Wetlands
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WETLANDS RESTORATION

Flood Exposure Affects Long-Term Tree Survival in Compensatory Mitigation Wetlands

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Abstract

Survival of planted trees is commonly used as a performance metric for compensatory mitigation wetlands. However, establishing floodplain forest through planting is difficult due to flood-induced tree mortality. We used multiyear tree census and hydrologic data from 17 compensatory mitigation projects in Illinois, USA, to relate planted and volunteer tree establishment to flood frequency, depth, and duration. Annual survival of planted trees decreased with greater annual maximum flood depth and duration. By the end of official compliance monitoring, sites with greater flood exposure had greater planted tree mortality. We resurveyed 10 sites that were at least 10 years old, and found that long-term tree survival was significantly lower in sites with greater flood exposure. Naturally colonizing trees differed in species composition from planted trees; specifically, wind dispersed species were well-represented among volunteer trees, whereas hard mast species were absent. There was no clear relationship between volunteer tree recruitment and measured flood variables. Across all sites, compliance with tree survival standards was poor, but influenced by hydrologic conditions. Current performance standards for tree survival may be unrealistic in restored wetlands that are exposed to long-duration floods. Regulators and practitioners should seek alternative methods to establish desired floodplain forest structure and function.

Keywords Floodplain forest · Hydrology · Monitoring · Performance standards · Reforestation · Tree recruitment

Introduction

Section 404 of the U.S. Clean Water Act, which is enforced primarily by the U.S. Army Corps of Engineers (Corps), prohibits the discharge of dredge or fill materials into waters of the United States, including wetlands. The current national policy goal is "no-net-loss" of wetland area and function (*Federal Register* 2008). Filling a wetland may be permitted if wetland impacts are unavoidable, but wetland destruction must be compensated for through restoration, creation, or enhancement of wetlands elsewhere. These compensatory mitigation wetlands are required to meet a set of site-specific performance standards approved by the Corps, usually within a five-year monitoring period (National Research Council 2001). Performance standards allow regulators to evaluate if mitigation projects are meeting objectives (Streever 1999; *Federal Register* 2008). A compensatory mitigation wetland is considered to be a success if, after a specified monitoring period, the site meets all performance standards stated in a permit or agreement between the regulatory agencies and the permittee. Corps District Engineers have broad discretion in developing performance standards for their districts (Urban 2008). Standards have been based on biological metrics, such as plant community characteristics, or abiotic metrics, such as hydrologic regime or soil characteristics (Environmental Law Institute 2004).

Impacts to forested wetlands are often compensated for by restoring forested wetlands on former agricultural land in river floodplains. Restoration of agricultural bottomland requires restoration of hydrological flows, for example through the removal of drain tiles or levees, followed by planting of flood-tolerant tree species. Success is evaluated, in part, based on survival of these planted trees (Breaux and Serefiddin

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1999; Streever 1999; Matthews and Endress 2008). For instance, project performance standards may specify a minimum percentage for tree survival by the end of the monitoring period. Tree survival rates at mitigation wetlands are not well documented in peer-reviewed literature. The few studies that have documented tree survival in mitigation wetlands have reported poor survival (Jarman et al. 1991; Pennington and Walters 2006; Matthews and Endress 2008; Van den Bosch and Matthews 2017). For example, Matthews and Endress (2008), in a review of 38 mitigation projects, found that 19 projects had performance standards requiring a minimum survival of planted trees by the end of site monitoring. Of these 19 projects, only four met their site-specific performance standard for planted tree survival by the final year of monitoring. Mitigation wetlands that do not meet tree survival criteria can be deemed unsuccessful by regulatory agencies, and the party responsible for mitigation must replant trees, often at great expense, to achieve compliance.

Reforestation in floodplains can be particularly challenging due to disturbances from frequent or prolonged flooding (King and Keeland 1999). Planted trees, even those tolerant of intermittent flooding, suffer high mortality with prolonged inundation during the growing season. Saturation of the root zone prevents oxygen from reaching root tissues, leading to reduced root and shoot growth, root decay, reduced mycorrhizal biomass, loss of leaves, severely reduced photosynthetic rates and growth, reduced or forestalled reproduction, and eventually, plant mortality (Kozlowski 2002; Glenz et al. 2006). Although many wetland plants have several adaptations that allow them to tolerate anoxic soil conditions (Cronk and Fennessy 2001; Kozlowski 2002; Glenz et al. 2006), long-duration or deep flooding is particularly stressful for woody plants that usually occupy sites that flood intermittently or infrequently. Furthermore, additional stressors are associated with flooding, including sedimentation and mechanical disturbance to trees from floating debris (Bendix 1999; Bendix and Hupp 2000; Hughes et al. 2001; Richardson et al. 2007).

As is the case for survival of planted trees, hydrologic regime is the primary constraint on establishment and distribution of naturally colonizing trees in floodplains (Hupp and Osterkamp 1985; Bell 1997; Toner and Keddy 1997; Middleton 2000). Tree seedlings can establish in areas that are temporarily flooded, allowing forested wetlands to develop, but as the frequency or duration of flooding increases, forested wetlands transition to emergent wetlands that are dominated by flood-tolerant herbaceous species (Toner and Keddy 1997; De Jager et al. 2016). Dispersal limitation is also an important constraint on natural colonization in floodplain restorations, particularly for species that are desired by regulatory agencies, such as bottomland oaks (*Quercus* spp.) and hickories (*Carya* spp.) (Kruse and Groninger 2003, Battaglia et al. 2008). Despite the importance of natural colonization in floodplain afforestation projects, few studies have evaluated volunteer tree recruitment in compensatory mitigation wetlands. Natural tree colonization is not often assessed during mitigation site monitoring and is rarely used as a mitigation performance standard (Matthews and Endress 2008).

There is currently little guidance for tailoring wetland mitigation performance standards to site-specific hydrologic conditions, and we are unaware of previous studies that have evaluated how flooding influences the achievement of vegetation performance standards in compensatory mitigation wetlands. The goals of this study were to determine the effect of flood events on the establishment of planted and naturally recruiting trees in restored floodplain forests. This study consists of three specific objectives: (1) relate annual planted tree survival rates at compensatory mitigation wetlands to flooding, (2) relate long-term (>9 years) planted tree survival to site hydrologic regimes, and (3) evaluate natural recruitment of trees in compensatory mitigation wetlands. We expected that annual and long-term survival of planted trees would be lower in restored floodplain forests that were subject to frequent, deep, or long-duration floods.

Methods

Study Sites

The Illinois Department of Transportation (IDOT) has restored floodplain forest wetlands throughout Illinois as compensation for wetlands impacted during road construction and maintenance projects. The Illinois Natural History Survey (INHS) and the Illinois State Geological Survey (ISGS) collected vegetation, soils, hydrologic, and topographic data at the IDOT compensation wetlands to monitor for attainment of wetland criteria and performance standards.

Flood-tolerant tree species were planted at these sites following the restoration of wetland hydrology. Stock types included bareroot seedlings, 3- and 5-gal containerized stock, Root Production Method-produced stock (Forrest Keeling, Elsberry, MO, USA), and balled and burlapped saplings. Although Corps Districts often specify requirements for planted tree survival (e.g., U.S. Army Corps of Engineers Seattle District 2017), required stock types are not often specified. Planting requirements are not consistent among Corps Districts, and in many cases, decisions regarding stock types, stocking rates, planting methods, and management are made on a case-by-case basis. However, regulatory agencies often require the planting of hard-mast producing tree species such as oaks (*Quercus* spp.) and pecans (*Carya illinoinensis*). Stock types and planting methods varied among our study sites, and annual monitoring reports did not consistently report this information.

We considered 59 IDOT wetland compensation sites, established since 1992, for potential inclusion in this study. The basic criteria we used for site selection were that (1) the site is located within a floodplain and receives direct flooding or has a hydrology that is influenced by the flood regime of the adjacent stream, (2) water-level data were collected at least daily by ISGS to provide adequate resolution for quantifying flood exposure, (3) trees were planted at the site and tree survival was monitored annually by INHS, and (4) the sampling period for hydrologic and tree survival data overlapped for at least three years. Of the 59 sites initially considered for the analysis, 17 sites met these criteria. Sites were monitored between 1996 and 2011, and the overlapping duration of hydrologic and tree survival monitoring ranged from 3 to 8 years after initial wetland restoration activities were completed. Contributing drainage areas for the selected sites ranged from 6.5 to ~1.8 million km^2; thus, the sites represent a wide range of drainage area and correspondingly a wide range of flood regimes.

Hydrologic Data

Surface-water data used for the analysis were either collected by ISGS during wetland mitigation site monitoring or were obtained from online stream gaging databases maintained by the U.S. Geological Survey (USGS 2012) or Corps (USACE 2012). Data collected at the wetland mitigation site by ISGS were acquired with electronic water-level dataloggers set at sampling intervals ranging from daily to hourly. At two sites, we used stage records from nearby gaging stations to develop calibration curves and applied these curves to estimate the hydrograph to supplement incomplete on-site datasets (see Toner and Keddy 1997). We used the surface water data to evaluate flood characteristics for each flood event during the monitoring period at each site. While a variety of measures have been developed and used for relating hydrologic regime to ecological variables (e.g., see Richter et al. 1996, Toner and Keddy 1997), our intent was to evaluate the frequency, depth, and duration of inundation of the wetland plant community.

We selected a threshold elevation at each site as the minimum site elevation that indicated most of the site was flooded, thus excluding hydrologic fluctuations that were restricted to within on-site water features (i.e., fluctuations within ponded areas). To select the threshold elevation, hydrographs were visually examined to distinguish flood events from fluctuations within ponded areas and a minimum floodplain elevation was selected to filter out hydrograph peaks that were not associated with river flooding. We defined three hydrologic metrics: (1) annual flood frequency, or the number of flood events occurring in a given year; (2) annual maximum flood depth (meters) at the threshold elevation; and (3) annual maximum flood duration (days) at the threshold elevation.

In addition, we used a fourth measure of flood intensity as described in Ahmad and Ahmed (2003). For clarity, we use the term flood exposure index (FEI) to distinguish this measure from other measures of flood intensity based on stream discharge (e.g., Walling and Teed 1971). The formula for FEI is given as:

$$\mathrm{FEI} = D_{avg} \times R,$$

where D_{avg} is the average depth above the specified elevation threshold and R is the duration of the flood above the specified threshold elevation. The unit of FEI is meter-days (m.days). FEI was calculated for each flood event during each year of the monitoring period. We chose to use the annual maxima for FEI ($\mathrm{FEI}_{\mathrm{max}}$), flood depth, and flood duration as the independent variables for statistical analyses because these values represent the highest magnitude flood in a given year and therefore represent the flood event that has the maximum effect on planted trees at a mitigation site.

Tree Counts

INHS evaluates whether IDOT mitigation wetlands achieve site-specific performance standards related to vegetation establishment, including planted tree survival. Throughout the specified monitoring period for each mitigation wetland, INHS visited sites annually during late summer or early fall and tallied surviving trees by species. Planted trees were easily distinguished from volunteer trees based on species, size, and arrangement in rows. For this project, we compiled existing data from annual tree survival counts conducted by INHS. IDOT often replanted trees in response to mortality. We assessed annual planted tree survival based on the number of trees surviving each year relative to the number of trees alive in the previous year plus any newly planted trees.

We used generalized linear mixed-effects models with binomial errors and logit link (package lme4 in R 3.5.1 statistical software; Bates et al. 2015, R Core Team 2018) to evaluate the influence of annual flood disturbance on the proportion of planted trees surviving each year at each site. Mixed models are appropriate for data that are organized at more than one level (Singer 1998). In this case, the data are organized at two levels, with years nested within wetland sites. The response variable was proportion of trees surviving each year. Separate models were constructed, each with one of four predictor variables: annual flood frequency, annual maximum depth, annual maximum flood duration, and $\mathrm{FEI}_{\mathrm{max}}$. Flood variables are likely collinear; therefore, we did not include multiple flood variables in the same model. Site identity, and age nested within site, were included as a random factors in all models to account for underlying differences in tree survival among sites. Additionally, observation-level random effects were included to correct for overdispersion in the data

(Harrison 2014). Annual maximum flood duration and FEI_{max} were right-skewed and were log-transformed prior to analysis. We repeated these analyses separately for individual tree species that were planted in at least 10 of the 17 wetlands. These species included pecan (*Carya illinoinensis*), green ash (*Fraxinus pennsylvanica*), sycamore (*Platanus occidentalis*), swamp white oak (*Quercus bicolor*), and pin oak (*Quercus palustris*).

Wetland Revisits

During the summer of 2014, we revisited 10 IDOT mitigation wetlands, which ranged in age from 10 to 19 years, to assess long-term planted tree survival. At each site, at least two people walked parallel to one another, approximately 10 m apart, back and forth across the entire site, and tallied planted trees by species.

Monitoring of vegetation and hydrology by INHS and ISGS at these older mitigation wetlands had ended by 2014. Therefore, we did not have access to hydrologic monitoring data for these sites through 2014. However, we were able to relate long-term planted tree survival to typical site flooding regimes by averaging hydrologic variables across all years for which hydrologic data were available (at least three years).

To quantify recruitment by naturally colonizing (volunteer) trees, we established three representative sampling transects at each of 11 mitigation wetlands that ranged in age from 9 to 19 years. We measured tree diameter-at-breast-height (DBH) of all woody stems greater than 5-cm DBH in a 10 × 50-m plot along each transect. Using tree tallies and DBH measurements, we calculated the stem density (number of stems ha^{-1}) and basal area (m^2 ha^{-1}) for each planted and volunteer tree species at each site. In addition, we determined straight-line distance between the edge of each mitigation wetland and the nearest mature forest patch identified on aerial imagery in a Geographic Information System.

We used generalized linear models (in R 3.5.1; R Core Team 2018) to describe the relationship between proportion of planted trees surviving in 2014 and site age, as well as annual averages for flood frequency, maximum depth, maximum duration, and FEI_{max}. We used quasibinomial errors, to correct for overdispersion in the data, and a logit link. We related density and basal area of volunteer trees to site age, distance from mature forest, and annual averages for flood frequency, maximum depth, maximum duration, and FEI_{max} using linear regression.

Results

Tree survival in IDOT mitigation wetlands each year decreased as annual maximum flood duration and depth increased (Table 1). Flood exposure index (FEI_{max}), which combines both depth and duration, was the best predictor of annual tree survival (Table 1, Fig. 1a).

Annual survival of *C. illinoinensis*, *F. pennsylvanica*, and *Q. bicolor* decreased significantly as FEI_{max} increased, indicating that these species suffered greater mortality during years with deeper and/or more prolonged flooding (Table 1, Fig. 1). Annual survival of *P. occidentalis* and *Q. palustris* was not significantly related to annual flood exposure (Table 1).

By the final year of mitigation site monitoring, planted tree survival was approximately 36% across all species in all sites. With the exception of *C. illinoinensis*, survival rates were similar among the most frequently planted species: *C. illinoinensis* - 30%, *F. pennsylvanica* - 47%, *P. occidentalis* - 47%, *Q. bicolor* - 47%, and *Q. palustris* - 46%. Planted tree survival in this final year of monitoring was significantly lower in wetlands with greater maximum flood depth, averaged across all years up to and including the final year of monitoring (Table 2).

During the summer of 2014, we reevaluated planted tree survival at 10 IDOT mitigation wetlands that were at least 10 years old. In all cases, tree survival declined, sometimes dramatically, between the end of the mitigation site monitoring period and 2014 (Fig. 2). Tree survival through 2014 decreased significantly as average annual flood frequency, average maximum annual flood duration, and average annual FEI_{max} increased (Table 2, Fig. 2).

In 2014, volunteer trees greatly exceeded planted trees in terms of stem density, basal area, and number of species (Table 3). However, species composition differed between naturally colonizing trees and planted trees. Natural colonization was dominated by wind-dispersed, early-colonizing species such as *Salix nigra*, *Acer saccharinum*, *Populus deltoides*, *F. pennsylvanica,* and *Salix interior.* We observed no colonization by *Quercus* spp. or *Carya* spp. in the sampled plots. In contrast, *Quercus* spp., along with *Betula nigra*, were well-represented among planted trees.

Stem density, basal area, and species richness of naturally colonizing trees were not clearly related to flood exposure, site age, or distance from the nearest mature forest (Table 2). Total basal area of naturally colonizing trees increased significantly as average maximum flood depth increased (Table 2). However, this effect was due entirely to a single outlier site with very high volunteer basal area, and this relationship was no longer significant after the outlier was removed.

Discussion

Planted Tree Survival and Flooding

The overall planted tree survival rate in compensatory mitigation wetlands was low (36%). Tree mortality was clearly

Table 1 Results of binomial mixed models (coefficient estimates and associated *z* values) to evaluate the influence of flood disturbance on annual planted tree survival

		Flood frequency (number of events)†	Maximum flood depth (m)	ln (Maximum flood duration [days])	ln (FEI_{max} [m.days])
Proportion of planted trees surviving (*n* = 76 obs.)	*z*	–	−1.61	−1.45	−2.20*
	estimate	–	−0.479	−0.298	−0.471
Proportion of *C. illinoinensis* surviving (*n* = 44 obs.)	*z*	−0.30	−0.47	−3.53***	−3.11**
	estimate	−0.044	−0.268	−1.319	−1.192
Proportion of *F. pennsylvanica* surviving (*n* = 56 obs.)	*z*	0.10	−3.20**	−2.86**	−3.71***
	estimate	0.013	−1.41	−0.965	−1.190
Proportion of *P. occidentalis* surviving (*n* = 48 obs.)	*z*	0.77	−1.77	−0.80	−1.61
	estimate	0.103	−0.570	−0.199	−0.396
Proportion of *Q. bicolor* surviving (*n* = 69 obs.)	*z*	−2.03*	−1.19	−1.93	−2.70**
	estimate	−0.234	−0.646	−0.585	−0.840
Proportion of *Q. palustris* surviving (*n* = 59 obs.)	*z*	−0.71	−0.73	−0.20	0.17
	estimate	−0.057	−0.364	−0.050	0.049

*$p < 0.05$, **$p < 0.01$, ***$p < 0.001$

†Model for proportion of planted trees surviving failed to converge

related to flood exposure during individual years, and survival rate continued to decline after the end of the mandatory site monitoring periods, particularly in those sites with the greatest exposure to flooding. These results are consistent with other studies that have shown tree mortality to be greater in sites with greater exposure to flooding (Acker et al. 2003; Ernst and Brooks 2003; Howard 2012).

Flood exposure appears to impose a “ceiling factor” effect on planted tree survival; survival can be either high or low at sites with lesser flood exposure, but survival is low at sites with greater flood exposure. Thus, other factors are likely important for determining tree survival in sites with less flood exposure. For example, site preparation, the size of planted trees, and management varied among sites and may have influenced survival rates. Furthermore, factors such as herbivory and competition with naturally colonizing vegetation have been shown to influence planted tree growth and survival in other studies (Groninger et al. 2004; Jacobs et al. 2004; Hovick and Reinartz 2007), and may have contributed to tree mortality in our study sites. We did not have data on these additional factors, and we are unable to assess their importance relative to flooding in this study.

Depth and duration of inundation were more important than flood frequency in determining tree survival. Most woody species are tolerant of brief flooding, but tissue damage accumulates with prolonged flooding and soil anoxia (Glenz et al. 2006). Deeper flooding is more stressful because deeper water prevents oxygen diffusion from the atmosphere and prevents the resupply of oxygen from leaves and lenticels to the roots (Hook 1984; Glenz et al. 2006). Thus, the combination of prolonged inundation of soils and deep inundation is particularly stressful for plants, reflected in our study by the fact that FEI_{max} often explained more variation in tree survival than maximum depth or duration alone. Flood frequency, although less important in this study, has been shown to be important elsewhere (Glenz et al. 2006). For instance, with more frequent flooding, trees have a shorter recovery time between subsequent floods (Toner and Keddy 1997).

Mortality in response to flooding appears to be episodic in compensatory mitigation wetlands. Our results and other studies (e.g., Yin 1998; Acker et al. 2003; Damasceno-Junior et al. 2004) indicate that tree mortality may be low in most years, but it rises sharply in flood-exposed sites during years with extreme flooding. For example, Yin et al. (2009) reported that the 1993 flood on the Mississippi River resulted in a 57% decrease in of the number of trees and a 33% decrease of total tree basal area in Upper Mississippi River floodplain study sites. These results suggest that in similar afforestation projects, in the absence of continual replanting, the number of surviving planted trees will ratchet downward after each major flood.

Mortality rates were greater for *C. illinoinensis* compared to the other four species in this study. This finding is consistent with experimental research that has shown differences among species in flood tolerance. For example, Krzywicka et al. (2017) reported poor survival in response to flooding for planted seedlings of *Juglans nigra* and *C. illinoinensis*, intermediate for *Q. palustris*, and greatest survival for *Q. bicolor*. Kabrick et al. (2012), in an experimental planting study, found that *C. illinoinensis* had high survival in response to flood treatments but suffered extensive stem dieback. *Quercus palustris*, *Q. macrocarpa*, and especially *J. nigra* were also sensitive to flooding treatments, whereas *Q. bicolor* was found to be flood tolerant (Kabrick et al. 2012). Since

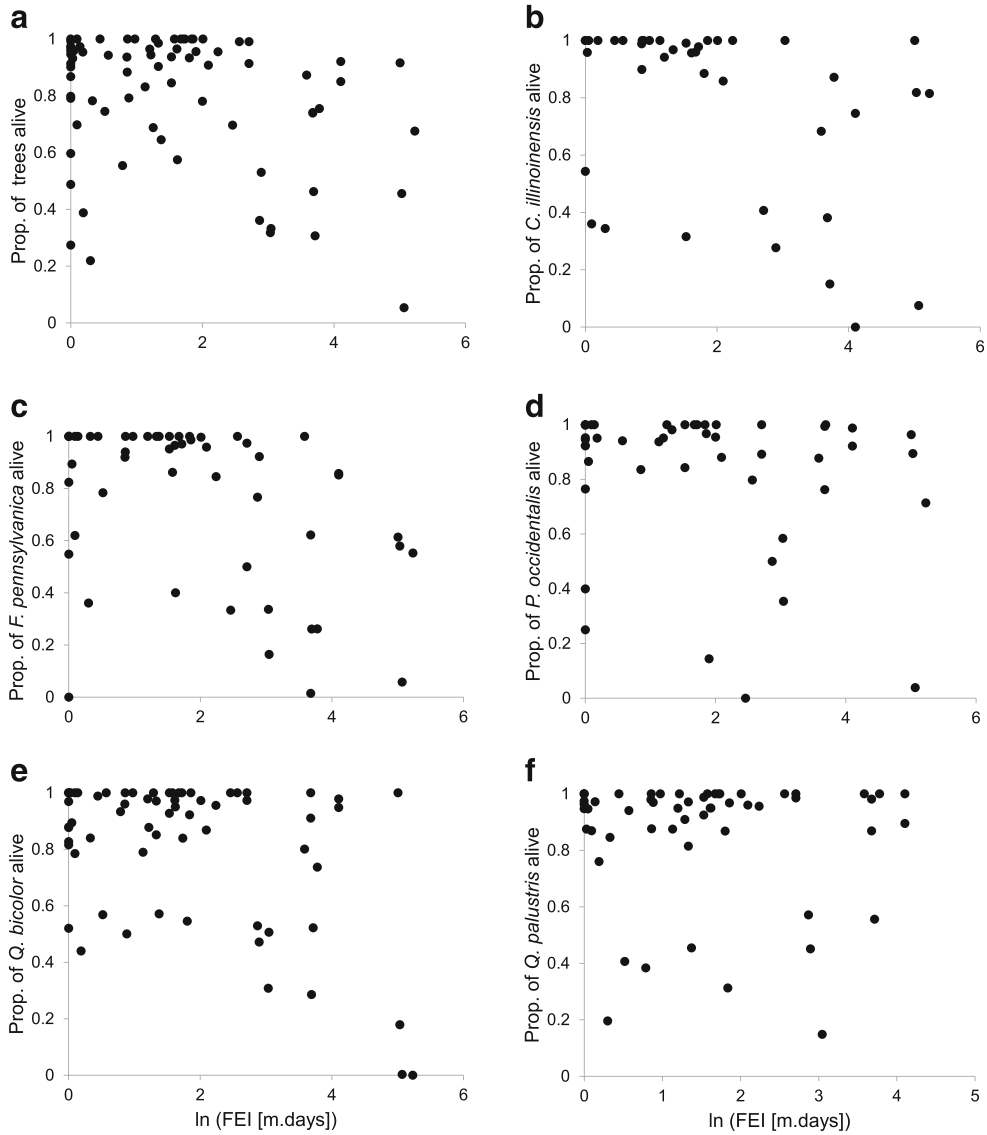

Fig. 1 Annual proportion of planted trees surviving vs. log-transformed annual flood exposure index (FEI_{max}), for all planted tree species (**a**) and species planted at more than ten mitigation wetlands (**b**–**f**)

species-specific flood tolerance seems to vary widely among studies and sites, it is difficult to define precise flood tolerance ratings or planting recommendations. However, broadly defined tolerance ratings have been developed (Teskey and Hinckley 1977; Hook 1984) and may serve as a guide for planting trees in compensatory mitigation wetlands.

Volunteer Tree Establishment

Flood disturbance sets back ecological succession and allows establishment of new species in floodplain communities. Floods destroy existing vegetation and deposit propagules and fresh sediments that provide new sites for seed germination and seedling establishment (Naiman and Décamps 1997; Bendix and Hupp 2000; Richardson et al. 2007). After agricultural abandonment or natural disturbances such as major floods, early succession of forested wetlands in the eastern United States is dominated by a few species of light-seeded, easily dispersed trees such as *Salix* spp. and *Populus deltoides* (Middleton 2003; Yin et al. 2009). These same species often dominate the woody community of recently restored floodplains (DeBerry and Perry 2012).

Compared with species that produce numerous wind-dispersed seeds, hard-mast species do not readily colonize restorations or naturally regenerating forest stands (Shear et al. 1996; Battaglia et al. 2008; Yin et al. 2009). Hard-mast tree species provide valuable food resources for wildlife, making these trees a priority for floodplain forest restoration (Shear et al. 1996), and regulatory agencies often require the planting of hard-mast producing tree species in mitigation wetlands. Our data on natural colonization in mitigation wetlands, and the lack of recruitment by hard-mast species such as

Table 2 Results of models to evaluate the influence of average annual flood disturbance, years since restoration, and distance to nearest forest on long-term tree survival (binomial models) and volunteer tree recruitment (linear models)

		Flood frequency (number of events)	Maximum flood depth (m)	ln (Maximum flood duration [days])	ln (FEI_{max} [m.days])	Years since restoration	Distance to nearest forest (m)
Proportion of planted trees surviving at the end of site monitoring	t	−0.32	−2.86*	−0.20	−1.03	–	–
	estimate	−0.023	−0.818	−0.037	−0.181	–	–
Proportion of planted trees surviving in 2014	t	−2.45*	−1.74	−2.98*	−2.80*	−2.18	–
	estimate	−0.490	−0.499	−0.571	−0.515	−0.257	–
Volunteer tree basal area	$F_{1,9}$	0.1	18.6**	0.07	1.85	1.04	0.27
	estimate	−0.565	3.375	4.483	3.868	1.693	0.058
	r^2	0.011	0.674	0.008	0.171	0.103	0.029
Volunteer tree stem density	$F_{1,9}$	0.05	2.98	0.94	0.00	1.13	0.88
	estimate	−21.730	469.596	−219.939	−12.729	94.895	−2.824
	r^2	0.001	0.249	0.094	0.000	0.111	0.089
Volunteer tree species richness	$F_{1,9}$	0.82	0.31	0.52	0.99	0.89	0.71
	estimate	0.233	−0.489	−0.474	−0.602	0.242	−0.007
	r^2	0.083	0.034	0.055	0.099	0.090	0.073

$*p < 0.05$, $**p < 0.01$

Quercus spp. and *Carya* spp., are consistent with this observation and support the contention that if these species are desired, they should be planted.

Contrary to expectations, we found little effect of flooding on volunteer tree recruitment. Other studies have found hydrology to be a strong filter for woody species recruitment. For example, sites on the Upper Mississippi River floodplain that were flooded for longer than 40% of the growing season were almost entirely dominated by *Acer saccharinum* and had low woody species diversity (De Jager et al. 2012). Several studies have shown that flood exposure, or elevation relative to a nearby river or stream, is a strong predictor of woody species composition in floodplain forests (Hall and Harcombe 1998; Bendix 1999; Battaglia et al. 2002; Turner et al. 2004; Loučková 2012; Marks et al. 2014). We found

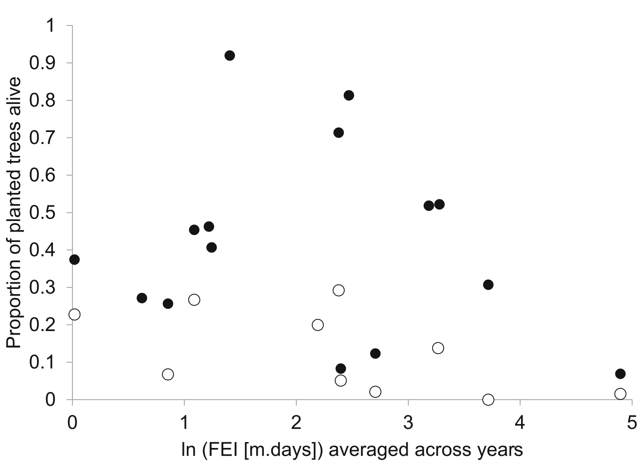

Fig. 2 Proportion of planted trees surviving in mitigation wetlands at the end of site monitoring (filled circles) and in 2014 (open circles) vs. log-transformed, average annual flood exposure index (FEI_{max})

volunteer tree recruitment to be highly variable among sites, and it is likely that unmeasured factors such as seed limitation and herbivory, in addition to flood exposure, affected volunteer recruitment.

Implications for Restoration Practice and Compensatory Mitigation

In 2008, the U.S. Environmental Protection Agency and the Corps issued updated rules for wetland compensation that clarified the use of ecological performance standards. The new rules require that mitigation plans contain ecological performance standards defined as "…observable or measurable physical (including hydrological), chemical and/or biological attributes that are used to determine if a compensatory mitigation project meets its objectives" (*Federal Register* 2008, p. 19672). Additionally, the new rules outline other principles for ecological performance standards such that they need to be objective and verifiable and based on best available science, and that they should consider hydrologic variability (*Federal Register* 2008).

Given the poor survival of planted trees observed in this and other studies, and the clear relationship between flooding and planted tree mortality, it is unrealistic to expect that species such as oaks and pecans can be successfully established at all mitigation wetlands. Typical tree survival performance standards may be unachievable at many mitigation wetlands (Matthews and Endress 2008; Van den Bosch and Matthews 2017), particularly given the conflicting demands for restoring sites that are wet enough to meet jurisdictional wetland criteria over the entire site while simultaneously supporting floristically diverse, high-quality plant communities. Performance

Table 3 Basal area and stem density of planted and volunteer tree species in mitigation wetlands

Species	Average basal area (m^2 ha^{-1})		Average stem density (stems ha^{-1})	
	planted	volunteer	planted	volunteer
Acer negundo		0.05		13.94
Acer saccharinum	0.75	1.17	12.12	134.55
Betula nigra	0.50		56.36	
Carya illinoinensis	0.10		9.70	
Cercis canadensis		<0.01		0.61
Cornus drummondii		<0.01		1.21
Fraxinus pennsylvanica	0.14	0.40	12.12	67.27
Juniperus virginiana		0.01		0.61
Liquidambar styraciflua	0.03	0.04	8.49	3.64
Morus alba		<0.01		0.61
Platanus occidentalis	0.06	0.36	15.15	20.00
Populus deltoides	0.14	1.81	2.42	132.73
Quercus bicolor	0.15		32.72	
Quercus lyrata	0.03		3.64	
Quercus palustris	0.19		20.00	
Salix amygdaloides		0.08		3.64
Salix interior		0.36		47.27
Salix nigra		4.67		222.42
Ulmus americana		0.01		3.03
All species	2.09	8.95	172.73	651.52

standards are unlikely to be achieved by continually replanting hard-mast species in locations where previously planted trees have died due to prolonged flooding. Not all restoration goals are achievable at every site. As such, mitigation performance standards must be realistically attainable given likely hydrologic conditions at a site.

Planting appropriate species in appropriate locations is critical for successful reforestation. We recommend monitoring hydrology and analyzing existing stage data, if available, at proposed mitigation sites prior to tree planting. These data could be combined with baseline information on the relationship between hydrology and vegetation communities in natural reference wetlands to identify appropriate planting locations within and among sites and to set more realistic performance standards for mitigation wetlands (e.g., Johnson et al. 2014). In addition to identifying appropriate planting locations, proper restoration management can reduce tree mortality. Larger individuals are less susceptible to flood stress, so investing in larger trees at the outset of restoration may improve success (Lin et al. 2004; Stanturf et al. 2004). Similarly, tall herbaceous plants such as giant ragweed (*Ambrosia trifida*) should be controlled to eliminate competition and shading, which allows planted trees to grow fast enough to escape some flood risk (Hall and Harcombe 1998; Stanturf et al. 2004).

As an alternative to exclusively planting hard-mast species, planting fast-growing species such as cottonwoods and silver maples does provide some benefits for restoration. First, even these species will not readily colonize reforestation sites unless nearby seed sources are available, so planting can increase overall stocking rates and tree diversity (Allen 1997; Groninger 2005; Lockhart et al. 2006). Second, fast-growing species hasten the development of vertical forest structure, providing wildlife habitat, including habitat for nesting birds (Twedt et al. 2002). Third, rapid canopy closure and the development of a shaded understory may prevent the establishment of aggressive, invasive plants such as reed canarygrass (*Phalaris arundinacea*) (Peralta et al. 2017). It should also be recognized that natural tree colonization can supplement planting, even in sites with extreme flooding where planted hard-mast species are unlikely to persist.

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