**Hydrology and soil magnetic susceptibility as predictors of planted tree survival in a
restored floodplain forest**

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Abstract

Flooding in floodplain forests is an important abiotic constraint on tree recruitment, as well as on planted tree survival and growth in restorations. Nevertheless, trees are often planted in floodplain restorations without regard to a site's hydrologic context, resulting in poor survival. There is a need for improved tools for identifying critical abiotic factors that control tree growth and mortality at reforestation sites. We planted 400 bareroot tree seedlings of four commonly planted species in plots along five 100-m transects along a hydrologic gradient in a recently restored wetland to determine the effect of hydrology on planted tree survival. We evaluated the effect of exposure to flooding on survival and growth for two growing seasons. We also evaluated the use of soil magnetic susceptibility (MS) as a proxy for soil drainage and predictor of tree survival and growth. Soil MS is easily measured and mainly reflects the concentrations of ferrimagnetic minerals, which can dissolve with iron reduction in poorly drained soils. In the first year, the overall survival rate of the planted seedlings was 61%. By the end of the study period, survival had declined to 25%. Of the four species planted, *Quercus bicolor* survived best, followed by *Quercus palustris* and *Carya illinoensis*. No *Juglans nigra* seedlings survived to the end of the study. Duration of inundation and species identity were important predictors of growth and survival; as duration of inundation increased, height growth and probability of survival for each species decreased. Soil MS was not strongly correlated with either flood duration or elevation and was not an effective predictor of tree survival at this site, but might be a useful tool to guide planting in areas with a more pronounced hydrologic gradient. This research can help provide higher precision tree planting in accordance with species' natural distribution across soil moisture gradients, ultimately leading to greater planting success in restorations.

Keywords: bottomland forest; flooding; restoration; seedling; tree growth; wetland mitigation

1. Introduction

Inundation in floodplain forests is an important constraint on tree establishment (Middleton 2000, Battaglia et al. 2002), as well as on planted tree survival and growth in restorations (Pennington and Walters 2006, Kabrick et al. 2012). Flooding decreases oxygen availability and light at the soil surface (Teskey and Hinckley 1977, Baskin and Baskin 1998), which can lead to poor seedling survival. Floodplain forests are often the focus of tree planting and restoration efforts, especially in the southern United States (Clewett and Lea 1990, Sharitz 1992, Noss et al. 1995, Stanturf et al. 2001). However, current approaches to floodplain restoration often ignore site-specific hydrologic variability, resulting in unexpected mortality of planted species and failure to achieve restoration goals (King et al. 2006, Pennington and Walters 2006). Since individual tree survival, along with overall species composition and diversity, is determined primarily by hydrology in floodplain forests (Toner and Keddy 1997, Turner et al. 2004), in-depth knowledge of the hydrologic regime at a restoration site is essential for reforestation success. In addition to soil inundation, other site conditions influencing tree establishment and survival include light availability and herbivory (Menges and Waller 1983, Lin et al. 2004, Turner et al. 2004). Understanding these factors, particularly flooding, and their effects on tree survival and growth in a site-specific context will aid restoration managers in their reforestation efforts.

Qualitative flood tolerance rankings have been used to describe species' ability to survive a certain depth of flooding over a number of days (Bell and Johnson 1974, Teskey and Hinckley 1977, Hook 1984). Flood tolerance differs among species because of their diverse adaptations for

enduring inundation (Kozlowski 1982a, 1982b, 1984, Keddy and Ellis 1985), such as developing adventitious roots to facilitate oxygen diffusion (Teskey and Hinckley 1977). However, a complete evaluation of flood tolerance for all tree species is not available (Lin et al. 2004). Furthermore, these ratings do not take into consideration other environmental factors that could influence tree species' establishment and survival (Battaglia et al. 2004). Thus, qualitative ratings of flood tolerance have limited practical utility for restoration planning, and there is a need for improved quantitative tools for predicting tree growth and mortality in the context of a particular reforestation site.

Currently, there are very few simple tools to measure hydrologic conditions in restored wetlands. Piezometers and unlined observation wells are used to measure hydrology, but they can be expensive and time-consuming (Faulkner et al. 1989, Thompson et al. 2012). In contrast, soil magnetic susceptibility (MS), a proxy for soil drainage at sites with similar soil parent materials (Grimley and Vepraskas 2000, Grimley et al. 2004), is easy to measure at a reasonable cost, is quantitative, and may be a useful tool for restoration managers when planning tree plantings. Under anaerobic soil conditions concentrations of ferrimagnetic minerals, typically magnetite and maghemite, are dissolved (Grimley and Arruda 2007, Lu et al. 2012). The concentrations of these magnetic minerals control soil MS. Although other soil forming factors, such as parent materials, can affect soil MS (de Jong et al. 2000, Blundell et al. 2009), MS has been shown to reliably reflect drainage conditions at sites where other local factors are relatively similar. Under such conditions, soil MS can be useful as a guide for matching individual tree species' requirements to site-specific soil moisture regimes. Measurements of soil MS have primarily been used in geoarchaeology and soil science, but have not been used extensively in restoration practice. Previous research on the use of soil MS in wetlands suggests that soil MS

could be applicable in ecological restoration (Grimley et al. 2008, Wang et al. 2008, Simms and Lobred 2011). For example, soil MS has been used successfully to delineate wetland boundaries by distinguishing between upland and wetland soils (Grimley et al. 2004, Simms and Lobred 2011). In a study surveying the distribution of naturally occurring trees in forested areas of east-central Illinois, higher soil MS was associated with well drained soils, which are suitable to upland species, whereas lower soil MS was prevalent in poorly drained or waterlogged soils and associated with flood-tolerant species (Grimley et al. 2008, Wang et al. 2008). The present study is the first to compare soil MS to current hydrologic data and planted tree survival and growth in a restoration context as a test of concept for future reforestation efforts.

We addressed two objectives. The first objective was to determine how the growth and survival of planted tree seedlings of four species varied with local hydrologic conditions in a recently reforested floodplain. Although we expected that different tree species would vary in their response to a soil saturation gradient, in general, we expected that planted tree growth would be least and mortality would be greatest in areas with prolonged flooding. The second objective was to determine whether soil MS could be used as a proxy for soil drainage to guide the planting of tree seedlings. Given that soil parent material and surface textures are relatively similar across the study site, we expected that soil MS would be a suitable proxy for current soil drainage and thus, a good predictor of planted tree survival.

2. Materials and Methods

2.1. Study Area

The project site is within the 50.9-ha Sugar Camp Creek wetland mitigation bank in Franklin County, Illinois (Fig. 1). This study was conducted in the southwestern corner of the

mitigation bank, which was restored by the Illinois Department of Transportation (IDOT) beginning in 2013. To re-establish wetland hydrology, IDOT filled all on-site ditches, blocked outlets to Sugar Camp Creek, lowered pre-existing levees along the creek, constructed low berms along the perimeter of the site, removed culverts within the site, excavated portions of the site, and installed four fixed-threshold spillways (IDOT 2009). The soils mapped in the study area are hydric, frequently flooded Bonnie silt loam and non-hydric Belknap silt loam (Preloger 2003, Pociask and Shofner 2007). Prior to modification for agricultural activity, the extent of the hydric soil at the Sugar Camp Creek site indicated that most of the site was a wetland area in the past (Pociask and Shofner 2007). There are also remnant floodplain forest patches to the east of the site.

Fig. 1. LiDAR-based elevation map of the study site showing plot locations along five transects. The map in the lower right shows the study site location within Illinois, USA.

2.2. Tree planting and monitoring

In late May 2014, we planted 400 bareroot tree seedlings (20 to 74 cm in height), which were obtained from the Mason State Tree Nursey in Topeka, IL, along five 100-m transects (Fig. 1). Soil conditions along transects spanned a gradient from very wet to wet-mesic, and soil was undisturbed by IDOT's site preparation activities. Along each transect, we established one 2-m x 2-m plot every 20 m in which we planted 16 individuals total, four of each of the following species: swamp white oak (*Quercus bicolor* Willd.), pin oak (*Quercus palustris* Muenchh.), pecan (*Carya illinoensis* [Wangenh.] K. Koch), and black walnut (*Juglans nigra* L.). These species are among the most commonly planted tree species in floodplain restorations in central U.S. (Kabrick et al. 2012). Each seedling was marked with an aluminum tag with identifying information. Baseline height and stem diameter at 18-cm height were measured for each seedling at the time of planting. In September 2014, height and stem diameter of surviving seedlings were re-measured and any signs of herbivory were noted. Sampling procedures were repeated in May and August of 2015.

Variables that could either limit or enhance the survival and growth of seedlings were measured within the plots. Illinois State Geological Survey (ISGS) has been monitoring ground and surface water elevations at the Sugar Camp Creek site since 2005, which provided a detailed and precise record of hydrologic variation. The ground elevation at each plot was determined by superimposing the point layer for the plots (determined by GPS) on a LiDAR-based digital elevation model (ISGS 2015) and using the 'Extract' tool in ArcGIS v. 10.1. Surface-water data were collected at hourly sampling intervals using a pressure transducer data logger. The elevation for the surface-water data logger was measured using survey grade GPS to assign a reference elevation so that surface water elevation at each plot could be calculated. Total flood duration and consecutive flood duration at each plot were determined by tallying the consecutive

hours that water levels exceeded the ground elevation of each plot based on water levels from the hydrograph. Both the total cumulative duration and the maximum continuous duration of inundation at each plot were tallied from June 2014 until August 2015. Light penetration through the plant canopy was measured using a LI-COR LI-250A Light Meter at 1 m above the ground and at the soil or water surface, depending on whether the plot was inundated at the time of collection. These readings were then relativized to light measurements collected under full sunlight, under no canopy vegetation.

2.3. Soil magnetic susceptibility and grain size determinations

Soil magnetic susceptibility was measured in the field in May 2015 on the soil surface with a portable Bartington MS2 meter and MSD loop (Bartington Instruments, Oxford, UK). MS was measured in the center of each plot on soil surfaces that were gently smoothed with work boots and cleared of any loose plant litter. This methodology, similar to Grimley et al. (2004) and Simms and Lobred (2011), maximizes contact between the MSD loop and the mineral soil and minimizes local variability. Three soil MS readings per plot were measured using SI units ($\times 10^{-5}$) and averaged.

We analyzed soil grain size distributions using a Malvern Mastersizer 3000 laser diffractometer (Malvern Instruments, Malvern, UK) to determine whether some portion of soil MS differences among plots could be caused by differences in parent material grain size fractionation. Ten surface soil samples (0 to 5 cm depth) were collected from random plots along the five transects. We passed each sample through a 2-mm sieve to remove roots and large organic material. A 2-g subsample of soil with 5 ml of Na-hexametaphosphate solution (25.0 g dissolved in 500 ml deionized water) were shaken vigorously in a centrifuge tube to provide

dispersion. The subsample was then transferred to the wet sampler of the diffractometer. Grain size data on the tenth percentile, median, ninetieth percentile, as well as the volume weighted means were compiled using Mastersizer 3000 software version 2.20.

Surface soil MS readings were also collected proximal to mature trees in a remnant floodplain forest patch adjacent to the site (~440 m from tree plantings) to establish means and ranges of soil MS values representative for naturally distributed mature trees of various species. Using methods similar to Wang et al. (2008), we measured the soil MS values for 21 individuals of six species, most with similar flood tolerance ratings to planted tree species in the restored wetland. Soil MS values from the remnant forest were compared with those in the planted tree plots.

2.4. Statistical analysis

Statistical analyses were performed using SAS 9.3 (SAS Institute 2011). Growth and survival of each species were modeled as functions of time and position along the hydrologic gradient. For survival, we calculated both nonparametric and parametric survival functions. Nonparametric functions, such as the product-limit Kaplan-Meier estimator, do not assume the shape of the survival function but are unable to determine the magnitude of the effects of covariates on survival, whereas parametric functions are able to account for explanatory variables in regression analyses (Beckage and Clark 2003). We used the product-limit Kaplan-Meier estimator as the nonparametric survival function to determine the survival probability of the four planted tree species during the study (PROC LIFETEST, SAS Institute 2011). For the parametric function, survival of seedlings was related to predictor variables using generalized linear mixed models along with a binomial distribution and a logit link function in SAS (PROC

GLIMMIX, SAS Institute 2011). Alternative models were constructed, and the Akaike Information Criterion (AIC) was used to determine which model best explained survival (Burnham and Anderson 2002). Each time period was analyzed separately. Alternative models included a null model and models with combinations of the main effects of species identity, soil MS, elevation, cumulative duration of inundation, maximum consecutive duration of inundation, light availability at water level, and light availability at 1 m above ground surface. Plot identity was included in all models as a random effect. Because of multicollinearity among predictor variables, we did not include all possible combinations of variables in the models. In particular, we did not include duration of inundation, elevation, or soil MS in the same models because of expected relationships among those variables. Specified models are listed in Appendix A. Growth was measured as each surviving individual's change in height and diameter from time of planting to time of recording. Each time period was analyzed separately because of the decreasing sample size of surviving individuals. Growth was modeled as a function of predictor variables using general linear mixed models with a normal distribution in SAS (PROC GLIMMIX, SAS Institute 2011). AIC model selection was used to determine which among a set of alternative models best described planted tree growth in each time period. Alternative models included a null model and models with combinations of the main effects of species identity, soil MS, elevation, cumulative duration of inundation, maximum consecutive duration of inundation, herbivory (yes/no), light availability at water level, and light availability at 1 m above ground surface (Appendix A). Plot identity was included as a random effect in the models.

3. Results

3.1. Site inundation

Seedlings were planted in late May 2014, and were censused in September 2014, May 2015, and August 2015. Toward the end of the growing season during the first time period after planting (June to September 2014), a relatively high flood peak in the stream adjacent to the site (Fig. B.1) inundated the plots from 8 to 72 cumulative days. During the second time period, October 2014 to April 2015, Sugar Camp Creek exceeded the bank full elevation several times, resulting in longer periods of inundation, between 41 and 153 cumulative days (Fig. B.1). In 2014, Sugar Camp Creek flooded the study site 9 times. The annual average, based on 11 years of monitoring, is between 4 and 5 floods. Although the period after planting was unusually wet, and plots at lowest elevation were flooded for a considerable duration during the first year (up to 225 days), hydrologic conditions were not atypical for floodplain restorations in the region. For comparison, average annual duration of inundation for 23 other floodplain restorations in Illinois was 50.9 cumulative days (range 0.3 to 285.4 days; G.E. Pociask and J.W. Matthews, unpublished data). During the third time period, from May to August 2015, plots were inundated for 3 to 89 cumulative days, but Sugar Camp Creek had fewer and smaller flood peaks as compared to the first time period (Fig. B.1).

3.2. Planted tree survival

At the end of the growing season in the first year, out of the 400 planted seedlings, 245 (61%) survived. *Quercus bicolor* had the highest survival rate (90%), followed by *Q. palustris* (64%), *C. illinoensis* (48%), and *J. nigra* (43%) (Fig. 2). At the beginning of the second growing, a total of 157 (39%) seedlings survived, and by the end of the second growing season in the second year, only 99 (25%) of seedlings survived (Fig. 2). *Quercus bicolor* had the highest survival rate (52%), followed by *Q. palustris* (30%) and *C. illinoensis* (17%). None of the *J.*

nigra seedlings survived. Survival functions, estimated using the nonparametric product-limit Kaplan-Meier estimator, indicated that seedling survival differed among species, with *Q. bicolor* having the highest annual survival probability, followed by *Q. palustris* and *C. illinoensis* (Fig. C.1).

Fig. 2. Number of surviving planted trees from June 2014 to August 2015. CI = *Carya illinoensis*, JN = *Juglans nigra*, QB = *Quercus bicolor*, QP = *Quercus palustris*.

Seedling survival was also modeled using generalized linear mixed models (Table 1). Models that incorporated species identity and some measure of, or proxy for, hydrology best described the differences in survival for the planted trees. In all three time periods, the best model included the main effect of species identity, and in time periods 2 and 3 the best model also included an effect of flood duration (Tables 1, A.1, A.2, A.3). Predicted survival probabilities of each planted tree species increased as elevation increased (Fig. 3), indicating that planted tree mortality was greatest in low elevation areas that experienced prolonged inundation.

Table 1. AIC model selection for planted tree survival in each time period

Date	Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i
Sept. 2014	Species	1	400	430.77	0	1	0.213
	Consecutive duration 1 (-), Species	2	400	431.33	0.56	0.756	0.161
	Light at 1 m (-), Species	2	400	431.56	0.79	0.674	0.143
	Light at water level (+), Species	2	400	431.77	1.00	0.607	0.129
	Elevation (+), Species	2	400	431.79	1.02	0.600	0.128
	Total duration 1 (-), Species	2	400	431.98	1.21	0.546	0.116
	Soil MS (+), Species	2	400	432.10	1.33	0.514	0.109
May 2015	Consecutive duration 1&2 (-), Species	2	400	353.74	0	1	0.457
	Total duration 1&2 (-), Species	2	400	355.01	1.27	0.530	0.242
	Elevation (+), Species	2	400	355.29	1.55	0.461	0.210
Aug. 2015	Total duration 1,2,3 (-), Species	2	300	324.84	0	1	0.368
	Elevation (+), Species	2	300	325.20	0.36	0.835	0.307
	Consecutive duration 1,2,3 (-), Species	2	300	325.78	0.94	0.625	0.230

Note: Only those models with more support than the null model and AIC $w_i > 0.1$ are shown; all models are shown in Appendix A. Plus or minus in parentheses indicates direction of effect for continuous variables. Total duration 1 and Consecutive duration 1 were from June to September 2014; Total duration 1&2 and Consecutive duration 1&2 were from June 2014 to April 2015; Total duration 1,2,3 and Consecutive duration 1,2,3 were from June 2014 to August 2015. The sample size in August 2015 is smaller than the previous two periods because of the complete mortality of *Juglans nigra*. The models are listed with their number of parameters (*K*), sample size (*N*), AIC, Δ AIC, likelihood, and the AIC weight (w_i).

Fig. 3. Predicted probabilities of survival of planted trees versus elevation for time periods 1 (A), 2 (B), and 3 (C) using PROC LOGISTIC (SAS Institute 2011). Shaded bands represent 95% confidence limits. Species included *Carya illinoensis* (CI), *Quercus bicolor* (QB), *Quercus palustris* (QP), and *Juglans nigra* (JN). *Juglans nigra* was excluded in time period 3 due to lack of survival.

3.3. Planted tree growth

For seedlings that had survived until the end of the growing season in the first year, *Q. bicolor* had significantly increased in height from May to September 2014 (one sample *t*-test: $t = 2.27$; $df = 89$, $p = 0.025$), whereas *C. illinoensis* had significantly decreased in height ($t = -2.47$, $df = 47$, $p = 0.017$) because of herbivory or stem dieback (Fig. 4). At the beginning of the growing season in the second year, *Q. palustris* ($t = -2.81$, $df = 47$, $p = 0.007$) and *C. illinoensis* ($t = -2.09$, $df = 25$, $p = 0.047$) significantly decreased in height from October 2014 to April 2015 (Fig. 4). By the end of the study, only *Q. bicolor* had significantly increased in height from May to August 2015 ($t = 2.99$, $df = 51$, $p = 0.004$; Fig. 4).

Fig. 4. Average change in height of surviving planted trees from September 2014 to August 2015. CI = *Carya illinoensis*, JN = *Juglans nigra*, QB = *Quercus bicolor*, QP = *Quercus palustris*. Error bars represent standard errors.

Models that incorporated species identity and some measure of, or proxy for, hydrology best described the changes in height. In all three time periods, the best model included species identity, but the variable associated with hydrology varied among time periods: elevation in time

period 1, consecutive flood duration in time period 2, and soil MS in time period 3 (Tables 2, A.4, A.5, A.6). As expected, height increases were greater in plots at higher elevation, plots with shorter duration of inundation, and in plots with greater soil MS values. Herbivory as a main effect was only included in candidate models for time period 3 because of missing data in time periods 1 and 2. A model including herbivory and species identity received some support in time period 3; evidence of herbivory was associated with smaller seedling height (Tables 2, A.6).

Table 2. AIC model selection for planted tree height in each time period

Date	Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i	R^2
Sept. 2014	Elevation (+), Species	2	245	1089.74	0	1	0.257	0.098
	Light at 1 m (+), Species	2	245	1090.09	0.35	0.839	0.215	0.100
	Total duration 1 (-), Species	2	245	1090.12	0.38	0.827	0.212	0.095
	Consecutive duration 1 (-), Species	2	245	1090.23	0.49	0.783	0.201	0.095
May 2015	Consecutive duration 1&2 (-), Species	2	157	827.29	0	1	0.379	0.144
	Total duration 1&2 (-), Species	2	157	827.71	0.42	0.811	0.308	0.141
	Elevation (+), Species	2	157	828.05	0.76	0.684	0.259	0.140
Aug. 2015	Soil MS (+), Species	2	99	541.40	0.00	1	0.218	0.160
	Herbivory, Species	2	99	541.84	0.44	0.803	0.175	0.157
	Species	1	99	542.37	0.97	0.616	0.134	0.135
	Light at 1 m (+), Species	2	99	542.45	1.05	0.592	0.129	0.151

Note: Only those models with more support than the null model and AIC $w_i > 0.1$ are shown; all models are shown in Appendix A. Plus or minus in parentheses indicates direction of effect for continuous variables. Total duration 1 and Consecutive duration 1 were from June to September 2014; Total duration 1&2 and Consecutive duration 1&2 were from June 2014 to April 2015. The models are listed with their number of parameters (*K*), sample size (*N*), AIC, Δ AIC, likelihood, AIC weight, (w_i) and R^2 .

Stem diameters increased for all four species at each time period (Fig. 5). The model with the main effect of light availability at the water (or soil) level best explained the change in planted tree diameter for each time period (Tables 3, A.7, A.8), but the relationship between diameter increase and light availability was unexpectedly negative. A model that included soil MS as a predictor of change in planted tree diameter was better than a null (intercept only) model in time periods 1 and 2 (Tables 3, A.7, A.8). None of the other models were able to predict

changes in planted tree diameter better than a null model. Species identity was not included in the selected models for changes in diameter, suggesting that increases in diameter were consistent across species.

Fig. 5. Average change in diameter of surviving planted trees from September 2014 to August 2015. CI = *Carya illinoensis*, JN = *Juglans nigra*, QB = *Quercus bicolor*, QP = *Quercus palustris*. Error bars represent standard errors.

Table 3. AIC model selection for planted tree diameter in each time period

Date	Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i	R^2
Sept. 2014	Light at water level (-)	1	245	-341.14	0	1	0.468	0.117
	Soil MS (-)	1	245	-339.17	1.97	0.373	0.175	0.028
May 2015	Light at water level (-)	1	157	-127.35	0	1	0.353	0.103
	Soil MS (-)	1	157	-126.16	1.19	0.552	0.194	0.025
Aug. 2015	Light at water level (-)	1	99	-49.68	0	1	0.159	0.078

Note: Only those models with more support than the null model and AIC $w_i > 0.1$ are shown; all models are shown in Appendix A. Plus or minus in parentheses indicates direction of effect for continuous variables. The models are listed with their number of parameters (*K*), sample size (*N*), AIC, Δ AIC, likelihood, AIC weight (w_i), and R^2 .

3.4. Soil magnetic susceptibility and grain size

Soil MS values were generally low across the study site, ranging from 6×10^{-5} to 12×10^{-5} SI units (mean = 8×10^{-5} SI units). The soil texture was dominantly fine-grained and silty (Bonnie silt loam series). Volume percentage of sand ranged from 5 to 33% (mean = 13%), volume percentage of clay ranged from 8 to 40% (mean = 14%), and median grain size ranged from 7 to 29 μm (mean = 17 μm). None of the three grain size parameters were significantly correlated with field-measured soil MS (volume percentage of sand, $r^2 = 0.044$; volume percentage of clay, $r^2 = 0.025$; median, $r^2 = 0.002$), suggesting that parent material texture was not a significant factor in soil MS changes across the study site. Soil MS was not significantly correlated to elevation ($r = 0.19$, $p = 0.37$) or cumulative duration of inundation ($r = -0.14$, $p = 0.52$) (Fig. 6). However, these correlations were influenced by an outlier plot with a high soil MS value relative to other plots. When this outlier was removed soil MS was significantly correlated to both elevation ($r = 0.47$, $p = 0.02$) and cumulative duration of inundation ($r = -0.41$, $p = 0.04$).

Our findings from the experimental plantings regarding flood tolerance were consistent with observations in the adjacent remnant floodplain forest. Soils beneath mature *Q. bicolor* trees had the lowest average soil MS value of the six tree species investigated (Table 4). Soils beneath mature individuals of other tree species, such as *Carya ovata* (Mill.) K.Koch, *Carya laciniosa* (Mill.) K.Koch, *Carya tomentosa* Sarg., and *Quercus alba* L., which have similar flood tolerance ratings to *C. illinoensis* and *J. nigra* (Bell and Johnson 1974, Teskey and Hinckley 1977, Hook 1984), had higher average soil MS values (Table 4).

Fig. 6. Relationship between soil MS and total flood duration (A) and elevation (B).

Table 4. Average soil MS values for soil beneath mature individuals of six tree species in adjacent remnant forest

Species	N	Average Soil MS ($\times 10^{-5}$ SI units)
<i>Carya ovata</i>	4	10.8
<i>Carya laciniosa</i>	4	10.3
<i>Carya tomentosa</i>	3	13.3
<i>Quercus alba</i>	2	14.0
<i>Quercus bicolor</i>	4	8.4

4. Discussion

Duration of inundation, along with time and species identity, were the best predictors of the survival of the planted tree species in this recently restored floodplain. Similarly, duration of inundation and species identity best predicted the changes in height for the planted trees. *Quercus bicolor* was the only species with individuals that survived in the lowest elevation areas, indicating that it was the most flood-tolerant of the four species. Soil MS was correlated to both flood duration and elevation, but only after removal of an outlier plot. As elevation increased and flood duration decreased, soil MS increased, suggesting that soil MS is a potential proxy for near-surface hydrology. The landscape relief (< 0.2 m) and the range of soil MS values recorded in this study were small, and we suspect that soil MS may be a more effective proxy for hydrology at sites with a wider range in elevation and soil drainage, based on prior work (Williams and Cooper 1990, Grimley et al. 2004, Wang et al. 2008, Lu et al. 2012). Our findings support the importance of flood duration on survival and growth of planted trees, and suggest that soil MS might be further investigated as a proxy for soil drainage in restored bottomland sites with similar surface soil textures.

4.1. Planted tree survival

A primary objective of this research was to determine how survival of planted tree seedlings varied with local hydrologic conditions in a recently reforested floodplain. We predicted that planted tree mortality would be least in areas with prolonged inundation even though different tree species would vary in their response to a soil saturation gradient. Consistent with these predictions, duration of inundation was the primary predictor of planted tree survival.

The results of this study reinforce those of previous studies that have reported that elevation, which, in part, defines the water regime, is critical in determining tree seedling survival in floodplain restorations (Battaglia et al. 2000, McLeod et al. 2000, Middleton 2002, Battaglia et al. 2004). Despite the small change in elevation across our study site, hydrologic conditions varied considerably among plots, with seedlings at lower elevations experiencing a more than 4-fold increase in the duration of inundation during the first year relative to seedlings at higher elevation. Even small shifts in elevation (< 30 cm) in wetlands have strong effects on plant performance and overall community composition (Vivian-Smith 1997, Peach and Zedler 2006), as well as planted tree survival (Pennington and Walters 2006).

Species identity was also an important determinant of tree survival, with *Q. bicolor* having the greatest probability of survival across all three time periods. Previous research on flood tolerance ratings, as summarized by Middleton (2002), has suggested that *Q. bicolor* and *Q. palustris* are moderately tolerant of flooding, whereas *C. illinoensis* and *J. nigra* are less tolerant. Results of this study suggest that *Q. bicolor* may be more tolerant than *Q. palustris*, at least at the seedling stage. Restoration strategies in restored forested wetlands incorporate planting heavy-seeded species such as oaks (*Quercus* spp.) and pecans (*Carya* spp.). Planting *Q. bicolor* in these areas can ensure greater survival of planted trees. On the other hand, planting only *Q. bicolor* could lead to low diversity restorations (Allen 1997, Middleton 2002). Instead, a strategy of planting flood tolerant tree species specifically in low elevation areas, while leaving gaps in between planted individuals to promote the passive establishment of other species (Middleton 2002), could lead to more successful and diverse floodplain restorations.

Overall tree survival in this study was relatively low compared to other studies that have reported two-year survival of trees planted in floodplains (45-99%, depending on the species;

Pennington and Walters 2006, Matthews and Pociask 2015), but was similar to median five-year survival rates for planted trees in Illinois floodplain forest restorations (31%, Matthews and Endress 2008). Poor survival was likely caused by the small size of the seedlings and the atypically wet conditions during the first year of the study.

4.2. *Planted tree growth*

Another goal of this study was to understand how hydrologic conditions affected the growth of planted tree seedlings. As with survival, we predicted that tree growth would be least in areas with prolonged flooding. Our results differed between height and diameter of the planted trees. Inundation was an important predictor of changes in planted tree height, likely because of reduced growth and stem dieback in areas with prolonged inundation. Our findings for growth, however, should be taken with reservations because of the small changes in the diameter growth of seedlings over the course of the study. Mitsch and Rust (1984) found that there was a general absence of a correlation between growth of moderately flood-tolerant trees and measures of flooding duration, whether the flooding occurred during the growing season or the entire year, because of a potentially non-linear relationship between the two. The impacts of flooding on tree growth are difficult to establish because flooding can be both positive, by supplying nutrients and water, and negative, by creating an anaerobic root zone (Mitsch and Rust 1984).

Of the four planted tree species, only *Q. bicolor* had a positive average change in height across all three time periods, whereas *Q. palustris*, *C. illinoensis* and *J. nigra* mostly decreased in height because of herbivory or dieback and resprouting. This observation is consistent with other studies that have found that flood intolerant tree species experience stem dieback or a decrease in shoot growth because of flooding (Dickson et al. 1965, Kozlowski 1984, Frye and Grosse 1992,

Blom et al. 1994, Ewing 1996, Kabrick et al. 2012). In contrast, diameter growth was positive for all four species. Increased diameter may have been a compensatory response to decreased stem height and the loss of the terminal bud. Furthermore, increases in diameter can occur in flood-tolerant species as they generate more intercellular gaps and less dense cells, which allow for oxygen transport (Frye and Grosse 1992). Yet the diameter growth in all four species was more strongly associated with light availability than flooding. However, the observed inverse relationship between diameter increase and light availability and the poor fit of the other models indicate that planted tree diameter was not well explained by the measured predictor variables.

In time period 3, there was an effect of herbivory, along with species identity, on changes in planted tree seedling height. Only *C. illinoensis* suffered obvious damage by vertebrate browsers (deer and possibly rabbits) during this time period. Although not included in our alternative models, there is potential for flooding to interact with herbivory, for example by making seedlings more or less apparent or by changing the movement or behavior of herbivores. Previous studies have shown that herbivores, such as beavers, other rodents, deer, and rabbits, can inflict considerable browsing damage upon woody plants (De Steven 1991, Conner et al. 2000, McLeod et al. 2000). Herbivore damage to planted trees could be prevented through the use of tree shelters (Conner et al. 2000, McLeod 2000, McLeod et al. 2000), but the shelters themselves can damage planted trees in floodplain settings where the shelters often get mangled by flood debris (personal observation).

4.3. Soil magnetic susceptibility

A second objective of this study was to determine whether soil MS could be used as a proxy for soil drainage to help guide planting of tree seedlings for restoration. Previous research,

regionally and globally, has consistently found that soil MS correlates well with hydric soil field indicators, landscape position, or mapped soil drainage class (de Jong et al. 2000, Grimley et al. 2004, Wang et al. 2008, Blundell et al. 2009, Simms and Lobred 2011, Lu et al. 2012). The Sugar Camp Creek site consists entirely of poorly drained alluvial soils (Bonnie silt loam) with low MS values ($< 12 \times 10^{-5}$ SI). Regional studies indicate that surface soil MS typically varies between about 7 and 85×10^{-5} SI across landscapes in glaciated Central Lowlands (Grimley et al. 2004, 2008). Particularly low MS values in the Bonnie soil series likely result from enhanced magnetite dissolution caused by a combination of relatively low soil pH and anoxic (reducing) soil conditions (Grimley and Arruda 2007). Variables that can potentially complicate the use of soil MS as a proxy for soil drainage include surface soil texture variation (Williams and Cooper 1990, de Jong et al. 2000), and changes in parent material composition (Grimley et al. 2004, Blundell et al. 2009). Surface soil texture was relatively uniform at the Sugar Camp Creek site and consisted of fine-grained alluvium (mainly resedimented loess).

Based on prior comparisons between tree species distributions and soil MS in Illinois (Grimley et al. 2008, Wang et al. 2008), we anticipated that soil MS would be a suitable proxy for soil moisture conditions and thus a good predictor of planted tree survival. Inconsistent with our predictions, however, soil MS did not significantly increase with increasing elevation or decreasing flood duration. Soil MS at the study site was also a relatively poor predictor of planted tree growth and survival. Soil MS did provide the best correlation with planted tree height ($r^2 = 0.16$) after two growing seasons (Aug. 2015), but was a poor predictor in earlier time periods. The use of soil MS as a predictor of tree growth and survival at the study site is likely limited by (1) the minimal areal scope and topographic relief, and (2) possible anthropogenic effects of soil disturbance (i.e., grading and excavation), additions, or compaction. We suspect

that the first rationale is the most significant because the study site has < 0.2 m of relief within a poorly drained alluvial floodplain. At the study site, soil MS from one plot was an outlier with respect to plot elevation (Fig. 6) and could reflect some anthropogenic disturbance; correlations with elevation and flood duration improved when this outlier was removed from the dataset. In addition, field MS measurements have inherent imprecisions ($\sim \pm 1 \times 10^{-5}$ SI) because of irregularities in the soil surface that limit smooth contact with the MSD loop, as well as background uncertainty in the MS2B instrumentation. This imprecision becomes statistically significant where the range in MS is small—here, varying only between 6×10^{-5} SI and 12×10^{-5} SI across the study site. Better predictability may be found in future experimental tree growth studies that encompass a more complete hydrosequence from poorly drained hydric soils (such as Bonnie silt loam) to moderately well or well drained soils. In the nearby remnant woodland (~ 1 m relief), mean soil MS, measured adjacent to individual trees (Table 4), corresponded well with known flood tolerances of these species. Thus, there is certainly potential for using soil MS as a predictor of the survival of planted trees; yet, additional studies are clearly needed.

5. Conclusions

Results from this study show the importance of fine-scale hydrology for the survival and growth of planted trees. Additional long-term studies on planted tree growth and survival in restored bottomland forests in response to hydrology and other abiotic factors are necessary to establish how local environmental features, individually and in conjunction with one another, may be influencing planted tree survival and growth. Further studies on soil MS are also needed to determine how consistent the results are for other tree species, soil types, geological terrains, and geographic zones. At sites such as Sugar Camp Creek which have fairly uniform surface soil

textures, soil MS might be further developed as a rapid tool that can lead to greater precision in tree planting according to individual species' natural distribution across hydrologic gradients. This has implications for wetland reforestation projects, where planted tree survival is one of many performance standards used to evaluate a project's success (Pennington and Walters 2006, Matthews and Endress 2008). Understanding how flooding affects planted tree survival and growth, along with developing tools such as soil MS that can be used to easily and quickly distinguish areas suitable for tree planting, is essential for success in restored floodplain forests.

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Appendix A. Full AIC model selection tables for planted tree survival and growth

Appendix B. Figure of stream gauge data for the study site

Appendix C. Nonparametric product-limit Kaplan-Meier estimates for planted trees

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650 **FIG. 1. Alternate grayscale version for print edition only**

651
652 Fig. 1. LiDAR-based elevation map of the study site showing plot locations along five transects.
653 The map in the lower right shows the study site location within Illinois, USA.

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APPENDIX A. Full AIC model selection tables for planted tree survival and growth

Table A.1. AIC model selection for planted tree survival in Time 1

Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i
Species	1	400	430.77	0	1	0.213
Consecutive duration 1 (-), Species	2	400	431.33	0.56	0.756	0.161
Light at 1 m (-), Species	2	400	431.56	0.79	0.674	0.143
Light at water level (+), Species	2	400	431.77	1.00	0.607	0.129
Elevation (+), Species	2	400	431.79	1.02	0.600	0.128
Total duration 1 (-), Species	2	400	431.98	1.21	0.546	0.116
Soil MS (+), Species	2	400	432.10	1.33	0.514	0.109
Null	1	400	502.45	71.68	0.000	0.000
Consecutive duration 1 (-)	1	400	502.99	72.22	0.000	0.000
Light at 1 m (-)	1	400	503.26	72.49	0.000	0.000
Elevation (+)	1	400	503.44	72.67	0.000	0.000
Light at water level (+)	1	400	503.45	72.68	0.000	0.000
Total duration 1 (-)	1	400	503.63	72.86	0.000	0.000
Soil MS (+)	1	400	503.88	73.11	0.000	0.000

Note: Total duration 1 and Consecutive duration 1 were from June to September 2014. Plus or minus in parentheses indicates direction of effect for continuous variables.

663 Table A.2. AIC model selection for planted tree survival in Time 2

Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i
Consecutive duration 1&2 (-), Species	2	400	353.74	0	1	0.457
Total duration 1&2 (-), Species	2	400	355.01	1.27	0.530	0.242
Elevation (+), Species	2	400	355.29	1.55	0.461	0.210
Species	1	400	358.93	5.19	0.075	0.034
Soil MS (+), Species	2	400	359.08	5.34	0.069	0.032
Light at water level (+), Species	2	400	360.91	7.17	0.028	0.013
Light at 1 m (-), Species	2	400	360.93	7.19	0.027	0.013
Consecutive duration 1&2 (-)	1	400	514.48	160.74	0.000	0.000
Total duration 1&2 (-)	1	400	515.77	162.03	0.000	0.000
Elevation (+)	1	400	515.79	162.05	0.000	0.000
Null	1	400	519.71	165.97	0.000	0.000
Soil MS (+)	1	400	519.95	166.21	0.000	0.000
Light at water level (+)	1	400	521.68	167.94	0.000	0.000
Light at 1 m (-)	1	400	521.70	167.96	0.000	0.000

664 Note: Total duration 1&2 and Consecutive duration 1&2 were from June 2014 to April 2015.

665 Plus or minus in parentheses indicates direction of effect for continuous variables.

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671 Table A.3. AIC model selection for planted tree survival in Time 3

Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i
Total duration 1,2,3 (-), Species	2	300	324.84	0	1	0.368
Elevation (+), Species	2	300	325.20	0.36	0.835	0.307
Consecutive duration 1,2,3 (-), Species	2	300	325.78	0.94	0.625	0.230
Species	1	300	329.33	4.49	0.106	0.039
Soil MS (+), Species	2	300	330.39	5.55	0.062	0.023
Light at 1 m (+), Species	2	300	330.78	5.94	0.051	0.019
Light at water level (+), Species	2	300	331.31	6.47	0.039	0.014
Total duration 1,2,3 (-)	1	300	356.44	31.60	0.000	0.000
Elevation (+)	1	300	356.70	31.86	0.000	0.000
Consecutive duration 1,2,3 (-)	1	300	357.38	32.54	0.000	0.000
Null	1	300	360.87	36.03	0.000	0.000
Soil MS (+)	1	300	361.93	37.09	0.000	0.000
Light at 1 m (+)	1	300	362.33	37.49	0.000	0.000
Light at water level (+)	1	300	362.86	38.02	0.000	0.000

672 Note: Total duration 1,2,3 and Consecutive duration 1,2,3 were from June 2014 to August 2015.

673 Plus or minus in parentheses indicates direction of effect for continuous variables.

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679 Table A.4. AIC model selection for planted tree height in Time 1

Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i
Elevation (+), Species	2	245	1089.74	0	1	0.257
Light at 1 m (+), Species	2	245	1090.09	0.35	0.839	0.215
Total duration 1 (-), Species	2	245	1090.12	0.38	0.827	0.212
Consecutive duration 1 (-), Species	2	245	1090.23	0.49	0.783	0.201
Species	1	245	1092.78	3.04	0.219	0.056
Light at water level (-), Species	2	245	1093.84	4.10	0.129	0.033
Soil MS (+), Species	2	245	1094.42	4.68	0.096	0.025
Light at 1 m (+)	1	245	1102.39	12.65	0.002	0.000
Elevation (+)	1	245	1104.61	14.87	0.001	0.000
Total duration 1 (-)	1	245	1104.95	15.21	0.000	0.000
Consecutive duration 1 (-)	1	245	1105.18	15.44	0.000	0.000
Null	1	245	1106.66	16.92	0.000	5.435E-05
Light at water level (-)	1	245	1107.98	18.24	0.000	2.809E-05
Soil MS (-)	1	245	1108.65	18.91	0.000	2.009E-05

680 Note: Total duration 1 and Consecutive duration 1 were from June to September 2014. Plus or
681 minus in parentheses indicates direction of effect for continuous variables.

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684 Table A.5. AIC model selection for planted tree height in Time 2

Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i
Consecutive duration 1&2 (-), Species	2	157	827.29	0	1	0.379
Total duration 1&2 (-), Species	2	157	827.71	0.42	0.811	0.308
Elevation (+), Species	2	157	828.05	0.76	0.684	0.259
Light at 1 m (+), Species	2	157	832.83	5.54	0.063	0.024
Soil MS (+), Species	2	157	834.01	6.72	0.035	0.013
Species	1	157	834.95	7.66	0.022	0.008
Light at water level (-), Species	2	157	835.39	8.10	0.017	0.007
Total duration 1&2 (-)	1	157	840.18	12.89	0.002	0.001
Consecutive duration 1&2 (-)	1	157	840.53	13.24	0.001	0.001
Elevation (+)	1	157	840.71	13.42	0.001	0.000
Light at 1 m (+)	1	157	843.03	15.74	0.000	0.000
Soil MS (+)	1	157	845.44	18.15	0.000	4.344E-05
Null	1	157	845.64	18.35	0.000	3.930E-05
Light at water level (-)	1	157	845.65	18.36	0.000	3.911E-05

685 Note: Total duration 1&2 and Consecutive duration 1&2 were from June 2014 to April 2015.

686 Plus or minus in parentheses indicates direction of effect for continuous variables.

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692 Table A.6. AIC model selection for planted tree height in Time 3

Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i
Soil MS (+), Species	2	99	541.40	0.00	1	0.218
Herbivory, Species	2	99	541.84	0.44	0.803	0.175
Species	1	99	542.37	0.97	0.616	0.134
Light at 1 m (+), Species	2	99	542.45	1.05	0.592	0.129
Elevation (+), Species	2	99	542.95	1.55	0.461	0.100
Total duration 1,2,3 (-), Species	2	99	543.01	1.61	0.447	0.098
Consecutive duration 1,2,3 (-), Species	2	99	543.30	1.90	0.387	0.084
Light at water level (+), Species	2	99	544.36	2.96	0.228	0.050
Herbivory	1	99	548.00	6.60	0.037	0.008
Light at 1 m (+)	1	99	552.39	10.99	0.004	0.001
Null	1	99	552.69	11.29	0.004	0.001
Total duration 1,2,3 (-)	1	99	553.92	12.52	0.002	0.000
Elevation (+)	1	99	553.94	12.54	0.002	0.000
Soil MS (+)	1	99	554.01	12.61	0.002	0.000
Consecutive duration 1,2,3 (-)	1	99	554.04	12.64	0.002	0.000
Light at water level (+)	1	99	554.66	13.26	0.001	0.000

693 Note: Total duration 1,2,3 and Consecutive duration 1,2,3 were from June 2014 to August 2015.

694 Plus or minus in parentheses indicates direction of effect for continuous variables.

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696 Table A.7. AIC model selection for planted tree diameter in Time 1

Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i
Light at water level (-)	1	245	-341.14	0	1	0.468
Soil MS (-)	1	245	-339.17	1.97	0.373	0.175
Null	1	245	-337.99	3.15	0.207	0.097
Light at water level (-),Species	2	245	-336.88	4.26	0.119	0.056
Total duration 1 (-)	1	245	-336.32	4.82	0.090	0.042
Light at 1 m (+)	1	245	-336.17	4.97	0.083	0.039
Elevation (+)	1	245	-336.08	5.06	0.080	0.037
Consecutive duration 1 (-)	1	245	-336.04	5.10	0.078	0.037
Soil MS (-), Species	2	245	-334.80	6.34	0.042	0.020
Species	1	245	-333.69	7.45	0.024	0.011
Total duration 1 (-), Species	2	245	-332.03	9.11	0.011	0.005
Light at 1 m (+), Species	2	245	-331.83	9.31	0.010	0.004
Elevation (+), Species	2	245	-331.79	9.35	0.009	0.004
Consecutive duration 1 (-), Species	2	245	-331.74	9.40	0.009	0.004

697 Note: Total duration 1 and Consecutive duration 1 were from June to September 2014. Plus or
698 minus in parentheses indicates direction of effect for continuous variables.

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704 Table A.8. AIC model selection for planted tree diameter in Time 2

Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i
Light at water level (-)	1	157	-127.35	0	1	0.353
Soil MS (-)	1	157	-126.16	1.19	0.552	0.194
Null	1	157	-125.05	2.30	0.317	0.112
Light at water level (-), Species	2	157	-124.15	3.20	0.202	0.071
Light at 1 m (+)	1	157	-123.36	3.99	0.136	0.048
Total duration 1&2 (-)	1	157	-123.20	4.15	0.126	0.044
Consecutive duration 1&2 (-)	1	157	-123.16	4.19	0.123	0.043
Elevation (+)	1	157	-123.02	4.33	0.115	0.040
Soil MS (-), Species	2	157	-122.83	4.52	0.104	0.037
Species	1	157	-121.81	5.54	0.063	0.022
Light at 1 m (+), Species	2	157	-120.07	7.28	0.026	0.009
Total duration 1&2 (-), Species	2	157	-120.01	7.34	0.025	0.009
Consecutive duration 1&2 (-), Species	2	157	-119.96	7.39	0.025	0.009
Elevation (+), Species	2	157	-119.81	7.54	0.023	0.008

705 Note: Total duration 1&2 and Consecutive duration 1&2 were from June 2014 to April 2015.

706 Plus or minus in parentheses indicates direction of effect for continuous variables.

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708 Table A.9. AIC model selection for planted tree diameter in Time 3

Predictor variables in model	<i>K</i>	<i>N</i>	AIC	Δ AIC	Likelihood	AIC w_i
Light at water level (-)	1	99	-49.68	0	1	0.159
Null	1	99	-49.43	0.25	0.882	0.140
Soil MS (-)	1	99	-49.37	0.31	0.856	0.136
Herbivory	1	99	-48.87	0.81	0.667	0.106
Consecutive duration 1,2,3 (-)	1	99	-47.90	1.78	0.411	0.065
Light at 1 m (+)	1	99	-47.83	1.85	0.397	0.063
Total duration 1,2,3 (-)	1	99	-47.73	1.95	0.377	0.060
Elevation (+)	1	99	-47.67	2.01	0.366	0.058
Light at water level (-), Species	2	99	-47.23	2.45	0.294	0.047
Species	1	99	-46.93	2.75	0.253	0.040
Soil MS (-), Species	2	99	-46.48	3.20	0.202	0.032
Herbivory, Species	2	99	-45.83	3.85	0.146	0.023
Consecutive duration 1,2,3 (-), Species	2	99	-45.39	4.29	0.117	0.019
Light at 1 m (+), Species	2	99	-45.32	4.36	0.113	0.018
Total duration 1,2,3 (-), Species	2	99	-45.23	4.45	0.108	0.017
Elevation (+), Species	2	99	-45.19	4.49	0.106	0.017

709 Note: Total duration 1,2,3 and Consecutive duration 1,2,3 were from June 2014 to August 2015.

710 Plus or minus in parentheses indicates direction of effect for continuous variables.

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APPENDIX B. Figure of stream gauge data for the study site

Fig. B.1. Stream gauge data for the study area at the Sugar Camp Creek site from January 2013 to September 2015. The red box indicates time period 1 (June-Sept 2014), the blue box indicates time period 2 (Oct 2014-April 2015), and the green box indicates time period 3 (May-Aug 2015). The orange line represents the average bank full elevation along Sugar Camp Creek in the vicinity of the study site.

720 **APPENDIX C.** Nonparametric product-limit Kaplan-Meier estimates for planted trees

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 722 Fig. C.1. Product-limit survival estimates for all four planted tree species from June 2014 to
 723 August 2015. *Quercus bicolor* (QB) had the highest survival probability, followed by *Quercus*
 724 *palustris* (QP), *Carya illinoensis* (CI), and *Juglans nigra* (JN).