

Long-Term Response of Stream and Riparian Restoration at Wilson Creek, Kentucky USA

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ABSTRACT

Degradation, impoundment, and channelization of streams is a global problem. Although stream restoration projects have increased in recent years, post-restoration, long-term monitoring is rare. In 2003, a channelized section of Wilson Creek (Nelson Co., Kentucky) was restored by creating a meandering channel, re-connecting the channel to its floodplain, and planting native riparian species: giant cane and bottomland forest species. Our main objective was to conduct a ten-year post-restoration assessment to determine long-term restoration outcomes of channel water quality, growth of trees planted in the riparian area, and soil development. Water quality, soil, and tree data collected in 2013–2015 was compared to 2004–2006 data. Quality of water parameters changed over time: sulfate, magnesium, calcium, potassium, alkalinity, pH, iron, and temperature decreased, whereas dissolved oxygen increased. Overall, soil pH, extractable ammonium, extractable nitrate, total carbon (TC), and total nitrogen (TN) increased over time. Effects were observed in restored riparian areas for pH, extractable ammonium, and TC; while TC and TN exhibited depth-dependent interactions. The carbon-nitrogen ratio in these soils significantly decreased over time for the reference sites, and the treatments recovered to near reference level. *Platanus occidentalis* (American sycamore) and *Fraxinus pennsylvanica* (green ash) individuals had higher survival (80% and 79%, respectively) than individuals of *Quercus palustris* (pin oak; 22%). Shelter and herbicide treatments had no effect on tree survival or height growth; however, height growth varied by species. *Platanus occidentalis* exhibited a greater than five-fold increase, *F. pennsylvanica* slightly increased, and *Q. palustris* decreased in height growth. Overall, water and soil quality improved over time at the restoration site, while tree survival and height growth exhibited species-specific outcomes.

Keywords: herbicide, riparian planting, soil quality, tree shelters, water quality

Restoration Recap

- We revisited a section of Wilson Creek, Kentucky, which was restored in 2003 with goals of reestablishing native riparian vegetation, enhancing floodplain connectivity, and improving water quality.
- We documented improved water quality and soil conditions at the restoration site. Floodplain design features that increase floodplain and channel interactions, including lower banks, adjacent pools, and channel meanders, are a possible mechanism for the improved conditions.
- Our research suggests soil pH and effects of co-planting species, specifically, fast growing pioneer species, should be considered before allocating funds for planting *Quercus palustris* at restoration sites.
- Tree shelters (protective plastic or mesh tubing) and herbicides delivered some short-term benefits but did not convey long term benefits at our restoration site. Therefore, long-term restoration outcomes should be considered when investing in what may be only short-term successes.
- Our restoration outcomes support the argument for longer term monitoring efforts, particularly with respect to water and soil properties, tree species selection and shelter/herbicide use. Subsequent adaptive management (e.g., replanting and thinning) may be necessary if restoration goals are not achieved.

al. 2007, Palmer et al. 2007) and worldwide (Nakamura et al. 2006). In the U.S., more than one billion dollars per year is allocated to stream restoration efforts (Bernhardt et al. 2005). Although the number of stream restoration projects has increased, monitoring efforts of these restored streams are lacking (Bash and Ryan 2002, Bernhardt et al. 2005). As of 2003, only 10% of stream restoration projects nationwide were monitored post-restoration (Bernhardt et al. 2005), though this estimate varies by region and may be low due to lack of written project monitoring records (Palmer et al. 2007). The percentage of projects monitored in the southeast region, where our study is based, is higher than the national average with a range from 11.2% in Kentucky to 47.5% in South Carolina (Bernhardt et al. 2005).

Professionals in the field of stream restoration have been encouraging monitoring, specifically long-term monitoring, since the beginning of the discipline (Kondolf and Micheli 1995, Bernhardt et al. 2005, Palmer et al. 2007). Kondolf and Micheli (1995) suggest a monitoring window of at least ten years post-restoration. A meta-analysis of restored wetland ecosystems by Moreno-Mateos et al. (2012) revealed a quick recovery for hydrologic function; however, in contrast, biological structure and biogeochemical functioning recovered to only 77% and 74%, respectively, after 50–100 years when compared with reference sites. Therefore, longer monitoring timescales (> 10 years) may be necessary to assess full ecosystem recovery at a site.

Riparian restoration commonly focuses on restoring both hydrologic or floodplain connectivity (Ward et al. 1999, Tockner and Stanford 2002) and a vegetated riparian area (Bernhardt et al. 2005) in an effort to improve water quality. Riparian flooding has positive effects on diversity of fish assemblages (Galat et al. 1998, Sommer et al. 2001, Sullivan and Watzin 2009) and has also been linked to positive responses in herpetofauna, zooplankton, plants, and birds (Galat et al. 1998). Healthy vegetated riparian zones provide flood control, help to maintain water quality, and create wildlife habitat (Zedler 2003, Richardson et al. 2011). In addition, riparian vegetation interacts with streams through chemical (e.g., nitrogen removal) and biogeochemical processes (e.g., litter decomposition), and physical processes (e.g., water flow patterns and erosion control; Dosskey et al. 2010). This vegetation also contributes essential nutrients to soils (Hafner and Groffman 2005) and to the stream channel through leaf litter and plant debris (Dosskey and Bertsch 1994). In-stream debris from riparian zones can decrease stream velocity, allowing for deposition of organic matter and sediments and enhancing key chemical and biological processes (Vannote et al. 1980, Harmon et al. 1986, Zhang and Mitsch 2007). Moreover, addition of in-stream debris at restoration sites has been shown to increase the retention of coarse particulate organic matter (CPOM) which enhances instream habitat and improves biotic breakdown of CPOM (Lepori et al. 2005).

Because riparian vegetation has an overwhelming effect on restoration outcomes, recent research has focused on the effectiveness of tree shelters (protective plastic or mesh tubing) and herbicides to increase riparian tree seedling survival and growth (West et al. 1999, Conner et al. 2000, Dubois et al. 2000, Sweeney et al. 2002, Sweeney and Czapka 2004, Andrews et al. 2010). This new body of research, except Andrews et al. (2010—observed height growth increase, not overall survival increase), documents an overall increase in tree survival and height growth with the use of shelters and herbicides. Tree shelters function to reduce herbivory on vulnerable seedlings and may improve microclimate favorability for survival and height growth (Tuley 1983, 1985), while herbicides are applied around the planted seedlings to reduce competition from weeds and improve seedling establishment (Dubois et al. 2000, Sweeney et al. 2002, Sweeney and Czapka 2004).

A channelized section of Wilson Creek, Kentucky was restored in 2003 with goals of reestablishing native riparian vegetation, enhancing floodplain connectivity, and improving water quality (Andrews et al. 2011). Previous studies documenting the Wilson Creek response to restoration focused on riparian re-vegetation planting techniques, water quality improvement, and soil properties. In early assessments, stream velocity at Wilson Creek decreased contributing to higher nutrient uptake rates which combined to reduce downstream nutrient flushing (Bukaveckas 2007). Andrews et al. (2010) found a high overall survivorship of planted seedlings (83–93%, $n = 648$) after two years and concluded that techniques for improving seedling survival, such as the use of tree shelters or herbicide, were not needed. When survivorship was split out by species, however, Andrews et al. (2010) observed that *Quercus palustris* survival (pin oak; 74%, $n = 216$) was lower than *Fraxinus pennsylvanica* (green ash; 93%, $n = 216$) and *Platanus occidentalis* (American sycamore; 97%, $n = 216$). While survival was not influenced by sheltering or herbicide use, tree height growth was influenced by several factors including plot row position, use of tree shelters, herbicide application, and species (Andrews et al. 2010). Further, Andrews et al. (2011) reported 61% survival of transplanted native *Arundinaria gigantea* (giant river cane grass) and documented changes in water quality, specifically increases in temperature and alkalinity and decreases in nitrate, chloride, sodium, and potassium in the restored reach when compared with the reference reach. Early soil data from the restored forest and cane plots indicated an increase in total carbon and total nitrogen over time, suggesting rapid recovery of the disturbed riparian area that was attributable to vegetation development and floodplain connectivity (Andrews et al. 2011). In addition, two years post-restoration, extractable nitrate and ammonium levels in the restored soils were not significantly different from those in the soils in the reference plots (Andrews et al. 2011).

In this current study, our main objective was to resample the Wilson Creek restoration site ten years after stream restoration, with a focus on channel water quality, the status of riparian tree plantings, and soil development. We hypothesized that soil properties at our site would continue to improve over time with increased floodplain connectivity and organic material deposition (Kaushal et al. 2008, Roley et al. 2012). In all, many of the parameters examined seemed to be responding favorably to the restoration activities two years post-restoration; we hypothesized that this positive trend would continue. Wilson Creek provides an excellent opportunity for long-term post-restoration monitoring.

Methods

Site Description

Wilson Creek is a third order tributary of the Rolling Fork River in the greater Salt River watershed, which borders Bernheim Arboretum and Research Forest, a privately-owned forest in Bullitt County, KY. It is located in Kentucky's Knobs-Norman Upland, Level IV ecoregion, within the larger interior plateau, Level III ecoregion (Woods et al. 2002). Steep, dissected knobs and narrow valleys characterize the region. Valley bottoms are generally composed of lacustrine sediments, Ordovician limestone, shale, and dolomite (Woods et al. 2002). Floodplain soils are comprised of Inceptisols (Endoaquepts, Eutrudepts), Entisols (Fluvaquents), and Mollisols (Hapludolls; Woods et al. 2002). Temperature and moisture regimes for the region are described as Mesic/Udic with floodplains considered Aquic/Udic (Woods et al. 2002). The Wilson Creek watershed, draining 1200 ha, consists of a mixture of forest, agriculture, pasture, and residential areas.

Restoration and Experimental Design

Completed in December 2003, a straightened, forested, 1-km portion (6.5 ha floodplain area) of Wilson Creek (37.873758° N, -85.596835° W) was reworked into a meandering channel with a gravel bottom using Rosgen (1996) techniques. Restoration adjacent to the stream channel included floodplain terracing, pool creation, and erosion control measures (Anti-Wash Geojute™, Belton Industries, Belton, SC, and burlap fabric). In March 2004, the 6.4-ha riparian and floodplain area was planted with bare root seedlings of bottomland forest species (> 13,000), wetland herbaceous plugs (2,000) and seeded with native forbs (Andrews 2006). Two separate sets of experimental plots were established alongside these plantings. Nine 225-m² streamside experimental plots (A–I, with A being most upstream) were established with three plots per treatment: undisturbed forested reference, native *A. gigantea* and bottomland hardwood forest (*P. occidentalis*, n = 36; *Q. palustris*, n = 36; and *F. pennsylvanica*, n = 36). The plots most upstream (A–C) were selected as undisturbed reference

plots for comparison with two potential types of riparian vegetation: cane and forest. These plots (A–I) were used exclusively for soil and water analyses, and reference plots were selected to allow comparison with original vegetation for the site. Native *A. gigantea* plots were added to this experiment as a prospective riparian restoration species. Canebreaks, thick stands of giant river cane, were historically an important part of riparian ecosystems of Kentucky, where they functioned to stabilize stream banks (Campbell 1985). In addition, eighteen separate plots (10 × 10 m) were established on both sides of the stream channel with three replicates of six treatments: control (no treatment), herbicide (Rodeo®, Dow Agrosiences, Indianapolis, IN), Tubex® shelters (Treessentials, Duluth, MN), continental mesh shelters (Farm Forestry Co., Shropshire, England), Tubex® + herbicide, and continental mesh + herbicide. Two types of tree shelters were placed over the seedlings in this study: Tubex®, a hard, semitransparent plastic tube and continental mesh, a flexible plastic mesh tube. These shelters degrade overtime, however how quickly they degrade is determined by environmental conditions. These plots were planted parallel to the stream channel in rows A–F (A closest to the stream bank) with a random mixture of *P. occidentalis*, n = 216, *F. pennsylvanica*, n = 216, and *Q. palustris*, n = 216. A full project site description is reported in Andrews et al. (2010, 2011).

Site Measurements

Water samples were collected from the center of the stream channel adjacent to each plot (A–I) on a monthly basis between March 2004 and May 2006 and between February 2013 and January 2014 (May 2013 was not sampled). Sampling and analytic protocols followed Standard Methods for the Examination of Water and Wastewater (Greenberg et al. 1992, Andrews et al. 2011). Calibration standards, blanks (de-ionized water), replicates, and spikes were used to confirm data quality. Overall, seventeen water quality variables were analyzed in-house, including specific conductivity, chloride (Cl), sodium (Na), nitrate (NO₃N), ammonium (NH₄N), total organic carbon (TOC), phosphate (PO₄), iron (Fe), sulfate (SO₄), magnesium (Mg), calcium (Ca), potassium (K), alkalinity, manganese (Mn), pH, temperature, and dissolved oxygen. Temperature and dissolved oxygen measures were obtained during each site visit using an YSI 556 model (Fondriest Environmental, Inc., Fairborn, OH).

Duplicate soil cores from the upper 0–10 cm were collected once from each plot (A–C, F–I) in November 2004 and plots (A–I) in October 2015. The cores were separated at 0–5 and 5–10 cm and analyzed for the independent depths and as a composite of the entire 10 cm profile. On each sampling date, four duplicate sample sets were collected at random from each plot—at the stream bank, and 5, 10, and 15 m from the stream bank. One soil sample set was air dried and sieved through a 2-mm screen for

Table 1. Mean (\pm standard error) stream water quality attributes at Wilson Creek, Kentucky. For the 2004–2006 data means followed by different lowercase letter indicate significant differences among treatments in the 2004–2006 sampling period ($\alpha = 0.05$); $n = 20$ sample dates, monthly between March 2004 and May 2006. For the 2013–2014 data there were no observed differences among within the 2013–2014 sampling period ($\alpha = 0.05$); $n = 11$ sample dates, monthly between February 2013 and January 2014 (exception May 2013). Significant differences between the two sampling periods were observed within individual treatments: * indicates significant decrease over the ten-year period; ** indicates significant increase over the ten-year period ($\alpha = 0.05$).

Water Variable	2004–2006 Wilson Creek Water Quality Data			2013–2014 Wilson Creek Water Quality Data		
	Reference	Restored		Reference	Restored	
		Cane	Forested		Cane	Forested
Temperature ($^{\circ}$ C)	13.60 \pm 4.00 ^a	15.70 \pm 4.00 ^b	15.60 \pm 4.00 ^b	8.81 \pm 1.30*	10.44 \pm 1.50*	10.24 \pm 1.40*
pH	8.00 \pm 0.09 ^a	8.10 \pm 0.09 ^a	8.20 \pm 0.09 ^a	7.40 \pm 0.06*	7.50 \pm 0.07*	7.50 \pm 0.07*
Alkalinity (HCO ₃ ⁻) (mg/L)	547.00 \pm 36.00 ^a	618.00 \pm 36.00 ^b	618.00 \pm 35.00 ^b	290.00 \pm 9.00*	290.00 \pm 11.00*	289.00 \pm 10.00*
Specific Conductance (μ S/cm)	259.00 \pm 14.00 ^a	278.00 \pm 18.00 ^a	275.00 \pm 16.00 ^a	391.00 \pm 17.00	395.00 \pm 18.00	397.00 \pm 17.00
Cl(mg/L)	12.60 \pm 2.20 ^b	8.00 \pm 2.20 ^a	8.00 \pm 2.20 ^a	9.60 \pm 1.20	7.40 \pm 0.90	7.90 \pm 1.00
SO ₄ ⁻ (mg/L)	34.80 \pm 2.70 ^a	34.40 \pm 2.70 ^a	32.30 \pm 2.60 ^a	21.20 \pm 1.10*	20.30 \pm 1.20*	20.80 \pm 1.20*
Mg (mg/L)	31.50 \pm 5.10 ^a	20.40 \pm 5.10 ^a	20.00 \pm 4.90 ^a	11.50 \pm 0.40*	11.40 \pm 0.40*	11.40 \pm 0.40*
Ca (mg/L)	36.90 \pm 8.00 ^a	38.50 \pm 8.00 ^a	35.20 \pm 7.80 ^a	22.20 \pm 0.80*	21.60 \pm 0.90*	21.80 \pm 0.80*
Na (mg/L)	8.40 \pm 1.10 ^b	5.10 \pm 1.10 ^a	5.10 \pm 1.00 ^a	5.50 \pm 0.30	4.80 \pm 0.20	4.80 \pm 0.20
K (mg/L)	4.90 \pm 0.80 ^b	2.80 \pm 0.70 ^a	2.90 \pm 0.70 ^a	2.50 \pm 0.10*	2.40 \pm 0.10*	4.80 \pm 0.20*
DO (mg/L)	11.70 \pm 1.60 ^a	11.90 \pm 1.60 ^a	12.50 \pm 1.60 ^a	14.30 \pm 0.90**	12.70 \pm 0.80**	12.90 \pm 0.80**
NO ₃ -N (mg/L)	0.63 \pm 0.16 ^b	0.29 \pm 0.16 ^a	0.31 \pm 0.16 ^a	0.41 \pm 0.05	0.24 \pm 0.04	0.24 \pm 0.04
NH ₄ -N (mg/L)	0.03 \pm 0.03 ^a	0.05 \pm 0.03 ^a	0.03 \pm 0.03 ^a	0.05 \pm 0.01	0.05 \pm 0.01	0.04 \pm 0.01
Total Fe (mg/L)	0.03 \pm 0.07 ^a	0.05 \pm 0.07 ^a	0.08 \pm 0.07 ^a	0.03 \pm 0.01*	0.05 \pm 0.02*	0.04 \pm 0.02*
Total Mn (mg/L)	0.12 \pm 0.03 ^a	0.10 \pm 0.03 ^a	0.12 \pm 0.03 ^a	0.03 \pm 0.01	0.09 \pm 0.03	0.08 \pm 0.03

pH, percent total carbon (TC), and percent total nitrogen (TN), and the other was used to determine gravimetric water content, extractable NO₃N, and NH₄N. Sampling and analytic protocols were performed in-house and followed standard soil analysis methods (Mulvaney, 1996, Andrews et al. 2011). Quality checks of replicates, analytical blanks, and manufacturer's standards were used in conjunction with these analyses.

Tree height growth was measured and survival was recorded for each tree species (*P. occidentalis*, *F. pennsylvanica*, and *Q. palustris*) in each of the eighteen experimental plots in 2004, 2005, 2006, and 2013. Since Andrews et al. (2010) summarized results from 2004–2005, we present data from 2006 and 2013. Tree height was measured to the top of the living stem with a telescoping measuring pole. Browse and beaver damage were also recorded. Percent survival and height growth were calculated by treatment and by species. Experimental cane and forest plots (D–I) were not surveyed for growth and survival in 2013 because plot markers were missing and planted seedlings could no longer be distinguished from recruited seedlings.

Data Analyses

Water quality data were checked for normality and transformed (ln, log, or square root transformation) to optimize normality. Data were analyzed by ANOVA using PROC MIXED (SAS 9.3, SAS Institute, Carey, NC) with time

(days from start of sampling), restoration treatment, and time \times treatment interaction modeled as fixed effects, and replicates modeled as a random effect. The SP (POW) covariance structure was selected because it provided the best fit for uneven intervals between sampling dates. Water quality data (2013–2014) were also analyzed for downstream gradient effects using PROC GLM (SAS 9.3). Significant differences detected by ANOVA were further tested using pairwise differences of LS means (pdiff statement).

Means of soil analysis parameters were calculated for each plot and analyzed by ANOVA using PROC MIXED (SAS 9.3), with year, treatment, depth, and interactions modeled as fixed effects and replicates modeled as a random effect.

Tree height growth data were averaged by species within plot. Tree height means were analyzed by ANOVA using PROC MIXED (SAS 9.3) with herbicide treatment, shelter treatment, species, year, row and all treatment \times species \times year interactions modeled as fixed effects, year modeled as the repeated statement, and replicate modeled as a random effect. Significant differences detected by ANOVA were tested using pairwise differences of LS means (pdiff statement). Survival proportions were calculated for each species within plot. Survival proportions were analyzed using PROC GLIMMIX (SAS 9.3), with treatment, species, year, and treatment*species*year modeled as fixed effects and replicate modeled as a random effect.

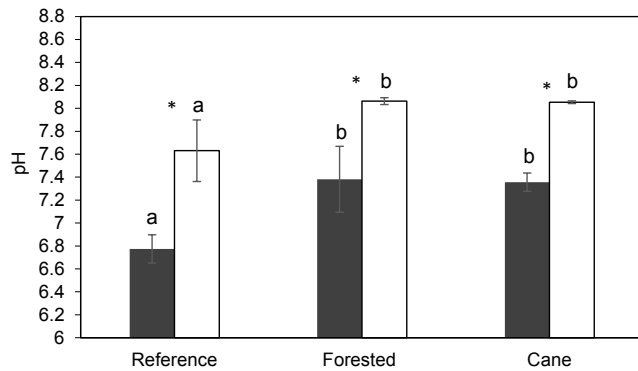


Figure 1. Soil pH results from 0–10 cm depth soil cores collected within reference, forested, and cane plots in July 2004 (black bars) and 2015 (white bars) at Wilson Creek, Kentucky, USA. Data means with the same letter are not significant within years ($\alpha = 0.05$). Asterisks (*) indicate significance between years ($\alpha = 0.05$).

Results

Water

Although Andrews et al. (2011) found differences in water quality between treatments in the 2004–2006 sampling period, we observed no treatment effects in the water samples for plots A–I for the 2013–2014 sampling period (Table 1). However, a time effect in water quality between the two sampling period was observed. Of the water quality variables analyzed for the two sampling periods (2006 and 2013–2014), eight significantly decreased over time, including SO_4 (ANOVA; $F_{1,240} = 73.46, p < 0.001$), Mg (ANOVA; $F_{1,240} = 99.25, p < 0.001$), Ca (ANOVA; $F_{1,240} = 19.05, p < 0.001$), K (ANOVA; $F_{1,240} = 25.68, p < 0.001$), alkalinity (ANOVA; $F_{1,240} = 345.26, p < 0.001$), pH (ANOVA; $F_{1,240} = 122.85, p < 0.001$), Fe (ANOVA; $F_{1,19} = 6.23, p = 0.022$), and temperature (ANOVA; $F_{1,255} = 10.71, p = 0.001$); while dissolved oxygen increased (ANOVA; $F_{1,232} = 4.13, p = 0.043$; Table 1). Conductivity, Cl, Na, nitrate, ammonium, total organic carbon, phosphate, and manganese were statistically similar during the two sampling periods. In 2013–2014, nitrates significantly decreased (ANOVA; $F_{1,8} = 1.07, p = 0.006$) from upstream to downstream; conversely pH increased (ANOVA; $F_{1,8} = 0.75, p = 0.036$). In addition, temperature and dissolved oxygen, though not statistically significant, had increasing and decreasing downstream trends, respectively.

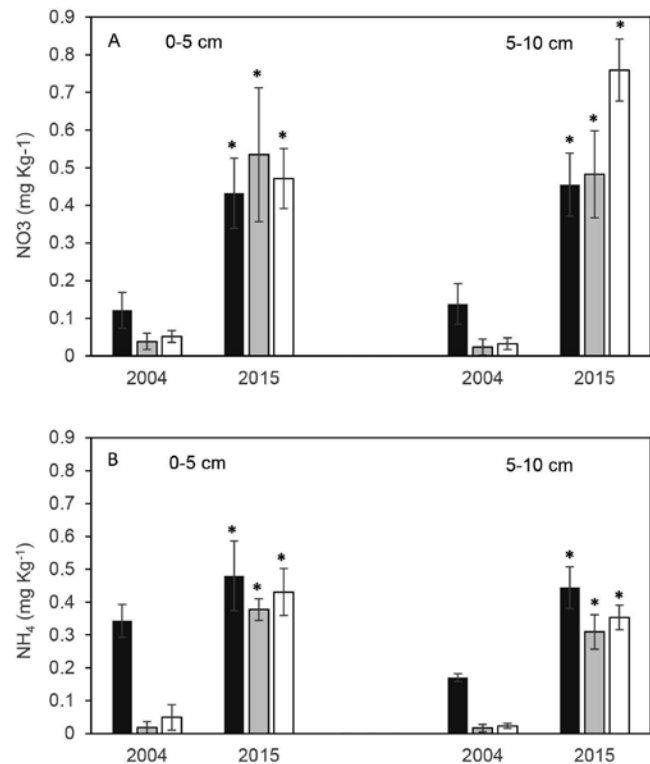


Figure 2. Soil A) extractable nitrate and B) extractable ammonium concentrations from 0–5 cm and 5–10 cm depth soil cores collected within reference (black bars), forested (gray bars), and cane (white bars) plots in November 4 2004 and December 2015 at Wilson Creek. Asterisks (*) indicate significant difference between years ($\alpha = 0.05$).

Soil

Overall, across all sites (reference, forest, cane), soil pH (ANOVA; $F_{1,8} = 73.11, p < 0.001$; Figure 1), extractable NH_4N (ANOVA; $F_{1,8} = 77.70, p < 0.001$), extractable NO_3N (ANOVA; $F_{1,8} = 164.75, p < 0.001$; Figure 2), TC (ANOVA; $F_{1,8} = 86.10, p < 0.001$), and TN (ANOVA; $F_{1,8} = 21.07, p = 0.002$; Figure 3) increased over time. In addition, extractable NH_4N (Figure 2) and TC (Figure 3) in treatments (forest, cane) remained below reference plot levels, while pH in treatments (forest, cane) persisted above reference plot levels (Figure 1), regardless of year. Soil depth effects were observed for both TC and TN. Specifically, TC decreased with depth (t-test; $t = 3.19, p = 0.008$), and

Table 2. Tree seedling survival and growth by shelter treatment between 2006 and 2013 at Wilson Creek, Kentucky.

Shelter Treatment	Mean Survival 2006	Mean Survival 2013	Mean Height Growth (cm) 2006 ± SE	Mean Height Growth (cm) 2013 ± SE
Control	0.82	0.67	20.2 ± 6.8	264.3 ± 43.9
Herbicide	0.76	0.68	25.2 ± 8.7	128.6 ± 26.5
Tubex	0.81	0.69	80.4 ± 7.8	231.9 ± 30.9
Continental	0.85	0.56	73.6 ± 7.2	286.1 ± 50.6
Tubex × herbicide	0.75	0.48	85.0 ± 9.9	338.3 ± 52.9
Continental × herbicide	0.92	0.53	58.6 ± 9.1	330.9 ± 68.3

TN increased over time (t-test; $t = -5.17$, $p < 0.001$) in the upper soil layer but did not change over time in the lower soil layer (Figure 3). The soil carbon-nitrogen ratio decreased over time for the reference plots and leveled to near reference plot ratios in the treatments (forest, cane; Figure 3).

Tree Survival and Height Growth

The first two rows (A, B) of trees were removed from analysis due to flooding and mitigation activities that occurred in 2004 and 2005 leading to high mortality (see Andrews et al. 2010); an equal number of each tree species was removed. Shelter treatment had no effect on short term (Andrews et al. 2010) or long term survival or height growth (Table 2). However, survival of all tree species declined over time. *Quercus palustris* survival, in particular, decreased by greater than 50% after the initial post-restoration measurements (Table 3). After ten years, *P. occidentalis* (80%) and *F. pennsylvanica* trees (79%) had a much higher rate of survival than *Q. palustris* (22%); though, overall tree survival at the site was 60%. Long-term height growth responses were not significantly influenced by herbicide, shelter treatments, or row position. However, there were significant long-term height growth differences observed between each species and within one species, *P. occidentalis* (t-test; $t = -11.43$, $p < 0.001$). In 2013, mean *P. occidentalis* height was 5.5 times their 2006 mean height (2006, 87.8 cm \pm 5.2 cm; 2013, 477.9 cm \pm 27.5 cm). In comparison, *F. pennsylvanica* increased only 1.5 times in mean height (2006, 67.8 cm \pm 5.8 cm; 2013, 103.1 cm \pm 14.3 cm), and *Q. palustris* decreased in mean height (2006, -1.2 cm \pm 4.4; 2013, -19.8 \pm 11.0) likely due to observed main stem dieback (Table 3). In 2013, tree height was significantly different among species, with *P. occidentalis* taller than *F. pennsylvanica* (t-test; $t = -8.26$, $p < 0.001$) and *Q. palustris* (t-test; $t = -9.66$, $p < 0.001$); and *F. pennsylvanica* significantly taller than *Q. palustris* (t-test; $t = 2.44$, $p = 0.021$; Table 3).

Discussion

Determining success of stream restoration projects is often hindered by inadequate post-restoration monitoring, which can be due to limited funding and inconsistent regulatory monitoring criteria (Bash and Ryan 2002, Bernhardt et al. 2005). Our ten-year post-restoration results at

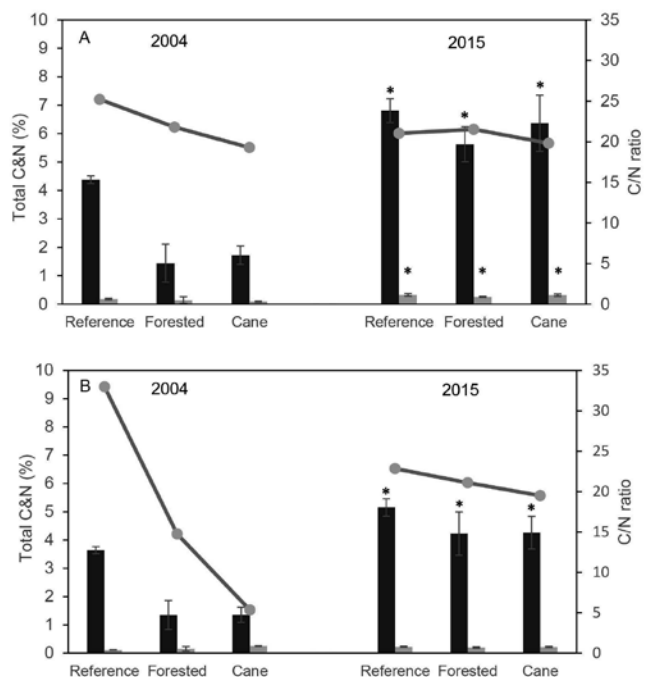


Figure 3. Percent total carbon (TC; black bars) and total nitrogen (TN; gray bars) on left y-axis at A) 0–5 cm and B) 5–10 cm soil depth. Carbon Nitrogen Ratio (C/N Ratio) is the right y-axis and represented by gray lines. Asterisks (*) indicate significant difference between years ($\alpha = 0.05$).

Wilson Creek varied substantially from the initial two-year post-restoration results in water quality, soil characteristics, and tree survival and height growth, which supports arguments for longer term monitoring at restoration sites (Bernhardt et al. 2005, Palmer 2007). Altogether, our long-term monitoring results at the Wilson Creek restoration site were encouraging, as expected. Compared to the original two-year post-restoration monitoring data, the water and soil quality at the site improved substantially to near reference levels, while tree survival and height growth responses were varied and species dependent.

Initial disturbances caused by restoration activities can cause a spike or a decline in water quality parameters due to weathering and erosion of disturbed soils and rock. For example, Andrews et al. (2011) documented significant increases in water temperature and alkalinity and decreases in NO_3N , Cl, Na and K in the Wilson Creek restored reach when compared with the reference reach. Our long-term monitoring revealed a significant decrease in levels of several key water quality parameters and an increase in

Table 3. Seedling survival and growth of three riverine forest tree species between 2006 and 2013 at Wilson Creek, Kentucky. Different lowercase letters indicate significant differences in survival or growth among the three species ($\alpha = 0.05$). * indicate significant differences in survival or growth for individual species between sample years.

Species	Mean Survival 2006	Mean Survival 2013	Mean Height Growth (cm) 2006 \pm SE	Mean Height Growth (cm) 2013 \pm SE
<i>Platanus occidentalis</i>	0.91 ^a	0.80 ^a	87.8 \pm 5.2 ^a	477.9 \pm 27.5 ^{a*}
<i>Fraxinus pennsylvanica</i>	0.87 ^a	0.79 ^a	67.8 \pm 5.8 ^a	103.1 \pm 14.3 ^b
<i>Quercus palustris</i>	0.67 ^b	0.22 ^{b*}	-1.2 \pm 4.4 ^a	-19.8 \pm 11.0 ^c

dissolved oxygen, indicating an overall improvement in water quality over the ten-year period; these results are consistent with the time expected for pelagic freshwater system recovery outlined in Jones and Schmitz (2009). Vigorous growth by some of the planted forest species and observed canopy closure by 2013 provided thermal protection in the riparian area. Lower stream temperature in the absence of oxygen-consuming stream pollutants would correspond with a rise in dissolved oxygen concentrations, which was observed. Re-vegetation of the riparian zone and increased floodplain connectivity, similar to that described by Dosskey et al. (2010) may also contribute to nutrient reduction via plant assimilation or sorption in soil. Although water quality improved overall, restoration legacy effects are still apparent in upstream to downstream differences, specifically for NO_3N and pH. Although canopy closure has occurred in the riparian zone, tree height and widths are not yet sufficient to fully shade the stream channel in the restored section of Wilson Creek. As such, light conditions are favorable for in-stream periphyton production, which may have contributed to the observed decline in nitrate (Gentry 2007). Further, increased floodplain connectivity downstream likely augments surface and ground water interactions (Bukaveckas 2007), consequently decreasing nitrate and elevating pH, due to limestone dissolution in the floodplain.

The removal of topsoil during restoration can influence soil properties (Bruland et al. 2006, Unghire et al. 2011). In 2004, the restored section was comprised of bare mineral soil with little groundcover and organic matter (leaf litter or woody debris). At that time, carbon and nitrogen levels in the restored section were much lower than levels observed in the reference (Andrews et al. 2011). Half of the soil parameters we tested in the experimental plots (nitrate, TN and C/N ratio), initially statistically lower than reference sites, are currently essentially equal to reference plot levels, indicating significant soil improvement and development. While extractable ammonium and TC has also increased substantially since 2004, showing improvement, more time may be needed for these soil characteristics to achieve reference levels; research by Craft et al. 2002, suggests a period of 70–90 years for full streamside soil development of a created marsh in North Carolina. Additionally, we observed a higher pH in treatment soils, which may be attributable to continued weathering of bed materials or the addition of organic plant material leading to the biological decarboxylation of organic ions (Yan et al. 1996, Xu et al. 2006). Alternatively, we observed a significant decline in stream water pH between the two sampling periods (Table 1), which may have influenced soil pH conditions. As observed by Agouridis et al. (2012) circumneutral (pH 6.6–7.5) water infiltrating through soils with high carbonate content can increase mineral weathering and release alkalinity, resulting in an increase of the soil pH. Although, we did not record higher ammonium

and nitrate in the upper surface layer of the soil as Schnell and King (1994) observed, a stratified layer effect was observed for TC. Amount of precipitation (Sextstone et al. 1985) and sampling season (Andrews et al. 2011) can affect soil characteristics; however, rainfall averages between the two years sampled did not vary widely and the plots were sampled in the same season across years. It is important to note, however, that our sampling was limited to one event between the two time periods, narrowing the scope of our conclusions. Future soil sampling during different seasons may help clarify our soil results.

Contrary to research by Sweeney and Czapka (2004), we saw no legacy effects of treatments on tree height growth; therefore, the initial advantages of shelters and herbicide (Andrews et al. 2010) were not sustained long-term for our system. As expected, survival of all three of our planted tree species declined over time; however, most concerning is the steep drop in the survival of *Q. palustris* seedlings over the ten-year period. *Q. palustris*, a shade intolerant, slow growing species, is commonly planted at restoration sites; therefore, it is critical to ascertain whether these efforts are successful over the long term. *Platanus occidentalis*, a rapidly growing pioneer species, is especially adapted to disturbed areas in that it has considerable resprouting ability, tolerates full sun, and has a large adult tree size (Witmer and Immel 1977, Loehle 2000). At Wilson Creek, we observed significant natural colonization of *P. occidentalis* seedlings alongside planted seedlings. The lack of available sunlight due to the spacing of our plantings (Clatterbuck et al. 1987, Oliver et al. 1990) and rapid growth of *P. occidentalis* seedlings (planted and naturally colonized), in combination with continual flooding (Hook 1984), and high soil pH (Stanturf et al. 2004), likely contributed to the low survival of *Q. palustris* seedlings and the stunted height growth of *F. pennsylvanica* seedlings at our site. While deer browse activity on planted seedlings is a common concern (Rooney and Waller 2003), we did not observe extensive browse activity on any of our three tree species; however beaver activity on our planted sycamore seedlings was observed and lead to extensive resprouting, adding to the shading pressure on *Q. palustris* seedlings.

Conclusions

Floodplain connectivity design features, including lower banks, adjacent pools, and channel meanders, likely functioned to increase floodplain and channel interactions and improved water quality and soil conditions at our site. Overall tree survival at the site was good, however, the paucity of oak seedlings after 10 years indicates *Q. palustris* was not suitable for the initial site prescription. To maintain a diverse riparian area at the Wilson Creek site, *P. occidentalis* plantings need to be thinned around the *F. pennsylvanica* and remaining *Q. palustris* trees to release them from light and resource competition.

Overall, our restoration outcomes support the argument for longer term monitoring efforts and subsequent adaptive management of sites, particularly with respect to soil properties, tree selection/thinning, and shelter/herbicide use. Specifically, our research suggests soil pH and co-planting species should be considered before allocating funds for planting late successional species in the initial round of planting at restoration sites. Restoration funds may be better utilized by phasing plantings with an initial introduction of early successional species to stabilize the riparian area and subsequent thinning and late successional species planting ten years later, when the soil has recovered from the disturbance and the banks are effectively stabilized. In addition, expensive tree shelters and herbicides may not convey long term benefits at all restoration sites, and long-term restoration outcomes should be considered when investing in what may be short-term successes. However, with respect to soil and water quality improvement, augmenting restoration projects with floodplain connectivity design features is worth the extra expense.

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