SOIL AND WATER CHARACTERISTICS IN RESTORED CANEBRAKE AND FOREST RIPARIAN ZONES[1]

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ABSTRACT: The degradation of streams has been widespread in the United States. In Kentucky, for instance, almost all of its large streams have been impounded or channelized. A restoration project was initiated in a channelized section of Wilson Creek (Nelson Co., Kentucky) to return its predisturbance meandering configuration. A goal of the project was to restore the native riparian corridor with giant cane and bottomland forest species. The objective of this study was to evaluate the use of giant cane in riparian restoration and to compare water quality and soil attributes between restored cane and forested communities. Comparison of data to replicated sites of similar size in undisturbed upstream areas (control) was also examined to evaluate restoration success. Vegetation establishment was initially hindered by frequent flooding in 2004, but mean survival was good after two growing seasons with rates of 80 and 61% for forest and cane plots, respectively. Results showed an improvement in stream water quality due to restoration activities. Significant differences between the cane and forested plots in shallow groundwater dissolved oxygen, NO_3^--N, NH_4^+-N, and Mn concentrations suggest that soil redox conditions were not similar between the two vegetation types. Retention and transformation of carbon (C) and nitrogen (N) within the restored riparian system also differed by vegetation treatment; however, both communities appeared to be advancing toward conditions exhibited in the control section of Wilson Creek.

(KEY TERMS: riparian restoration; water quality; nutrient dynamics; giant cane.)

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INTRODUCTION

The United States (U.S.) has more than 5.5 million kilometers of rivers and streams that, in conjunction with their riparian corridors, have great economic, social, cultural, and environmental value (FISRWG, 2001). Riparian corridors are complex ecosystems that effectively perform a number of functions such as improving water quality (Parkyn *et al.*, 2003; Dosskey *et al.*, 2010), providing habitat for aquatic and terrestrial plants and animals (Tufekcioglu *et al.*, 2003) and supplying carbon (C) and nitrogen (N) in the form of litter and woody debris (Hafner and Groffman, 2005).

However, human activities have had a significant impact on both streams and riparian corridors in the U.S., affecting nearly 98% of these systems (Benke, 1990; Baron *et al.*, 2003). Restoring streamside

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forests is requisite to restoring disturbed stream systems as it aims to improve water quality within a given watershed by permitting an increase in nutrient uptake by plants and microbes, increasing in the physical filtration of suspended solids, and by providing a source for C and N input (Sweeney, 1992; Naiman and Décamps, 1997; Tabacchi *et al.*, 1998; Sweeney *et al.*, 2002, 2004).

While the influence of various forested communities on riparian restoration and function has been studied extensively (such as Parkyn *et al.*, 2003), much less information is available for nonforest vegetation types found within these corridors. Woody perennial grasses [vetiver grass (*Vetiveria zizanioides*), giant cane (*Arundinaria gigantea*) and other temperate bamboo species] may be an important species in riparian restoration because of their rapid growth, compact stem and root morphology, and resprouting ability (Dattilo and Rhoades, 2005). Grasses are commonly used for soil erosion control and streambank stabilization throughout the wet, humid, and semi-arid tropics (Young, 1989; National Research Council, 1993). Additionally, temperate bamboos have been suggested as a potential species for revegetation of riparian corridors in North America (Miles, 1998; Harker *et al.*, 1999; Schoonover and Williard, 2003; Cirtain *et al.*, 2004).

Giant or river cane [*A. gigantea* (Walt.) Muhl.] is one of only three bamboo species native to the U.S. (Triplett and Clark, 2009). Giant cane occurs from Florida to eastern Texas, and to the north from southeastern Missouri to Virginia (Marsh, 1977; Judziewicz *et al.*, 1999). Dense stands of cane (canebrakes) once covered vast areas of southeastern North America prior to European settlement (Campbell, 1985; Platt and Brantley, 1997). Canebrakes were most commonly found on floodplain terraces where they occurred beneath sparse forest canopies; they also occurred within canopy openings in upland forest and savannas (Farrelly, 1984; Campbell, 1985; Platt and Brantley, 1997). Though cane still grows in small patches throughout its range, canebrakes approaching the size of those observed prior to European settlement are rare. It has been estimated that 85-98% of giant cane canebrakes in the U.S. have been lost and as a result these ecosystems have been designated endangered (Noss *et al.*, 1995) and are a priority for conservation and restoration (Platt *et al.*, 2001).

High culm density (16-26 culms/m^2), rapid lateral spread (up to 6 m/year), and height growth (8 m/year) (McClure, 1973; Marsh, 1977) make *A. gigantea* a logical species of choice for riparian buffer zones. Giant cane's compact network of rhizomes benefits aquatic resources through streambank stabilization, sediment retention, and bioaccumulation

of nutrients and toxins (Welsch, 1991; Geyer *et al.*, 2000; Lee *et al.*, 2003). For example, Schoonover and Williard (2003) observed reduced groundwater (GW) nitrate levels in Illinois riparian zones that were planted with *A. gigantea* due to enhanced plant nitrogen uptake and microbial denitrification.

Straightening of stream channels (channelization) was commonly practiced in the U.S. in the mid-1900s for the benefits of flood protection and agricultural drainage (Black and Fisher, 2001). Channelization can affect the stream ecosystem through increased flow velocity, accelerated erosion of the stream channel (incision), loss of habitat features (pools, riffles), diminished connection to the floodplain, and reduced nutrient retention and hyporheic exchange (Bernhardt *et al.*, 2005). In 2003, a straightened and incised channel was restored to its original morphological configuration using natural channel design techniques (Rosgen, 1996). The overall aim of this project was to evaluate the influence of channel restoration activities on water quality. As a part of this objective, revegetation efforts that complement channel design parameters for improving water quality were examined. Additionally, the project measured soil nutrient and carbon pools between cane, forest, and control plots to evaluate biogeochemical change within the restored riparian corridor.

MATERIALS AND METHODS

Site Description

The Wilson Creek riparian restoration site is located in central Kentucky's Knobs Region (37°52′ N, 85°36′ W) on the western border of the Bluegrass section of the Interior Low Plateaus Physiographic Province (Quarterman and Powell, 1978). It is a third order stream and a tributary of the Rolling Fork River in the Salt River watershed. Wilson Creek forms the southeastern border of the Bernheim Arboretum and Research Forest. Deep valleys and bedrock, mostly composed of Ordovician limestone and shale, characterize the region. Soils have been classified as fine-loamy, mixed, mesic, Dystric Fluventic Eutrochrepts, with medium fertility, moderately rapid permeability, moderate water holding capacity, and deep rooting zones (Dattilo, 2003).

The Wilson Creek Restoration Project

Sections of Wilson Creek were straightened in the late 19th or early 20th Century by European settlers

to maximize the amount of arable land in the valley bottom. This transformation resulted in a confined stream channel with low sinuosity which increased flow velocity and channel incision. The incised channel contained a bedrock streambed with limited pool and riffle zones. Channelization of the stream also resulted in a reduction in GW levels in the floodplain alluvium, increased bank erosion and decreased channel bar stability. Thus, stream channelization increased sediment input into the stream and simultaneously reduced the capacity of the stream to retain sediment and nutrients by disconnecting it with the floodplain (Bukaveckas, 2007).

Restoration activities were focused on a 1 km section of Wilson Creek and a 6.5 ha floodplain area which had been maintained as a tall fescue (*Festuca arundinacea*) hay field. By redirecting Wilson Creek into its previous drainage, it was projected that the stream would be reconnected to its floodplain, causing an increase in both floodplain flooding and GW levels that create and support adjacent wetlands, and an alluvial gravel streambed. The stream design followed techniques outlined by Rosgen (1996) with ranges for bankfull dimensions, meander belt width and radius, and channel slope attained from reference reaches on the forest. The morphometry and location of the designed channel were determined in part by historical documents, topographic surveying, and excavations to locate the former streambed alluvium. Floodplain terracing was completed with a bulldozer while pools were excavated with a backhoe. Riffles were lined with gravel taken from the channelized reach, and a combination of Anti-Wash Geojute™ (Belton Industries, Belton, SC, USA) and burlap fabric was used to stabilize banks. Channel construction and erosion control activities were completed in December 2003.

In March 2004, reestablishment of a native riparian corridor was undertaken with the use of giant cane (*A. gigantea*), as well as native forbs, wetland herbaceous species, and bottomland forest species. Forested tree species used in this project included, but were not limited to, American sycamore (*Platanus occidentalis*), green ash (*Fraxinus pennsylvanica* var. *subintegerrima*), pin oak (*Quercus palustris*), boxelder (*Acer negundo*), black willow (*Salix nigra*), dogwood (*Cornus* sp.), and northern spicebush (*Lindera benzoin*), all common in Bernheim riparian zones. Over 13,000 bare-root seedlings, 2,000 wetland herbaceous plugs and several hundred pounds of seed were planted in the reshaped 6.5 ha floodplain area. All planted species were native to central Kentucky. Images from the site (S1-S6) showing the progress of the restoration and vegetative response may be found in the online version of this article.

Experimental Design

Nine experimental plots, 225 m^2 in area and approximately 1 m from the streambank, were established at Wilson Creek in 2004. In the restoration corridor, three plots were planted entirely with giant cane and three were planted with a mixture of forest hardwood species (American sycamore, pin oak, and green ash). Three control plots of similar size and arrangement were established in an undisturbed section of Wilson Creek <1 km upstream from the restoration area. Planted treatment plots were positioned adjacent to riffles and were separated by distances of 50-200 m. Control plots were separated by a distance of approximately 0.3 km. Cane culms were transplanted in clumps (root balls containing three or more culms) on 1.5 m centers using methods described by Dattilo and Rhoades (2005). Seedlings were planted in the forested plots on 2 m centers. Tubex® tree shelters (Treessentials, Duluth, MN, USA) (1.5 m tall) were placed over the seedlings after planting and attached with plastic lock ties to oak stakes that were anchored into the soil.

Site Measurements

Hydrology and Water Quality. Stream water level was continuously monitored using a capacitance logger (WL-40; Remote Data Systems, Whiteville, NC, USA) at the streamside edge of each plot. HOBO® (Onset Computer Corporation, Bourne, MA, USA) water temperature probes were used to semicontinuously (4 hour intervals) monitor stream water temperature. A HOBO tipping bucket was installed in the restored area to measure precipitation.

Shallow GW wells were used to collect water samples from the saturated zone in each plot. The GW sample wells were constructed by hand. A trench was dug in the saturated zone to a depth of approximately 5 cm below the bottom of an adjacent stream channel cross-section, or to bedrock. A line-level and rod were used to verify that the desired trench depth was achieved. A 1.5 m well casing (5 cm diameter) slotted along its entire length was placed in the trench parallel to the stream channel. This well casing was then connected to a "standpipe" out of which water samples were collected. In all cases, standing water was observed in the trench prior to backfilling with excavated soil.

Water samples were collected from the GW wells and the stream channel at each plot on a monthly basis between March 2004 and May 2006. Sampling, preservation and analytic protocols were performed in accordance to procedures outlined in the Standard Methods for the Examination of Water and Wastewa-

ter (Greenberg *et al.*, 1992). To ensure data quality, a system of calibration standards, analytical blanks (de-ionized water), replicates (every 10 samples), and spikes were used.

Groundwater wells were purged with a peristaltic pump before sampling. Field measurements of pH, oxidation-reduction potential, electrical conductivity (EC), dissolved oxygen (DO), and temperature were performed using a YSI® (YSI, Inc., Yellow Springs, OH, USA) environmental monitor (610 D Model) for all stream water and GW samples. EC values were converted to specific conductance at a reference temperature of 25°C. The YSI® was calibrated prior to each sample date following instructions in the Operator's manual.

Analysis of nitrate (NO_3^--N), and ammonium (NH_4^+-N) was performed by colorimetric analysis using a Bran+Luebbe Autoanalyzer (Bran+Luebbe, Analyser Division, Norderstedt, Germany). Continuous-flow multi-test methods for NO_3^--N and NH_4^+-N [MT7/MT8 (EPA 353.2) and MT15/16 (EPA 350.1), respectively] were used. Sulfate (SO_4^{2-}) and chloride (Cl^-) concentrations were determined by means of a quantitative ion chromatography procedure on a Dionex Ion Chromatograph (IC) 2000 (Dionex Corporation, Sunnyvale, CA, USA).

Measurements of sodium (Na^+), potassium (K^+), calcium (Ca^{2+}), and magnesium (Mg^{2+}) concentrations were made with a GBC SDS 270 Atomic Adsorption Spectrophotometer (GBC Scientific Equipment, Hamphsire, IL, USA). An ICP-OES – Varian Vista-Pro CCD Simultaneous (Varian Instruments, Santa Clara, CA, USA) was used to measure total iron (Fe) and manganese (Mn). Alkalinity (HCO_3^-) was determined on an Orion pH meter (Thermo Fisher Scientific, Waltham, MA, USA) and auto titrator with a titrant endpoint pH of 4.6.

Woody Debris. Each plot was surveyed for coarse woody debris (CWD) during summer 2004 and summer 2005. In the summer of 2004, a complete enumeration of all large woody debris (LWD – >10 cm diameter) within each plot was performed, and in summer 2005, the survey was performed for LWD and small woody debris (SWD – 5-10 cm diameter). Each piece of CWD was assigned a decay class (1-sound, 2-intermediate, or 3-decayed), and one measurement of length and three measurements of diameter were taken following the procedures of McClure *et al.* (2004). All pieces of CWD were tagged with a button-capped nail which had identification information to maintain a record of previously measured pieces and also for possible future CWD recruitment studies. For each piece of CWD, the mean diameter and the length was used to calculate volume (cylindrical volume, cm³), and this volume

together with the density (g/cm³) was used to calculate the mass. The mean density value (0.45 g/cm³) used was representative of the species type and an intermediate decay class (McClure *et al.*, 2004).

Woody debris samples were oven dried (60°C) and weighed. Total carbon (TC) and total nitrogen (TN) contents were measured on subsamples of approximately ≤0.1 g on a LECO® CHN-2000 Carbon (C), Hydrogen (H), and Nitrogen (N) Analyzer (LECO Corporation, St. Joseph, MI, USA). Calibration curves were validated daily by using a LECO standard (EDTA-LECO®) and quality control was done by running an EDTA standard every 10 subsamples.

Soil. Duplicate soil sample sets of the upper 0-10 cm were collected five different times between 2004 and 2006 from each plot. On each sampling date, four duplicate sample sets were collected at random from each plot – at the streambank, and 5, 10, and 15 m from the streambank. One soil sample set was air dried and sieved through a 2 mm screen for pH, TC, and TN, whereas the other was used to determine gravimetric water content and extractable NO_3^-. Soil pH was measured in a 1:1 soil-water suspension by an Orion Model 250A pH meter. NO_3^- was extracted from soils with 100 ml of 1 M KCl solution to 10 g of soil (adapted from Mulvaney, 1996), and measured through a modified colorimetric method using a Bran-Luebbe Autoanalyzer. Soil TC and TN per plot were determined on ≤0.2 g subsamples using the LECO® CHN 2000 Analyzer. A system of analytical blanks, manufacturer's standards, and replicates (every 10 samples) were used for these analyses.

Soil moisture measurements were taken monthly at a depth of 0-25 cm with an Environmental Sensor® Moisture Point instrument (Model MP-917: Soil Moisture Instrument; Gabel Corporation, Victoria, British Columbia, Canada).

Data Analysis

Water chemistry parameters were analyzed using linear regression models for a 2 × 3 × 4 [stream water *vs.* groundwater × treatment (cane *vs.* forested *vs.* control) × season (winter, spring, summer, fall)] experimental design for longitudinal (temporal trends from right after restoration to later post restoration) data (PROC MIXED). The models included all main-effects and two-way interactions. These models were used to determine any differences in the measured water quality parameters between the control and restored plots (cane and forested). These tests were also used to determine significant differences in the measured water variables between the GW and

stream water within the control and the restored plots. Seasonal interactions were also accounted for in these models.

Soil parameters including temperature and moisture were analyzed using repeated measures with a one-way treatment structure (PROC GLM). Soil pH, TC, TN, ammonium, and nitrate were analyzed using repeated measures in a randomized block (edge, 5, 10, 15 m) design (PROC MIXED). Seasonal interactions were accounted for in these models (one dimension repetitiveness). Statistical significance was established where $p < 0.05$ in all cases. All statistical models were performed using SAS Version 9.1.3 (SAS, 2002).

Coarse woody debris (LWD and SWD) were compared between control, cane, and forested plots using Mann-Whitney U tests.

RESULTS AND DISCUSSION

Hydrology, Precipitation and Vegetation Establishment

Following construction of the restored reach, average annual rainfall for 2004 (133.2 cm) was similar to the 20 year precipitation average (132.4 cm), whereas average rainfall was well below the 20 year average in 2005 (96.9 cm) (Figure 1). Even though the yearly average for 2004 was normal, several major precipitation events in May, July, and November resulted in flooding (Figure 2). The control reach over-topped its banks 19 days (2% of time) during the 28 month period shown, whereas the restored section (cane and mixed forest plots) exhibited out-of-bank flow conditions for 119 days (14% of time).

Seed was broadcast throughout the riparian zone in the fall of 2003 for establishment of ground cover for erosion control. Unfortunately, germination rates were low and the area had to be reseeded in April 2004. The May 2004 rain events were especially damaging because topsoil and the recently broadcast seeds were washed away before sufficient cover was established. As such, the numerous out-of-bank events caused significant erosion of streambanks and floodplain soil and severe damage to the two planted rows of cane and forested species closest to the stream channel. During the drought of 2005, water remained pooled in the control area, while the restored segment went completely dry. It should be noted that since part of the overall goal of this project was to increase floodplain interaction (Bukaveckas, 2007), the restored section of Wilson Creek was given relatively low banks, so even with normal levels of precipitation, out-of-bank events were expected to occur more frequently.

Mitigation activities were performed in spring 2005 to repair the damaged areas and to further protect the entire streambank. Dozer activity to re-grade the bank and to install Anti-Wash Geojute™ fabric resulted in the loss of most seedlings and cane clumps planted in the two rows closest to the stream. As such, information pertaining to those affected rows was not evaluated. After two growing seasons (2004 and 2005), the forested bottomland riparian community establishment was successful, with a
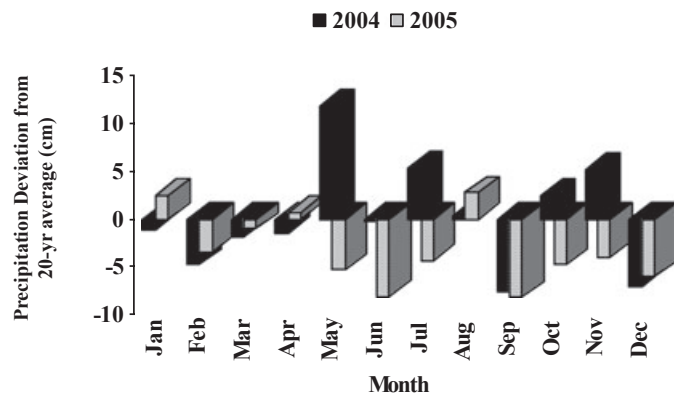

FIGURE 1. Deviation of Monthly Precipitation for 2004 and 2005 From the 20-Year Average at the Bernheim Arboretum and Research Forest, Nelson Co., Kentucky.
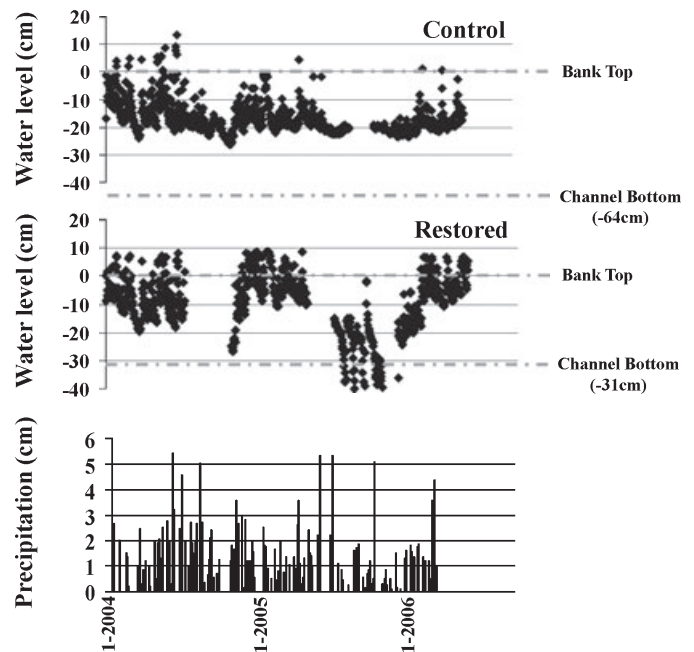

FIGURE 2. Average Daily Stream Water Level for 2004-2006. Zero elevation refers to the height at bankfull and points above that indicate a flooded condition.

mean survival of 80% (range: 62-95%). Cane establishment was only moderately successful (range: 34-58%), after the first growing season. At the end of the second growing season, mean cane survival increased slightly to 61% (range: 58-66%). As noted elsewhere, transplanted culms bearing multiple basal buds may take root under favorable site conditions, but initiation of lateral growth is inconsistent and slow (Platt and Brantley, 1993; Dattilo and Rhoades, 2005). Seemingly, some rhizomes survived the flood damage of 2004 and were capable of re-sprouting during the drier second year. In addition, spread of new growth away from the transplanted clump was observed. Initial culm dieback or culm mortality was presumably due to the frequent flooding.

Water Chemistry

Stream Quality. Stream water quality parameters were examined to assess the water quality change in Wilson Creek due to restoration activities (Table 1). The cane and forested restoration plots were not statistically different for any of the measured stream water variables, but significant differences were observed between the control and restored segments of Wilson Creek for some parameters. Given that the restored segment was downstream of the control, changes in stream quality are presumably due to the overall effect of restoration at the

stream continuum level and not due to individual treatments at the plot level.

Average stream water temperature was found to be significantly lower ($p < 0.05$) in control reaches when compared with the restored reach. Based on the two year study period, mean stream water temperature of the control segment was 13.6°C, whereas the mean temperature in the restored section was 15.7°C. The temperature influence on stream water of the reference section of Wilson Creek can be attributed to the shading effect given by the already established streamside forest in this area as compared with the very open canopy in the restored area. Similar findings have been noted in other riparian restoration projects (Sweeney, 1992; Anbumozhi *et al.*, 2005), and temperature differences are expected to become less significant as the canopy develops.

Restoration activities did not significantly influence DO levels in the stream. The mean stream water DO concentration ranged from 11.7 mg/l in the control area to 12.2 mg/l in the restored area. These DO levels suggest that this stream system was in a highly oxidized state and that the biological oxygen demand was likely low.

The bedrock of Wilson Creek is composed mainly of limestone and high alkalinity levels in stream water was expected. Mean alkalinity was found to be significantly different between the restored treatment plots (618 mg/l) and the control (547 mg/l). This difference is likely due to the exposed soils, and unweathered bed material that was placed in the restored stream channel. With time, it is expected that alkalinity levels in the restored section will decrease as readily dissolved bicarbonate materials are removed and sedimentation decreases.

Stream water Cl^-, Na^+, K^+, and NO_3^--N concentrations were also found to be significantly different between the control and restored sections of the stream. The Cl^- concentration decreased from 12.6 to 8.0 mg/l from the control reach to the restored reach. Mean Na^+ concentration ranged from 8.4 mg/l in the control to 5.1 mg/l in the restored reach, whereas average K^+ concentration was found to be 4.9 mg/l in the control and 2.9 mg/l in the restored section. The NO_3^--N concentration in Wilson Creek's stream water exhibited a significant reduction from 0.63 mg/l in the upstream control reach to 0.30 mg/l in the restored downstream reach. Hydrologic mixing of stream water with GW beneath and adjacent to the channel can have a dramatic affect on the composition and concentration of dissolved ions in stream water (Duff and Triska, 2000), as well as influence many controlling variables of nitrogen cycling (Findlay, 1995; Duff and Triska, 2000). As noted by Bukaveckas (2007), the decline in Cl^- concentrations is likely due to dilution with GW. Decreases in Na^+ and K^+ may

TABLE 1. Mean (standard error) Stream
Water Quality Attributes at Wilson Creek.

Water Variable	Control	Restored	
		Cane	Forested
Temperature (°C)	13.6 (4.0)[a]	15.7 (4.0)[b]	15.6 (4.0)[b]
pH	8.0 (0.09)[a]	8.1 (0.09)[a]	8.2 (0.09)[a]
Alkalinity (HCO_3^-) (mg/l)	547 (36)[a]	618 (36)[b]	618 (35)[b]
Specific conductance (μS/cm)	259 (14)[a]	278 (18)[a]	275 (16)[a]
Cl^- (mg/l)	12.6 (2.2)[b]	8.0 (2.2)[a]	8.0 (2.2)[a]
SO_4^{2-} (mg/l)	34.8 (2.7)[a]	34.4 (2.7)[a]	32.3 (2.6)[a]
Mg^{2+} (mg/l)	31.5 (5.1)[a]	20.4 (5.1)[a]	20.0 (4.9)[a]
Ca^{2+} (mg/l)	36.9 (8.0)[a]	38.5 (8.0)[a]	35.2 (7.8)[a]
Na^+ (mg/l)	8.4 (1.1)[b]	5.1 (1.1)[a]	5.1 (1.0)[a]
K^+ (mg/l)	4.9 (0.8)[b]	2.8 (0.7)[a]	2.9 (0.7)[a]
DO (mg/l)	11.7 (1.6)[a]	11.9 (1.6)[a]	12.5 (1.6)[a]
NO_3^--N (mg/l)	0.63 (0.16)[b]	0.29 (0.16)[a]	0.31 (0.16)[a]
NH_4^+-N (mg/l)	0.03 (0.03)[a]	0.05 (0.03)[a]	0.03 (0.03)[a]
Total Fe (mg/l)	0.034 (0.07)[a]	0.052 (0.07)[a]	0.083 (0.07)[a]
Total Mn (mg/l)	0.118 (0.03)[a]	0.097 (0.03)[a]	0.119 (0.03)[a]

Notes: Means followed by the same letter are not significantly different ($\alpha = 0.05$); $n = 20$ sample dates, monthly between March 2004 and May 2006.

also be attributed to dilution or to sorption on exchange sites on recently exposed soil and rock surfaces. Restored meanders and low banks reduced stream velocity and increased floodplain interaction likely contributed to the observed reduction in nitrate by enhancing surface-subsurface water interactions that lead to more denitrification and/or enhanced nutrient spiraling (Newbold, 1992; Bukaveckas, 2007). Alternatively, increased instream productivity by periphyton may have also contributed to this decline (Gentry, 2007).

Stream Water and Groundwater Interactions. Although there were no significant differences between stream water collected adjacent to the cane and reforested plots, differences were detected between the two treatments in shallow GW (Table 2). Average pH was significantly different between the stream and GW in both treatments and the control. For the different treatments, GW pH in the cane plots was found to be lower than the GW pH in the control and forested plots. Significant differences ($p < 0.05$) were also found for alkalinity and Ca between the stream and GW for the cane plots, but not for the control and forested plots. Calcium and alkalinity concentration were also significantly higher in GW of the cane plots than the forested plots (Table 2). These results may be due to the cane plots having a longer time of saturation (or flooding) than the forested plots. Mean soil moisture (0-10 cm depth) was found to be 4.9 and 4.2% less in the forested and

control plots, respectively, over that measured in the cane plots, which supports the suggestion that the cane plots were more saturated. Limestone dissolution and alkalinity production rates increase as the partial pressure of carbon dioxide (pCO_2) increases (Stumm and Morgan, 1996). Atmospheric pCO_2 (0.0003 atm) is much lower than that observed in flooded or waterlogged soils (0.05 to 0.3 atm) because diffusion of gases is much slower through water than air (Lindsay, 1979). As such, the formation of carbonic acid in the cane plots has resulted in a lower pH, dissolution of limestone, and subsequent release of alkalinity and Ca.

Mean DO concentrations in stream waters were significantly higher than GW concentrations for all three treatments. Based on the different treatments, DO levels were lowest in GW for the cane (3.3 mg/l), followed by the forested (5.1 mg/l), and then the control (6.8 mg/l). This pattern correlates well with moisture and pH data as discussed above and could have an effect on other redox sensitive elements and compounds.

Ammonium concentrations, though minimal, were shown to be significantly higher in GW than in the stream waters, whereas nitrate was found to be significantly higher in the stream water compared with GW (Tables 1 and 2). Similar relationships between NO_3^--N and NH_4^+-N have been noted in the literature (Triska *et al.*, 1989; Duff and Triska, 1990; Hendricks and White, 1991; Valett *et al.*, 1996; Storey *et al.*, 2004). Since NH_4^+-N is oxidized to NO_3^--N in environments where sufficient levels of oxygen are present (Duff and Triska, 1990), it would stand to reason that NH_4^+-N concentrations may be greater in GW and NO_3^--N concentration would be higher in stream waters. Also, in the absence of sufficient oxygen, reduction of NO_3^--N (denitrification) to nitrogen gas may occur, as bacteria use NO_3^--N as a terminal electron acceptor, thus, further decreasing the NO_3^--N levels in the GW. Following the GW DO gradient exhibited between treatments, mean NO_3^--N concentrations were found to be significantly lower in the cane plots (0.12 mg/l) than the forested (0.24 mg/l) and the control (0.35 mg/l) plots, likely as a result of varying levels of denitrification.

Redox sensitive elements, Fe and Mn, were found at higher concentrations in GW compared with stream water (Tables 1 and 2). The higher Mn concentrations observed in the cane can be attributed to longer or more frequent flooding periods in this area. As cane establishment was not as successful as the forested riparian community, it is believed that the lack of root networking in the cane plots may be the reason for the seemingly more reducing environment as compared with the forested plots. The cane plots had a higher water content (as indicated earlier)

TABLE 2. Mean (standard error) Groundwater Quality Attributes at Wilson Creek.

Water Variable	Control	Restored	
		Cane	Forested
Temperature (°C)	16.0 (4.2)[a]	16.2 (4.1)[a]	15.6 (4.0)[a]
pH	7.6 (0.1)[b]	7.2 (0.1)[a]	7.5 (0.1)[b]
Alkalinity (HCO_3^-) (mg/l)	638 (67)[ab]	774 (52)[b]	608 (36)[a]
Specific conductance (μS/cm)	250 (38)[a]	260 (23)[a]	235 (25)[a]
Cl^- (mg/l)	5.3 (3.7)[a]	5.0 (2.8)[a]	3.7 (2.2)[a]
SO_4^{2-} (mg/l)	36.3 (5.5)[b]	37.7 (3.9)[b]	28.2 (2.7)[a]
Mg^{2+} (mg/l)	22.8 (10.9)[a]	22.5 (7.6)[a]	15.9 (5.1)[a]
Ca^{2+} (mg/l)	42.3 (13.3)[ab]	63 (10.2)[b]	39.4 (8.0)[a]
Na^+ (mg/l)	5.6 (2.2)[a]	4.1 (1.6)[a]	3.5 (1.1)[a]
K^+ (mg/l)	3.8 (1.6)[a]	1.2 (1.2)[a]	1.7 (0.8)[a]
DO (mg/l)	6.8 (2.0)[b]	3.3 (1.7)[a]	5.1 (1.6)[b]
NO_3^--N (mg/l)	0.35 (0.29)[b]	0.12 (0.24)[a]	0.24 (0.29)[b]
NH_4^+-N (mg/l)	0.36 (0.05)[b]	0.10 (0.07)[a]	0.25 (0.03)[b]
Total Fe (mg/l)	0.235 (0.18)[a]	0.282 (0.08)[a]	0.359 (0.07)[a]
Total Mn (mg/l)	0.199 (0.03)[ab]	0.287 (0.04)[b]	0.165 (0.03)[a]

Notes: Means followed by the same letter are not significantly different ($\alpha = 0.05$); $n = 20$ sample dates, monthly between March 2004 and May 2006.

TABLE 3. Coarse Woody Debris and Soil Carbon
and Accumulation as Influenced by Vegetation Treatment.

Treatment	LWD (g/m^2/year)	SWD (g/m^2/year)	Soil C (g/m^2/year)
Control	954[a]	144[a]	334
Restored forested	299[b]	122[b]	818
Restored cane	393[b]	45[b]	681

Notes: LWD, large woody debris; SWD, small woody debris. Treatments followed by the same letter are not significantly different ($p = 0.05$). SWD < 10 cm diameter; LWD >10 cm diameter.

and thus, lower oxygen content, potentially resulting in greater reducing conditions.

Coarse Woody Debris

A total of 87 pieces of CWD were tagged across treatments. Seventy-five percent of the measured CWD fell into the intermediate decay class. The total mass of CWD (both LWD and SWD) varied under the different vegetation treatments (Table 3). The mass of LWD and SWD was greatest in the control area and significantly higher than levels observed in the forested and cane plots. SWD was higher in the forested treatment than the cane treatment, but not significantly different. LWD exhibited the opposite trend where the cane treatment was higher than the forested treatment, but not significantly different. The total amount of woody debris in the control (1,098 g/m^2/year) was about 2.5-fold higher than that observed in the forested (421 g/m^2/year) and cane (438 g/m^2/year) treatments. As such, the TC and TN contained within woody debris were greatest in the control section, and also approximately 2.5-fold higher in the forested plots as compared with cane plots. The C:N ratios for woody debris, however, were similar for all treatments with ratios of 51:1, 48:1, and 47:1 for control, forested, and cane treatments, respectively.

It has been shown that CWD may substantially contribute to both C and N pools within forest ecosystems (Hafner and Groffman, 2005; Hefting *et al.*, 2005; Wilcke *et al.*, 2005). Research has also revealed that CWD temporarily stores C and N and may alter C and N availability and transformations especially in the underlying soil and thus influences their cycling in stream ecosystems (Hafner and Groffman, 2005; Wilcke *et al.*, 2005). Particulate organic C from partly decomposed plant material at an early stage of decomposition has been characterized as a transitional stage in humification that provides substrates for microbes (Franzuelebbers *et al.*, 2000; Six *et al.*, 2002) and influences aggregation (Denef *et al.*, 2004; John *et al.*, 2005; Pulleman *et al.*, 2005). Moreover,

soil organic matter fractions with turnover times of years to decades, such as particulate organic C, often respond more rapidly to changes in the soil organic C pool than stabilized mineral-associated C fractions (Cambardella and Elliot, 1992; Carter *et al.*, 2003; Handayani *et al.*, 2008). Thus, an available pool of CWD could facilitate soil development and microbial colonization in the restored riparian area. Even though the restored forest and cane plots are young, results indicate that they are recruiting woody debris at a rapid pace (approximately 60% less than the mature control area in just two years). The low bank design used for restoring Wilson Creek likely enhanced debris recruitment through the large number of out-of-bank events it experienced (Figure 2). Moreover, retention of debris in the terrestrial environment would allow for increased N and C processing in riparian soils.

Soil

There was a significant difference in pH between the control (7.1) and restored (cane – 7.5 and forested – 7.6) segments of Wilson Creek riparian zone, as well as significant seasonal variation. Higher pH in the restored section is likely due to weathering of fresh surfaces of bed material.

Soil NO_3^- and NH_4^+ were found to be significantly higher in the control areas than in the restored plots during the first four sampling events (November 2004 to February 2006), but not significantly different between the restored cane and forested plots (Figure 3). N pools were higher in NH_4^+ than NO_3^-, which would be expected if reducing conditions were prevalent. Soil NO_3^- concentrations also varied significantly by season with the highest levels occurring during summer and lowest during winter. Nitrification rates were likely elevated in the summer of 2005 due to a low water table, higher temperatures, greater aeration, lower soil moisture content, and increased microbial activity. Conversely, lower NO_3^- levels during the wet (winter/spring) months are indicative of a reduction in nitrification. Although not significantly different, NH_4^+ levels in the cane plots were consistently higher than levels observed in the forested plots, which supports other data indicating that the cane site was more reduced.

As the restoration progresses, it would be expected that the microbial population would increase and ultimately, the restored system would function similarly to the control system in retaining inorganic nitrogen sources via plant immobilization, microbial immobilization, and denitrification. Interestingly though, our results indicated that approximately two years after restoration, soil NO_3^- and NH_4^+ levels in the restored
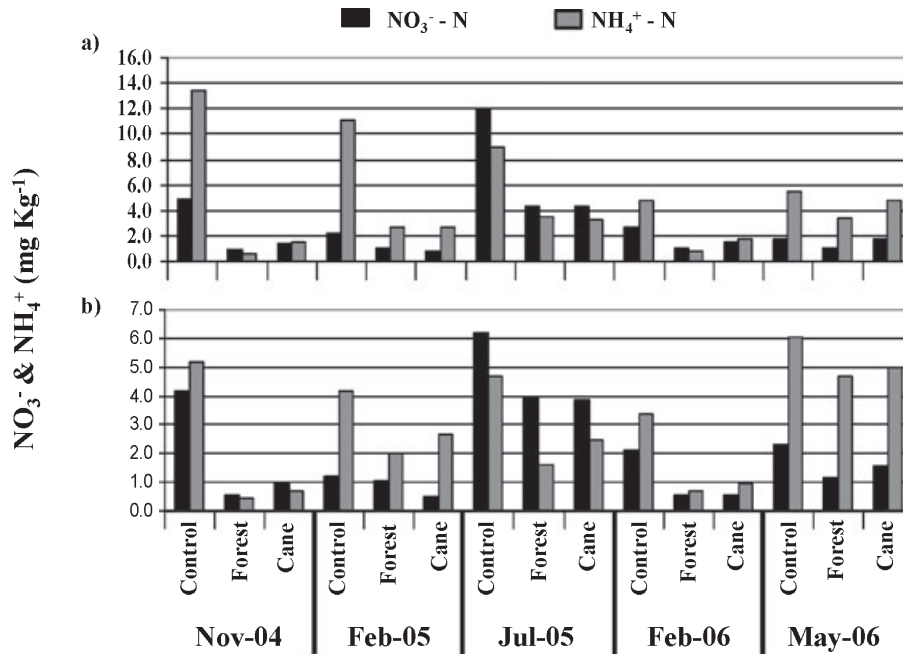
FIGURE 3. Soil Nitrate and Ammonium Concentrations in the (a) 0-5 cm and (b) 5-10 cm Depths With Respect to Different Vegetation Treatments. Significant differences in nitrate and ammonium were observed between the control and restored sections for all but the May 2006 sampling event ($p < 0.05$). Nitrate and ammonium were similar in both treatments and the control in the May 2006 samples. No significant differences were observed between the restored cane and forested plots for all sampling events.

plots were becoming similar to the control plots. In May 2006, no significant differences between the control and restored sections were observed. This may indicate that the planted community is exerting more influence on soil moisture conditions and nutrient transformations as they mature, or this may be indic-

ative of drier soil conditions and less frequent flooding that was observed in 2006.

Carbon and nitrogen dynamics within the soil are influenced by litterfall, root turnover, and microbial activity. Increased vegetation growth and productivity following restoration typically results in larger
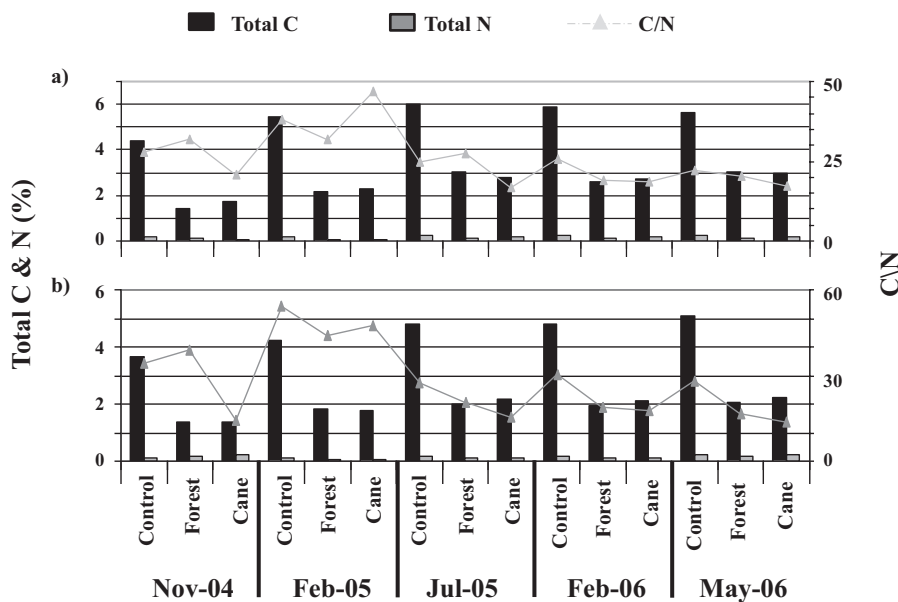

FIGURE 4. Total C, N and C:N Ratio of Soil in the (a) 0-5 cm and (b) 5-10 cm Depths With Respect to Different Vegetation Treatments. Significant differences for C and N were observed between the control and restored sections but not between cane and forested plots for all sampling events.

and more stable inputs from roots and leaf litter (Dosskey *et al.*, 2010). After two years, total soil C and N percent were found to be significantly higher in the control plots compared with restored riparian community treatments, and varied considerably between the seasons (Figure 4).

The C:N ratios were significantly different between seasons (winter – the highest and spring – the lowest), but not between treatments. The average soil C:N ratio at Wilson Creek was found to be 31:1 for the control areas and 27:1 and 23:1 for the restored forested and restored cane plots, respectively, which corresponds well to the gradient of woody debris and litter found on the sites (Table 3). Higher C:N ratios in the control plots may be linked to more of the C in the control areas being in a more resistant (not easily decomposed) form such as lignin. Lower soil C and N percent levels observed in the restored plots were probably due to lower carbon inputs from root turnover, litterfall, and microbial activity. TC and TN were highest during the spring season and lowest during the winter, presumably due to limited microbial activity in the winter (most likely because of soil temperature).

Accumulation of C was observed over the course of the study in both the control and restored sections (Table 3). However, changes in soil C were found to be more than 2× higher in the restored section (forested – 818 g/m^2/year and cane – 681 g/m^2/year) as compared with the control section (334 g/m^2/year). This difference corresponds well to the findings of Handayani *et al.* (2008) who examined soil C pools at the Wilson Creek restoration site. Handayani *et al.* (2007) found that particulate organic C, C-associated with macroaggregates (>2 mm), and the amount of macroaggregates were strongly affected by plant species in the restored treatments, and showed that early changes in soil properties were reflected in labile C pools and soil structure. The cane and forested sites had 45% higher particulate organic C and 64 and 23% more macroaggregates for the 0.25-2 mm and >2 mm size class, respectively, compared with the control sites.

CONCLUSION

Restoration activities at Wilson Creek had an influence on stream chemistry as temperature and alkalinity levels increased in the restored reach, whereas NO_3^--N, Cl, Na, and K levels were reduced. Greater incident radiation, due to lack of riparian cover, and higher temperatures in the restored reach may have provided more suitable conditions for in-stream algae and bacteria growth (periphyton), which

may have led to the observed increase in nutrient processing. In addition, stream features created using natural channel design techniques (meanders, pools, low banks) reduced flow velocities and increased floodplain interaction which likely contributed to the nutrient loss via dilution with GW or by enhancing surface-subsurface water interactions that led to more denitrification and/or enhanced nutrient spiraling.

Prior to restoration, the channelized stream was disconnected to its floodplain (via incision) and stream functions were modified, one of which would be nutrient processing in the terrestrial system. The low banks were a restoration strategy to offset the impacts of channelization and channel incision. Frequently flooding, rapid recruitment of CWD, and increases in soil TC and TN with time suggest that the floodplain in the restored section of Wilson Creek has become more active and indicates that this system is advancing toward conditions exhibited by the control. We believe that reconnecting the stream to its floodplain was a major factor that contributed to improved water quality in the restored section.

The extent of biochemical transformation and translocation in the floodplain will be influenced not only by the biochemical makeup of water entering the system, but also by physical and chemical attributes associated with vegetation in the riparian area (Norris, 2001). The riparian vegetation plays a critical role in the restoration process by providing roots that contribute to the structural stability of sediments; dissolved organic carbon input from litter, exudate secretion, and root turnover; and microsites that are essential for microbial colonization. As such, the design and assessment of stream restoration projects should consider the importance of the riparian vegetation for its part in establishing and maintaining linkages between the stream and its floodplain as they are a key component for reestablishing ecosystem structure and function.

Although substantial research has been performed on examining the role of forested vegetation on riparian restoration and function, little information exists on the utility of cane as a suitable restoration species. Structural attributes of riparian canebrakes are known to provide wildlife habitat for a wide array of mammals, birds, reptiles, and insects (Meanley, 1972; Eddleman *et al.*, 1980; Kilgo *et al.*, 1996; Thomas *et al.*, 1996) including several cane-dependent species (Remsen, 1986; Platt *et al.*, 2001). In addition, water quality benefits have been linked to *A. gigantea* (Schoonover and Williard, 2003; Blattel *et al.*, 2009). The establishment of both forested and cane riparian species at Wilson Creek was initially successful with survival rates of 80 and 61%, respectively. Although cane exhibited a higher mortality than the forested species, new growth from rhizomes was observed

after two growing seasons. The influence of the differing riparian communities on stream water quality was not obvious; however, significant differences in shallow GW and soil development between the cane and forested plots were observed. We speculate that the active-floodplain conditions and frequent flooding observed in 2004 contributed to limited root development in cane. Poor initial development linked with higher mortality likely influenced soil saturation in the cane plots due to decreased water usage over that exhibited by forested species. Significant differences between the cane and forested plots in groundwater DO, NO_3^--N, NH_4^+-N, and Mn concentrations indicates that redox conditions were not similar between the vegetation types. Continued monitoring of soil saturation and root development differences between the two vegetation types is needed to determine if this is a short-term effect related to early development or something that will persist as influenced by physiological differences between cane and riparian forested species. We also recognize that the system is immature and additional monitoring is needed to fully evaluate the role of these two vegetation communities on the recovery of Wilson Creek.

SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Figure S1. Wilson Creek riparian area prior to restoration.

Figure S2. Recently transplanted giant cane (*Arundinaria gigantea*) clumps (root balls containing three or more culms).

Figure S3. Forest riparian species establishment adjacent to the restored Wilson Creek channel.

Figure S4. Giant cane growing in the Wilson Creek riparian area.

Figure S5. Transplanted seedlings growing out of the tree shelters.

Figure S6. View of the Wilson Creek riparian restoration area in June 2006.

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