THE EFFECTS OF STREAM RESTORATION ON WOODY RIPARIAN VEGETATION IN THE NORTHWESTERN NORTH CAROLINA MOUTAIN REGION: A COMPARATIVE STUDY OF RESTORED, DEGRADED, AND REFERENCE STREAM SITES

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ABSTRACT

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Anthropogenic impacts have significantly degraded streams and rivers worldwide. In the past three decades, stream restoration has become increasingly common for addressing issues of waterway degradation. An important component of stream restoration projects is riparian management. Riparian areas are critical to the functioning of stream and river ecosystems and re-vegetation is almost ubiquitous to restoration measures. Re-vegetation is frequently associated with restoration of ecosystem function, ecosystem services, landscape connectivity, and biodiversity. However, monitoring of long-term riparian re-vegetation trajectories is not a mandatory part of the restoration process. Too frequently, collection of vegetation data is neglected. Such databases have the potential to provide useful information about restoration outcomes and ultimately inform best management practice.

This research examines the effects of stream restoration on woody riparian plant communities on headwater streams in the mountains of northwestern North Carolina. Twenty-seven sites were examined within three groups: reference, restored, and degraded

sites. The average age of restored sites was four years since project implementation. Degraded sites were rural agricultural or residential headwater stream sites that could merit restoration and reference sites were sections of headwater streams with intact forest on both sides of the channel. Field-based sampling documented woody species structure and composition in three geomorphic positions (i.e., channel bed, channel bank, top of bank) on two transects per site. Woody structure at restored sites was compared to reference and degraded sites by calculating site level metrics (i.e., species richness, stem density, basal area, percentage canopy cover), and by assessing community composition using multivariate analysis and ordination analysis. Channel structure was also assessed using channel width and percentage channel bed canopy cover metrics.

Restored and degraded sites had similar species richness, stem density, basal area, percentage canopy cover, and channel structure. Restored and reference sites were similar in species richness and stem density, but not basal area, percentage canopy cover, or channel structure. Species dominance differed among all treatments. Degraded conditions were dominated by small-statured, opportunistic species. Restored sites were characterized by the shrub species used for re-vegetation and some opportunistic species associated with degraded sites. At reference sites, typical regional riparian forest conditions were present. Overall species composition showed a distinct pattern for reference conditions that was different from both degraded and restored sites. Degraded and restored sites were not compositionally distinct from one another. These data suggest that currently restoration projects on low-order streams in the mountains of northwest North Carolina do not yet resemble regional reference conditions.

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CHAPTER 1

INTRODUCTION

Stream restoration has become common practice for addressing issues of waterway degradation. Restoration methods range from riparian re-vegetation to large-scale redesign of stream channels. For three decades, United States social and political will for stream restoration has steadily increased (Doyle et al., 2008). Thousands of projects have been implemented and billions of dollars have been spent on United States stream restoration projects (Bernhardt et al., 2007; Doyle et al., 2008; Hobbs, 2007; Wohl et al., 2005). There has been a recent and widespread call for better integration of modern scientific understanding of fluvial ecosystems into the application of stream restoration projects (Bernhardt et al., 2007; Wohl et al., 2005). Despite an increasingly advanced scientific understanding of fluvial environments and processes, the science and practice of stream restoration remain relatively isolated fields (Bernhardt et al., 2007; Wohl et al., 2005). Thus, the ecological outcomes of stream restoration projects remain relatively un-documented (Bernhardt et al., 2005).

In this chapter, I discuss first the function and successional pathways of woody riparian plant communities. Next, I examine the characteristics and causes of stream and riparian area degradation. And in the final section of this chapter, I investigate the origins of stream restoration, the prevalence of ecological restoration and riparian re-vegetation, as well as the status of project monitoring and assessment.

1.1 Woody Species and Riparian Function

Woody vegetation has important and complex physical effects on stream channel morphology. As alluvial material moves through a stream channel, rates of erosion, transportation, and deposition are influenced by velocity of water (Knighton, 1998). Roughness is a common measure of flow resistance and studies have shown vegetation to be a primary source of velocity dissipation and diversion in stream channels (Nepf & Vivoni, 2000). Masterman & Thorne (1992) demonstrated that variation in vegetation height, density, and flexibility influenced bank shear strength by increasing reach scale roughness in more vegetated areas. Heavy vegetation, most commonly associated with humid regions, can divert in-channel flow paths, initiate meanders, and resist channel widening. In sections of a channel with less vegetation, channel incision and widening can be common (Corenblit, Tabacchi, Steiger, & Gurnell, 2007).

Vegetative root systems minimize erosion of alluvial material, thus facilitating bank cohesion and landform creation and stabilization (Corenblit et al., 2007; Downs & Gregory, 2004). Early research, such as Smith (1976) and Zimmerman, Goodlett, & Comer (1967), demonstrated that denser root networks result in less erodible stream banks. These study results showed stream channel width-depth ratios to be lower (i.e., narrower, deeper, more incised) for grassy reaches than forested reaches, suggesting that stream banks characterized by shallow but mat-like root systems of grass communities denote most geomorphic stability. However, Davies-Colley (1997) and Gregory & Gurnell (1988) discussed the tendency of tree roots, by penetrating to lower levels of bank alluvium than grasses, to increase the vertical shear strength of the channel banks. Thus, tree cover reduced the propensity for bank undercutting erosion patterns. Hickin & Nanson (1984) demonstrated, on western

Canadian rivers, that un-vegetated alluvial stream sites erode at twice the rate of naturally forested stream sites.

Woody riparian vegetation is important for creating and maintaining aquatic habitat conditions and water quality. Riparian buffer zones have long been recognized for non-point source pollution control, from both upland and aquatic inputs, via nutrient filtration and retention (Lowrance, 1997; Lowrance et al., 1984; Malard, Tockner, Dole-Olivier, & Ward, 2002; Peterjohn & Corell, 1984; Sabater et al., 2003). Hanson, Groffman, & Gold (1994), Lowrance et al. (1984), and Peterjohn & Correll (1984) showed mature riparian forests (30-70 years of age) to form a dynamic but stable buffering system that reduced agricultural non-point source pollution and sustained water quality. The function of riparian ecosystems is particularly important on small streams, where fluvial and terrestrial ecosystems interact frequently and influentially (Lowrance, 1997; Naiman, Décamps, & McClain, 2005). Low-order streams account for over three-fourths the total stream length in the United States (Leopold, Wolman, & Miller, 1964). Increased fluvial and terrestrial interaction in small stream riparian plant communities facilitate more efficient pollutant and dissolved nutrient removal from water and soils (Alexander, Smith, & Schwartz, 2000).

Another important function of riparian forests is the contribution of wood to stream channels. The amount of wood in any stream is affected by forest type, succession stage, disturbance history, decomposition rate, and channel size (Downs & Gregory, 2004). In-stream wood influences the shaping of the stream channel by affecting channel roughness and the water and sediment routing in the fluvial corridor (Davis & Gregory, 1994; Gregory & Davis, 1992). Thus, wood affects bank stability, location of channel change, sediment storage, and the development of the pool-riffle sequence in mountain streams (Downs &

Gregory, 2004). In-stream wood also affects stream ecology. Due to the abundance of wood, leaves, twigs, and fruit in typical small stream channels, woody inputs constitute a significant proportion of in-stream nutrients (Bilby, 2003). In-stream wood also captures fine, nutrient-rich organic matter and sediment in stream channels, thus reducing rapid material transport and promoting local nutrient availability for biological processing (Bilby, 2003). Habitat creation results from the presence of wood in streams. On small high-gradient streams, wood inputs form dams creating a step-pool profile where water velocity slows upstream of the obstruction and plunge pools and riffle sequences are created downstream. Up to 90 % of forested, small stream pools have been attributed to woody inputs (Dolloff & Warren, 2003). Numerous studies have shown wood-created, in-stream features to be critical habitat for both invertebrate and vertebrate organisms, where food, refuge, and reproduction strategies take place (Benke & Wallace, 2003; Dolloff & Warren, 2003).

Riparian vegetation influences stream temperature, which affects community processes such as nutrient cycling and productivity and aquatic species metabolic rates, physiology, and life history strategies (Poole & Berman, 2001). Fluctuating stream temperature can cause behavioral and physiological changes in aquatic species and permanent temperature shifts can make streams inhabitable for native species (Poole & Berman, 2001). The physical structure of riparian vegetation acts as insulation from external drivers of stream temperature (i.e., solar radiation and wind), as well as serving to regulate stream temperature by affecting microclimatic conditions via biologic functions such as evapo-transpiration (Johnson, 2004). In shallow, low-order streams, shading is likely the most influential contribution of riparian vegetation in relation to stream temperature

(Johnson, 2004; Poole & Berman, 2001; Sinkrot & Stefan, 1993). Riparian plant community height, density, and distance to the stream influence shade, regulating solar inputs and thus stream temperature.

1.2 Determinants of Reach-Scale Vegetation Patterns

Riparian habitats host diverse plant assemblages adapted to the disturbance and stress characteristic of the fluvial system. Riparian vegetation continually adjusts to the effects of hydrologic processes, producing multiple states that persist from years to centuries (Corenblit et al., 2007; Naiman et al., 2005). Thus, floodplain ecology is typically described in terms of shifting patch mosaics, where the relationship of the hydrology and the patch (e.g., geomorphic position) influences biological communities. Reach-scale vegetation structure and composition are strongly influenced by the elevation of the patch in relation to the river. Ward, Tockner, Arscott, & Claret (2002) discussed the significance of riparian elevation at three scales: (1) longitudinal elevation from headwaters to the sea, (2) lateral elevation from stream center (e.g., thalweg) to uplands (e.g., riparian terraces), (3) lateral elevation in relation to topographic features (e.g., bars, islands, levees, swales). Patch position in relation to these scales influences the magnitude and frequency of hydrologic processes interacting with terrestrial ecosystems, thus affecting plant species distribution and spatial arrangement among patch types (Hupp & Osterkamp, 1996; Kalliola & Puhakka, 1988).

Riparian vegetation communities uniquely balance environmental stress and physical impacts of disturbance. Physical processes of transportation and deposition, sediment removal, plant submersion and destruction, and seed dispersal all affect reach scale vegetation dynamics (Corenblit et al., 2007). In this environment, stress and disturbance regulate the intensity of species competition. Patch proximity to the stream determines the level of impact to which vegetation must be adapted, in order to survive. Thus, riparian vegetation communities exhibit structure and composition that reflect the patch proximity to the stream (Bendix & Hupp, 2000). For example, Hupp (1982) demonstrated, in the humid, temperate climate of Passage Creek, Virginia, that riparian patch age (i.e., successional stage) was driven most by inundation frequency and degree to which the plants endured flood damage. Vegetation patches that flooded most frequently tended to host young, opportunistic species that use disturbance as a mechanism for colonization. These same patches also hosted mature vegetation tolerant of flooding, such as shrubby species characterized by stems resilient to flood-damage. On the Cedar River, Iowa, Kupfer & Malanson (1993) found that riparian vegetation communities in areas of high flood frequency were distinct from more upland assemblages of species. Dominant species in the most flood-prone areas were typically young colonizers not found in the forest interior. Rather than developing toward mature forest conditions, the riparian zone perpetuated early-successional patterns due to higher light levels, moisture availability, and competition-eliminating disturbance regime provided by more frequent and intense flooding.

Riparian terraces, the part of riparian zone most infrequently disturbed by flooding, may host species intolerant of damage and/or inundation. Hupp (1983) showed that on Passage Creek, Virginia, the more elevated floodplain species assemblages tended to be less tolerant of flood damage and more tolerant of periodic inundation. In low flood frequency zones, stages of succession may be governed most by ecological factors such as ageing and forest gap dynamics (Corenblit et al., 2007). On the Cedar River, Iowa, Kupfer & Malanson (1993) found that upland riparian vegetation communities were commonly associated with later stages of succession. Thus, in riparian areas the species competition that drives upland forest succession is mediated by varying degrees of fluvial disturbance.

In degraded hydrologic systems, altered flood regimes are common and have significant effects on riparian communities. Altered flood regimes most often are caused by the presence of a dam. Flow alteration commonly changes high and low flow levels. Thus, the variability associated with the natural flood regime is often diminished in an altered system (Poff et al., 1997). In riparian areas, these changes can resemble either more constant levels of inundation or a drastic range of human-controlled peak flows.

Altered disturbance (i.e., flood) regimes affect riparian vegetation assembly and succession patterns. For example, Cowell & Dyer (2002) studied vegetation and hydrology patterns on the Allegheny River, Virginia, at a wilderness area river reach with an upstream dam. Here, the human-created absence of flood events resulted in continual inundation of once flood-prone landforms. The change in environmental conditions caused a shift in riparian species composition, where early succession patches had not been initiated since construction of the dam. Sycamore and silver maple, typical pioneer species in this region, rely on greater light levels in flood-impacted zones for regeneration. In the absence of flooding, these two species established a mature, closed-canopy patch in which altered light availability prevented self-replacement by these species.

Cowell & Dyer (2002) also demonstrated that exotic species tolerant of both low light and inundation were replacing typical native early succession species on floodplains. It is widely recognized that there is correlation between disturbance regimes and the occurrence of invasive species (Richardson et al., 2007). In modified riparian systems, native vegetation life-history strategies can be compromised. Affected riparian areas can be susceptible to changes in vegetation assemblages or patterns of dominance caused by exotic species or environmental shifts (Richardson et al., 2007). As such, fluvial processes are critical determinants of riparian vegetation spatial arrangements and temporal trajectories.

1.3 Human Impacts: Stream Degradation

Globally, humans have altered many stream and river ecosystems. In riparian systems, natural disturbance is part of the overall system function where alteration is catalyst for ecosystem revitalization. Thus, riparian ecosystems are naturally dynamic. However, anthropogenic stress often exceeds the capacity of the fluvial system to recover from disturbance. This degradation is frequently persistent and compounding, thus causing changes in the fluvial system that are debilitating. In riparian plant communities, symptoms of degradation include reduced biodiversity, altered productivity, susceptibility to disease, reduced efficiency of nutrient cycling, and increased dominance of exotic and opportunistic species (Naiman et al., 2005).

Riparian area degradation is caused by many anthropogenic factors. Naiman et al. (2005) described four broad types of human-induced stress that affect riparian areas: flow regulation, pollution, climate change, and land use. Dam and levee construction, channelization, and water extraction are common regulators of stream flow. These affect the water table, flood regime, and aquatic and terrestrial ecosystem interactions. Polluted waterways have added toxic materials or nutrients that can increase or decrease riparian productivity, as well as alter community assemblages. Climate change is characterized by temperature and precipitation regime shifts. Resulting regionally specific environmental gradient changes are expected to change riparian communities. Land use change alters vegetative cover, thus changing the ecosystem dynamics of riparian areas.

Direct impacts to riparian areas are common. Riparian areas have traditionally been zones of intense human use, such as vegetation clearing, channelization, and livestock trampling. Removal of streamside vegetation affects ecosystem dynamics including habitat, diversity, water temperature, and structure (Johnson, 2004; Johnson & Jones, 2000; Jones, Helfman, Harper, & Bolstadt, 1999; Poole & Berman, 2001). Channelization affects bank stability and induces accelerated channel evolution which, in turn, affects patterns of vegetation development (Hupp, 1992). Wildlife and livestock also tend to congregate in riparian zones. This causes bank erosion, damage to vegetation, and adds pollution to streams (Rinne, 1988; Roath & Krueger, 1982; Sarr, 2002).

At the watershed scale, land use change is likely the most influential human impact that can cause riparian degradation. Land use change is the leading cause of habitat fragmentation and loss in fluvial and terrestrial ecosystems worldwide. In the United States, it is estimated that greater than 70 % of riparian forests have been removed (Palmer, Allen, Meyer, & Bernhardt, 2007; Wohl, Palmer, & Kondolf, 2008). Land use in the southeastern United States is characterized by intense agricultural development, with metropolitan areas that are currently some of the most quickly growing regions in the country (Sudduth, Meyer, & Bernhardt, 2007). As a result, greater than one-third of streams and rivers in the southeastern United States are listed as polluted or impaired (U.S. Environmental Protection Agency [USEPA], 2006).

Watershed land use strongly affects riparian ecosystem health. The Hubbard Brook Ecosystem Study, in New Hampshire, demonstrated the effects of forest cutting on fluvial ecosystems. Likens, Bormann, Johnson, Fischer, & Pierce (1970) analyzed a Hubbard Brook watershed after the forest was cut, the felled vegetation not removed, and two year herbicide treatment applied to prevent re-growth. They found a 39 % increase in annual runoff and significant increase of most major ions in stream water. Explanation for the exponential increase of dissolved nutrients in runoff was disruption of the nitrogen cycle, where nutrients are rapidly flushed from the ecosystem instead of being conserved by the forest. Similarly, researchers have shown that nutrient losses in agricultural catchments are consistently higher than in forested catchments (Johnson, Richards, Host, & Arthur, 1997; Omernik, Abernathy, & Male, 1981).

1.4 Human Impacts: Stream Restoration

Prevalence and practice

For over 30 years, the science and practice of stream restoration have been developing methods to improve degraded conditions in the physical fluvial environment and re-establish healthy fluvial ecosystem function. River channel management originated in control and utilization of the power of water for human benefit (e.g., dams, levees, and channelization; Downs & Gregory, 2004). Over time, development and use of stream channels and floodplains has disrupted hydrologic processes to a degree that now necessitates stream and river restoration for the preservation and conservation of water resources (Downs & Gregory, 2004). Primary focus of early stream restoration projects included pollution control and water quality improvement, fish and wildlife protection and habitat improvement, securing flow at dam sites, and faulty engineering work mitigation (Downs & Gregory, 2004). Stream restoration was founded in hydrologic and hydraulic engineering practice used for dam building, levee construction, and channelization, and thus initial rehabilitative trends were primarily static technological solutions (e.g., constructed of rock and concrete) that on many levels failed to strike balance with a natural river system's tendency to change over time (Downs & Gregory, 2004; Leopold, 1977).

In the United States, stream restoration has become a common freshwater management response to widespread altered and degraded conditions that characterize streams and rivers. In the past decade, much literature has cited a growing social awareness of waterway degradation and shifting of initiatives toward restoring biodiversity and ecosystem function (Bernhardt et al., 2005; Bernhardt et al., 2007; Palmer et al., 2007; Wohl et al., 2005). Social demand for ecosystem restoration, backed by significant political will and governmental funding, has grown a stream restoration industry (Bernhardt et al. 2007; Cunningham, 2002; Doyle et al. 2008; Palmer et al. 2007). Restoration burgeoning has also attracted significant interest from diverse fields of the scientific community (Bernhardt et al., 2007; Committee on Applied Fluvial Geomorphology [CAFG], 2004; Wohl et al., 2005). Combined, these interests have developed into the field of restoration ecology which produces research striving to develop and promote informed restoration practice.

The National River Restoration Science Synthesis (NRRSS) working group was formed in 2001 to assess the field of stream restoration from a multidisciplinary, scientific point-of-view (Bernhardt et al., 2005). The NRRSS conducted a survey of stream restoration practice that included close to 800 data sources and compiled information on approximately 37,000 stream restoration projects (Bernhardt et al., 2005). Despite being incomplete due to the numbers of local and non-profit projects that remain un-documented, the result of the NRRSS survey is the most effective existing synthesis of United States stream restoration statistics (Bernhardt et al., 2005; Lake, Bond, & Reich, 2007; Wohl et al., 2008). The NRRSS survey indicated exponential growth of stream restoration activity for all regions of the United States and that annual restoration expenditures in the United States exceeded one billion dollars (Bernhardt et al., 2005). According to Bernhardt et al. (2007), the only subjective field in the survey was project goals (i.e., motivations, intents, or purposes) which were divided into 13 categories: aesthetics/recreation/education, bank stabilization, channel reconfiguration, dam removal/retrofit, fish passage, floodplain reconnection, flow modification, in-stream habitat improvement, in-stream species management, land acquisition, riparian management, stormwater management, and water quality management. Projects were often were comprised of multiple goals, with riparian management, water quality management, and in-stream habitat improvement being the most common (Bernhardt et al., 2007; Palmer et al., 2007; U.S. Geological Survey Center for Biological Information [USGS CBI], 2006). Interestingly, riparian management was the most frequently cited project goal and ranked fourth among national project intent spending.

According to NRRSS data, restoration of ecological process and function, biodiversity, connectivity, or historic conditions were often stated by practitioners as objectives of restoration projects (Palmer et al., 2005). Riparian vegetation is vital to hydrologic ecosystem function, as well as a stream or river's function within its catchment (Lake et al., 2007). In particular, riparian vegetation links water quality, channel stability, biotic habitat and diversity, and aquatic ecosystem function to adjacent ecosystems. Although it was the primary project goal in only 8 % of NRRSS documented projects, riparian management was nearly ubiquitous as a component of restoration projects (Bernhardt et al., 2007). Re-vegetation of riparian areas via planting seedlings or live stakes was almost universally implemented in NRRSS projects to address ecosystem function and connectivity, as well as stream bank structure and stability (Bernhardt et al., 2007; USGS CBI, 2006).

Re-vegetation methods

Live stakes (e.g., cuttings of trees or shrubs) have become a common vegetative medium for ecological restoration projects. Certain species possess functional traits that enable them to root from planted clippings (Davy, 2002; Schiechtl & Stern, 1996). One advantage of this methodology is that large numbers of live stakes can be propagated rapidly in horticultural situations for use in restoration projects. As such, live stake cultivation and planting is economical compared to the cost of growing and planting trees and efficient compared the uncertainty of direct seeding germination.

Salix sericea and *Cornus amomum* are the most commonly planted species, in the form of live stakes, for restoration re-vegetation in northwestern North Carolina (Doll et al., 2003; North Carolina Forest Service [NCFS], 2008). Many species in the Salicaceae family and other specific species (i.e., *Cornus amomum* and *Physocarpus opulifolius* in western North Carolina) display functional traits making them well suited for use in stream restoration projects. These traits include the production of a large number of early-season, wind-dispersed seeds, high seedling growth rates, fast regeneration from broken stems, and dense root systems that serve to anchor the plants in alluvial material (Karrenberg, Edwards, & Kollmann, 2002). In highly dynamic systems, such as fluvial corridors, it has been acknowledged that restoring broad goals such as an ecosystem function or functional group presence is maybe more realistic and achievable than restoring endemic species or specific regional vegetative community types (Palmer, Ambrose, & Poff, 1997; Suding & Gross, 2006).

Goals and measures

The use of riparian restoration plantings is often based on an assumed link between re-vegetation of stream banks and restoration of biological and ecological function and process (Lake et al., 2007; Parkyn, Davies-Colley, Halliday, Costley, & Croker, 2003). As such, re-vegetation relies on a presumption that restoring physical conditions and processes will initiate ecosystem recovery capable of reversing or changing the trajectory of degraded riparian conditions (Jansson, Nilsson, & Malmqvist, 2007; Katz, Stromberg, & Denslow, 2009; Palmer et al., 1997). Currently, little data exist to support the assertion that riparian restoration re-establishes complex levels of historic ecosystem function and species diversity (Bernhardt et al., 2007; Jansson et al., 2007; Palmer et al., 2005; Parkyn et al., 2003; Wilkins, Keith, & Adam, 2003).

Understanding intact ecosystem function is requisite for successful ecological restoration (Hobbs, 2007; Lindenmayer et al., 2008). Targets for re-vegetation are commonly to return a degraded system to a pre-disturbance condition or historic state (Downs & Gregory, 2004; Hobbs, Higgs, & Harris, 2009; Palmer, Falk, & Zedler, 2006). In the United States, pre-disturbance is typically defined as a “natural” condition that existed before European settlement (Jackson & Hobbs, 2009). This definition, however, raises significant questions as to what conditions are appropriate restoration targets and whether achieving them is possible. It may be that streamside land use legacies, often characterized by intense, long-term development and deforestation, are more influential than restoration efforts (Katz et al., 2009; Lake et al., 2007). Alternatively, present riparian vegetation assembly may be adapted to a changing climate regime or an altered disturbance regime, where environmental conditions are very different from pre-settlement ecosystems (Hobbs et al., 2009; Katz et al., 2009; Poff et al., 1997; Seastedt, Hobbs, & Suding, 2008). Species composition and function could be completely transformed from historic conditions, having new combinations of species or different functional properties (Hobbs et al., 2009). Even ecosystems of the recent past may not be sustainable in the modern environment (Jackson & Hobbs, 2009). Thus, restoring to historic states is uncertain at best.

Existing, intact riparian areas, functioning within similar environmental gradients as candidate restoration sites, may be more effective and attainable guides for setting restoration trajectory goals. These reference conditions are indicators of current, region-specific target forest conditions where channel conditions and biological communities are more intact (Katz et al., 2009; Palmer et al., 2005). White & Walker (1997) described four sources of reference data: (1) current data from the proposed restoration site, (2) historical data from the proposed restoration site, (3) current data from reference sites, and (4) historical data from reference sites. Although obtaining each of these levels of reference data may not be possible, such comprehensive data collection has the most potential to reveal region-specific patterns of assembly, succession, and even how disturbance regimes are likely to influence the area.

Plant community structure and species composition are useful measures for reference data. Structure and composition are indicators of riparian vegetation assembly and succession. Rheinhardt et al. (2009) developed structure and composition data on 219 low-order forested reaches in the United States drainage basins of the Delaware River, Chesapeake Bay, and Albemarle/Pamlico Sound, to determine target states for riparian restoration. Reference site data aided these researchers in developing strategies for restoring degraded riparian areas. One conclusion was that presence of key species at degraded sites could affect the likelihood of restoration success or delineate restoration as less of a priority (Rheinhardt et al., 2009). For example, key species present at degraded sites could facilitate either sustained dominance of degraded conditions or unaided riparian forest recovery (Hobbs et al., 2009).

Structure and composition are also useful indicators of how re-vegetated restoration sites are maturing in comparison with reference sites (Harris, 1999; Rheinhardt et al., 2009). Katz et al. (2009) compared structural vegetation metrics and community composition at groundwater recovery restoration sites and reference sites on the lower San Pedro River, Arizona. After six years of data collection, they were able to discern that structure and composition were similar to reference conditions for one restoration site and different for another. On the Cumberland Plain, Sydney, Australia, vegetation structure and composition were compared among degraded, restored, and reference riparian stream reaches (Wilkins et al., 2003). Ordination of site type and species composition did not differentiate between restored sites and degraded sites, and showed restored site trajectory to be different than the composition of reference vegetation. Structurally, there was some evidence of increasing similarity between restored and reference vegetation. But overall, the results showed that 10 year old restored plant communities did not resemble naturally existing vegetation.

Current status of monitoring and assessment

Monitoring and assessment is widely recognized as critical to understanding recovery trajectories of restored stream sites and whether restoration practice is, in fact, achieving healthy, functional ecological outcomes (Bernhardt et al., 2007; Hobbs, 2005; Palmer &

Allan, 2006; Palmer et al., 2007). Modern scientific understanding of fluvial ecosystems is becoming more and more sophisticated, yet long-term databases of project-specific information are necessary for scientific evaluation of restoration outcomes. Insufficient assessment of restoration projects, both before and after completion, impedes our understanding of the short- and long-term ecosystem effects of stream restoration (Bernhardt et al., 2007; Palmer et al., 2007; Tullos, Penrose, & Jennings, 2009).

In relation to the numbers of projects being implemented, NRRSS project results revealed little existing long-term data telling of the effectiveness of restoration projects (Bernhardt et al. 2007; USGS CBI, 2006). In fact, only 10 % of NRRSS collected restoration project records cited any form of monitoring and projects that did indicate monitoring rarely included specific monitoring information (Bernhardt et al., 2005). The primary reasons surveyed restoration practitioners stated as cause for insufficient project monitoring were lack of time and funding. Bernhardt et al. (2007) argue that dearth of incentives and requirements for documenting restoration project outcomes are debilitating the ability to understand a restoration's ecological success or failure. For example, many North Carolina restoration projects are funded by the North Carolina Clean Water Management Trust Fund (CWMTF). The CWMTF does not specifically fund water quality monitoring, but water quality funding can be obtained in the granting process via matching contributions (CWMTF, 2009). Such matching contributions are allowed only if water quality improvements are part of the project intent and the funding match is necessary for completion of project objectives (CWMTF, 2009). No other restoration project monitoring objectives are specifically addressed in the CWMTF application guidelines.

According to NRRSS surveys, when stream restoration data were collected the definitions and objectives of monitoring were highly variable (Bernhardt et al., 2007). Bernhardt et al. (2007) cited *permit monitoring* (i.e., regionally specific requisites for state and federal project permit acquisition), *implementation monitoring* (i.e., evaluation of the functional effectiveness of structural or vegetation measures), and *outcome monitoring* (i.e., assessment of bigger picture project success in relation to overall project goals) as the most applicable descriptors of assessment practice. In NRRSS surveys, there was often little distinction as to whether assessments were directed toward permitting, project implementation, or documenting outcomes. Poorly defined objectives can result in projects that are not cost-effective restoration strategies. NRRSS surveys indicated that quantitative data were used to evaluate project success in 59 % of projects that did monitor. Further, 29 % of these projects did not use existing quantitative data to evaluate success and 47 % gauged the effectiveness of restoration only with qualitative assessment (Bernhardt et al., 2007).

1.5 Research Rationale

Stream restoration re-vegetation measures are often implemented under the assumption that ecological dynamics are being restored, without mandatory monitoring to document restoration outcomes. In North Carolina from 1993-2004, NRRSS statistics documented over 500 stream restoration projects (USGS CBI, 2006). However, many additional local and non-profit stream restoration projects have been implemented and are not included in the NRRSS restoration database (Bernhardt et al., 2007). Forty-eight percent of NRRSS database North Carolina projects included some form of riparian management

(USGS CBI, 2006). In North Carolina, NRRSS project records that include cost values totaled $272,228,057 for riparian area management initiatives. In fact, riparian area management ranked third among North Carolina stream restoration expenditures, after water quality management and land acquisition, indicating social and political recognition of the importance of riparian areas to overall stream ecosystem health and function (USGS CBI, 2006).

Despite re-vegetation being nearly universal in stream restoration projects that include riparian management measures, only 36 % of NRRSS surveyed practitioners in North Carolina report project monitoring or assessment (USGS CBI, 2006). This NRRSS monitoring statistic does not specify what restoration measures are being monitored by the practitioners that do conduct project assessments. Dearth of specific monitoring objectives and data collection results in an incomplete understanding of whether stream restoration measures are successfully fulfilling ecological goals. Specifically, little is known about the degree to which restoration re-vegetation measures are, in fact, promoting the ecological functions of intact riparian buffers.

This study examined the effects of stream restoration (including riparian woody plant re-vegetation) on nine low magnitude streams in the mountain region of northwestern North Carolina. These streams are headwaters for three southeastern United States watersheds (New River, Watauga River, and Nolichucky River) and present a valuable landscape for developing regional reference reach conditions, as much of the region is comparatively undeveloped. Specific research questions were: (1) Does stream restoration, and re-vegetation in particular, change degraded riparian conditions and (2) Is there indication of riparian re-vegetation changing riparian conditions to more closely resemble regional reference conditions? This study used a replicated, comparative sampling design to evaluate the effects of restoration on woody vegetation. Vegetation structure and composition were measured at reference, restored, and degraded stream sites on the same stream and were compared to infer change in site structure, species dominance, and composition.

CHAPTER 2

METHODS

2.1 Study Area

This study was conducted on nine North Carolina headwater streams in the Blue Ridge Province of the southern Appalachian Mountains. Three streams were located in Ashe County, four streams were located in Watauga County, and two streams were located in Avery County. Stream site topography is mountainous, with average elevation ranging from 858 m to 1146 m. All nine drainage areas are classified as low-order, and thus contain no tributaries of equal or greater size than the studied stream reaches.

Physical Region

The Blue Ridge Province of the Appalachian Mountains is over 965 km long, extending from southern Pennsylvania to northern Georgia (Patton, 2008). The North Carolina section of the Blue Ridge Province is its widest point (105 km) and is comprised of the Unaka Mountains, the Black Mountains, and the Blue Ridge Mountains. A distinguishing feature of the Blue Ridge Province is the prominent east-facing scarp, which attains maximum elevation of 1220 m close to Boone, North Carolina (Patton, 2008). The study streams are headwaters for the north and west draining aspects of the Blue Ridge Province. These streams contribute to the New, Gauley, and Kanawha drainage systems and the Nolichucky and Tennessee drainage systems. Both river systems flow into the Ohio River which feeds the Mississippi River.

Climate

Climate, vegetation, and soils vary throughout the Blue Ridge Mountains, with elevation as the primary driver. In Boone, North Carolina from 1971-2000, average July maximum temperature was 24.4 °C and average July minimum temperature was 15 °C (Southeast Regional Climate Center [SERCC], 2007). Average January maximum temperature was 8.9 °C and average January minimum temperature was -5 °C (SERCC, 2007). In comparison with other regions of the southeastern United States, temperatures for northwestern North Carolina are temperate in the summer and cold in the winter.

Northwestern North Carolina receives some of the highest levels of precipitation east of the Mississippi River (Carbone & Hidore, 2008). Average annual precipitation for Boone, North Carolina from 1971-2000 was 149.9 cm (SERCC, 2007). The Blue Ridge Mountains are positioned in the path of weather patterns originating from both the Atlantic Ocean and the Gulf of Mexico (Carbone & Hidore, 2008). The result is significant orographic lifting and precipitation.

Vegetation

The woody vegetation of the Blue Ridge Province is characterized by varying species assemblages that correspond to environmental gradients. At an elevation range of 858-1146 m, dominant forest community types in the study area are mixed hardwood assemblages (Carbone & Hidore, 2008). Wofford & Chester (2002) describe the potential woody vegetation of the study area as Appalachian oak forest, typical northern hardwood assemblage (*Acer-Betula-Fagus-Tsuga*), and spruce-fir assemblage (*Picea-Abies*) at highest elevations.

2.2 Study Design

Nine rural, headwater streams in the northwestern mountain region of North Carolina were chosen to assess the effects of stream restoration on riparian vegetation (Figure 1). Each stream was chosen based on the existence of a rural stream restoration project that included riparian plantings as part of stream restoration. Local restoration practitioners were contacted and asked to provide site information for stream restoration projects on low-order streams. Cooperating entities were ENV-Environmental Consulting Inc., Foggy Mountain Nursery, National Committee for the New River, and North Carolina Cooperative Extension Service (Watauga County Center; Table 1). Site information was gathered primarily via practitioner interviews, landowner interviews, and restored stream site visits. Approximately 15 separate projects were considered for inclusion in the study. Ten projects met the low-order stream classification criteria, were considered to exist in a comparable elevation gradient (i.e., one that would have similar vegetation composition) of 825-1150 m, and were accessible based on landowner permission. Nine streams were successfully sampled before the end of the 2008 growing season (Table 1).

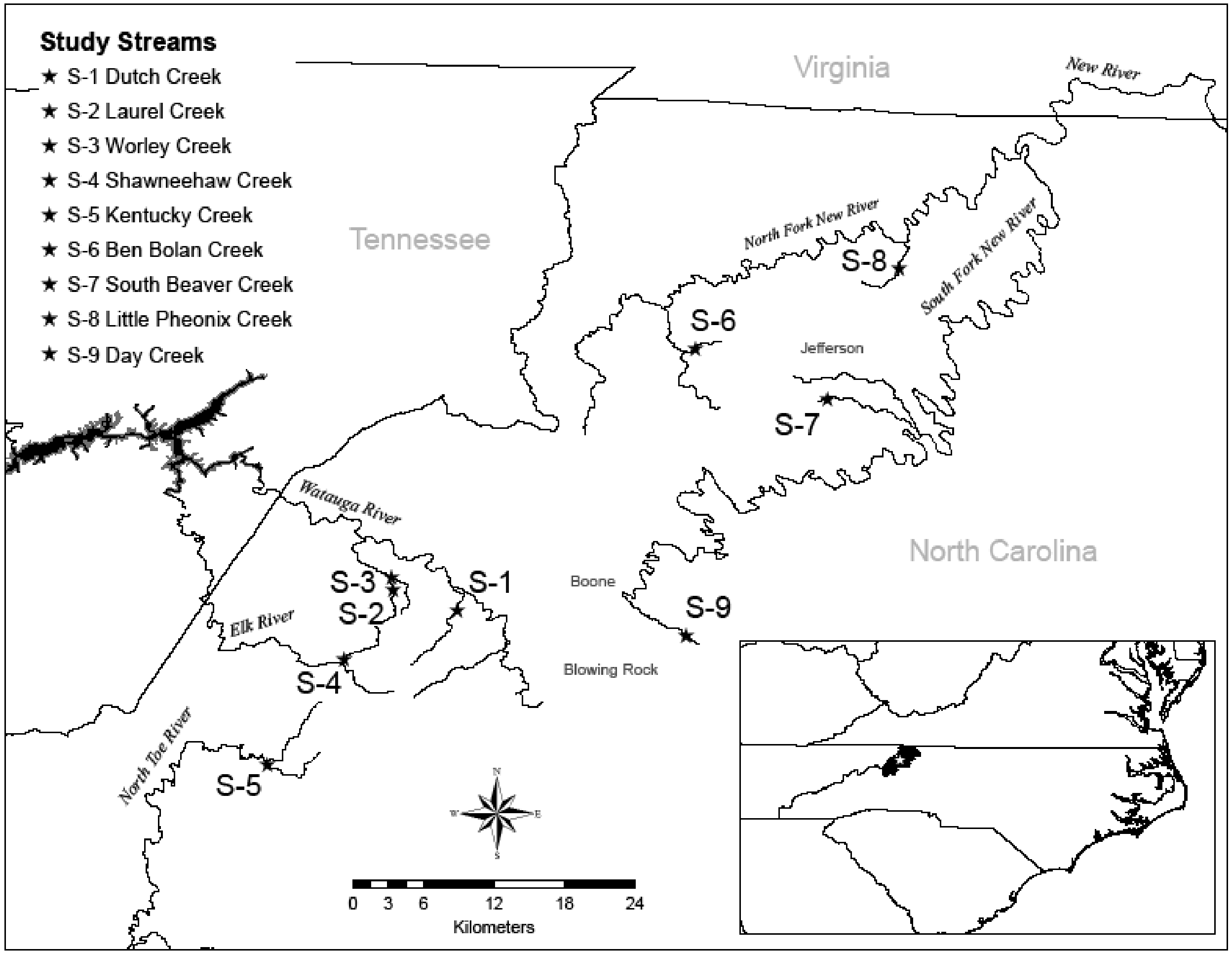


Figure 1. Study area map. Study area includes three counties in northwestern North Carolina: Ashe, Avery and Watauga. Stream sites were nine low-order headwater streams. A reference and degraded site was located on the same stream as each restored site, totaling 27 study sites.

Table 1. Restoration project information. Data are based on May, 2008 interviews with practitioners and agencies† involved with stream restoration projects, in the North Carolina counties of Ashe, Avery, and Watauga. Vegetation planting information provided as available.

Site	Date Restored	Engineer	Practitioner	Re-veg Methods	Basin	Reach Length (m)	Basin Area (ha)	Mean Elev. (m)
Dutch Creek	2001	NCSU	Shamrock	live stake, bare root trees	Watauga	457.2	74.2	858
Laurel Creek	2003	Buck Engineering	ENV	live stake, seedlings, seeded rye-millet cover, grass mats	Watauga	457.2	24.5	1045
Worley Creek	2003	Buck Engineering	ENV	live stakes, seedlings, seeded rye-millet cover, grass mats	Watauga	152.4	13.7	1049
Shawneehaw Creek	1999 2001	NCSU	North State	live stakes, bare root trees, container shrubs and trees	Watauga	426.7	31.9	1146
Kentucky Creek	2004	NCSU	North State	live stakes, transplants, containers, grass mats, brush mattresses	Nolichucky	243.8	45.7	1117
Ben Bolan Creek	2006	Foggy Mtn. Nursery	Foggy Mtn. Nursery	live stakes, bare root trees, shrubs	New	313.9	21.1	983
South Beaver Creek	2005	Foggy Mtn. Nursery	Foggy Mtn. Nursery	live stakes, bare root trees	New	329.2	7.9	1045
Little Pheonix Creek	2008	Foggy Mtn. Nursery	Foggy Mtn. Nursery	live stakes	New	152.4	23.6	859
Day Creek	2006	Foggy Mtn. Nursery	Foggy Mtn. Nursery	live stakes	New	62.2	9.0	1019

† ENV-Environmental Consulting Inc., 3764 Rominger Rd., Banner Elk, NC, 28604, (828) 297-6946.
Foggy Mountain Nursery, 2251 Ed Little Rd., Creston, NC, 28615, (336) 977-2958.
National Committee for the New River, P.O. Box 1480, West Jefferson, NC, 28694, (336) 982-6267.
NC Cooperative Extension Service, 971 W. King St., Boone NC, 28607, (828) 264-3061.

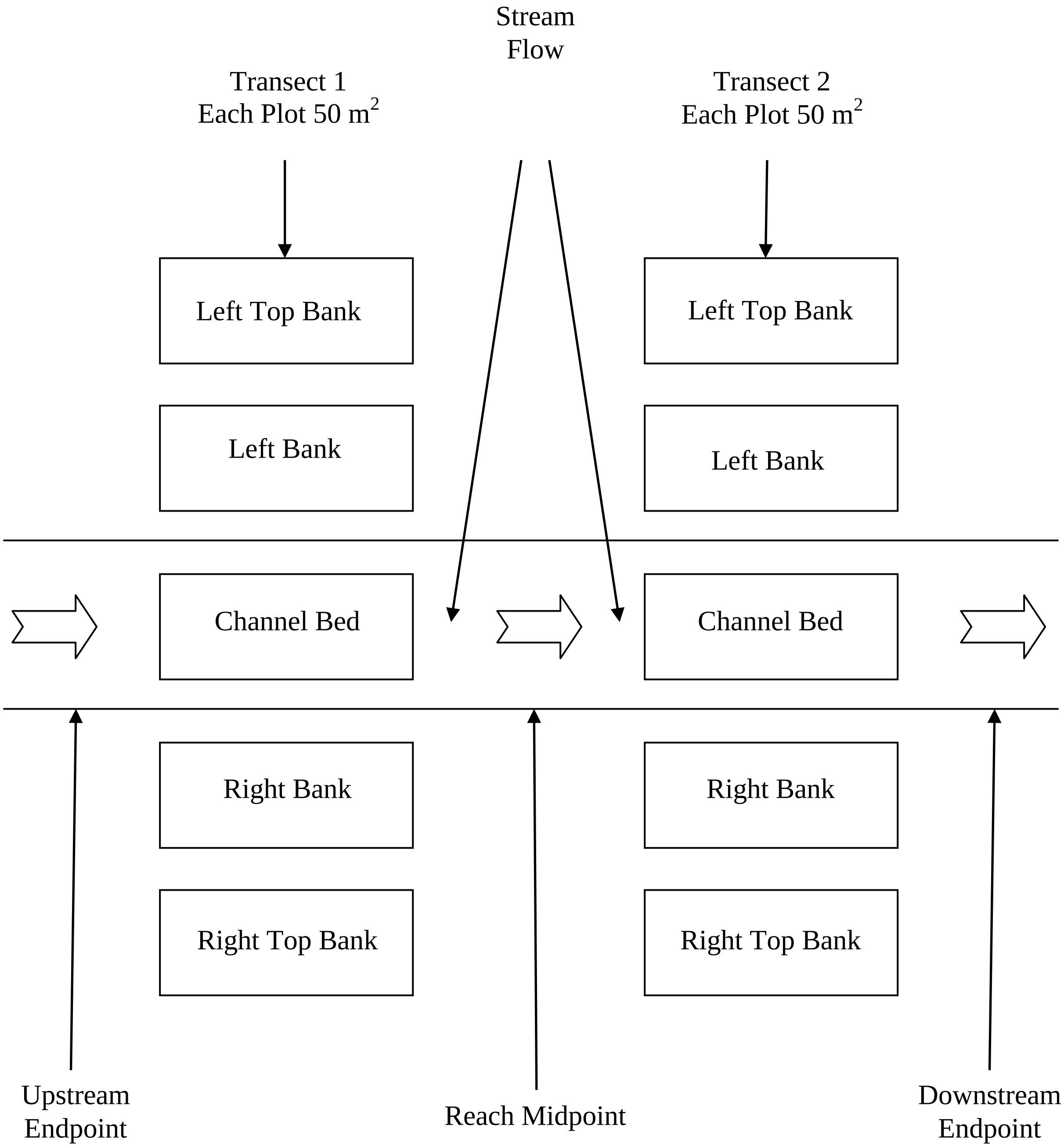
A comparative study was developed by matching the restored reaches with both degraded and reference low-order headwater reaches. Degraded and reference sites were identified and verified for access by pairing six inch aerial photos of Ashe and Watauga

Counties (N.C. Floodplain Mapping Program [NCFMP], 2005) with county tax parcel GIS layers (Ashe County GIS Department [AC GIS], 2007; Watauga County GIS Department [WC GIS], 2007) in a geographic information system (Arc GIS 9.2, 2007). No recent high resolution aerial photography was available for Avery County, and as a result sites were selected in the field. Degraded reaches were defined as rural, mostly remnant agricultural sections of streams that could merit some form of restoration based on current restoration practices and local practitioner assessment. Reference reaches were defined as stream sections where mature forested conditions currently existed on both sides of the stream. Degraded and reference sites were located on the same stream and were the same length as restored sites (Table 1). All sites were within an 825-1150 m elevation gradient in three northwestern North Carolina headwater watersheds (i.e., New River, Nolichucky River, and Watauga River; Table 1). A geographic information system (Arc GIS 9.2, 2007) was used to generate drainage basin areas (ha), using pre-processed, 2007 LiDAR data for Ashe, Avery, and Watauga Counties (N.C. Department of Transportation [NC DOT], 2008). The downstream endpoints of each of the 27 study sites were used as pour points for watershed calculations.

2.3 Sampling Design

All stream site coordinates were captured with a GPS, at an upstream endpoint, a midpoint, and a downstream endpoint. Degraded and reference site reach length was measured to match the restored site on the same stream. Sites ranged in length from 62.2 m (Day Creek) to 457.2 m (Dutch and Laurel Creeks). At all stream sites, two transects equidistant from the reach midpoint were established perpendicular to the channel, and spanning from top of the left bank to the top of the right bank (Figure 2). Channel width, from water's edge to water's edge, was measured at each transect. On each transect, five 50 m^2 vegetation sampling plots were placed in specific geomorphic positions: channel bed, left bank, top of left bank, right bank, top of right bank (Figure 2). The default plot shape was 5 m wide by 10 m long (50 m^2). Plot shape, however, was ultimately determined by the character of the plot's position. For example, if a particular section of bank was narrower than 5 m then a 2.5 m wide by 25 m long (50 m^2) plot was established. Thus, 10 plots on two transects of the stream were sampled at each of the 27 study sites. All 10 plot coordinates, at each site, were captured with a GPS.

Figure 2. Sampling design. Each study site was divided into two transects, consisting of 10 total plots. Each sampling plot measured 50 m^2, although length and width dimensions varied depending on the physical characteristics of the landform.



2.4 Vegetation Sampling

Woody vegetation data were recorded at the plot level. Field sampling methods for this study were based partly on the Carolina Vegetation Survey or North Carolina Ecosystem Enhancement Program Protocol for Recording Vegetation (Lee, Peet, Roberts, & Wentworth, 2006). Data collected were woody species identification, stem size, stem count, and canopy cover. Plant species were differentiated and samples collected in the field. Over 400 woody vegetation samples were pressed, dried, and stored at the Appalachian State University, I.W. Carpenter Jr. Herbarium. Samples were identified using Weakley (2008) and Wofford & Chester (2002). All woody stems present in each plot were measured at 10 cm above the ground and recorded as members of seven size classes in centimeters: 0-1 cm, 1-2.5 cm, 2.5-5 cm, 5-10 cm, 10-20 cm, 20-40 cm, and >40 cm. Canopy cover was measured at one meter above ground level using a concave spherical densiometer (Lemmon, 1956) at two, random points in each of the 10 plots per site. Percentage canopy cover for each species visible in the densiometer was recorded for both north and south directions at each canopy cover sampling point. Stream channel width in meters was also recorded at both transects of each site.

2.5 Analysis

Channel width

Channel width measurements were analyzed to discover existing differences in stream channel dimensions based on reference, restored, or degraded treatment. Channel width data were analyzed using SAS 9.1 (SAS Institute Incorporated, 2007). The two measurements recorded at each site were averaged and a mixed model Analysis of Variance

(ANOVA) was used to indicate difference in treatment. Channel width was the response variable, stream (i.e., each of the nine study streams) was used as the random factor, and site treatment type (i.e., reference, restored, degraded) was used as the fixed factor for analysis of the channel width metric. Pair-wise comparisons among site types were used to determine the relationship of one treatment to another. Differences were analyzed with 95 % confidence interval *t*-tests ($\alpha = 0.05$), using Tukey-Kramer adjustment.

Vegetation structure

Vegetation data were analyzed to indicate patterns in species richness, stem density, basal area, and canopy cover using SAS 9.1 (SAS Institute Incorporated, 2007). Due to study-wide absence of living woody vegetation in stream channels, the channel bed position was omitted from vegetation metric calculations. The number of woody species encountered in study plots was totaled by site and standard error calculated. Plot total basal area and stem density values were used to calculate site means and standard errors. Site percentage canopy cover values (excluding the channel bed position) and channel bed percentage canopy cover values (using only channel bed position) were averaged by measurement direction, sampling point, and plot to calculate site means and standard errors. The Shapiro-Wilk test was then used to test the metrics for departures from normality. Only basal area data were not normally distributed and, as a result, were LOG10-transformed to meet the assumptions of a normal distribution. Untransformed basal area data were presented in the results.

Vegetation metrics were compared using a two factor, mixed model ANOVA. The response variables were the site species richness totals and site stem density, basal area, site canopy cover, and channel bed canopy cover means. Here, stream was used as the random factor and site treatment type was used as the fixed factor for analysis of each vegetation metric. Pair-wise comparisons were used to determine similarity or difference between site type vegetation metrics. Differences between treatment pairs were analyzed with 95 % confidence interval *t*-tests ($\alpha = 0.05$), using Tukey-Kramer adjustment.

Dominance

Species importance values were calculated for the vegetation data, as measures of relative species dominance for the different riparian site type communities (Kuers, 2005). Plot level species importance values were calculated based on the relative basal area, relative stem density, and relative percentage canopy cover data. These plot level values were averaged for each species at each site, omitting the channel bed position. Primarily, sites were compared using the 10 and five most dominant species at reference, restored, and degraded site types.

Composition

Woody riparian vegetation composition was analyzed using non-metric multidimensional scaling (NMDS), based on Sorensen distance, using species importance values for the woody plant species found at the 27 stream study sites. Ordination analysis was conducted using PC-ORD version 5 software (McCune & Mefford, 1999). Following the methods of McCune & Grace (2002), random starting configuration and autopilot mode was used for ordination.

CHAPTER 3

RESULTS

3.1 Riparian Vegetation

Riparian environment

Riparian vegetation metrics of species richness, basal area, stem density, and canopy cover showed no effect for the stream variable but did indicate some treatment effects. There was a significant effect of site treatment (i.e., reference, restored, degraded) on species richness (*df* = 2.23, *F*-value = 4.25, *p*-value = 0.0269). Mean and SE of site type total species richness was 21 ± 1.7 at reference sites, 20 ± 3.3 at restored sites, and 11 ± 2.9 at degraded sites (Figure 3a). According to pair-wise tests, the only significant difference was reference site species richness being higher than degraded site species richness (Table 2). Restored site species richness was not significantly different from reference sites and marginally significantly different from degraded site (Table 2).

There was a significant effect of site treatment (i.e., reference, restored, degraded) on basal area (*df* = 2.23, *F*-value = 24.83, *p*-value <.0001). Mean and SE of site type basal area was 68.6 m^2/ha ± 7.6 at reference sites, 11.5 m^2/ha ± 3.5 at restored sites, and 5.5 m^2/ha ± 3.1 at degraded sites (Figure 3b). Pair-wise results for mean basal area showed reference site basal area to be significantly higher than that of both restored and degraded sites (Table 2). Restored and degraded sites were marginally significantly different (Table 2).

There was not a significant effect of site treatment (i.e., reference, restored, degraded) on stem density (*df* = 2.23, *F*-value = 0.65, *p*-value = 0.5312). Mean and SE of site type stem density was 23,875 stems/ha ± 3,454 at reference sites, 24,275 stems/ha ± 2,818 at restored sites, and 16,861 stems/ha ± 3687 at degraded sites (Figure 3c). Pair-wise test results showed mean stem density was not significantly different at reference, restored, and degraded sites (Table 2).

There was a significant effect of site treatment (i.e., reference, restored, degraded) on riparian canopy cover (*df* = 2.23, *F*-value = 25.63, *p*-value = <.0001) Mean and SE of site type canopy cover was 66.9 % ± 1.4 at reference sites, 30.8 % ± 3.6 at restored sites, and 19.5 % ± 3.0 at degraded sites (Figure 3d). Reference site percentage canopy cover was significantly different than that of both restored and degraded site types (Table 2). There was no significant difference between degraded and restored site type canopy cover (Table 2).

Channel environment

Stream channel structure metrics of channel width and channel bed canopy cover, showed no effect for the stream variable but did indicate treatment effects. There was a significant effect of site treatment (i.e., reference, restored, degraded) on channel width (*df* = 2.23, *F*-value = 8.87, *p*-value = 0.0014). Mean and SE of site type channel width was 5.3 m ± 0.8 at reference sites, 3.2 m ± 0.5 at restored sites, and 2.7 m ± 0.4 at degraded sites (Figure 4a). Pair-wise comparisons showed that reference site channel width was significantly different from both degraded and restored site types (Table 2). Degraded and restored site channel widths were not significantly different (Table 2).

There was a significant effect of site treatment (i.e., reference, restored, degraded) on channel bed canopy cover (*df* = 2.23, *F*-value = 9.89, *p*-value = 0.0008). Mean and SE of site type channel bed canopy cover was 64.7 % ± 3.5 at reference sites, 17.4 % ± 4.0 at restored sites, and 14.1 % ± 4.4 at degraded sites (Figure 4b). Pair-wise tests showed reference site channel bed canopy cover was significantly different from restored and degraded site type channel bed canopy cover (Table 2). There was no difference between restored and degraded sites (Table 2).

(a)

(b)

(c)

(d)

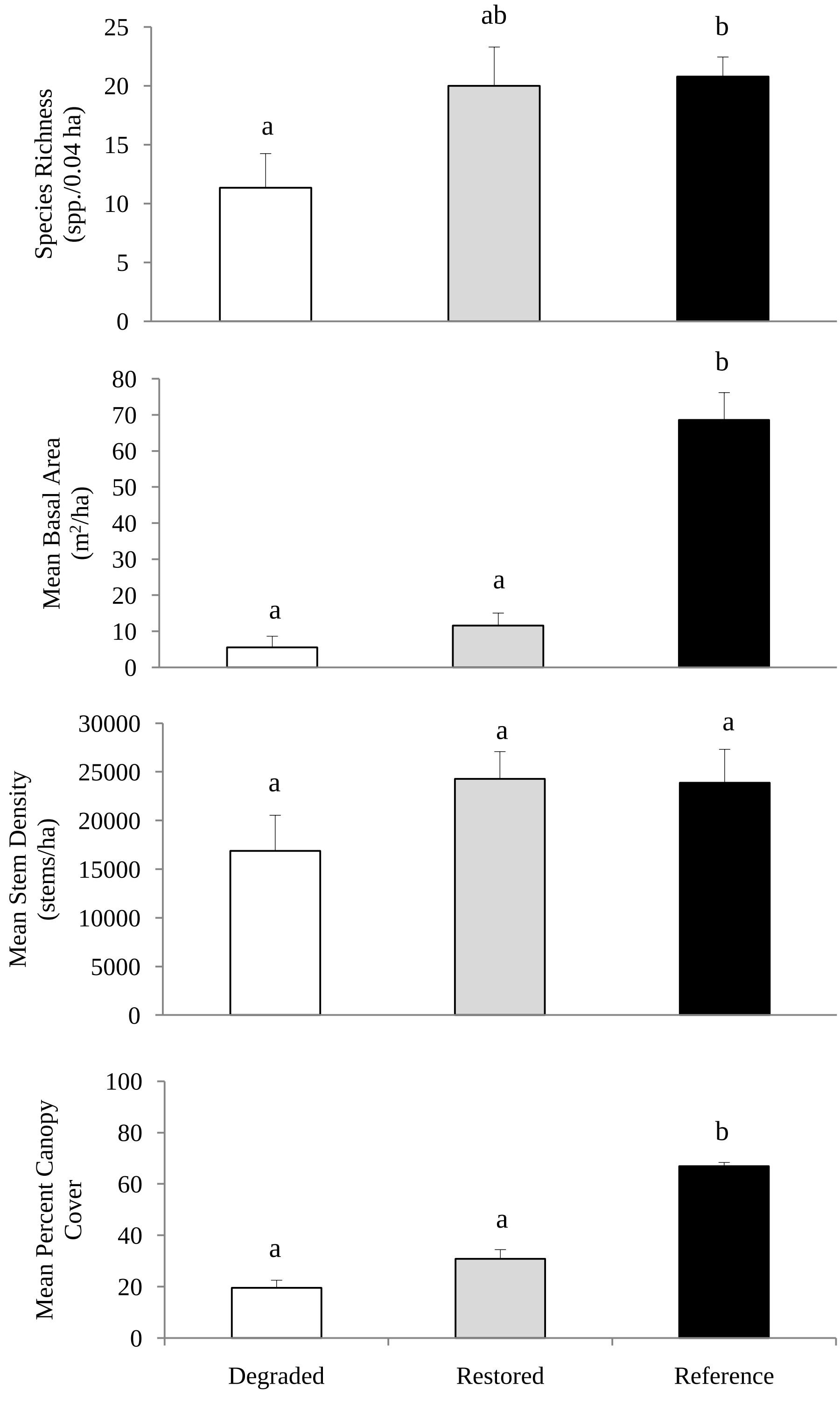


Figure 3. Woody riparian vegetation structure. Sampled area includes left top bank, left bank, right bank, and right top bank plot positions (0.04 ha). (a) Mean site type species richness. (b) Mean site type basal area. (c) Mean site type stem density. (d) Mean site type canopy cover. Bars with different superscripts are significantly different according to a mixed model ANOVA, experiment-wise α = 0.05. Random factor = stream, n = 9. Fixed factor = treatment type, n = 3 (with nine sites per type).

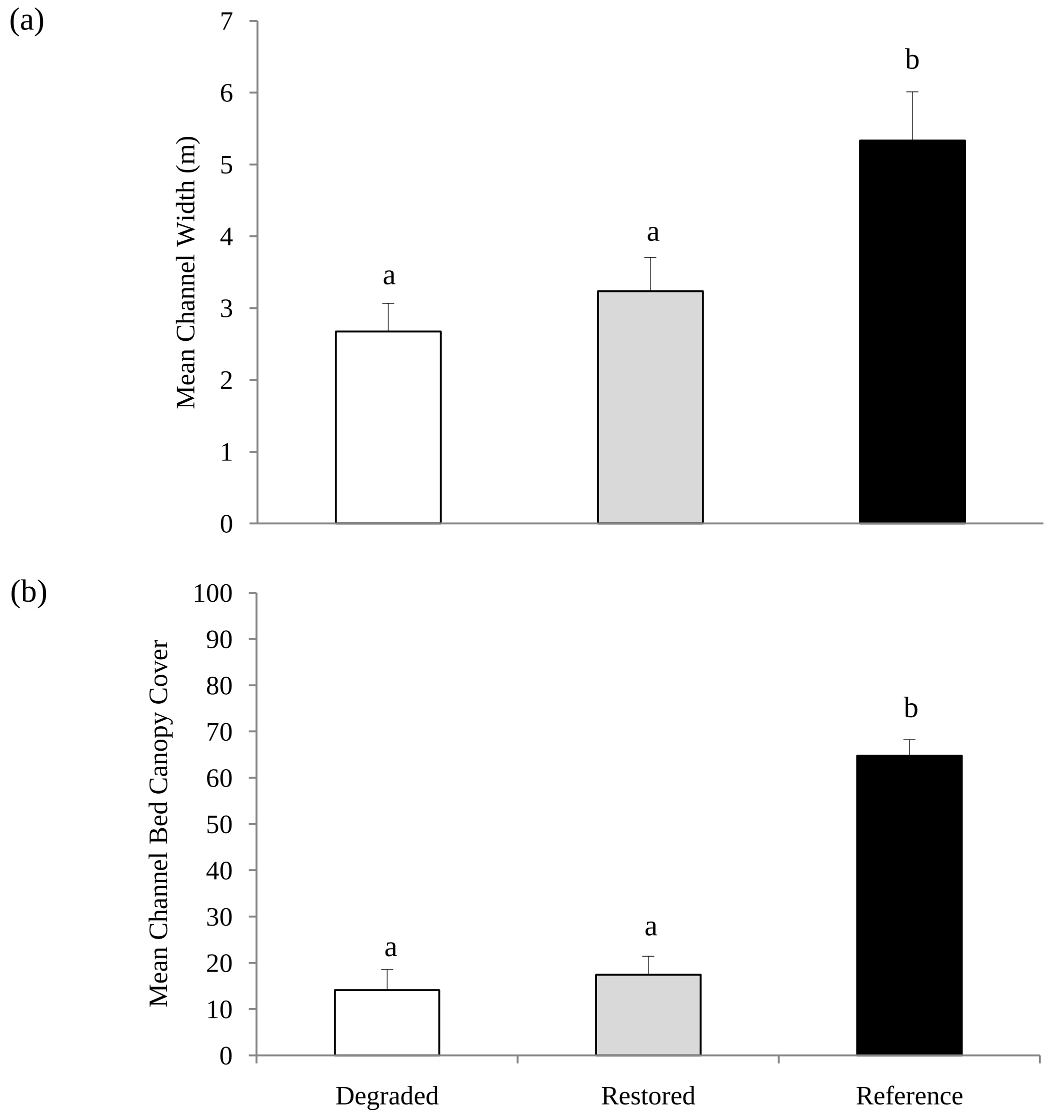


Figure 4. Stream channel structure. (a) Mean channel width. Values are averages of channel width in meters, at both transects per site. (b) Mean site type channel bed canopy cover. Sample area includes only the channel bed position (0.01 ha). Bars with different superscripts are significantly different according to a mixed model ANOVA, experiment-wise α = 0.05. Random factor = stream, *n* = 9. Fixed factor = treatment type, *n* = 3 (with nine sites per type).

Table 2. Vegetation metrics pair-wise tests. Mixed model ANOVA tests, where experiment-wise α = 0.05, were used to determine difference in vegetation metrics among three site types. Vegetation metrics used were species richness, basal area, stem density, canopy cover, channel width, and channel bed canopy cover. Site types were reference (N), restored (R), and degraded (D). Tukey-Kramer adjusted *p*-values were used to determine difference in site type metrics among pairs. Significantly different *p*-values are in bold font.

	Site Type	Site Type	Estimate	*SE*	*df*	*t*-value	Adjusted *p*-value
Riparian metrics							
Species richness (spp./0.04ha)	N	R	0.7778	3.6	23	0.22	0.9746
	N	D	-9.4444	3.6	23	-2.63	**0.0386**
	R	D	-8.6667	3.6	23	-2.41	0.0608
Basal area (m^2/ha)	N	R	0.8374	0.2	23	4.47	**0.0005**
	N	D	-1.3028	0.2	23	-6.95	**<.0001**
	R	D	-0.4654	0.2	23	-2.48	0.0522
Stem density (stems/ha)	N	R	-2.0000	36.6	23	-0.05	0.9984
	N	D	-35.0694	36.6	23	-0.96	0.6093
	R	D	-37.0694	36.6	23	-1.01	0.5758
Site canopy cover (%)	N	R	36.1422	6.9	23	5.22	**<.0001**
	N	D	-47.4213	6.9	23	-6.85	**<.0001**
	R	D	-11.2792	6.9	23	-1.63	0.2537
Channel metrics							
Channel width (m)	N	R	2.0978	0.7	23	3.15	**0.0119**
	N	D	-2.6561	0.7	23	-3.99	**0.0016**
	R	D	-0.5583	0.7	23	-0.84	0.6828
Channel bed canopy cover (%)	N	R	0.7645	0.2	23	3.33	**0.0080**
	N	D	-0.9701	0.2	23	-4.22	**0.0009**
	R	D	-0.2056	0.2	23	-0.89	0.6492

3.2 Species Composition

Overall, there were 90 woody plant species present at all site types (Appendix B). According to the USDA PLANTS database, 83 of the 90 total species were listed as native species (U.S. Department of Agriculture Natural Resources Conservation Service [USDA NRCS], 2009). The seven species listed as non-native were *Budleja davidii*, *Celastrus orbiculatus*, *Lespedeza cuneata*, *Malus pumila*, *Pyrus communis*, *Rosa multiflora*, and *Salix babylonica* (Appendix B). *Rosa multiflora* was present at all site types (Appendix C). The USDA PLANTS database describes *R. multiflora* as an invasive species for several states.

However, the states of closest relation to this study's region, North Carolina, Tennessee, and Virginia, do not list *R. multiflora* as invasive (USDA NRCS, 2009). Reference sites had 64 species present, with 63 listed as native and one listed as non-native (USDA NRCS, 2009; Appendix C). Restored sites had 65 species present, with 61 listed as native and four listed as non-native (USDA NRCS, 2009; Appendix C). Degraded sites had 46 species present, with 41 listed as native and five listed as non-native (USDA NRCS, 2009; Appendix C).

Species importance

Patterns of species dominance varied among site types. The dominant 10 species (and their average percentage importance values) at reference sites comprised a total importance value of 84.7 %. These were *Rhododendron maximum* (30.4 %), *Betula alleghaniensis* (22.4 %), *Acer rubrum* (12.6 %), *Liriodendron tulipifera* (4.8 %), *Fagus grandifolia* (3.5 %), *Hamamelis virginiana* (3.2 %), *Tsuga canadensis* (2.6 %), *Quercus rubra* (2.2 %), *Prunus serotina* (1.5 %), and *Betula lenta* (1.5 %) (Table 3; Appendix C). The 10 most dominant species (and their percentage importance values) at restored sites comprised a total importance value of 59.2 %. The 2 most dominant of the 10 dominant species at restored sites were *Salix sericea* (12.9 %) and *Cornus amomum* (10.3 %) (Table 3; Appendix C), which are the two most commonly planted species for riparian restoration in the North Carolina mountain region. The remaining 8 of the 10 most dominant species at restored sites were *A. rubrum* (7.4 %), *Clematis virginiana* (6.8 %), *Rubus argutus* (5.3 %), *B. alleghaniensis* (4.6 %), *R. multiflora* (3.6 %), *Aesculus flava* (3.3 %), *P. serotina* (2.7 %), and *Betula nigra* (2.3 %) (Table 3; Appendix C). The 10 most dominant species (and their percentage importance values) at degraded sites comprised a total importance value of 51.3

%. These were *R. argutus* (14.3 %), *B.alleghaniensis* (8.0 %), *S. sericea* (6.4 %), *C. virginiana* (5.2 %), *R. multiflora* (4.3 %), *Q. rubra* (3.6 %), *A. rubrum* (3.0 %), *A. flava* (2.6 %), *Salix babylonica* (2.3 %), and *Sambucus canadensis* (1.6 %) (Table 3; Appendix C).

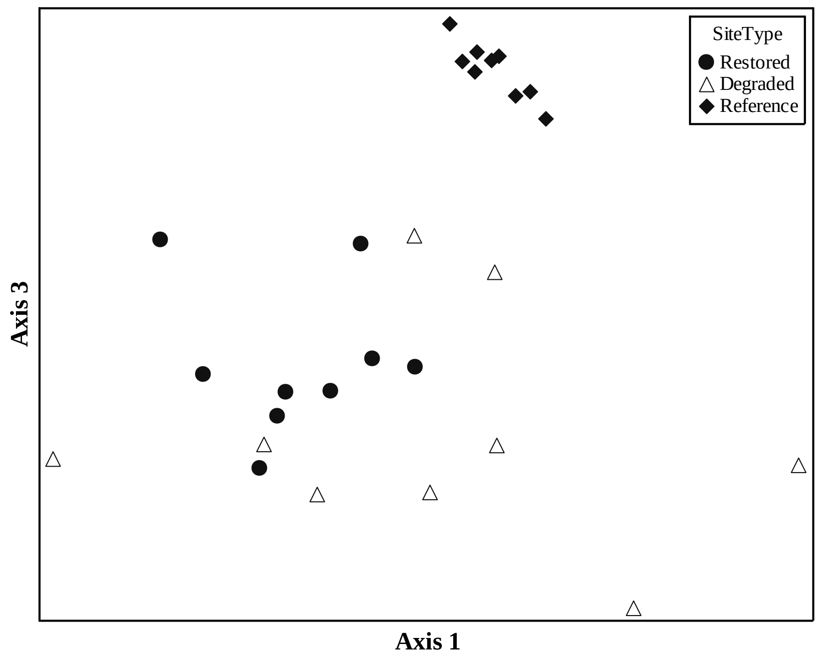
Table 3. Site type species dominance. Species listed are the 10 most dominant species per site type (i.e., reference, restored, and degraded). Scores are importance values for sites. Importance values are averages of relative stem density, basal area, and canopy cover metrics. Bold font indicates that a species is among the 10 most dominant of that site type.

Species name	Reference	Restored	Degraded
Rhododendron maximum	**30.4**	0.1	0.0
Betula alleghaniensis	**22.4**	**4.6**	**8.0**
Acer rubrum	**12.6**	**7.4**	**3.0**
Liriodendron tulipifera	**4.8**	2.0	1.0
Fagus grandifolia	**3.5**	0.0	1.3
Hamamelis virginiana	**3.2**	0.2	----
Tsuga canadensis	**2.6**	0.0	0.4
Quercus rubra	**2.2**	1.4	**3.6**
Prunus serotina	**1.5**	**2.7**	0.3
Betula lenta	**1.5**	0.4	----
Aesculus flava	1.3	**3.3**	**2.6**
Salix sericea	----	**12.9**	**6.4**
Cornus amomum	----	**10.3**	----
Rubus argutus	0.2	**5.3**	**14.3**
Clematis virginiana	0.0	**6.8**	**5.2**
Rosa multiflora	0.1	**3.6**	**4.3**
Betula nigra	0.0	**2.3**	----
Sambucus canadensis	0.0	0.9	**1.6**
Salix babylonica	----	----	**2.3**

Community composition

NMDS ordination of the 27 site-species importance combinations produced a three-dimensional solution, with final stress = 11.79, instability = 0.00030, and Monte Carlo Test p = 0.0196. The solution accounted for 81.7 % of the cumulative variability (Axis 1 = 28.5 %, Axis 2 = 21.9 %, and Axis 3 = 31.3 %). The ordination space shows reference site community composition to be a distinct group, with degraded and restored sites showing no clear organizational patterns (Figure 5). That is, reference sites group closely together while there is large variation in community composition at restored and degraded sites (Figure 5).

(a)



(b)

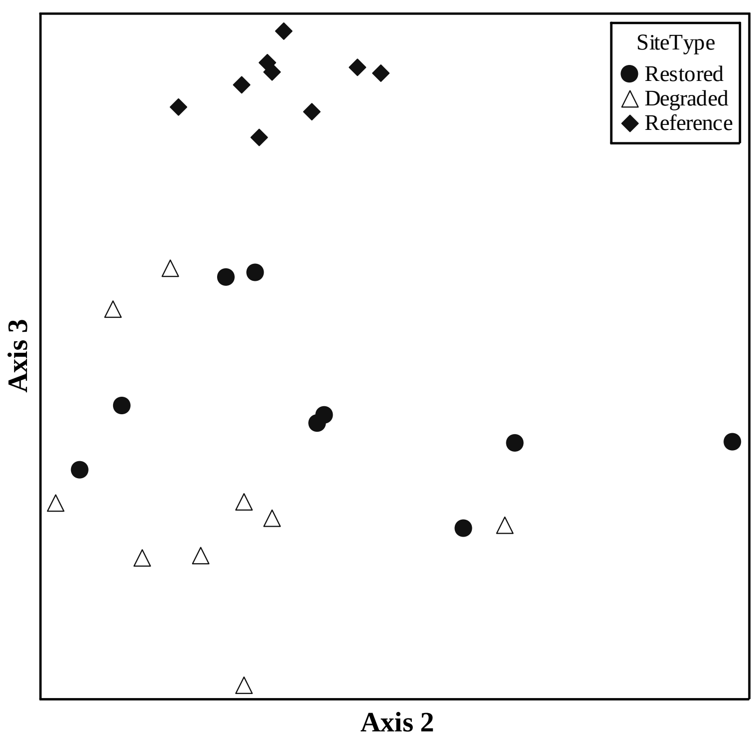


Figure 5. Community composition. These are the results of a three-dimensional NMDS ordination of all 27 stream site and type combinations in species space, based on frequency of occurrence of 90 woody species in study plots. (a) Axes 1 and 3. (b) Axes 2 and 3. Site types refer to study site treatment (i.e., reference, restored, and degraded).

CHAPTER 4

DISCUSSION

4.1 Target Conditions

Overall, reference sites were relatively distinct in terms of vegetation structure and composition. Reference sites were forested with an average of 21 species per site, mean basal area of 68.6 m^2/ha, and mean stem density of 23,875 stems/ha (Figure 3a-c). Reference site riparian area canopy cover was 67 % (Figure 3d). Mean site channel width was 5.3 m and the canopy cover of reference site channel bed was 64.8 % (Figure 4). Reference sites displayed a clear pattern of species composition. Ordination showed study-wide reference sites grouping closely, indicating similar species composition among reference sites (Figure 5). The 10 most dominant species at reference sites comprised 85 % importance of the total species (Appendix C), thus these 10 species were very influential in determining these regional reference conditions.

Rhododendron maximum dominated reference site species composition, with an importance value of 30.4 % (Table 3; Appendix C). *Rhododendron maximum* is an evergreen shrub or small tree. In the southeastern United States, *R. maximum* can grow 12 m tall and 7.5 m wide (Anderson, 2008). Multiple crooked stems grow from a single root crown (Anderson, 2008). These stems can reach 0.3 m in diameter and can live to be 100 years old (Anderson, 2008). Dominant presence of *R. maximum* is consistent with results from other Blue Ridge Mountain physiographic province research at Coweeta Hydrologic

Laboratory, North Carolina (Elliot, Boring, Swank, & Haines, 1997; Hedman & van Lear, 1995). *Rhododendron maximum* is a species that characterizes millions of acres of southern Appalachian forest community understory (Anderson, 2008), and is typical of southern Appalachian forests and riparian areas in particular. However, large-scale changes in forest structure, composition, and land use over the last century have increased *R. maximum* frequency (Monk, McGinty, & Day, 1985). The species commonly forms dense thickets that allow little light to reach the forest floor, which makes survival for other species difficult. For example, Baker & van Lear (1998) found that riparian area plots with high *R. maximum* stem density had an average of six species, compared to 26 species found on plots with lower stem density.

Reference site tree assembly was dominated by *B. alleghaniensis* and *A. rubrum* (Table 3; Appendix C). Secondary importance for reference site trees was shared by *L. tulipifera*, *F. grandifolia*, *T. canadensis*, *Q. rubra, P. serotina,* and *B. lenta* (Table 3; Appendix C). The overall assemblage pattern exhibited at these sites is typical of an early- to mid-successional stage of forest development for southern Appalachian riparian areas, at elevations up to 1800 m (Coladonato, 1991; Sullivan, 1994; Tirmenstein, 1991a).

Reference riparian sites displayed high importance of early-successional tree species. The reference riparian forests of this study likely began the currently existing stage of restructuring either after widespread early 20th Century logging or after periodic flood-induced disturbance of riparian areas. *Betula alleghaniensis* was the most dominant tree species at reference sites (Table 3; Appendix C). In southern Appalachian forests, *B. alleghaniensis* seedlings exhibited the highest growth rates of any species in disturbance associated gaps, but generally did not successfully compete with advance regeneration of other mixed hardwood species (Erdmann, 1990; White, MacKenzie, & Busing, 1985). The presence of *B. alleghaniensis* may be indicative of fluvial disturbance, as its seeds contain a water-soluble germination inhibitor (Sullivan, 1994). It is, therefore, an early-succession species that requires full light conditions to germinate and is commonly associated with forest gaps of all kinds (Sullivan, 1994). *Acer rubrum* is also a prolific species associated with many forest cover types, in many regions (Tirmenstein, 1991a). *Acer rubrum* produces large numbers of seeds, sprouts readily from the stump, and is moderately tolerant of shade, thus enabling it to compete with other species and successfully colonize a variety of disturbed sites (Walters & Yawney, 1990). At Coweeta Hydrologic Laboratory, North Carolina, increased importance of *A. rubrum* after clear-cutting was likely due to very high levels of seed production and sprouting (Elliot et al., 1997). *Acer rubrum* typically does not persist into late-successional stages of forest development, as it is often overtopped by faster growing species such *as Q. rubra* (Tirmenstein, 1991a).

Tree species of secondary importance at reference sites were *L. tulipifera*, *F. grandifolia*, *T. canadensis*, and *Q. rubra* (Table 3; Appendix C). *Liriodendron tulipifera* sprouts and grows more quickly than most other Appalachian forest tree species (Elliot et al., 1997). These traits enable *L. tulipifera* to attain early dominance in full light conditions (Griffith, 1991). Seventeen years after clear-cutting in a watershed at Coweeta Hydrologic Laboratory, North Carolina, *L. tulipifera* was the most dominant species in both cove hardwood and mixed oak hardwood forests (Elliot et al., 1997). *Liriodendron tulipifera* and *A. rubrum* often share co-dominance of early succession stands in the southern Appalachian region (Beck & Hooper, 1986; Elliot et al., 1997; Phillips & Shure, 1990). The presence of *Q. rubra*, a moderately shade-tolerant, mid-seral species (Tirmenstein, 1991b), *F. grandifolia* a climax species whose seedlings prefer a moderate shade and grow slowly under a hardwood canopy (Runkle, 1981), and *T. canadensis* a climax species that is very shade-tolerant and can survive years of light suppression (Carey, 1993) are all suggestive of possible mid- to late-successional patterns in this region (Table 3; Appendix C).

4.2 Restored Site Conditions Compared To Reference Site Conditions

Restored sites did have some similarity to reference conditions. Species richness and stem density were similar in restored and target conditions (Figures 3a & 3c; Table 2). Tree species dominance also bore some resemblance to target tree composition. *Acer rubrum* and *B. alleghaniensis* were the dominant tree species at reference sites, after the shrub *R. maximum* (Table 3; Appendix C). *Acer rubrum* and *B. alleghaniensis* were also influential at restored sites (Table 3; Appendix C). In southern Appalachia, numerous studies have documented *A. rubrum* and *B.alleghaniensis* establishing in early-succession forests after disturbance (Beck & Hooper, 1986; Elliot et al., 1997; Erdmann, 1990; Phillips & Shure, 1990; White et al., 1985). It is possible that the presence of these trees at restored sites is an indicator of early-succession, with some characteristics that resemble reference conditions. As a whole, however, restored sites were structurally and compositionally very different from reference conditions. Basal area and percentage canopy cover were much lower at restored site riparian areas than at reference sites (Figures 3b & 3d), indicating either a very different type of vegetation or stage of development.

Channel conditions also differed between reference and restored sites. At restored sites, channel width was smaller and channel bed canopy cover was lower than at reference sites (Figure 4; Table 2). This suggests that riparian function relevant to aquatic conditions differs between the two site types. Researchers have demonstrated the relationships between vegetative cover and channel morphology. McBride, Hession, & Rizzo (2008) documented channel widths and in-stream woody debris of two tributaries to the Sleepers River, northwestern Vermont, that were non-forested in 1966 and had re-established forest canopy by 2004. The results showed that reforested reaches were significantly wider than when measured in 1966. The three study reaches with the oldest forest cover had the widest stream profile and the non-forested reaches were significantly narrower.

The geomorphic processes responsible for wider stream channels in forested riparian areas have been the subject of debate. Important phenomena include scouring around in-stream large woody debris, debris dams, and tree-throw sites (McBride et al., 2008; Murgatroyd & Ternan, 1983; Zimmerman et al., 1967). In riparian areas where streamside vegetation has been removed, grasses may grow more readily. Grass cover can stabilize cobble bars with dense root networks and tends to retain sediment, which in turn can cause channel narrowing (Davies-Colley, 1997; Trimble, 1997). As such, it seems possible that wider stream channels in reference reaches of this research could be linked to greater channel bed and riparian canopy cover, which results from greater riparian tree height and density.

As well, restored site composition was different from reference site composition. Compared to the distinct compositional pattern at reference sites, restored site composition was highly variable (Figure 5). Restored sites were dominated by *S. sericea* and *C.amomum* (Table 3; Appendix C), both species commonly used for re-vegetation. These species are native to riparian areas of the North Carolina mountain region (Radford, Ahles, & Bell, 1964; Weakley, 2008; Wofford & Chester, 2002). However, *S. sericea* and *C. amomum* are not typical of reference forests (Table 3; Appendix C). Secondary importance at restored sites was primarily influenced by *C.virginiana*, *R. argutus*, and *R. multiflora*, which were not prevalent at reference sites (Table 3; Appendix C). These three species were also dominant species in degraded conditions (Table 3; Appendix C). These species can grow quickly into dense thickets of shrub vegetation, with numerous, slender, and flexible stems. At restored sites, the dominants *S. sericea* and *C. amomum* and co-dominants *C. virginiana*, *R. argutus*, and *R. multiflora* combine to form dense, many-stemmed, shrub and vine thickets that can reach a maximum height of approximately 4 m (USDA NRCS, 2009). The characteristics of the restored site species assemblage were very different from reference sites. According to the USDA NRCS (2009), woody riparian assemblages dominated by *B. alleghaniensis, A. rubrum, and F. grandifolia* typically have mature canopy heights of approximately 25 m with a *R. maximum* understory of approximately 8 m tall.

4.3 Restored Site Conditions Compared To Degraded Site Conditions

Degraded site woody riparian vegetation can be described as a sub-shrub thicket dominated by opportunistic species. Compared to reference and restored sites, degraded sites were characterized by woody species that have numerous stems, with low basal area, and low percentage canopy cover (Figure 3b-d). Interestingly, degraded and restored sites were structurally very similar. Species richness, basal area, stem density, and canopy cover were all similar between restored and degraded conditions (Figure 3; Table 2). Channel width and percentage channel bed canopy cover were also similar for restored and degraded conditions (Figure 4; Table 2).

As well, degraded site dominant species had noticeable resemblance to restored site dominant species. *Salix sericea* was dominant at both degraded and restored sites (Table 3;

Appendix C). Excluding restoration planted species (i.e., *S. sericea*, *C. amomum*), 6 of the 10 most dominant species at degraded sites were also among the 10 dominant species at restored sites (Table 3; Appendix C). *Clematis virginiana*, *R.multiflora*, and *R. argutus* were among the five most dominant species at degraded sites and among the seven most dominant species at restored sites (Table 3; Appendix C). These three species are typical early-succession species and are known to heavily colonize old fields and remnant agricultural sites, as well as after forest disturbance (Cain & Shelton, 2003; Munger, 2002; USDA NRCS, 2009). Although *R.multiflora* and *R. argutus* are primarily associated with more open upland habitat, *C. virginiana* is equally likely to occur in either uplands or wetlands (Appendix B). These species commonly grow together and each is considered invasive in some region of the United States (USDA NRCS, 2009), especially *R. multiflora* which is widely regarded as invasive or a noxious weed (Munger, 2002).

The fundamental difference between degraded and restored sites was the change in species dominance after restoration. At restored sites, *S. sericea* and *C. amomum* replaced degraded site assemblage of *C. virginiana*, *R. multiflora*, and *R. argutus* (Table 3; Appendix C). However, degraded conditions do seem to persist after restoration as degraded site dominant species retained influential secondary importance at restored sites (Table 3; Appendix C). As well, restored and degraded site species composition was highly variable. Ordination results showed some overlap in degraded and restored site composition, but an overall mixed pattern between the two treatments (Figure 5). Such variability, especially compared to reference site composition, suggests that many different assemblage patterns could define both degraded and restored site species composition.

4.4 Restoration Success

This study used reference conditions to define target conditions for riparian restoration. Based on our data, stream restoration projects did not yet resemble target conditions. At restored sites, basal area, canopy cover, channel width, and channel bed canopy cover were all very different from reference sites (Figures 3b & 3d; Figure 4; Table 2). Current composition and species dominance at restored sites were also very different than the composition and dominance of reference sites (Table 3; Figure 5; Appendix C).

It is possible that differences between reference and restored sites were due to the difference in vegetation age between the site types. Reference site assemblage suggests that these sites were likely no greater than 100 years old, while the average age of restored sites was only four years since restoration completion. Assuming reference site vegetation assembly is possible at restored sites, riparian forests will likely gain biomass with age (Brinson et al., 2006), and it is logical to expect basal area and canopy cover to develop as trees mature. Interestingly, several species that were compositionally important at reference sites were also present with moderate importance at restored sites. *Acer rubrum*, *B. alleghaniensis*, and *L. tulipifera* were the three most dominant trees at this study's regional riparian reference sites (Table 3; Appendix C). These species were also present with moderate importance at restored sites (Table 3; Appendix C). In the case of *A. rubrum* and *L. tulipifera*, species importance values were higher than at degraded sites (Table 3; Appendix C). Thus, there is indication that restored sites may possess structural and compositional traits that will succeed in becoming more similar to reference conditions in time.

Another factor influencing restoration success is the stability of the degraded condition. Restoration success depends on whether successional pathways have changed to support restored conditions. In contrast, restoration failure might result from an inability to permanently disrupt the feedbacks that perpetuate degraded site dominant vegetation communities. Multiple equilibrium ecological theory suggests that the environmental feedbacks that maintain an ecosystem in a degraded condition are likely very different than those supporting intact forest conditions (Suding & Gross, 2006). Degraded ecosystem dynamics can be resilient to restoration measures (Suding & Gross, 2006). For example, Prober, Thiele, & Lunt (2002) demonstrated a positive relationship between annual exotics in Australian grassy woodlands and soil nitrogen cycling. In this particular scenario, achieving restoration re-vegetation success required focus on nitrate-dependent chemical transitions between annual and perennial understory species. Strong positive feedbacks between vegetation that characterizes degraded sites and a site's environmental qualities can override restoration efforts.

Failure to disrupt feedbacks perpetuating degraded conditions can impede restoration efforts by causing successional pathways to return restored sites to degraded state patterns of structure and composition. The results of this study do indicate possible reassembly of degraded site species composition. *Rubus argutus, B. alleghanienis, S. sericea, C. virginiana*, and *R. multiflora* are the five most dominant species at degraded sites (Table 3; Appendix C). These 5 species are all in the 10 most dominant species at restored sites, and *S. sericea*, *C. virginiana*, and *R. argutus* are among the five most dominant species at restored sites (Table 3; Appendix C). The relatively young age of restoration projects makes it uncertain whether the successional trajectory of restoration projects supports restoration targets or is, in fact, regressing toward degraded conditions. Based on the species dominance results of this study, it seems very possible that restored sites could reestablish degraded vegetation conditions.

On the other hand, restored site structure and composition could be characterized as a new anthropogenic vegetation community type. Novel, human-created ecosystems are defined as ecosystems structurally and/or compositionally different from existing or historical environmental conditions (Hobbs et al., 2009). Such ecosystems may include non-native species, or native species occurring in different combinations and abundances from previously existing conditions (Hobbs et al., 2009; Lindenmayer et al., 2008). In particular, change in land use presents windows of opportunity for changes in vegetation assembly. In the case of this study, abandoned agricultural sites could potentially foster cultivars, remnant tree and shrub growth, weedy and exotic species, as well as early-successional woody species growth. Restoration re-vegetation mixes these possible plant combinations with species planted for specific function, thus creating significant opportunity for novel ecosystem establishment.

Our results strongly suggest that stream restoration re-vegetation has resulted in the creation of a novel ecosystem on headwater streams in northwestern North Carolina. Restored site basal area, canopy cover, and composition were very different from reference sites (Figures 3b & 3d; Table 2; Table 3; Figure 5; Appendix C). Restored site species composition also had important differences from degraded sites. Restored sites were dominated by *S. sericea* and *C. amomum* (Table 3; Appendix C), which are both species planted to promote riparian function. At degraded sites, *S. sericea* was present at half the importance of restored sites and *C. amomum* was absent altogether (Table 3; Appendix C). Degraded site dominant species, characterized by disturbance-adapted, opportunistic, and invasive traits, were of secondary importance at restored sites. The main difference between restored and degraded site dominant species is that restored site dominant species are planted to ensure stream bank stability. Although the structural vegetation data showed restored and degraded sites to be structurally similar (Figure 3; Table 2), restored site species dominance was distinct from degraded sites (Table 3; Appendix C). Thus, the change in dominance from weedy, early-successional species to species planted for restoration function may constitute a restoration success, where this community type persists as a stable novel ecosystem on low-order headwater riparian areas in northwestern North Carolina.

CHAPTER 5

CONCLUSION

Degraded riparian conditions are frequently the result of cleared vegetation. Stream bank erosion, habitat loss, altered nutrient levels, and exotic species invasion are some problems associated with loss of riparian vegetation. Stream restoration is commonly implemented to address degraded riparian conditions. In particular, re-vegetation of riparian areas is common practice attempting to restore ecological health and function. Re-vegetation is often performed under the assumption that restoration sets the stage for reestablishing specific ecological communities and ecological function, thus remedying issues associated with stream degradation.

Vegetation community restructuring theories have been developed to help guide ecological restoration practice. Munro, Fisher, Wood, & Lindenmayer (2009) discussed two theories that are commonly used to define the trajectory of ecosystem recovery after re-vegetation. First is the foster ecosystem hypothesis. Here, native forest regeneration is kindled by a nurse-plant function. Nurse-plants are restoration plantings that provide structure, which aids the colonization of regional understory vegetation. This theory assumes that understory species existing in nearby forest sites are able to migrate to the restored site via wind, water, or animal dispersion. In particular, re-vegetation structure should provide habitat for regional fauna, which, in turn, will facilitate seed importation to the site via feces and hitchhiking. Munro et al. (2009) also discussed the diversity-resistance theory. Two commonly used indicators of ecological health are native species diversity and presence of exotic species. The diversity-resistance theory hypothesizes that high native plant diversity will facilitate a site's ability to resist exotic species invasion. Verification of either of these theories would be valuable knowledge for the science and practice of ecological restoration.

However, there is currently a lack of information to lend validity to these theories. Based on the Munro et al. (2009) research, very few studies exist telling of whether the various ecological restoration re-vegetation measures are in fact achieving theorized ecological goals. In particular, they found very few studies on how re-vegetation planting structure develops over time and no studies pertaining to weed cover or exotic species richness in re-vegetated areas. Information about the development of restoration plantings is a valuable resource with potential to more efficiently direct restoration practice.

This study looked at stream restoration, in particular. I compared patterns of riparian vegetation structure, composition, and dominance at reference, restored, and degraded sites, on nine low-order headwater streams in the northwest North Carolina mountain region. Overall, reference conditions were very different than either restored or degraded riparian conditions. Restored and degraded conditions, however, had striking similarities and one fundamental difference. Possible explanations for these patterns are (1) the difference in reference and restored conditions is due to the relatively young age of restored sites compared to reference sites, (2) restoration has failed to establish stability and these riparian conditions may revert to a degraded state, or (3) restoration methods, and re-vegetation in particular, are responsible for creating new assemblages of vegetation in riparian areas that are best described as novel human-created ecosystems.

The results of this study demonstrate that re-vegetation monitoring has the potential to provide clearer understanding of the effects of restoration practice. This research provided a useful illustration of current reference, restored, and degraded vegetation patterns in riparian areas in northwestern North Carolina. Such knowledge helps develop further questions regarding restoration trajectories. Specifically, are restored sites becoming more like reference or degraded conditions, or are they unique? This research would be greatly strengthened by both pre-restoration and continuing post-restoration data collection. Monitoring is essential to understanding restoration outcomes. Monitoring of restoration projects both before project implementation and for years after project completion is requisite for developing best-management practice.

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APPENDIX A

Stream Site Vegetation Metrics

Appendix A. Stream site vegetation metrics. Results show the mean and SE of structural vegetation metrics at each site study-wide. 27 stream sites are grouped by stream, date of restoration implementation, and site type. Metrics are species richness, stem density, basal area, percentage canopy cover, percentage channel bed canopy cover, and channel width.

	Date Rest.	Type	Species Richness (spp./0.04ha)	SE	Stem Density (stems/ha)	SE	Basal Area (m^2/ha)	SE	Canopy Cover (%)	SE	Channel Bed Canopy Cover (%)	SE	Channel Width (m)	SE
Dutch Creek	2001	Restored	18	21.4	16500	6455.2	1.4	0.9	23.8	10.8	0.2	0.0	6.0	1.0
		Degraded	24	37.3	48375	23056.6	11.3	5.2	30.5	10.1	17.7	17.5	4.9	0.4
		Reference	30	15.8	28075	10794.4	66.1	15.7	56.6	4.7	80.7	10.2	8.9	0.6
Laurel Creek	2003	Restored	39	11.3	29375	8164.3	5.6	3.4	48.0	12.8	39.2	0.3	4.1	0.6
		Degraded	25	14.6	23350	8886.6	8.3	7.5	24.4	10.7	6.8	2.5	4.0	0.5
		Reference	18	44.0	31850	10326.7	50.2	18.2	67.5	3.9	48.9	11.5	4.8	0.1
Worley Creek	2003	Restored	22	53.4	45750	10527.4	27.7	18.4	39.9	12.4	33.8	12.5	2.5	0.1
		Degraded	7	81.8	35200	6690.1	0.9	0.2	1.9	1.2	0.2	0.0	2.9	0.2
		Reference	17	15.3	14375	1497.3	78.6	23.0	67.3	4.8	58.2	9.7	1.9	0.1
Shawneehaw Creek	2001	Restored	10	12.4	4750	2270.5	13.7	11.8	13.5	4.8	2.9	2.8	2.3	0.2
		Degraded	5	2.0	525	261.7	0.0	0.0	4.4	2.3	0.2	0.0	2.0	0.1
		Reference	15	9.6	7000	1815.8	109.3	30.2	69.4	4.8	63.6	15.8	7.0	0.7
Kentucky Creek	2004	Restored	17	31.3	33400	8196.1	10.0	4.5	49.4	12.3	15.4	9.8	4.5	0.4
		Degraded	0	0.0	0	0.0	0.0	0.0	14.3	7.0	17.0	16.8	2.0	0.1
		Reference	24	13.4	14950	3069.1	68.8	26.2	59.6	2.7	56.5	1.2	7.0	0.0
Ben Bolen Creek	2006	Restored	27	21.2	38875	10672.9	8.3	2.2	24.8	5.7	5.8	5.6	3.0	0.1
		Degraded	8	13.9	3875	1401.0	0.1	0.0	47.4	5.6	43.5	8.5	2.2	0.1
		Reference	22	15.7	27725	7061.0	29.2	15.9	68.9	2.7	78.8	6.7	3.9	0.4
South Beaver Creek	2004	Restored	26	15.7	23750	6003.1	30.0	22.2	37.0	12.3	24.4	16.6	1.5	0.4
		Degraded	11	26.3	10950	4332.9	1.5	0.9	0.5	0.3	0.2	0.0	1.0	0.1
		Reference	26	46.0	58875	22502.2	26.6	5.9	65.6	3.6	58.0	9.8	5.2	2.7
Little Phoenix Creek	2008	Restored	15	14.7	17650	7428.7	3.2	1.9	4.9	4.7	3.6	3.5	3.2	1.0
		Degraded	17	24.4	28800	14234.1	0.6	0.3	49.7	7.7	37.1	12.7	2.9	0.5
		Reference	18	13.7	12975	2421.2	74.0	23.3	70.8	4.2	61.4	6.4	4.9	0.4
Day Creek	2006	Restored	6	29.0	8425	3336.5	3.9	1.5	36.0	9.7	31.4	7.3	2.2	0.1
		Degraded	4	5.5	675	591.5	26.9	26.9	2.8	1.7	4.5	4.3	2.4	1.3
		Reference	17	33.0	19050	3849.6	114.5	25.4	76.9	2.1	76.8	7.2	4.5	0.3

APPENDIX B

<u>Woody Species List and Information</u>

Appendix B. Woody species list. Woody riparian species present (n = 90) at 27 first order steam sites in northwestern North Carolina. Information listed for each species is scientific name, common name, growth habit, USDA wetland indicator status (WIS), and USDA native status (NS).

	Scientific name	Common name	Growth habit	WIS †	NS ††
1	*Abies fraseri*	Frasier fir	Tree	FACU	N
2	*Acer pensylvanicum*	Striped maple	Tree/Shrub	FACU-	N
3	*Acer rubrum*	Red maple	Tree	FAC	N
4	*Acer saccharum*	Sugar maple	Tree/Shrub	FACU-	N
5	*Acer saccharinum*	Silver maple	Tree	FACW	N
6	*Aesculus flava*	Yellow buckeye	Tree/Shrub	NI	N
7	*Aesculus glabra*	Ohio buckeye	Tree	FACU	N
8	*Alnus serrulata*	Hazel alder	Tree/Shrub	FACW+	N
9	*Amelanchier canadensis*	Canada sarvis	Tree/Shrub	FAC	N
10	*Aristolochia macrophylla*	Pipevine	Vine	NI	N
11	*Aristolochia tomentosa*	Dutchman's pipe	Vine	FAC	N
12	*Betula alleghaniensis*	Yellow birch	Tree	FACU+	N
13	*Betula lenta*	Sweet birch	Tree	FACU	N
14	*Betula nigra*	River birch	Tree	FACW	N
15	*Budleja davidii*	Butterfly bush	Shrub	NI	I
16	*Carya glabra*	Pignut hickory	Tree	FACU	N
17	*Carya ovata*	Shagbark hickory	Tree	FACU	N
18	*Carya tomentosa*	Mockernut hickory	Tree	NI	N
19	*Castanea dentata*	American chestnut	Tree	NA	N
20	*Celastrus orbiculatus*	Oriental bittersweet	Vine	NI	I
21	*Clematis virginiana*	Clematis virginiana	Vine	FAC+	N
22	*Clethra acuminata*	Sweet-pepper bush	Tree/Shrub	NI	N
23	*Cornus amomum*	Silky dogwood	Shrub	FACW+	N
24	*Cornus florida*	Flowering dogwood	Tree/Shrub	FACU	N
25	*Crataegus macrosperma*	Bigfruit hawthorn	Tree/Shrub	NI	N
26	*Crataegus pruinosa*	Waxyfruit hawthorne	Tree/Shrub	NI	N
27	*Crataegus punctata*	Dotted hawthorn	Tree/Shrub	NI	N
28	*Carpinus caroliniana*	American hornbeam	Tree/Shrub	FAC	N
29	*Fagus grandifolia*	American beech	Tree	FACU	N
30	*Fraxinus americana*	American, white ash	Tree	FACU	N
31	*Fraxinus pennsylvanica*	Green ash	Tree	FACW	N
32	*Hamamelis virginiana*	American witchhazel	Tree/Shrub	FACU	N
33	*Hydrangea arborescens*	Wild hydrangea	Shrub	FACU	N
34	*Hydrangea cinerea*	Ashy hydrangea	Shrub	NI	N
35	*Ilex ambigua*	Carolina holly	Tree/Shrub	NI	N
36	*Juglans nigra*	Black walnut	Tree	FACU	N
37	*Kalmia latifolia*	Mountain laurel	Tree/Shrub	FACU	N
38	*Lespedeza cuneata*	Sericea lespedeza	Subshrub/Forb	NI	I
39	*Lindera benzoin*	Northern spicebush	Tree/Shrub	FACW	N
40	*Liriodendron tulipifera*	Tulip poplar	Tree	FAC	N
41	*Magnolia acuminata*	Cucumber tree	Tree	NI	N
42	*Magnolia fraseri*	Mountain magnolia	Tree	FAC	N

Appendix B. Woody species list (continued).

	Scientific name	Common name	Growth habit	WIS [†]	NS [††]
43	*Magnolia macrophylla*	Big-leaf magnolia	Tree	NI	N
44	*Magnolia tripetala*	Umbrella tree	Tree	FAC	N
45	*Malus pumila*	Paradise apple	Tree	NI	I
46	*Morus rubra*	Red mulberry	Shrub	FAC	N
47	*Nyssa aquatica*	Water tupelo	Tree	OBL	N
48	*Nyssa sylvatica*	Blackgum	Tree	FAC	N
49	*Ostrya virginiana*	Hophornbeam	Tree/Shrub	FACU-	N
50	*Parthenocissus quinquefolia*	Virginia creeper	Vine	FAC	N
51	*Physocarpus opulifolius*	Common ninebark	Shrub	FAC-	N
52	*Pinus strobus*	White pine	Tree	FACU	N
53	*Platanus occidentalis*	Sycamore	Tree	FACW-	N
54	*Prunus pensylvanica*	Pin cherry	Tree	FACU	N
55	*Prunus serotina*	Black cherry	Tree	FACU	N
56	*Pyrus communis*	Common pear	Tree	NI	I
57	*Quercus alba*	White oak	Tree	FACU	N
58	*Quercus coccinea*	Scarlet oak	Tree	NI	N
59	*Quercus rubra*	Red oak	Tree	FACU	N
60	*Rhododendron calendulaceum*	Flame azalea	Shrub	NI	N
61	*Rhododendron catawbiense*	Catawba rosebay	Tree/Shrub	NI	N
62	*Rhododendron maximum*	Rosebay, great laurel	Tree/Shrub	FAC	N
63	*Rhus glabra*	Smooth sumac	Tree/Shrub	NI	N
64	*Robinia pseudoacacia*	Black locust	Tree	UPL	N
65	*Rosa multiflora*	Multiflora rose	Vine/Subshrub	UPL	I
66	*Rosa palustris*	Swamp rose	Subshrub	OBL	N
67	*Rubus argutus*	Southern blackberry	Subshrub	FACU+	N
68	*Rubus canadensis*	Smooth blackberry	Subshrub	NI	N
69	*Rubus occidentalis*	Black raspberry	Subshrub	NI	N
70	*Rubus odoratus*	Flowering raspberry	Subshrub	NI	N
71	*Salix babylonica*	Weeping willow	Tree	FACW	I
72	*Salix nigra*	Black willow	Tree	OBL	N
73	*Salix sericea*	Silky willow	Tree/Shrub	OBL	N
74	*Sambucus canadensis*	Common elderberry	Tree/Shrub	FACW-	N
75	*Sassafras albidum*	Sassafras	Tree/Shrub	FACU	N
76	*Smilax rotundifolia*	Roundleaf greenbrier	Shrub/Vine	FAC	N
77	*Spiraea alba*	Meadowsweet	Shrub	FACW+	N
78	*Spiraea japonica*	Japanese meadowsweet	Shrub	FACU+	N
79	*Spiraea virginiana*	Appalachian meadowsweet	Shrub	FACW	N
80	*Tilia americana*	American basswood	Tree	FACU	N
81	*Toxicodendron radicans*	Poison ivy	Shrub/Subshrub/Vine	FAC	N
82	*Tsuga canadensis*	Eastern, Canada hemlock	Tree	FACU	N
83	*Tsuga caroliniana*	Carolina hemlock	Tree	NA	N
84	*Ulmus americana*	American elm	Tree	FACW	N
85	*Vaccinium corymbosum*	Highbush blueberry	Shrub	FACW	N

Appendix B. Woody species list (continued).

	Scientific name	Common name	Growth habit	WIS [†]	NS [††]
86	*Vaccinium pallidum*	Blue Ridge blueberry	Shrub	NI	N
87	*Viburnum dentatum*	Southern arrowwood	Tree/Shrub	FAC	N
88	*Viburnum prunifolium*	Blackhaw	Tree/Shrub	FACU	N
89	*Vitis labrusca*	Fox grape	Vine	FAC+	N
90	*Vitis rotundifolia*	Muscadine	Vine	FAC	N

[†] **Indicator code**

OBL = Obligate Wetland. Occurs almost always (estimated probability 99%) under natural conditions in wetlands.

FACW = Facultative Wetland. Usually occurs in wetlands (estimated probability 67%-99%), but occasionally found in non-wetlands.

FAC = Facultative. Equally likely to occur in wetlands or non-wetlands (estimated probability 34%-66%).

FACU = Facultative Upland. Usually occurs in non-wetlands (estimated probability 67%-99%), but occasionally found on wetlands (estimated probability 1%-33%).

UPL = Upland. Occurs in wetlands in another region, but occurs almost always (estimated probability 99%) under natural conditions in non-wetlands in the regions specified. If a species does not occur in wetlands in any region, it is not on the National List.

NA = No agreement. The regional panel was not able to reach a unanimous decision on this species.

NI = No indicator. Insufficient information was available to determine an indicator status.

NO = No occurrence. The species does not occur in that region.

[††] **Native status**

N = native

I = introduced/exotic

APPENDIX C

Site Type Species Importance Values

Appendix C. Species importance scores. Values are species importance scores for degraded, restored, and reference sites. Values are sorted in descending order by site type and are averages of relative stem density, basal area, and canopy cover metrics.

Species Importance Values (IV) Degraded site flora (n = 46)	IV	Restored site flora (n = 65)	IV	Reference site flora (n = 64)	IV
Rubus argutus	14.3	*Salix sericea*	12.9	*Rhododendron maximum*	30.4
Betula alleghaniensis	8.0	*Cornus amomum*	10.3	*Betula alleghaniensis*	22.4
Salix sericea	6.4	*Acer rubrum*	7.4	*Acer rubrum*	12.6
Clematis virginiana	5.2	*Clematis virginiana*	6.8	*Liriodendron tulipifera*	4.8
Rosa multiflora	4.3	*Rubus argutus*	5.3	*Fagus grandifolia*	3.5
Quercus rubra	3.6	*Betula alleghaniensis*	4.6	*Hamamelis virginiana*	3.2
Acer rubrum	3.0	*Rosa multiflora*	3.6	*Tsuga canadensis*	2.6
Aesculus flava	2.6	*Aesculus flava*	3.3	*Quercus rubra*	2.2
Salix babylonica	2.3	*Prunus serotina*	2.7	*Prunus serotina*	1.5
Sambucus canadensis	1.6	*Betula nigra*	2.3	*Betula lenta*	1.5
Toxicodendron radicans	1.5	*Robinia pseudoacacia*	2.0	*Aesculus flava*	1.3
Fagus grandifolia	1.3	*Liriodendron tulipifera*	2.0	*Kalmia latifolia*	1.3
Rhododendron catawbiense	1.2	*Malus pumila*	1.5	*Tilia americana*	1.3
Robinia pseudoacacia	1.1	*Quercus rubra*	1.4	*Acer saccharum*	1.1
Liriodendron tulipifera	1.0	*Parthenocissus quinquefolia*	1.3	*Magnolia fraseri*	1.1
Cornus florida	0.9	*Ostrya virginiana*	1.0	*Tsuga caroliniana*	0.9
Celastrus orbiculatus	0.8	*Platanus occidentalis*	1.0	*Smilax rotundifolia*	0.8
Fraxinus americana	0.7	*Rhus glabra*	1.0	*Carya ovata*	0.7
Rosa palustris	0.6	*Acer saccharum*	0.9	*Ulmus americana*	0.5
Tsuga canadensis	0.4	*Pinus strobus*	0.9	*Parthenocissus quinquefolia*	0.5
Vitis labrusca	0.4	*Viburnum dentatum*	0.9	*Juglans nigra*	0.5
Prunus serotina	0.3	*Abies fraseri*	0.9	*Robinia pseudoacacia*	0.4
Physocarpus opulifolius	0.2	*Sambucus canadensis*	0.9	*Crataegus punctata*	0.4
Crataegus punctata	0.2	*Rosa palustris*	0.7	*Carya tomentosa*	0.3
Lespedeza cuneata	0.2	*Physocarpus opulifolius*	0.5	*Fraxinus americana*	0.3
Aesculus glabra	0.2	*Rubus odoratus*	0.5	*Magnolia acuminata*	0.3
Parthenocissus quinquefolia	0.2	*Fraxinus americana*	0.4	*Toxicodendron radicans*	0.3
Smilax rotundifolia	0.1	*Nyssa sylvatica*	0.4	*Crataegus macrosperma*	0.3
Ostrya virginiana	0.1	*Alnus serrulata*	0.4	*Carya glabra*	0.2
Rubus canadensis	0.1	*Betula lenta*	0.4	*Acer saccharinum*	0.2
Carya tomentosa	0.1	*Spiraea japonica*	0.3	*Quercus alba*	0.2
Fraxinus pennsylvanica	0.1	*Magnolia acuminata*	0.2	*Vitis rotundifolia*	0.2
Pyrus communis	0.0	*Pyrus communis*	0.2	*Rubus argutus*	0.2
Rhododendron maximum	0.0	*Morus rubra*	0.2	*Pinus strobus*	0.2
Quercus alba	0.0	*Hamamelis virginiana*	0.2	*Acer pensylvanicum*	0.2
Rubus occidentalis	0.0	*Toxicodendron radicans*	0.2	*Lindera benzoin*	0.2
Morus rubra	0.0	*Rubus occidentalis*	0.2	*Rubus odoratus*	0.1
Pinus strobus	0.0	*Salix nigra*	0.1	*Aristolochia macrophylla*	0.1

Appendix C. Species importance scores (continued).

Species Importance Values (IV)					
Degraded site flora (n = 46)	IV	Restored site flora (n = 65)	IV	Reference site flora (n = 64)	IV
Prunus pensylvanica	0.0	*Carya tomentosa*	0.1	*Rosa multiflora*	0.1
Sassafras albidum	0.0	*Rhododendron maximum*	0.1	*Ostrya virginiana*	0.1
Carya ovata	0.0	*Acer saccharinum*	0.1	*Spiraea virginiana*	0.1
Quercus coccinea	0.0	*Vaccinium corymbosum*	0.1	*Amelanchier canadensis*	0.1
Platanus occidentalis	0.0	*Rhododendron catawbiense*	0.1	*Crataegus punctata*	0.1
Spiraea japonica	0.0	*Spiraea alba*	0.1	*Viburnum dentatum*	0.1
Juglans nigra	0.0	*Smilax rotundifolia*	0.1	*Fraxinus pennsylvanica*	0.1
		Vitis labrusca	0.1	*Clethra acuminata*	0.1
		Crataegus macrosperma	0.1	*Sassafras albidum*	0.1
		Hydrangea arborescens	0.0	*Spiraea japonica*	0.0
		Fagus grandifolia	0.0	*Carpinus caroliniana*	0.0
		Hydrangea cinerea	0.0	*Vitis labrusca*	0.0
		Lindera benzoin	0.0	*Clematis virginiana*	0.0
		Sassafras albidum	0.0	*Magnolia macrophylla*	0.0
		Castanea dentata	0.0	*Vaccinium pallidum*	0.0
		Budleja davidii	0.0	*Vaccinium corymbosum*	0.0
		Quercus coccinea	0.0	*Ilex ambigua*	0.0
		Tsuga canadensis	0.0	*Betula nigra*	0.0
		Rubus canadensis	0.0	*Sambucus canadensis*	0.0
		Viburnum prunifolium	0.0	*Physocarpus opulifolius*	0.0
		Aristolochia tomentosa	0.0	*Cornus florida*	0.0
		Carya ovata	0.0	*Platanus occidentalis*	0.0
		Crataegus punctata	0.0	*Rhododendron calendulaceum*	0.0
		Nyssa aquatica	0.0	*Hydrangea arborescens*	0.0
		Magnolia tripetala	0.0	*Rhus glabra*	0.0
		Quercus alba	0.0		

APPENDIX D

Site Species Presence

Appendix D. Site species presence. Species are sorted by family. X indicates species presence at that site number. Site numbers represent 27 study sites: 1=Dutch Creek-Restored, 2=Dutch Creek-Degraded, 3=Dutch Creek-Reference, 4=Laurel Creek-Restored, 5=Laurel Creek-Degraded, 6=Laurel Creek-Reference, 7=Worley Creek-Restored, 8=Worley Creek-Degraded, 9=Worley Creek-Reference, 10=Shawneehaw Creek-Restored, 11=Shawneehaw Creek-Degraded, 12=Shawneehaw Creek-Reference, 13=Kentucky Creek-Restored, 14=Kentucky Creek-Degraded, 15=Kentucky Creek-Reference, 16=Ben Bolen Creek-Restored, 17=Ben Bolen Creek-Degraded, 18=Ben Bolen Creek-Reference, 19=South Beaver Creek-Restored, 20=South Beaver Creek-Degraded, 21=South Beaver Creek-Reference, 22=Little Phoenix Creek-Restored, 23=Little Phoenix Creek-Degraded, 24=Little Phoenix Creek-Reference, 25=Day Creek-Restored, 26=Day Creek-Degraded, and 27=Day Creek-Reference.

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27
Aceraceae																											
Acer pensylvanicum Linnaeus						X			X												X			X			
Acer rubrum Linnaeus		X	X	X	X	X	X	X	X				X		X	X	X	X	X	X	X			X	X		X
Acer saccharinum Linnaeus				X		X						X				X											
Acer saccharum Marshall										X		X												X			
Anacardiaceae																											
Rhus glabra Linnaeus	X		X				X																				
Toxicodendron radicans (L.) Kuntze		X	X	X	X										X					X	X	X	X	X			
Aquifoliaceae																											
Ilex ambigua (Michaux) Torrey						X																					
Aristolochiaceae																											
Aristolochia macrophylla Lambert						X						X						X									
Aristolochia tomentosa Sims				X																							
Betulaceae																											
Alnus serrulata (Aiton) Willdenow													X														

Appendix D. Site species presence (continued).

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27
Betulaceae (continued)																											
Betula alleghaniensis Britton	X		X	X	X	X	X		X			X	X		X	X	X	X	X		X	X	X	X	X		X
Betula lenta Linnaeus			X	X									X		X						X			X			X
Betula nigra Linnaeus	X		X							X																	
Carpinus caroliniana Walter															X												
Ostrya virginiana (Mill.) K. Koch					X	X	X				X				X	X			X		X						
Buddlejaceae																											
Budleja davidii Franchet				X																							
Caprifoliaceae																											
Sambucus canadensis Linnaeus	X		X	X	X		X	X					X		X	X			X	X							
Viburnum dentatum Linnaeus			X	X																				X			
Viburnum prunifolium Linnaeus																											
Celastraceae																											
Celastrus orbiculatus Thunberg											X																
Clethraceae																											
Clethra acuminata Michaux															X												
Cornaceae																											
Cornus amomum Mill.	X			X						X			X			X			X			X			X		
Cornus florida Linnaeus		X									X							X									
Ericaceae																											
Kalmia latifolia Linnaeus									X						X						X						X
Rhododendron calendulaceum (Michaux)																					X						

Appendix D. Site species presence (continued).

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27
Ericaceae (continued)																											
Rhododendron catawbiense Michaux																X			X							X	
Rhododendron maximum Linnaeus		X	X	X		X			X			X			X			X	X		X	X		X			X
Vaccinium corymbosum Linnaeus															X				X		X						
Vaccinium pallidum Aiton																		X									
Fabaceae																											
Lespedeza cuneata (Dumont) G. Don					X																						
Robinia pseudoacacia Linnaeus	X		X	X	X		X					X			X	X	X		X	X	X		X				
Fagaceae																											
Castanea dentata (Marshall) Borkhausen																X											
Fagus grandifolia Ehrhart		X	X	X								X			X			X	X					X			
Quercus alba Linnaeus			X	X	X																X						X
Fagaceae (continued)																											
Quercus coccinea Münchhausen		X		X																							
Quercus rubra Linnaeus		X	X	X	X	X	X		X			X	X		X	X	X	X	X		X		X	X			
Hamamelidaceae																											
Hamamelis virginiana Linnaeus									X			X			X	X		X			X			X			
Hippocastanaceae																											
Aesculus flava Solander	X	X	X	X	X		X		X			X	X			X	X	X	X		X		X	X			
Aesculus glabra Willdenow		X																					X				
Juglandaceae																											
Carya glabra (Mill.) Sweet			X									X						X									

Appendix D. Site species presence (continued).

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27
Juglandaceae (continued)																											
Carya ovata (Mill.) K. Koch			X	X		X			X												X		X	X			X
Carya tomentosa (Poiret) Nuttall	X						X											X	X	X							X
Juglans nigra Linnaeus		X	X			X												X			X						
Lauraceae																											
Lindera benzoin (Linnaeus) Blume							X								X									X			
Sassafras albidum (Nuttall) Nees				X	X		X																X	X			
Magnoliaceae																											
Liriodendron tulipifera Linnaeus	X	X	X	X	X	X	X		X			X				X		X	X			X	X	X	X		X
Magnolia acuminata Linnaeus															X			X	X		X			X			X
Magnolia fraseri Walter			X			X						X			X									X			
Magnolia macrophylla Michaux			X																								
Magnolia tripetala Linnaeus				X																							
Moraceae																											
Morus rubra Linnaeus	X	X																				X					
Nyssaceae																											
Nyssa aquatica Linnaeus, Walter				X																							
Nyssaceae (continued)																											
Nyssa sylvatica Marshall, Black							X												X								
Oleaceae																											
Fraxinus americana Linnaeus	X	X	X	X		X				X						X		X	X	X							X
Fraxinus pennsylvanica Marshall		X			X																						X

Appendix D. Site species presence (continued).

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27
Pinaceae																											
Abies fraseri (Pursh) Poiret										X									X								
Pinus strobus Linnaeus				X	X		X		X	X											X						X
*Tsuga canadensis (*Linnaeus) Carrière			X			X			X			X		X	X				X								X
Tsuga caroliniana Engelmann			X			X									X												
Platanaceae																											
Platanus occidentalis Linnaeus	X	X		X	X											X		X	X								
Vitaceae																											
Parthenocissus quinquefolia Linnaeus			X	X												X						X	X				
Ranunculaceae																											
Clematis virginiana Linnaeus		X	X	X	X		X	X					X		X	X			X	X		X	X		X		
Rosaceae																											
Amelanchier canadensis (Linnaeus) Medik															X												
Crataegus macrosperma Ashe													X								X						
Crataegus pruinosa (Wendland) K. Koch																					X						
Crataegus punctata Jacquin		X		X																	X						
Malus pumila Mill.																X											
Physocarpus opulifolius (Linnaeus) Maximowicz																X		X	X	X		X			X		
Prunus pensylvanica Linne					X																						
Prunus serotina Ehrend.	X	X	X	X	X	X	X			X		X	X		X	X			X		X						X

Appendix D. Site species presence (continued).

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27
Rosaceae (continued)																											
Pyrus communis Linnaeus		X											X														
Rosa multiflora Thunberg	X	X	X	X	X		X	X		X	X		X		X	X	X		X	X	X	X	X				
Rosa palustris Marshall								X					X									X					
Rubus argutus Link	X	X	X	X	X		X	X	X				X			X	X	X	X	X	X	X	X			X	
Rubus canadensis Linnaeus				X	X			X																			
Rubus occidentalis Linnaeus	X			X												X											
Rubus odoratus Linnaeus				X	X																X						
Spiraea alba Du Roi													X														
Spiraea japonica Linne	X	X	X	X					X										X						X		
Spiraea virginiana Britton									X																		
Salicaceae																											
Salix babylonica Linnaeus														X												X	
Salix nigra Marshall	X																										
Salix sericea Marshall	X	X		X	X		X	X		X	X		X			X			X			X	X		X		
Saxifragaceae																											
Hydrangea arborescens Linnaeus				X		X																					
Hydrangea cinerea Small																X											
Smilacaceae																											
Smilax rotundifolia Linnaeus				X	X		X		X							X	X	X			X		X	X			X
Tiliaceae																											
Tilia americana Linnaeus			X														X										

Appendix D. Site species presence (continued).

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27
Ulmaceae																											
Ulmus americana Linnaeus																											X
Vitaceae																											
Vitis labrusca Linnaeus						X							X									X	X				
Vitis rotundifolia Michaux									X			X						X									

VITA

Christopher Todd Kaase was born in Radford, Virginia, on August 4th, 1975. His parents are Robert and JoAnn Kaase, who now reside on Watauga Lake, in northeastern Tennessee. He spent his childhood in Wythe County, VA and moved to Boone, North Carolina in the fall of 1993 to attend Appalachian State University. In 1999, he was awarded a Bachelor's of Arts degree in English, concentrating in writing and with a minor in Spanish.

After graduation he began spring through fall seasonal work at the Mast Farm Inn restaurant and garden, in Valle Crucis, NC. Late winter layoffs were spent traveling and skiing the mountains of the American West and Europe. In 2002, Mr. Kaase joined the Taylor House Inn, LLC, a family business in Valle Crucis, NC. The Kaase family managed the responsibilities of the inn until 2006. In the fall of 2007, Mr. Kaase was accepted by the Geography and Planning Department at Appalachian State University and began study toward a Master of Arts degree. The M.A. was awarded in May 2010.

Mr. Kaase lives in Valle Crucis, NC with his partner Tina Houston and her daughter, Reid. Ms. Houston has been cooking in the High Country since 1997 and currently owns Reid's Catering Company.