maurepas Landbridge reforestation area (B), including annual planting sites (cohorts), Coastwide Reference Monitoring Stations (CRMS) and sub-regional areas of interest (circles).

(upper estuary) is cut off from the delta and represents the status of much of coastal Louisiana over the last 100 years. These two regions also represent two types of hydrologic restoration: reconnecting the Mississippi River to its adjacent wetlands versus only limiting saltwater intrusion.

Reforestation Timeline

The CFD is capable of discharging up to 212 m^3/s into a shallow receiving basin. Actual discharge is much lower, typically less than 28 m^3/s (CPRA 2024). About 35% of CFD flow goes directly through the receiving basin; the rest flows into a network of bayous, canals, and lakes to Breton Sound and the open Gulf (Henkel et al. 2023). Given the intermittent operation of the CFD and low discharge, sediment deposition was low. The first mudflats emerged in the receiving basin circa 2004 (Lopez et al. 2014). Whether this new land, which eventually stabilized and vegetated, could sustain coastal swamp forests was unclear. Nevertheless, new land presented a novel opportunity to attempt coastal swamp reforestation within the footprint of a river diversion, and because river diversions are a tool used to combat land loss and habitat degradation across coastal Louisiana (Peyronnin et al. 2013) the outcomes of reforestation in this area can be impactful regionally, as well as globally.

Coastal swamp reforestation began in Winter 2010/2011 and followed an annual approach for the next decade. Each planting season (Winter), a predetermined number of tree saplings (cohort) were planted in the vicinity of the CFD across multiple sites with the help of community volunteers and commercial crews (Figure 2A). Over time, the scale and complexity of plantings, including species assemblages increased (Table 1). In short, plantings increased from 100s of saplings per season for the first four seasons to 1000s for subsequent seasons, because monitoring showed positive reforestation outcomes. Similarly, plantings initially consisted of 100% *T. distichum* saplings. In 2011/2012 other species such as *N. aquatica*, *Fraxinus pennsylvanica* (green ash), *Acer rubrum* var. *drummondii* (Drummond's maple), and *N. sylvatica* (swamp tupelo) were introduced to increase the biodiversity of plantings and better replicate a typical Louisiana coastal swamp forest tree assemblage. Furthermore, planting efforts initially concentrated on newly emerged land in the CFD outfall area (< 2.5 km from diversion, NEAR). Later planting sites were farther from the diversion (2.5–4.0 km, MID; 4.0–7.0 km, FAR). After several seasons of planting and monitoring across the CFD region, plantings expanded into another region in the upper estuary on the MLB. PC began reforestation on the MLB in 2013/2014 (Figure 2B); simultaneously, reforestation activities continued in the CFD region. By the end of the 2020/21 planting season, ~80,000 tree saplings in total were planted in equal proportion across the two regions (CFD: 39,635; MLD: 40,230).

Reforestation Protocols

We chose the CFD and MLB regions for reforestation because key environmental markers (i.e., soil salinity), landscape features (i.e., open land), and access (i.e., landowner permission) presented a unique opportunity for coastal swamp reforestation. When mean soil salinity is < 2.5 ppt, it generally indicates an area may be suitable for wetland trees (Allen et al. 1997); however, in coastal Louisiana, these areas may also experience spikes in salinity up to 8–10 ppt. Annually, before each planting season, PC staff 1) scouted potential planting sites, 2) tagged a statistically significant subset of saplings with aluminum numbered tags, and 3) recorded baseline data.

Baseline data were recorded by 1) measuring height in meters (m) from ground level to the tip of woody stems using a telescoping metric meter stick, and 2) by measuring

Table 1. Total number and number of trees by species planted in the Caernarvon Freshwater Diversion area and on the Maurepas Landbridge from 2010–11 through 2020–21. Almost 80,000 tree saplings were planted.

Year	Planted	Tagged	*T. distichum*	*N. aquatica*	*A. rubrum*	*N. sylvatica*	*F. pennsylvanica*
Caernarvon Freshwater Diversion region							
2010–11	375	38	375	—	—	—	—
2011–12	925	189	925	—	—	—	—
2012–13	500	113	345	100	—	—	55
2013–14	600	135	400	200	—	—	—
2014–15	10,000	776	7,500	1,000	750	750	—
2015–16	5,250	444	3,307	788	787	368	—
2016–17	8,400	1,135	7,400	500	250	250	—
2017–18	3,600	266	2,800	500	250	250	
2018–19	6,650	0	6,650	—	—	—	—
2019–20	1,635	0	1,635	—	—	—	—
2020–21	1,700	0	1,700	—	—	—	—
Subtotal	39,365	3,092	33,037	3,138	2,037	1,368	55
Maurepas Landbridge region							
2013–14	100	26	100	—	—	—	—
2014–15	12,000	1,058	9,000	960	1,800	240	—
2015–16	8,000	374	6,200	800	650	350	—
2016–17	6,500	888	5,000	870	420	210	—
2017–18	5,500	0	4,400	550	550	—	—
2018–19	2,350	274	1,850	—	—	—	500
2019–20	3,080	0	3,080	—	—	—	—
2020–21	2,700	0	2,700	—	—	—	—
Subtotal	40,230	2,620	32,330	3,180	3,420	800	500
Total	79,865	5,712	65,367	6,318	5,457	2,168	555

diameter at breast height (DBH; cm) at 1.42 m above ground level (Magarik 2021) using digital tree calipers. No DBH measurements were recorded if saplings were less than 1.42 m in height. Bright flagging was then attached to tagged saplings and tag numbers were written on the flagging as secondary markers in case aluminum tags were lost during transport. Finally, we mixed tagged saplings with untagged saplings at staging areas on each planting site.

Saplings were planted at a density of ~493/ha or approximately 3–5 m apart (Hillmann et al. 2019). Herbivory by *M. coypu* can be problematic in coastal Louisiana (Kinler et al. 1998), and research has shown that outfitting saplings with tree protectors/herbivory excluders results in higher survival and growth (Allen 1995). Therefore, volunteers also protected each sapling by enclosing them within 45-cm (diameter) tree protectors (TreePro, West Lafayette, IN) fastened to 1.80-meter bamboo poles. After planting, 1) tagged sapling locations were recorded with a Garmin GPSmap 65, 2) the top of the tagged tree protectors were sprayed with bright tree paint (another secondary marker) as a visual cue to indicate the tagged tree had been accounted for, and 3) the GPS locations, tag numbers, baseline data, and other ancillary information (i.e., date planted, grower, site name, etc.) were integrated into a project database.

Reforestation Monitoring

Monitoring of tagged saplings occurred annually and began approximately one-year post-planting. Starting in mid-December, tagged saplings were located in the field by PC staff via GPS locations, tags, and secondary markers. Survival, height, and DBH were recorded upon locating each tree as previously described. Saplings with lost tags that could be positively identified via secondary markers received new tags and new numbers that were changed in the database. If a tagged sapling was not found, it was designated Did Not Find (DNF) in the database and not included in survival/mortality calculations for that season. If a tagged sapling was DNF for two years consecutively, it was presumed dead, marked as such in the database, and accounted for in survival and mortality calculations. If a DNF sapling was found later, it was reintegrated into the database.

Data Analyses

T. distichum saplings accounted for 82% of all individuals planted. There is a strong positive correlation (r = 0.98) between *T. distichum* growth and the growth of the collective species group. Therefore, all species are included in our analyses. We analyzed total survival and growth (height, DBH) in the CFD through winter 2019/2020 and in the MLB through winter 2018/2019 because that was

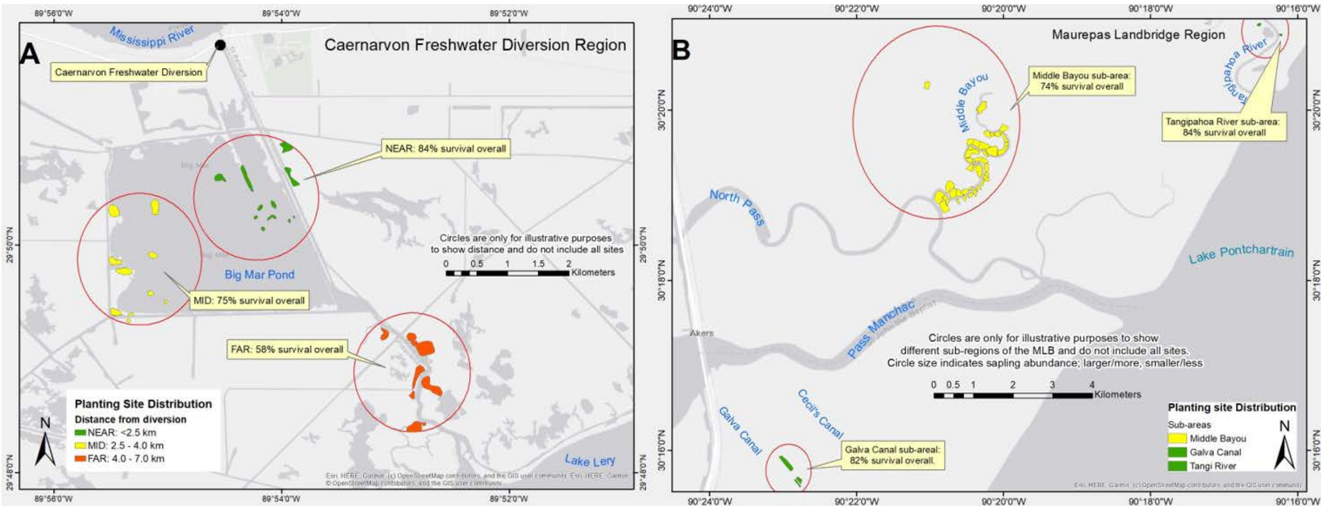

Figure 3. Around the Caernarvon Freshwater Diversion, tree/sapling survival for the 2019/20 monitoring season remained higher close to the diversion and lower farther from the diversion (A); on the Maurepas Landbridge, tree/sapling survival (2018/19) was less variable (B).

the last complete monitoring season in the MLB before COVID restrictions curtailed monitoring activities. We calculated survival by dividing the number of live-tagged saplings by the number of tagged saplings planted. We followed a similar procedure when constraining survival/mortality calculations to specific groups of saplings (i.e., sub-regions). Growth was calculated on individual trees by subtracting the previous seasons' measurement of height or DBH from the current measurement(s) and then averaging across planting cohorts for each region. We tested the normality of the data with Levene's test and then tested for statistical differences ($p < 0.05$) in growth (height, DBH) across age classes, planting cohorts, and regions. New tagged saplings were incorporated into the database annually; therefore, we developed a general linear model and performed the ANOVA procedure (SAS v. 9.4, SAS Institute, Cary, NC). If significant differences in growth were found, we performed post hoc analyses with Tukey's HSD test to determine where these differences occurred. Further, we interpolated growth data in ESRI ArcGIS 10.8 using the splining with barriers technique to visualize which areas within our regions of interest corresponded to high/low growth.

Results

Baseline data

In total 7.3% (n = 5,712) of planted saplings from 2011 to 2021 were tagged and baseline measurements recorded (Table 1). Mean baseline height in the CFD measured 1.42 m ± 0.42 SD (range 0.20 to 3.30 m) and 1.40 m ± 0.51 SD (range 0.14 to 2.40 m) in the MLB; mean baseline DBH was 0.66 cm ± 0.29 SD (range 0.05 to 1.60 cm) in the CFD and 0.72 cm ± 0.30 SD (range 0.10 to 2.00 cm) in the MLB.

Survival

Sapling survival was higher across the MLB than in the CFD region. After the 2019/2020 monitoring season, survival across 2- to 10-year-old trees was 63% in the CFD region (Table 1). Across planting cohorts/age classes, survival ranged between 12% to 95%. Irrespective of age, survival was higher closer to the CFD (NEAR: 84%) compared to farther from the CFD (FAR: 58%) (Figure 3A). Tree/sapling survival of 2- to 7-year-old trees was 82% on the MLB (Table 1). Survival was less variable across individual planting cohorts/age classes and ranged between 74% and 84% (Figure 3B).

Growth

Height. Height growth varied across years in the CFD region (ANOVA; $F_{7,7265} = 100.93$; $p < 0.00001$). The year with the least growth was 2016 to 2017 (0.19 m/yr ± 0.55 SD). The highest growth was 2013 to 2014 (0.69 m/yr ± 0.48 SD). Height growth was also variable on the MLB (ANOVA; $F_{6,7780} = 68.55$; $p < 0.00001$). The year with the least growth was 2019 to 2020 (–0.03 m/yr ± 0.21 SD). The year with the highest growth was 2013 to 2014 (0.44 m/yr ± 0.17 SD). Overall, height growth differed between the CFD (higher growth rates) and MLB (lower growth rates) (ANOVA; $F_{1,14825} = 2854.9$; $p < 0.00001$) but followed the same variable growth trends year to year (Figure 4A).

Diameter. Diameter growth was also variable across years in the CFD region (ANOVA; $F_{7,\,5490} = 66.39$; $p <$ 0.00001). The year with the least growth was also 2016 to 2017 (0.66 cm/yr ± 0.88 SD), and the year with the highest growth was 2014 to 2015 (1.72 cm/yr ± 0.99 SD). Diameter growth was also variable on the MLB (ANOVA; $F_{5,\,4329} =$ 67.78; $p < 0.00001$). The year with the least growth was 2019 to 2020 (–0.02 cm/yr ± 0.39 SD) and the year with

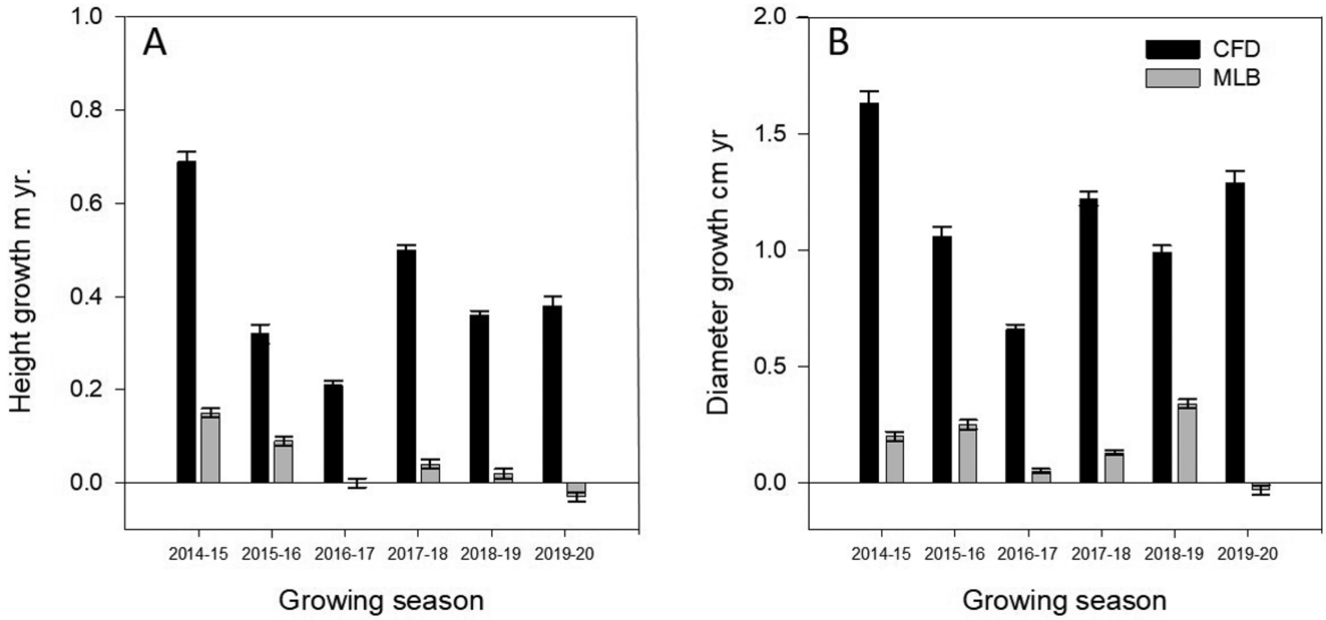

Figure 4. Height (A) and diameter (B) growth were higher around the Caernarvon Freshwater Diversion (CFD - black) compared to growth on the Maurepas Landbridge (MLB - grey); both varied annually.

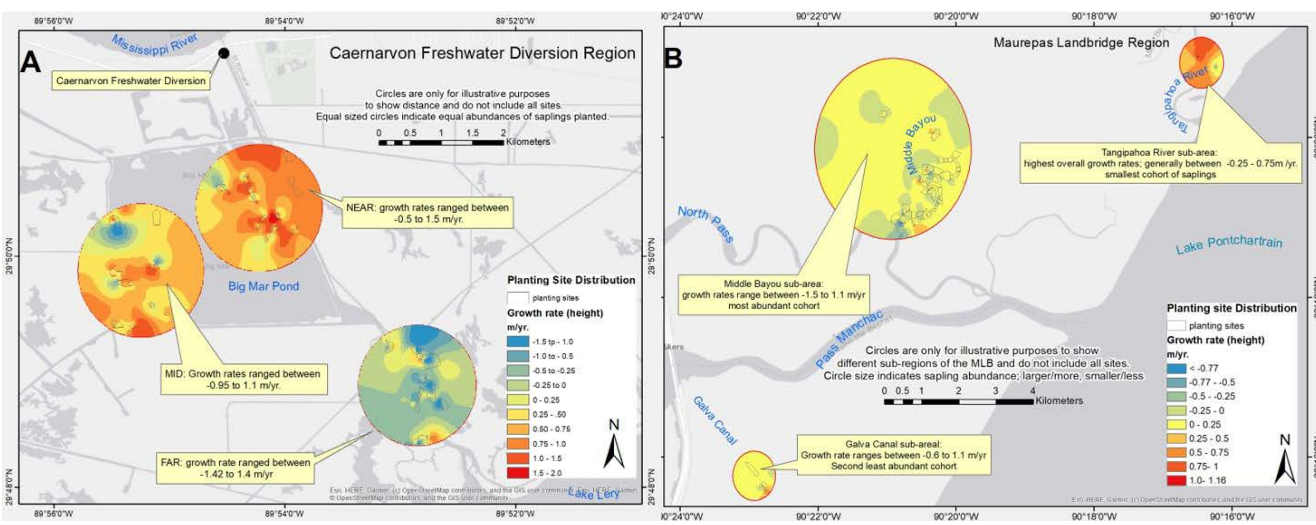

Figure 5. Around the Caernarvon Freshwater Diversion (A), height growth rates were higher closer to the diversion (redder) and decreased further from the diversion (yellows and blues); on the northern Maurepas Landbridge (B), growth rates were lower across most of the area (yellows and blues), but appear higher along the Tangipahoa River (redder).

the highest growth was 2018 to 2019 (0.34 cm/yr ± 0.50 SD). Overall, diameter growth differed between the CFD (higher growth rates) region and MLB (lower growth rates) (ANOVA; $F_{1,9820} = 3149.90$; $p < 0.00001$), but followed similar variable growth trends every year, except for 2018/19–2019/20 (Figure 4B).

Within Regions. In the CFD region, overall height growth was 0.36 m/yr ± 0.53 SD in 2018/2019. Closer to the CFD, growth rates were higher (NEAR: 0.47 m/yr ± 0.35 SD) compared to further from the CFD (FAR: 0.20 m/yr ± 0.36 SD) (Figure 5A). Height growth was 0.01 m/yr ± 0.23 SD on the MLB in 2018/2019. Across all MLB sites, growth rates were relatively uniform, yet were higher at sites further east, along the Tangipahoa River (0.37 m/yr ± 0.39 SD; Figure 5B).

Age. There was a difference in height by age classes across planting regions (Figure 6A). The CFD area trees/saplings were taller than those on the MLB, except for the first two years post-planting, when they had similar heights. Height was variable across age classes in the CFD region (ANOVA; $F_{9,9701} = 1843.5$; $p < 0.00001$) and followed a predictable trend of increasing height with age. Omitting height at

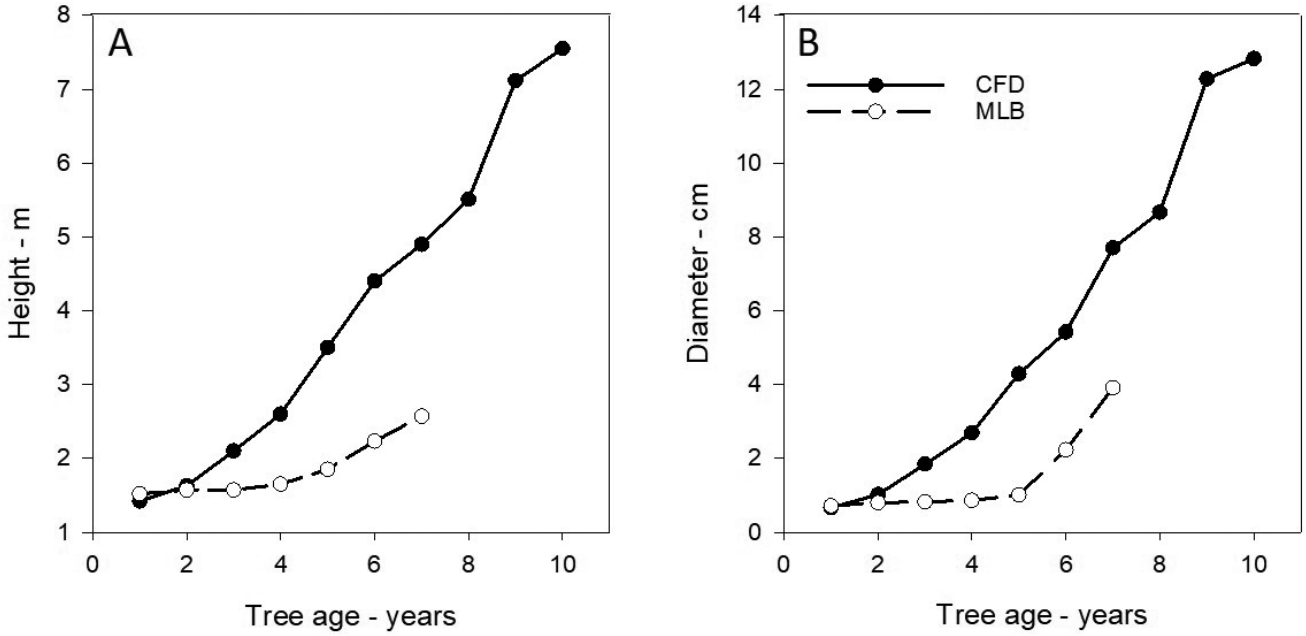

Figure 6. Around the Caernarvon Freshwater Diversion (CFD), tree height increased significantly with the age of trees, while the Maurepas Landbridge (MLB) trees grew slower (A); we observed the same pattern for diameter growth as trees aged (B).

baseline, the shortest individuals were 2-year-old saplings (1.62 m ± 0.47 SD; range 0.10 to 4.88 m). The tallest individuals were the 10-year-old trees (7.42 m ± 1.88 SD; range 2.1 to 11.0 m). Height across age classes was less variable on the MLB (ANOVA; $F_{6,11021} = 173.8$; $p < 0.00001$). The shortest individuals there were 2–3-year-old individuals, which did not differ and exhibited limited growth (1.54 m and 1.55 m ± 0.37 SD, range[s] 0.30 to 3.40 m and 0.42 to 3.82 m, respectively). The tallest trees were the 7-year-old trees (2.42 m ± 0.53 SD; range 1.60 to 3.30-m), the oldest trees in the MLB.

Similar patterns were observed for DBH by age classes; there was a difference in DBH by age classes across planting regions (Figure 6B). CFD area trees/saplings had thicker trunks than MLB area trees/saplings, except for the first two years post-planting when they were similar. The DBH was variable across age classes in the CFD region (ANOVA; $F_{9,7041} = 1106.5$; $p < 0.00001$) and generally increased with age. Omitting diameter at baseline, the smallest DBH was observed on the 2-year-old saplings (1.03 cm ± 0.51 SD; range 0.05 to 3.60 cm), and the biggest DBH was observed on the 10-year-old trees (11.85 cm ± 4.82 SD, range 0.70 to 20.90 cm). DBH across age classes was less variable on the MLB (ANOVA; $F_{6,6499} = 99.9$; $p < 0.00001$). The smallest DBH was observed on the 2–4-year-old individuals, which did not vary and exhibited limited growth (means 0.81–0.89 cm, 0.42 SD, range 0.08 to 3.20 cm). The biggest diameter was observed on the 7-year-old trees (3.20 cm ± 1.71 SD; range 1.18 to 6.50 cm), which are the oldest planted cohort on the MLB.

Discussion

Overview

We had one main goal: the reforestation of former coastal swamp forests. After 10 years, we observed landscape-scale differences in the survival and growth of planted saplings across different regions of the estuary (upper and lower). We reforested approximately 162 ha of coastal swamp and observed 63% and 82% survival depending on region, which is an improvement over previous coastal swamp reforestation outcomes. This coastal swamp reforestation case study indicated, 1) different outcomes between regions, and 2) differences that appear to coincide with hydrological modifications (i.e., river diversion, canal closure) that lowered soil salinity (Figure 7). However, data on previous reforestation efforts are sparse, and we did not track environmental factors, including elevation, inundation, soil nutrients, and salinity, pre- or post-planting because it was not the original goal, nor were those tasks funded. We did not design this reforestation effort as a research project, and therefore, we do not show a direct link between improved reforestation outcomes and hydrological modifications. We acknowledge this limitation. Current reforestation projects now include environmental surveys alongside planting activities and tree monitoring.

Nevertheless, we observed differences between regions. Lower survival and higher growth in the lower estuary (CFD) and higher survival and lower growth in the upper estuary (MLB), a trend observed consistently across years and metrics (survival, height, DBH). Differences in survival

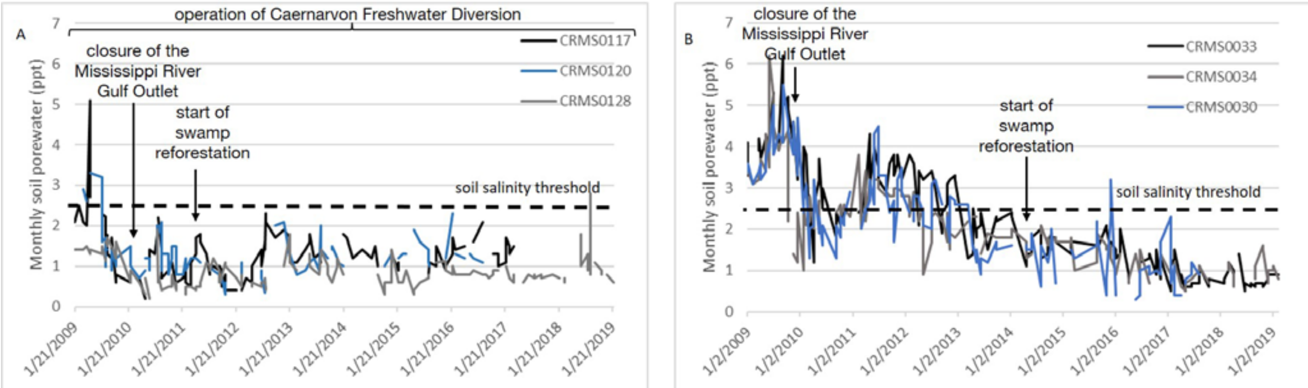

Figure 7. Porewater salinity in both reforestation areas has decreased over the past decade. Reforestation in the Caernarvon Freshwater Diversion area began two years after the closure of the Mississippi River Gulf Outlet (MRGO) canal and while the diversion was periodically run (A), while reforestation on the Maurepas Landbridge began five years after the MRGO closure (B).

may be explained by location in the estuary and lessons learned, and differences in growth rates may be impacted by exposure to oxygenated water and nutrients (i.e., river diversions).

Survival

Planting regions spanned the upper and lower estuary, despite being located only ~90 km apart, which made the observations compelling. Isolating the 1) environmental differences between regions and 2) regional lessons learned may help explain why reforestation outcomes varied. For instance, location in the estuary likely influenced survival around the CFD because of its proximity to the Gulf of Mexico (~45 km) and more direct contact with wind and storm surge. Tropical storms weaken as they move onto and travel across land (Farber 1987, Wamsley et al. 2020, Hlywiak and Nolan 2021). Three hurricanes (Isaac, Zeta, Ida) and multiple other storms crossed coastal SE Louisiana over the last decade, tracking close to both regions. Hurricanes impacted the CFD region first before the MLB (~110 km from the Gulf of Mexico) (Lopez et al. 2020). In the upper estuary, the MLB is more protected. This location likely contributed to higher survival and lower mortality on the MLB.

Lessons learned also impacted reforestation outcomes across regions. Trial and error during earlier planting seasons (CFD) improved outcomes during MLB reforestation. For instance, the first CFD area planting cohort (Table 1; 2010/2011) resulted in only 12% survival likely due to herbivory and unconsolidated soils. Herbivory excluders were consistently used thereafter along with planting exclusively on vegetated soils. These measures increased survival to > 50% by 2011/2012, even with Hurricane Isaac's storm surge (3.6 m; Lopez et al. 2020) impacting the CFD in 2012. Adding herbivory excluders came with a considerable cost, adding about $1.50/per tree planted. We accepted the additional cost because the tradeoff was much higher sapling survival. We now estimate the overall cost, including saplings, excluder device, stakes, and the associated planting costs (boat rental, gas, staff time, snacks, water, tools, gloves, etc.), but excluding monitoring, is approximately $20/tree.

The land around the CFD stabilized quickly, PC and its partners continued to plant saplings annually, and survival kept improving. Due to these generally positive outcomes, reforestation efforts scaled up in 2014/2015. Scaling up reforestation meant increasing the number of trees planted by an order of magnitude and expanding reforestation into the MLB. The first planting cohorts on the MLB benefitted from lessons learned over previous planting seasons in the CFD region. Other beneficial lessons included using a thicker gauge wire for tagged trees and using secondary markers on all tagged trees, which increased the recovery rate of tagged trees during monitoring. Since trees not found for two consecutive monitoring seasons were marked dead, these steps ultimately improved survival rates and decreased mortality rates. By the 2019/2020 season, the overall tree/sapling survival rate settled at 63% in the CFD region and 82% on the MLB.

Growth

Conversely, we observed higher growth rates in the CFD region and lower growth rates on the MLB. How well a species performs generally reflects how well that species tolerates multiple aspects of the physical and biological environment (Sagarin et al. 2006). Here, soil salinity, light availability, soil properties, competition, inundation, elevation, and nutrient availability are environmental factors (separately and in combination) that impact growth. Data from nearby Coastwide Reference Monitoring System (CRMS) stations (CFD: 0114, 0115, 0120; MLB: 0030, 0033, 0034; Figure 2) indicate some factors were similar across regions (i.e., soil salinity < 2.5 ppt, % organic matter 30–60%, bulk density 0.11–0.20 g cm^{-3}; CPRA 2022), and some were different (i.e., inundation frequency CFD: 50–90%; MLB: 70–90%; CPRA 2022). Research shows that inundation is a stressor for terrestrial vegetation, including coastal vegetation acclimated to wetter environments (Pezeshki et al. 1987). Allen et al. (1996) found reduced growth in "deep, > 1 m permanently flooded swamp." Multiple studies support this finding (Nash and Graves 1993, Myers et al. 1995, Gough and Grace 1998). Therefore, growth on the MLB may be limited due to longer inundation of the planting sites. The observation that MLB tree growth increases after several years of negligible growth (Figure 6) may indicate a period of adjustment to more inundation—a lag period. However, more research is needed to clarify this correlation, if it exists. In addition, the CRMS stations are not located directly on-site but are located 0.15 to 15 km from MLB planting sites, and the natural heterogeneity between individual planting sites across both regions confounds inundation estimates, especially around the CFD, where heterogeneity between sites is significant. Elevation surveys across individual planting sites within both regions would help clarify inundation estimates and whether, or to what extent, inundation impacts growth.

Another factor impacting growth may be exposure to fresh, oxygenated, sediment-rich river water with nutrients. The CFD region receives intermittent flow from the Mississippi River through the diversion, with a nitrogen load of approximately 715 Gg N/year (Tian et al. 2020). Exposure is intermittent because the diversion operation is highly regulated and governed by changes in surface water salinity in Breton Sound (Ko et al. 2017). In contrast, the MLB is only indirectly exposed to waters from the Blind, Amite, and Tickfaw rivers via Lake Maurepas, and is completely disconnected from the Mississippi River. The Lake Maurepas region is generally considered "nutrient-starved," with a 100-times lower nitrogen load than the Mississippi River (Lane et al. 2003, Shaffer et al. 2016). Research shows exposure to nitrogen increases growth (Dickson and Boyer 1972), resulting in freshwater wetland plants with more above- and belowground biomass (Lundberg et al. 2011, Hillmann et al. 2019). Hillmann et al. also found that nitrogen loading impacts *T. distichum* density, leading to increased resiliency during storms. Darby and Turner (2008) and others (Turner et al. 2009, Mozdzer et al. 2020) observed less growth and less belowground biomass, in particular, when nutrients were added to brackish wetland plants, which can result in top-heavy plants susceptible to uprooting during storms. Yet, the species assemblages in the CFD region are freshwater/intermediate species, including *Sagittaria lancifolia* L. (bulltongue arrowhead), *Typha* spp. L. (cattail), *Zizaniopsis miliacea* Marshall (giant cutgrass), and *Salix nigra* Marshall (black willow), which are thriving in the footprint of a river diversion. Further, a recent review indicated that exposure to oxygenated freshwater with nutrients and sediment (river water) might be the key to positive belowground biomass growth in the outfall area of river diversions (Elsey-Quirk et al. 2019). Less exposure to fresh river water on the MLB then possibly leads to less biomass production overall on the MLB or less growth.

A planned freshwater river diversion for the Lake Maurepas region is currently in the planning/engineering phase (CPRA 2023, Buras et al. 2018). The diversion would re-introduce Mississippi River water, flowing at a maximum rate of 56 m^3/s. When that happens, growth rates in the coastal swamp forests closest to the diversion will likely increase, but whether growth rates on the MLB will increase or to what degree is unclear. Some sections of the land bridge are > 20 km away from the proposed diversion, and our data indicate that at those distances the positive effects may be negligible (Figure 5). Either way, the current MLB dataset demonstrates pre-diversion conditions and will complement post-diversion monitoring.

Lessons learned, challenges, and benefits

Despite differences in survival and growth, we invested equal time, money, and energy in planning and executing the planting of ~40,000 saplings across each region. Each planting event was an opportunity for learning (Table 2), which made future reforestation efforts and monitoring more efficient and possibly more successful; they are an essential part of adaptive management. For instance, early on, a number of tagged trees lost their tags from corroded wires, which resulted in significant time spent searching for lost tags during monitoring. Using a thicker-gauge wire reduced the number of lost tags, improved monitoring efficiency, and became integrated into the planting protocols. In another example, we planted *N. biflora* for three seasons starting in 2014/15 to increase the overall biodiversity of reforestation plantings. Monitoring indicated low survival for *N. biflora* (< 35%), so we stopped planting that species because using resources (i.e., money) effectively is critical.

Yet, we did not resolve all the issues. Three of the biggest and most consistent challenges were 1) herbivory by *Odocoileus virginianus* (white-tailed deer) and destruction by *Sus scrofa* (feral pigs, Figure 8A), 2) and competition from other species (i.e., *Vigna luteola* [hairypod cowpea], Figure 8B), and 3) ill-timed inundation/flooding of newly planted saplings. Herbivory was a bigger problem in the CFD region compared to the MLB, and *M. coypu* was problematic only at specific sites around the CFD. Despite consistently using excluder devices, we observed > 50% mortality at one site where a commercial crew planted over 3,000 saplings. We observed that *M. coypu* burrowed underneath the excluder devices in soft soils and chewed the saplings' trunks, which killed them. *O. virginianus* herbivory and *S. scrofa* disturbance were a bigger problem at other sites. *O. virginianus* fed on sapling stems and leaves

Table 2. The following protocols and lessons learned helped improve reforestation efficiency and outcomes. They may be helpful for future swamp reforestation projects.

Lessons Learned for Reforestation of Coastal Swamp Forest	Category
Saplings at the convergence of canals are susceptible to more storm damage	Planting
Inundation from storms, tide, high diversion flow can wash away newly planted saplings	Planting
Plan for deer herbivory, hog damage and vegetative competition	Planting
Budget contingency money for extra planting days	Planting
Herbivory guards add about $1.50 to cost of each planted sapling	Planting
Specialized software (i.e., TerraFlex) increases plannting & monitoring efficiency	Planting
Boat use (type, number, rental, driver, gas) increases cost of planting events	Planting
Plant on stable ground with emergent vegetation	Planting
In outfall of a river diversion, planting when flow exceeds 155 m^3/s is dangerous	Planting
Use hebivory guards on all saplings	Planting
Plant saplings 3–5 meters apart	Planting
Longer bamboo stakes increase stability and are light	Planting
Identify monitored saplings with numbered alumninum tags	Monitoring
A thicker gauge wire (18/22) works best to fasten tags	Monitoring
Record tagged sapling locations with GPS technology (pref. < 1 m accurracy)	Monitoring
Mark tagged saplings with secondary indicators (i.e., flagging, paint)	Monitoring
Write tag numbers on flagging	Monitoring
Monitoring starts 1-year after planting	Monitoring
Larger/older trees outgrow wired tags—which need to be nailed	Adaptive Management
Henceforth, DBH is measured just above nailed tag	Adaptive Management
After 10-years annually, scale down monitoring to every 3 years	Adaptive Management
After 10-years annually, scale down monitoring to only survival and DBH	Adaptive Management
Herbivory guards break down after 3–4 seasons; prioritize debris pick-up	Adaptive Management
Continually evaluate site access in response to land building	Adaptive Management

Figure 8. Annual tree monitoring illuminated challenges (i.e., herbivory [A] and competition [B]), as well as benefits (i.e., coastal reforestation [C] and habitat restoration [D]) associated with this work. *Image credit Eva R. Hillmann.*

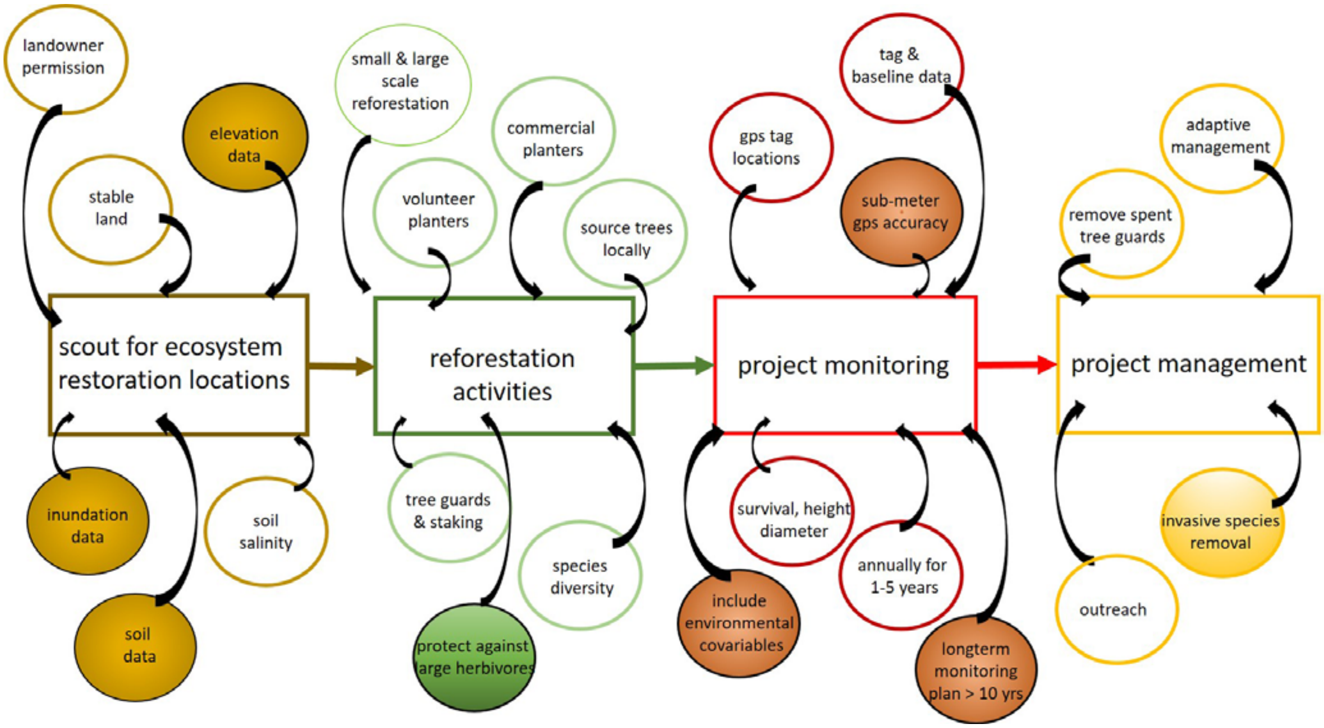

Figure 9. Proposed workflow and factors to consider for coastal swamp reforestation, including site selection, planting, monitoring and management. The direction of the arrows indicates the progression of the workflow. Open circles represent factors incorporated into the 2011–2021 reforestation project in coastal southeast Louisiana; solid circles represent factors not incorporated into the 2011–2021 reforestation project but that should be considered in future projects when possible.

that extended above the excluder devices, leaving saplings alive but nubbed and stressed. *S. scrofa* ran roughshod through several sites, leaving deep wallows and uprooted saplings. These impacts likely resulted in higher mortality at specific sites, which contributed to higher mortality in the CFD region compared to the MLB.

We also reported on positive outcomes. The most obvious benefits included 1) robust sapling growth resulting in healthy stands of young trees (Figure 8C), 2) trees producing seed pods, and 3) trees providing habitat for other species (Figure 8D). Many sites in the CFD region are developing into healthy coastal swamp forests and planted saplings are growing into established trees. In some cases, trees are now 5–7 m tall and exhibit features commonly associated with coastal swamp forests, including buttressed trunks, knees, and seed pods. Despite seed pod development on *T. distichum* trees, natural regeneration has not been observed. Also, slower growth on the MLB means fewer sites have developed into mature stands, but maturation is beginning to become apparent as evidenced by higher growth of the older planted trees on the MLB. Those trees are close to 3 m tall at some MLB sites. Further, across both regions, we observed many species, including birds, invertebrates, snakes, and frogs using individual planted saplings/trees as food, refuge, and habitat.

Conclusion

By observing the survival and growth of planted *T. distichum* saplings, as well as other wetland species, across different regions in the Pontchartrain Estuary, this work demonstrated four things: 1) more planted saplings survived in the upper estuary than in the lower estuary, 2) sapling survival in the lower estuary was higher near the river diversion and lower further from the diversion, 3) sapling growth was higher in the lower estuary and lower in the upper estuary, and 4) sapling growth was also higher closer to the diversion and lower further from the diversion. This work suggests that partial hydrological restoration leads to positive coastal swamp forest reforestation outcomes (Turner and Lewis 1996 and others); however, clarifying differences between regions (i.e., inundation frequency, soil nutrient content) to understand better these differences warrants further studies. We propose a framework for coastal swamp reforestation (Figure 9) that is built around the lessons learned from this and other regional reforestation efforts to guide and standardize future reforestation projects here and elsewhere. As coastal swamp reforestation continues to scale-up across coastal Louisiana and globally in similar estuaries, this framework should inspire comprehensive and collaborative regional protocols and coastal swamp reforestation databases that 1) track progress, 2) increase resource accessibility and 3) improve the overall efficiency of reforestation efforts.

Acknowledgments

We thank Valero Energy Corporation, Colonial Pipeline, Delacroix Corporation, Walton Family Foundation, Meraux Foundation, National Fish and Wildlife Foundation, Entergy, the Coastal Protection and Restoration Authority, Foundation for Louisiana, Chalmette Refining, American Water Resources Association, the Joe and Dorothy Dorsett Foundation, SITKA Gear and the Pontchartrain Institute for Environmental Studies for funding planting and/or monitoring activities. We thank the Restore the Earth Foundation, the St. Bernard Wetland Foundation, Lower 9th Ward Center for Sustainable Engagement and Development, and Common Ground Relief for providing trees for this project and the Coalition to Restore Coastal Louisiana and the St. Bernard Parish Coastal Division for organizing and funding numerous volunteer planting events. We thank the Delacroix Corporation, the heirs of Peter Moss, the Louisiana Department of Wildlife and Fisheries, Stinking Bayou, LLC, Hayden Warehouses, LLC, Williams Inc., Flood Protection Authority-East, and the Lake Borgne Levee District for access and permission to plant. Additionally, we thank Mike Farizo (Delacroix Corporation), Howard Callahan (Callahan Forestry), Dr. Robert Moreau and Haden Reno at Southeastern Louisiana University Turtle Cove Research Station, and Patrick Dufrene and the Tangipahoa Parish Port Commission, who provided significant logistical support. We thank numerous people who spent considerable time over the years in the marsh planting and monitoring trees with us, including Andrew Baker, Brianna Cervantes, Tasia Denapolis, Kevin DeSantiago, Michael Hopkins, Arnaud Kerisit, Marshall Kormanec, Eden Krolopp, Mary Grace Lemon, Patrick Smith, and Adam Songy. Finally, we acknowledge the many community volunteers—individuals who gave their time and energy to do their part to plant trees and reforest a portion of coastal Louisiana, including employees of Valero Energy Corporation, Colonial Pipeline, Marathon Corporation, and students from Loyola University (New Orleans, LA), Southeastern Louisiana University (Hammond, LA), Tulane University (New Orleans, LA), Chalmette High School (Chalmette, LA), and Ponchatoula High School (Ponchatoula, LA), as well as a contingent of faith groups organized by Helen Rose Patterson of the National Wildlife Federation. Thank you.

Author Contributions

JAL, TKH, and ERH contributed to idea development. ERH, DAB, SB, and KB collected data and managed all daily activities. ERH and KB analyzed data. ERH, DAB, SB, KB, TKH, and JAL contributed to manuscript writing.

Conflicts of Interest

The authors declare that we have no significant competing financial, professional, or personal interests that may have influenced the performance or presentation of the work contained in this manuscript.

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