Field Observations from Reforesting a *Typha*-Dominated Conifer Swamp in Southwest Michigan

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Abstract

Wetlands dominated by conifer trees are valued for wildlife, carbon sequestration, and commercial lumber production. Managers face unique challenges when reforesting these sites given the seasonally wet soils and variable hydrology. Black ash (*Fraxinus nigra* Marshall) is typically a component of these wetlands in the Lake States of the Eastern United States, but the species is disappearing due to the emerald ash borer (*Agrilus planipennis* Fairmaire, 1888). The loss of this species, competition from invasive plants such as cattail (*Typha* spp.), and climate change affect the delicate hydrology and subsequent tree regeneration in these sites. This article describes an observational study to assess success of planting native and nonnative tree species in a wetland area over 10 years. Northern white cedar (*Thuja occidentalis* L.) had the highest growth rate followed by eastern white pine (*Pinus strobus* L.). Growth of nonnative Atlantic white cedar (*Chamaecyparis thyoides* [L.] Britton, Sterns & Poggenb.) was higher than expected. Growth varied by species and tended to be influenced by duration of high water levels (especially during summer months). Canopy closure is expected to shade out competing vegetation an estimated 20 years after the time of planting.

Introduction

Wetlands dominated by woody evergreens, herein referred to as conifer swamps, are a relatively common forest type across the Great Lake States of the United States. Conifer swamps often host large densities of northern white cedar (*Thuja occidentalis* L.), tamarack (*Larix laricina* [Du Roi] K. Koch), white spruce (*Picea glauca* [Moench] Voss), and black spruce (*P. mariana* [Mill.] Britton, Sterns & Poggenb.). Wildlife use these tree species as a food source and winter cover (Curtis 1959, Doepker et al. 1990). These wetlands also provide habitat for rare plants (Epstein et al. 1999). Tree species, such as northern white cedar, also have strong cultural value to Native Americans (Meeker et al. 1993), such as the Anishinaabe people who have inhabited this region for thousands of years. Wetlands with healthy conifer trees have the advantage of year-round shade which can deter invasive plants from becoming established. For example, narrowleaf cattail (*Typha angustifolia* L.) or reed canarygrass (*Phalaris arundinacea* L.) are aggressive plants that can establish in gaps created by dead and dying overstory trees and reduce the overall biodiversity of the site (Hovik and Reinartz 2007). Conifer swamps are also threatened by human activities. For example, unintended consequences of poor restoration practices, such as tilling to increase soil aeration, can further degrade a site by altering water levels and lowering biodiversity (Cappiella et al. 2006). Many tree species that reside in these wetlands are predicted to decline, or experience range contraction, as a result of climate change (Prasad et al. 2007).

Black ash (*Fraxinus nigra* Marshall), a broad-leaved species that frequently inhabits conifer swamps across the Lake States, is being decimated by emerald ash borer (EAB, *Agrilus planipennis* Fairmaire, 1888), leading to rapid shifts in the hydrology (Slesak et al. 2014) and biodiversity of these forest types. Conifer species such as white spruce, northern white cedar, juniper (*Juniperus* spp.), pine (*Pinus* spp.), and tamarack are considered replacements to maintain tree cover on sites with declining black ash trees (Kesner and Nelson 2018, Palik et al. 2021). In the aftermath of extensive black ash dieback from EAB invasion, many wetlands are expected to revert from woodlands to grasslands dominated by *Typha* species and reed canarygrass (Bansal et al. 2019, Palik et al. 2012). These grasslands create new challenges for reforestation due to fluctuating water tables (Diamond et al. 2018), anaerobic soil conditions, and competition. Interestingly, leaf litter of conifer swamps releases less methane than wetlands dominated by *Typha* spp. (Emilson et al. 2018), so the post-EAB transition of these degraded sites to conifer-dominated swamps may play a unique role in carbon sequestration of this region.

Management tools and resources to revert *Typha*-dominated wetlands to conifer wetlands are scant. In addition, managers face numerous challenges such as identifying the optimal combination of tree species that can survive on these sites and procuring local seed sources. Changes in hydrology from shifting water tables can also interact with tree performance (Slesak et al. 2014) and are not well studied for any northern conifer species. Timing a reforestation effort to maximize survival is difficult, since most trees in the Northern United States are planted in the spring when water tables tend to be at a maximum from melting snow. Natural mounds are common in most unmanaged conifer swamps, and artificial mounds may improve success of planting upland tree species on lowland sites (Åkerstrom and Hånell 1997, Londo and Mroz 2001, Mehne and Mehne 2014, Reid 1985). The objectives of this report are to summarize observations from a reforestation effort on a wetland in southwestern Michigan, make suggestions for managers, and encourage future study to improve the success of reforestation on these valuable ecosystems.

Materials and Methods

Site Description

The site consists of a 4-ac (1.6-ha) conifer wetland on private property in Barry County, MI. The county is located at 42.5° N, 85.35° W in U.S. Department of Agriculture (USDA) plant hardiness zones 5b and 6a, corresponding to minimum temperatures of -15 to -10 °F (-26.1 to -23.2 °C) and -10 to -5 °F (-23.2 to -20.6 °C), respectively (USDA ARS 2012). The climate is continental with warm, humid summers, cold winters, and consistent precipitation year round (table 1). The site is 60 to 100 mi (100 to 167 km) south of the southern range edge of boreal forests in the northern Great Lakes region.

The vegetation on the study site is dominated by narrowleaf cattail, broadleaf cattail (*Typha latifolia* L.), and their hybrids, along with *Carex* spp., various forbs, reed canarygrass, and Phragmites spp. The grasses (*Typha, Phalaris*, and *Phragmites* spp.) are primary competitors to young trees and were 3- to 9-ft (1- to 3-m) tall across most of the site (figure 1). The wetland was classified as a northern hardwood swamp, but the Michigan Natural Features Inventory presettlement records indicate the likelihood of a prior conifer swamp with northern white cedar, black spruce, white spruce, and tamarack likely dominant before black ash became established (Comer et al. 1995). The wetland surrounding the site contains forested areas dominated by tamarack and black ash (figure 2).

Tree cover across the site was sparse and included tamarack, American elm (*Ulmus americana* L.), and eastern redcedar (*Juniperus virginiana* L.). Secondary species included quaking aspen (*Populus tremuloides* Michx.), black ash, eastern white pine (*Pinus strobus* L.), yellow birch (*Betula alleghaniensis* Britton), and red maple (*Acer rubrum* L.). Before EAB's arrival in 2012, black ash dominated portions of the site. The soil type for the entire wetland complex, including the study site, is Houghton muck, a poorly drained, deep organic soil. Holes dug in the wetland complex indicate that the organic matter is deeper than 6.5 ft (2 m) in some areas, and at least 3 ft (1 m) thick in most areas across the study site. Topsoil samples taken across the site had pH values of 5.6 to 5.9, and consisted predominantly of moderately decomposed to well-decomposed peat, originating from sedges or woody material.

Table 1. Normal monthly climate averages for the weather station in Hastings, MI, near the study area.

	January	February	March	April	May	June	July	August	September	October	November	December
High temperature °F (°C)	30 (-1)	34 (1)	44 (7)	57 (14)	69 (21)	78 (26)	82 (28)	80 (27)	73 (23)	60 (16)	47 (8)	1 (34)
Low temperature °F (°C)	15 (-9)	16 (-9)	24 (-4)	34 (1)	45 (7)	55 (13)	59 (15)	57 (14)	49 (9)	38 (8)	30 (-1)	21 (-6)
Rainfall in (cm)	2 (5)	2 (5)	2 (6)	3 (8)	10(4)	10(4)	4(9)	4(10)	4 (10)	3 (8)	3(8)	2 (6)
Snowfall in (cm)	19 (48)	13 (33)	7 (18)	2 (5)	0 (0)	0 (0)	0 (0)	0(0)	0 (0)	1 (3)	5(13)	16 (41)

Figure 1. In this overview of the planting site, taken in October 2020, the planted conifer trees are visible against the dense *Typha* species growing in the understory. (Photo by Alex Mehne)

Figure 2. Tamarack trees were prevalent along the periphery of the planting. (Photo by Alex Mehne)

Planting

In total, tree seedlings from more than 20 native and nonnative species were planted across multiple years (2011–2020) in December of each planting year (table 2). This article focuses on 11 species that had the highest survival, including five that are not native to the site, such as Atlantic white cedar (*Chamaecyparis thyoides* [L.] Britton, Sterns & Poggenb.), which grows in wetlands of the glaciated Northeastern and Southern United States (Laderman 1989). The number of seedlings planted per year and

Table 2. A variety of conifer species and seed sources were used at the study site in Michigan.

Species	Common name	Soil preference	Shade tolerance	Native biome	Seed source, rationale, and other details
Abies balsamea	Balsam fir	Upland	High	Eastern boreal forests	Native boreal species; may underperform because of expected range contraction with climate change; stock types and seed sources were variable.
Abies concolor	White fir	Upland	Medium	Western high-elevation forests	Upland fir species native to western States; favored by the horticulture industry; may be a replacement for balsam fir due to climate change; all stock types were 2-1.
Chamaecyparis thyoides	Atlantic white cedar	Lowland	Medium	Eastern temperate forests	New Jersey seed source; thrives in wetlands across the eastern seaboard where its habitat is threatened by development; not native to Michigan; grown from seed for 2 years in pots.
Juniperus virginiana	Eastern red cedar	Upland	Medium	Eastern temperate forests	Thrives in hot, continental climates; Michigan is its northern range edge; little is known about its performance in wetlands; all seedlings were transplanted from adjacent forests.
Picea abies	Norway spruce	Upland	Medium	Europe	Naturalized in Michigan; seed source unknown; stock types were variable.
Picea glauca	White spruce	Upland	Medium	Eastern boreal forests	Native boreal species; may underperform because of expected range contraction with climate change; stock types were variable.
Picea mariana	Black spruce	Lowland	Low	Eastern boreal forests	Native boreal species; may underperform because of expected range contraction with climate change; stock types were variable.
Picea sitchensis	Sitka spruce	Upland	Medium	Western high-elevation forests	Native to western States; grows on wetlands and uplands; best adapted to USDA zone 7 (Eckenwalder 2009) but may be cold hardy to mild winters (Sakai and Weiser 1973); western Washington seed source; all were 2-1 stock.
Pinus rigida	Pitch pine	Upland	Low	Eastern pine forests	Thrives in pine barrens in eastern States (e.g., NY and NJ); sometimes grows alongside Atlantic white cedar; not native to Michigan; all stock was 2-2.
Pinus strobus	Eastern white pine	Upland	Medium	Eastern temperate forests	Native to Michigan; occurs in adjacent stands, primarily uplands; stock types were variable.
Thuja occidentalis	Northern white cedar	Lowland	High	Eastern boreal forests	Native to Michigan and common in stands adjoining the study site; stock types were variable.

per species varied, and some were planted as replacements following mortality (table 3). Seedlings were planted at a spacing of approximately 7 by 7 ft (2 by 2 m). The seed sources for native species were a combination of local (adjacent counties) and nonlocal sources. All eastern redcedar seedlings, and a subset of balsam fir (*Abies balsamea* [L.] Mill.), white spruce, and eastern white pine seedlings, were transplanted from adjacent forests. Nursery-grown stock types were grown at a variety of nurseries. Roughly half of the seedlings were planted into mounds while the other half were planted directly into soil. Mounds were created using soil from the site (see Mehne and Mehne 2014). Dead planted trees were noted and replaced, either on the same planting spot or within 7 ft (2 m) of the original planting spot. Protective netting cones were placed around each planted tree to reduce incidence of browsing by white-tailed deer (*Odocoileus virginianus* [Zimmermann, 1780]) (figure 3). No herbicides were applied to the site.

Typha Removal

Two 0.1-ac (0.04-ha) plots were established for observing the effects of *Typha* removal on tree growth. *Typha* was removed mechanically (with brush cutters) on half of each plot and trees were measured from 2015 to 2017. The Wilcoxon rank sum test, a nonparametric t-test, was used to compare the removal and nonremoval plots.

Figure 3. Protective tubes were placed over the seedlings after planting to protect from herbivory. (Photo by Alex Mehne)

Tree Measurements

Tree height from the soil to the most distal living stem was measured with a telescoping height pole annually between November and January from 2012 to 2020 to the nearest centimeter. The difference in height growth from between the current (H_n) and previous year (H_{n-1}) is reported in figure 4. Relative growth was calculated as H_n / H_{n-1}; relative values greater than 1 indicate an increase in growth relative to the previous year. Surviving Sitka spruce (*Picea sitchensis* [Bong.] Carrière) trees (n = 39) were tracked for foliage loss (attributable to winter frost damage) each year.

Soil Temperature and Hydrological Assessments

Soil temperature was measured with a thermometer two to four times monthly during 2020 at each of three locations: south-facing shade, south-facing full sun, and north-facing full sun. Round dial thermometers affixed to metal stakes at different depths recorded belowground temperatures (figure 5). In addition, snow depth was measured weekly from November through January annually from 2012 to 2020.

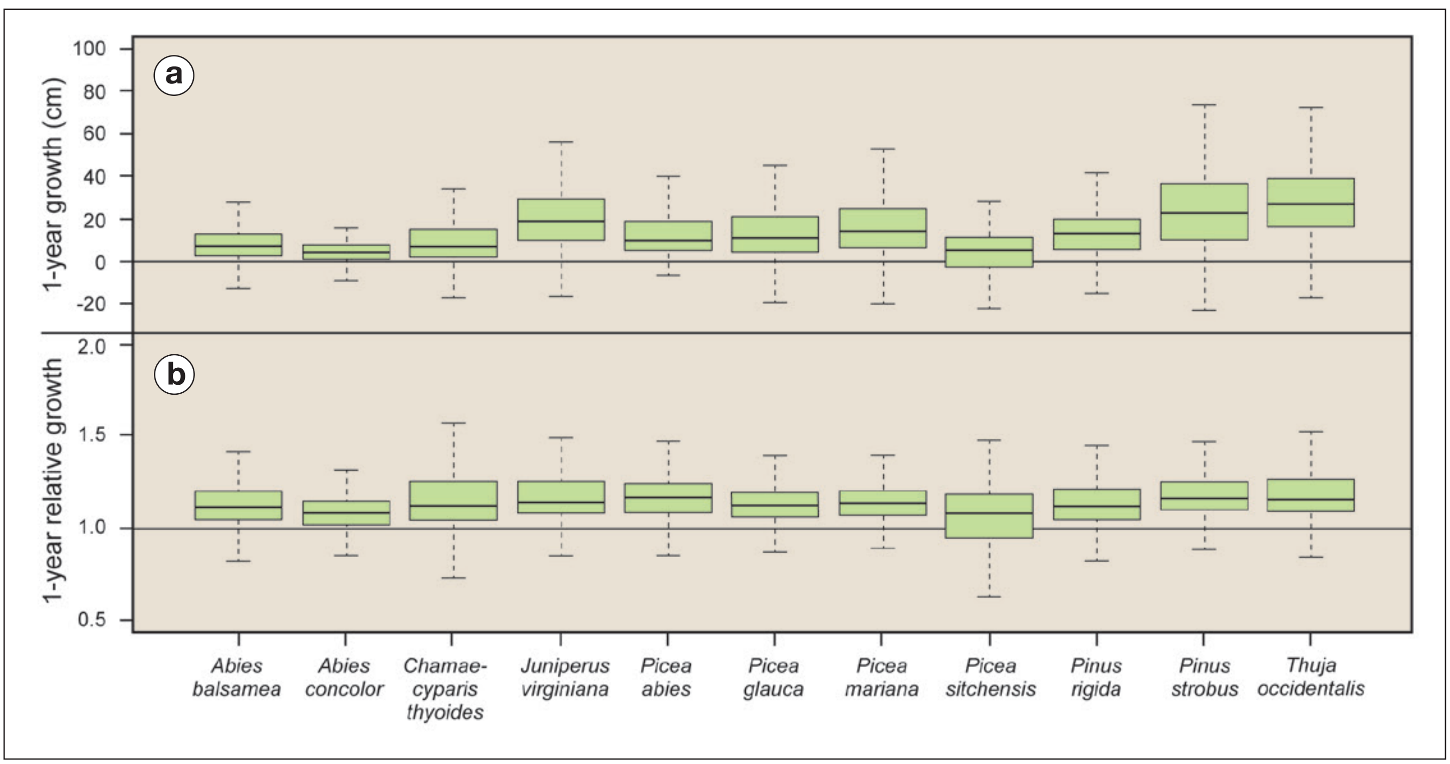

Figure 4. (a) Absolute height growth and (b) relative height growth from 2011 to 2020 for 11 tree species planted in a southwestern Michigan wetland. Midpoints represent median growth (annual increase in height). Outliers were omitted. The horizontal line represents no measurable growth. For a tree of height H (centimeters), growth of year n is defined as $H_n - H_{n-1}$ and relative growth is defined as H_n / H_{n-1}.

Figure 5. Water temperatures were monitored with round dial thermometers affixed to metal stakes at different depths to record belowground temperatures. (Photo by Alex Mehne)

Figure 6. Water levels were monitored with a measuring stick at 47 locations across the plantings. (Photo by Alex Mehne)

The study site included 47 pools set up for monitoring water table depth at random distances from each other. Water depth was measured weekly to the nearest 0.25 in (0.64 cm) at all pools (figure 6) during 2020. Pools that no longer had standing water were excavated and depth to the water table was recorded. Water table depth for each tree was estimated by georeferencing tree locations onto a raster calculated from an inverse-distance weight interpolation model.

The following hydrological variables were calculated to determine their association with tree growth by species: average and median water table heights, 5th and 95th quantiles for water table height, median water table height in July (driest month), and number of days water table height was above 0 (above the soil surface). Whether the tree was mounded or not mounded and whether the tree had been recently planted (within 3 years) was also noted. In addition, the number of days the water table height dropped below the soil surface to varying depths (<0, 0 to 5 in [0 to 12.7 cm], 5 to 10 in [12.7 to 25.4 cm], and >10 in [>25.4 cm]) was recorded. Kendall's rank correlation coefficient was used to observe the relationship between water table and 3-year height growth for six of the tree species. Mounding (with or without), initial tree height, and years since planting (>3 or <3 years) were included in the correlation analysis. All statistical analyses were run using R and QGIS software.

Results

Mortality occurred for most species during the first growing season after planting as reflected in the replacement rates (table 3). Survival data, however, were not closely tracked, so observations focused on the height of surviving individuals.

Tree Growth

Median height growth varied by species over time (figure 4). Eastern white pine and northern white cedar had the highest median annual growth. Trees with negative growth were shorter the second year due to dieback, infection, browsing, or frost heave. Minor frost damage occurred on select individuals

Table 3. The number of trees planted and the percentage replaced due to mortality varied by year and species.

Species	Common name	Number planted in 2011	Number planted in 2012	Percent replaced in 2012	Number planted in 2013	Percent replaced in 2013	Number planted in 2014	Percent replaced in 2014	Percent replaced in 2015	Number planted in 2011–2020
Abies balsamea	Balsam fir	92	89	13	6	6	0	6	4	187
Abies concolor	White fir	11	0	9	17	8	0	21	0	28
Chamaecyparis thyoides	Atlantic white cedar	0	7	0	82	19	43	5	11	132
Juniperus virginiana	Eastern red cedar	35	6	6	64	17	7	27	8	112
Picea abies	Norway spruce	1	1	0	0	30	28	6	9	30
Picea glauca	White spruce	20	131	20	95	30	4	3	3	250
Picea mariana	Black spruce	94	4	4	155	9	5	4	2	258
Picea sitchensis	Sitka spruce	0	103	14	0	18	0	7	16	103
Pinus rigida	Pitch pine	66	0	5	0	3	0	3	7	66
Pinus strobus	Eastern white pine	66	112	27	68	38	0	9	8	246
Thuja occidentalis	Northern white cedar	182	166	9	243	4	119	5	2	710
Sum /average %		567	619	10	730	1	206	9	6	2,122

of Atlantic white cedar in the fall for two of the years and Sitka spruce had extensive cold damage. For all surviving Sitka spruce, the number of trees with no signs of foliage loss from 2011 to 2020 was 24, 13, 3, 1, 3, 18, 32, 24, and 34 trees, respectively. Median height growth was negative in 2014 and 2015 but increased to 10 cm per year by 2020. The white pine weevil (*Pissodes strobi* W. D. Peck, 1817) affected eastern white pine, but overall height growth was not greatly diminished because lateral stems reclaimed apical dominance. In 2019, an unknown foliar disease on balsam fir resulted in partial to complete foliar loss in both healthy and unhealthy trees. Northern white cedar had the highest growth rates, with several individuals growing more than 18 in (46 cm) in a single year. Sitka spruce and white fir (*Abies concolor* [Gord. & Glend.] Lindl. ex Hildebr.), the two taxa from the Western United States, had some of the lowest growth rates in 2020. Across species and years, stock types taller than 28 in (70 cm) but shorter than 63 in (160 cm) had the highest annual height growth (figure 7).

Effects of *Typha* spp. Removal

Survival of planted tree seedlings was initially poor in the two *Typha*-removal plots but improved over time (data not shown). Height growth tended to be greater in plots where *Typha* had not been removed. These differences in height growth were significant in both east and west blocks ($p = 0.001$ and 0.092, respectively).

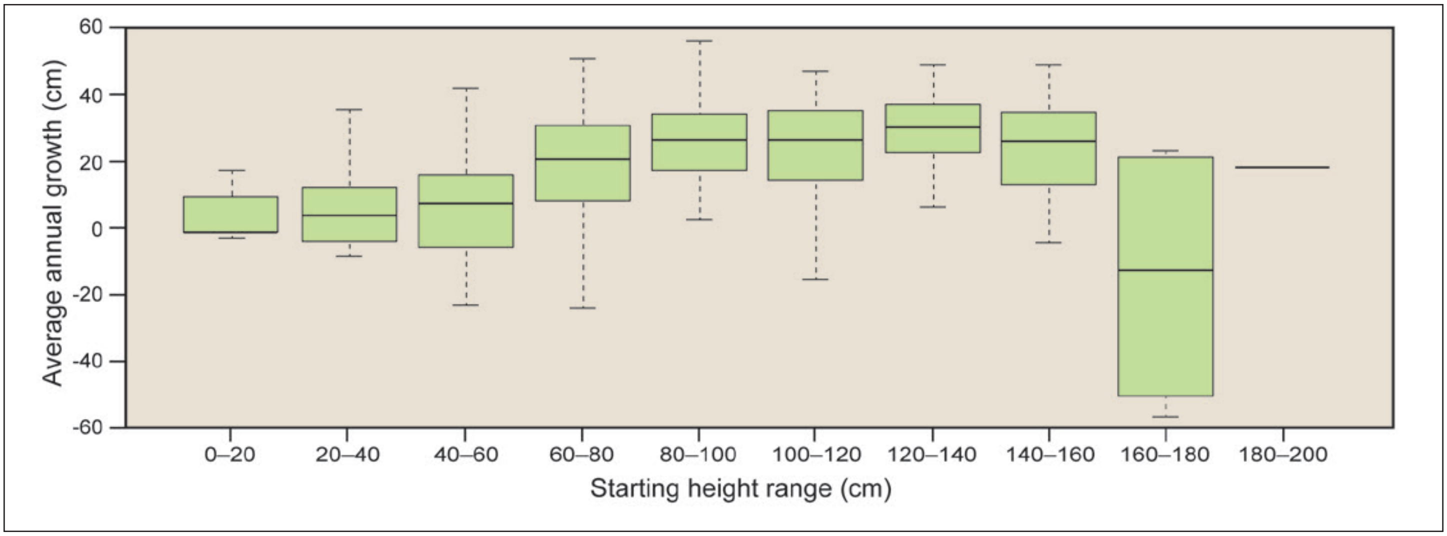

Figure 7. Median annual height growth averaged for all trees planted in a southwestern Michigan wetland from 2011 to 2020 and grouped by initial height class at planting. Starting height classes are in increments of 20 cm median height class.

Hydrology

Pools excavated at the study site displayed a range of hydrology types, ranging from consistently wet to relatively dry with water levels consistently below the soil line (figure 8). Wet pools had depths as high as 10 in (25 cm), while the driest pools were as low as 10 in (25 cm) below ground level. Pools also had high variance (water heights that varied from extremely high to low) and low variance (water heights that were stable) (figure 8). Most pools had water levels from 2 to 7 in (5 to 18 cm) below the soil surface most of the year. In general, hydrological variables were only weakly correlated with 3-year tree-growth parameters (table 4). Most species preferred drier microsites, indicated by negative coefficients for water height (especially during summer) and positive coefficients for the number of days with water greater than 10 in (25 cm) below the soil surface. These data indicate that a high water table in summer was associated with growth reductions. Eastern white pine, balsam fir, and white spruce growth rates

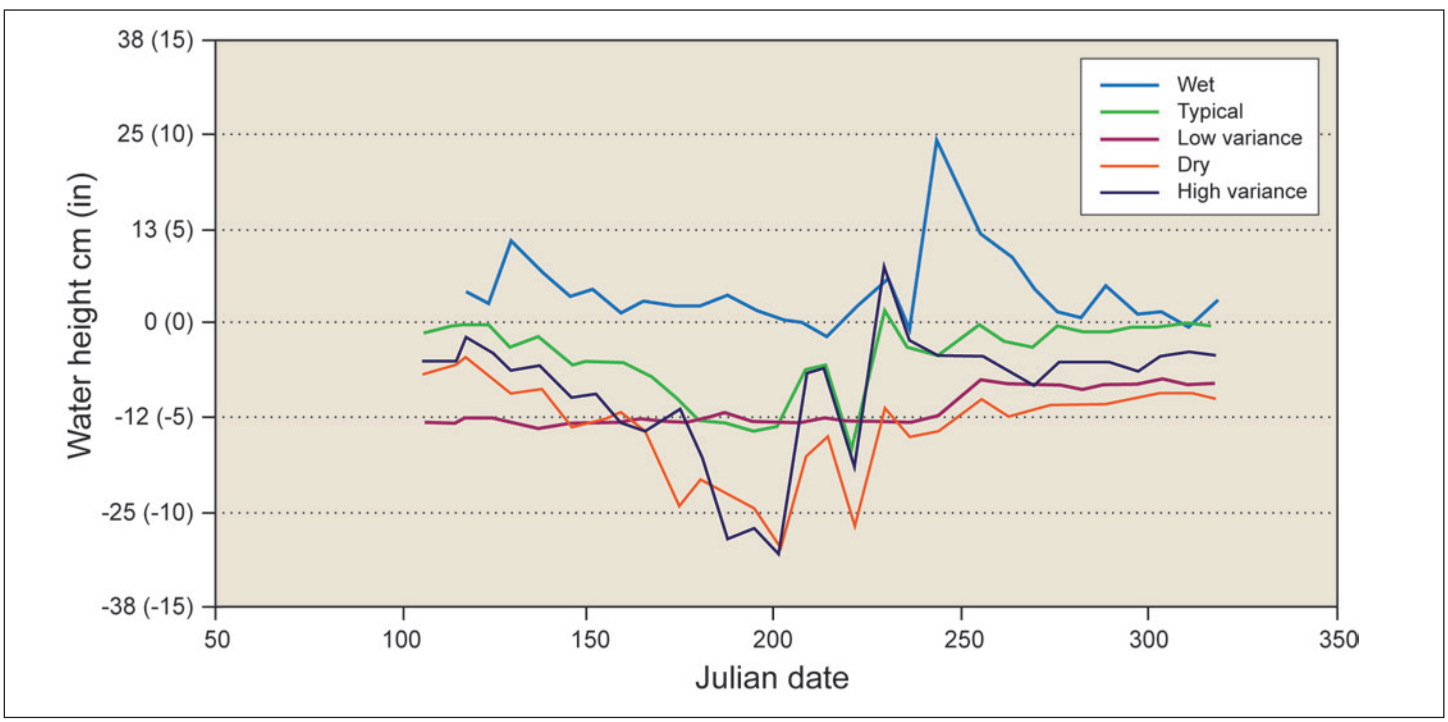

Figure 8. Water table height for five representative pools measured weekly in 2020. This graph shows pools with consistently high water tables (wet), consistently low water tables (dry), and a typical pool. In addition, some pools exhibited low variation in water depth while others exhibited high variance over the season.

Table 4. Kendall's rank correlation coefficient between 3-year growth for a subset of species and variables with p-values for significance in parentheses (NS=nonsignificant).

Species	Common name	Mound[a]	Within 3 years of planting[b]	Median water height[c]	Mean water height[c]	June-August median water table height	Number of days water height 0–5 in below soil	Number of days water height 5–10 in below soil	Number of days water height >10 in below soil
Abies balsamea	Balsam fir	0.11 (<0.01)	0.03 (NS)	-0.09 (<0.01)	-0.10 (<0.01)	-0.10 (<0.01)	-0.07 (<0.05)	-0.02 (NS)	0.11 (<0.01)
Juniperus virginiana	Eastern red cedar	0.08 (NS)	0.01 (NS)	-0.09 (<0.05)	-0.09 (NS)	-0.08 (<0.10)	-0.10 (<0.05)	0.11 (<0.05)	0.15 (<0.01)
Picea glauca	White spruce	0.10 (<0.01)	-0.17 (<0.001)	-0.01 (NS)	-0.01 (NS)	-0.01 (NS)	-0.08 (<0.05)	-0.05 (<0.10)	0.04 (NS)
Picea mariana	Black spruce	0.16 (<0.01)	0.01 (NS)	-0.17 (<0.001)	-0.18 (<0.001)	-0.18 (<0.01)	-0.15 (<0.01)	-0.04 (<0.010)	0.20 (<0.001)
Pinus strobus	Eastern white pine	0.13 (<0.01)	0.01 (NS)	-0.02 (<0.01)	-0.10 (<0.01)	-0.11 (<0.01)	-0.09 (<0.05)	-0.05 (NS)	0.13 (<0.01)
Thuja occidentalis	Northern white cedar	0.03 (NS)	0.11 (<0.001)	-0.07 (<0.01)	-0.08 (<0.01)	-0.10 (<0.01)	-0.06 (<0.01)	-0.02 (<0.10)	0.12 (<0.01)

[a] Mounds were artificially created; positive values mean growth was favorable on mounded versus unmounded sites.

[b] Positive values indicate growth of trees planted within 3 years was relatively high, while negative values indicate that growth was relatively low.

[c] Negative values for water height indicate that tree growth was higher under lower water conditions.

were correlated significantly and positively with mounding. Lastly, young northern white cedar trees (less than 3 years since planting) were positively associated with high growth. In contrast, white spruce trees that were within 3 years of planting showed reduced growth (table 4).

Soil and Water Temperatures

Soil temperatures during the 2020 growing season were consistently colder than ambient temperatures. Temperatures varied depending on depth and location (figure 9). The warmest temperature recorded in lowland soil at a depth of 3 in (8 cm) was 78 °F (25.6 °C) in 2020. Conversely, the warmest soil temperature at a depth of 3 in (8 cm) on upland forested soil with a

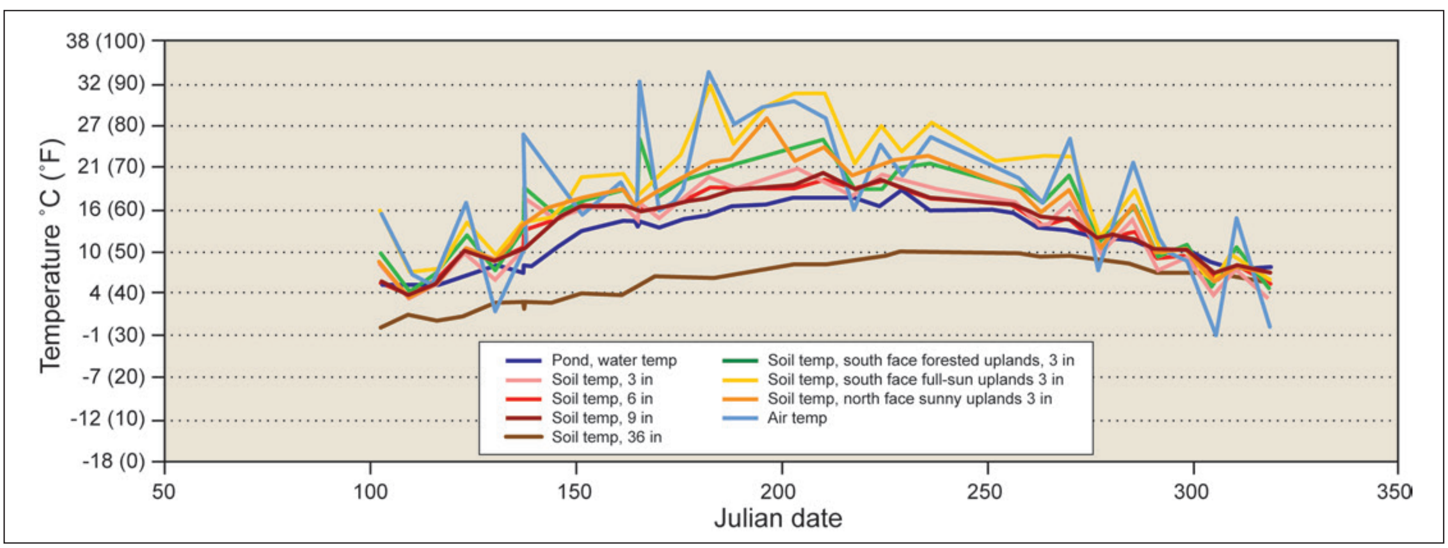

Figure 9. Temperatures (temp) were measured weekly during the 2020 growing season at multiple locations within the planting site, at varying soil depths, and in a nearby pond.

southern aspect reached 85 °F (29.4 °C). On non-forested (fully exposed) upland soils, temperatures reached 90 °F (32 °C) at 3 in (8 cm) deep. Water temperatures in the pools remained cool, with warmest temperatures ranging from 60 °F to 65 °F (15.6 °C to 18.3 °C) which was cooler than the "cold" treatment reported in Holland et al. (2003).

Discussion

The purpose of this observational study was to explore potential management options for converting a *Typha*-dominated wetland to a coniferous swamp. The benefits of a conversion include reducing methane outputs (from *Typha* decomposition), increasing carbon fixation, controlling water levels, and creating a more biodiverse ecosystem. Observations on subsequent performance of planted trees from a variety of species are aimed at providing baseline information that can be used for establishing tree cover on a converted wetland. This project also demonstrated the water table variability of these types of sites.

Variations Among Species and Seed Sources

Northern white cedar had the most annual height growth of the species studied, which was not surprising because this species is well matched to these sites (Johnston 1990). Six years after planting, many of the northern white cedar trees were taller than 96 in (244 cm). Trees planted in in 2013 had a median height of 118 in (300 cm) in 2020. This growth rate is much faster than this species generally grows in the wild, where trees often take 30 years to reach 118 in (300 cm) (Rooney et al. 2001). Relative growth rate of this species slowed as trees became larger but was still more rapid than that of other species, except eastern white pine.

Planted species were native to Michigan (balsam fir, white spruce, and black spruce) south of their typical range limits. Hydric conditions at the site were wetter than optimal, especially for seedlings that were reared in nurseries where water levels are controlled. Foliar diseases on balsam fir were observed on nursery stock. White spruce, an upland boreal species, eventually became established but likely experienced transplant shock in the first 3 years after planting. The need to replant white spruce decreased considerably in later years, suggesting the species was able to adapt its root system to various soils as found previously (Strong and La Roi 1983). Black spruce, commonly found in lowlands, had high survival and was relatively competitive on the site in spite of being moved southward. White spruce and black spruce grew faster than might be expected along their southern range edge and may have benefitted from the site's longer growing season compared to more northerly sites. While it is possible climate-related issues may not be observed until these trees are older, tree adaptation to warmer temperatures has been observed (Bermudez et al. 2021). Forest managers may seek out similar sites, with northerly aspects and cold microsites, to manage as reservoirs to maintain northern species on the landscape even as the climate warms.

Although Atlantic white cedar and pitch pine (*Pinus rigida* Mill.) were moved far west of their natural ranges, growth rates for both species were higher than expected. The seed source for Atlantic white cedar was poorly matched to the local climate (New Jersey versus Michigan), but the species typically thrives in high water tables (Golat and Lowry 1987). Atlantic white cedar had only minor cold damage and grew surprisingly well considering the stock type was younger than other species. These observations suggest that more inland habitats might be possible to conserve Atlantic white cedar populations that are otherwise largely coastal and in areas with dense anthropogenic pressure. Limited research exists on the establishment of this species even in the warmer part of its commercial range in the Carolinas (Hinsley et al. 1999). Pitch pine thrives in a climate warmer than hard pines native to Michigan and is also reported to grow in swamps alongside Atlantic white cedar. Pitch pine may be adapted to future warmer climates as demonstrated by successful regeneration through seeding in sand dunes in Ottawa County, MI (Reznicek et al. 2011).

Cold Damage

Some cold damage occurred on the nonnative trees which appeared as frost heave and/or damaged needles. Frost heave was generally only a temporary setback for the nonnative species (white fir and Atlantic white cedar), and negative growth rates were not frequent enough to have a notable effect on median growth. For Sitka spruce, needles became necrotic on some trees over the winter and were dropped in the spring. After

10 years, only 39 of the 103 planted Sitka spruce survived, nearly all of which experienced periodic foliar loss (mostly during the few years after establishment). The Sitka spruce originated from a coastal Washington seed source (plant hardiness zone 7 versus zone 5 for Michigan). White fir also exhibited slow growth, similar to native balsam fir, which wasn't unexpected since neither species are considered wetland specialists.

Variations Among Stock Types

Because researchers planted a wide variety of stock types and seed sources, the ability to recommend specific stock types and seed sources is limited. Most of the stock types were larger than those typically used for upland plantings (e.g., 2-2 compared with 1-0 or 2-0 stock types). These larger stock types are less economical but were expected to fare better under heavy competition at the site. Based on results from this study, stock types with heights of 24 to 47 in (60 to 120 cm) are recommended for similar wetland tree plantings. Tree seedlings in this size range showed better survival and were easier to transport and plant than those that that exceeded 47 in (120 cm) in height. Small seedlings were at a competitive disadvantage, while trees that exceeded 60 in (150 cm) experienced notable transplant shock.

Typha Removal

Typha spp. dominated this previously forested stand and were present in high densities (approximately 100 stems per yd^2 [120 per m^2]). The labor costs to remove stems mechanically from the entire area were prohibitive, so *Typha* was removed from two small plots as a pilot study. Shade-tolerant trees, such as northern white cedar and balsam fir, established successfully on plots without *Typha* removal, which is a desirable finding for private landowners with limited resources. Hovik and Reinartz (2007) reported a positive effect of removing reed canarygrass on tree growth, although the rooting habits of reed canarygrass and *Typha* are likely different.

Mounding

Mounding, as a strategy to create dry microsites in otherwise wet sites, was beneficial to some species, such as white spruce, in this study and in other studies (Hawkins et al.1995, Londo and Mroz 2001, McMinn 1983, McMinn et al. 1995). For other species, such as northern white cedar, no benefits of mounding were observed, which contrasts with Mehne and Mehne (2014) on the same site. Based on these contrasting results, additional data are needed to compare long-term survival and growth on mounded and unmounded microsites for conifers on these wetlands. Nonetheless, mounding may be helpful to create upland microsites in extremely hydric conditions. Well-decomposed mud, along with a mound size of about 1 gal (4.5 L) of mud per 1 ft (30 cm) of tree height, were observed to be optimal for tree growth.

Management Implications for Conifer Swamps

One of the landowner's management goals for this study site was to shade out invasive species. Observations suggest that the planted trees are becoming established in spite of the challenging hydrological conditions. Some of the first northern white cedar planted in 2002 (prior to the current study) have grown enough to begin shading out invasive *Typha* species. Sufficient shading may have occurred sooner if trees had been planted 4 ft (1.2 m) apart rather than 8 to 9 ft (2.4 to 2.7 m). Once northern white cedar was established, other shade-tolerant (and markedly more aggressive) shrubs such as red osier dogwood (*Cornus sericea* L.) also became established and appeared to outcompete *Typha* spp. If the planted trees continue growing at their current rate, approximately 70 percent of the study area will be shaded in another 5 to 7 years, which is about 20 years from the time of planting until canopy closure.

The study site was typical of conifer wetlands in the Midwestern United States, with high annual variability in hydrology. A relatively weak influence of hydrology on tree growth (table 4) may be due to uneven sample sizes, variable water, or other factors. Some trends were apparent, however. The length of time a given area remained wet was a better predictor for species performance than whether the area experienced pulses of extreme wet or dry conditions. Recommendations for land managers include evaluating planted trees at different times of the year, particularly in the summer when the water table is presumably at its lowest point, as well as monitoring across multiple years to make a complete determination on the planting's success or failure.

The cost to establish trees in the study was approximately four times the cost of tree planting on an optimal upland site due to the larger stock used and the labor required. When the recommendations from this study were repeated on a small-scale (<1 ac [0.4 ha]), followup planting, the cost was 2.7 times the expected cost for an optimal upland planting, which could be logistically feasible in some scenarios.

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