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Effects of riparian zone buffer widths on vegetation diversity in southern Appalachian headwater catchments

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ABSTRACT

In mountainous areas such as the southern Appalachians USA, riparian zones are difficult to define. Vegetation is a commonly used riparian indicator and plays a key role in protecting water resources, but adequate knowledge of floristic responses to riparian disturbances is lacking. Our objective was to quantify changes in stand-level floristic diversity of riparian plant communities before (2004) and two, three, and seven years after shelterwood harvest using highlead cable-yarding and with differing no-cut buffer widths of 0 m, 10 m, and 30 m distance from the stream edge. An unharvested reference stand was also studied for comparison. We examined: (1) differences among treatment sites using a mixed linear model with repeated measures; (2) multivariate relationships between ground-layer species composition and environmental variables (soil water content, light transmittance, tree basal area, shrub density, and distance from stream) using nonmetric multidimensional scaling; and (3) changes in species composition over time using a multi-response permutation procedure. We hypothesized that vegetation responses (i.e., changes in density, species composition, and diversity across the hillslope) will be greatest on harvest sites with an intermediate buffer width (10-m buffer) compared to more extreme (0-m buffer) and less extreme (30-m buffer and no-harvest reference) disturbance intensities. Harvesting initially reduced overstory density and basal area by 83% and 65%, respectively, in the 0-m buffer site; reduced by 50% and 74% in the 10-m buffer site; and reduced by 45% and 29% in the 30-m buffer site. Both the 0-m and 10-m buffer sites showed increased incident light variability across the hillslope after harvesting; whereas, there was no change in the 30-m and reference sites over time. We found significant changes in midstory and ground-layer vegetation in response to harvesting with the greatest responses on the 10-m buffer site, supporting our hypotheses that responses will be greatest on sites with intermediate disturbance. Ground-layer species composition differed significantly over time in the 0-m buffer and 10-m buffer sites (both $P < 0.0001$), but did not change in the 30-m buffer and reference sites (both $P > 0.100$). Average compositional dissimilarity increased after seven years, indicating greater within-stand heterogeneity (species diversity) after harvesting. These vegetation recovery patterns provide useful information for evaluating management options in riparian zones in the southern Appalachians.

1. Introduction

Riparian communities are critical components of forest ecosystems (Naiman and Decamps, 1997; Verry et al., 2004), serving as the interface between terrestrial and aquatic communities (Kominoski et al., 2013; Kuglerová et al., 2014). Studies suggest that the effectiveness of a riparian zone in promoting stream and ecosystem health is strongly related to the diversity and richness of a riparian community's vegetation (Sweeney et al., 2004; Hagan et al., 2006). More recent studies have investigated plant functional attributes (e.g., Mouillot et al., 2013), while others concluded that a diverse flora stabilizes ecosystem processes (e.g., Garnier et al., 2016); and some have linked biodiversity to ecosystem services (e.g., Durance et al., 2016). Changes to riparian plant community composition could lead to altered diversity and redundancy of plant functional trait distributions as a result of species declines (see review, Kominoski et al., 2013). In many regions, the native flora contribute to fulfill numerous stream ecosystem functions such as water quality enhancement, flood and erosion control, bank stability, wildlife uses, and in-stream root habitat (Sweeney et al., 2004; Boggs et al., 2016; Witt et al., 2016). As a result, there has been a strong push to protect riparian ecosystems

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from disturbances such as harvesting or development, through the use of buffers (Pielech et al., 2015; Cristan et al., 2016; Schilling, 2016). However, the boundaries of riparian areas are often hard to define and it can be difficult to determine acceptable buffer widths (Holmes and Goebel, 2011; Kuglerová et al., 2014), particularly for headwater streams (Alexander et al., 2007; Clinton et al., 2010).

Headwater (first- and second-order) perennial streams comprise about 50% of the total stream network length in most forest watersheds (Wipfli et al., 2007); and between 74% and 80% in the U.S. (Hill et al., 2014). Higher order stream systems may experience repeated flooding and develop terracing and alluvial deposits that can be linked directly to subsurface hydrology, geomorphology, and flood-plain development; characteristics that can be useful in delineating riparian zones (Verry et al., 2004). Characteristics of lower order streams can vary among geo-physiographic regions, and headwaters range from steep, swift, and cold montane streams to warm, low gradient streams (Meyer et al., 2007). For example, first-order streams in the U.S. Lake States can have a valley floor (from 37 to 87 m), narrow floodplains (4–16 m wide), and/or alluvial benches (Palik et al., 2012); features that easily distinguish riparian zones from the upland forest (Holmes and Goebel, 2011). In contrast, small perennial streams in montane headwater catchments often lack alluvial benches, have steeper sideslopes, and have a closed canopy cover relative to larger or low gradient streams; features that may diminish the distinction between riparian zones and the surrounding forest (Goebel et al., 2003; Dieterich et al., 2006; Hagan et al., 2006; Clinton et al., 2010). In addition, small southern Appalachian streams may not have riparian obligate vegetation communities, or riparian indicator plant species, such as seen in other regions, particularly obligate wetland species or those associated with larger tributaries (Zenner et al., 2012; De Steven et al., 2015). With the increasing emphasis on managing headwater riparian zones (Sanders and McBroom, 2013; MacDonald et al., 2014) and the differences among headwater catchments (Meyer et al., 2007; Hill et al., 2014), setting a standard sized buffer width across geo-physiographic regions becomes problematic.

In previous work on the Appalachian watersheds used in this study, Clinton (2011) concluded that a 10 m buffer width was adequate to protect water resources after upslope forest harvest; however, wider riparian buffers can potentially provide additional benefits such as wildlife corridors (Sweeney and Newbold, 2014) and unique habitat for flora and fauna (Richardson and Béraud, 2014). Riparian buffers may also influence vegetation responses in harvested watersheds outside of the buffer area by providing propagules and modifying micro-environmental changes (Dovčiak and Brown, 2014; MacDonald et al., 2014). Therefore, a better understanding of floristic responses to different riparian buffer widths can provide important information for management planning and protection of ecosystem services (Sweeney et al., 2004; Kuglerová et al., 2014; Hill et al., 2014; Sweeney and Newbold, 2014). To address this, we quantified changes in floristic composition on harvested areas that implemented three riparian buffer widths: 0 m (i.e., no riparian buffer), 10 m, and 30 m distance away from the stream edge, as well as an unharvested reference site. We hypothesized that stand-level vegetation responses (i.e., changes in density, species composition, and diversity across the hillslope) will be greatest on harvest sites with an intermediate buffer width (10-m buffer) compared to more extreme (0-m buffer) and less extreme (30-m buffer and no-harvest reference) disturbance intensities, as suggested by the intermediate disturbance hypothesis (*sensu* Connell, 1978; Biswas and Mallik, 2010). Our hypothesis is based on the premise that an intermediate-width buffer will create more heterogeneous post-harvest micro-environmental conditions that will facilitate a greater vegetation response as measured by changes in density, species composition, and diversity.

2. Methods

2.1. Site descriptions

Study sites were located in the Ray Branch watershed in the Nantahala National Forest of the Southern Appalachian Mountains in western North Carolina (35°15′N, 83°35′W). The area has abundant rainfall (mean, 1800 mm yr^{-1}) distributed evenly throughout the year. Less than 5 percent of total annual precipitation falls as snow or ice. Mean annual air temperature is 12.6 °C ranging from 3.3 °C to 21.6 °C in January and July, respectively (Laseter et al., 2012).

Four catchments (treatment sites) with 1st order perennial streams, similar topography, soils and vegetation were selected from the Ray Branch area (Fig. 1). Catchments were east-facing, approximately 10 ha in size and had stream gradients ranging from 7 to 23%. Soils are Evard-Cowee-Saunook, mostly Saunook predominating along riparian areas, and Evard in the uplands. Saunook are fine-loamy, mixed, mesic Humic Hapludults; and Evard are loamy, oxidic, mesic Typic Hapludults (Thomas, 1996). Elevation ranges from 1000 to 1200 m. Forest composition in each of the four sites had a dense canopy with mesophytic trees including *Acer rubrum* L., *Betula lenta* L., *Carya* spp., *Liriodendron tulipifera* L., *Quercus rubra* L., and *Tsuga canadensis* (L.) Carr. (see Supplementary Table A1) with a lush and diverse herbaceous layer (see Supplementary Table A2). Midstory vegetation consisted primarily of advanced *Quercus* spp. regeneration, with scattered *Pinus strobus* L. and *Tsuga canadensis*. Although often abundant in southern Appalachian riparian areas (Vandermast and Van Lear, 2002), very few evergreen shrubs (*Rhododendron maximum* L. and *Kalmia latifolia* L.) were present on the study sites (see Clinton et al., 2010 for detailed description of pretreatment conditions).

2.2. Experimental treatments and measurements

Our experimental treatments were part of a commercial harvest and could not be replicated. Hence, we used a Before-After/Control-Impact experimental design (BACI) to address the concern of pseudoreplication (Stewart-Oaten et al., 1986; van Mantgem et al., 2001). Each site was assigned to one of the following uncut buffer-width treatments: 0 m (no buffer, i.e., harvesting to the stream edge), 10 m distance from the stream edge, 30 m distance from the stream edge, and an unharvested reference site that spanned a distance of 60–80 m from stream edge to ridge. Pretreatment vegetation measurements (see below) occurred in June-July 2004. The three harvest sites were designated by the Nantahala National Forest to receive a two-age shelterwood prescription using primarily cable-yarding due to steep slopes. Cable-yarding also minimizes forest floor disturbance because logs are suspended above the ground during removal (Miller and Sirois, 1986). Leave-trees at each treatment site were marked, and all remaining unmarked standing timber was felled outside the designated buffer zone. Harvesting began in October, 2005. The goal of the prescription was to leave 5 m^2 ha^{-1} residual basal area; however, the resulting residual basal area across the hillslope was generally lower (Table 1). No trees were felled within the buffer zones and no losses were observed due to windthrow, ice or insects during our study period. The size of the harvested area was comparable among all treated catchments; 9.7 ha for the 0-m buffer site, 6.0 ha for the 10-m buffer site and 8.5 ha for the 30-m buffer site (Joan Brown, Nantahala Ranger District Silviculturist; personal communication).