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# The Effects of Riparian Management Zone Delineation on Timber Value and Ecosystem Services in Diverse Forest Biomes Across the United States.

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# THE EFFECTS OF RIPRIAN MANAGEMENT ZONE DELINEATION ON TIMBER VALUE AND ECOSYSTEM SERVICES IN DIVERSE FOREST BIOMES ACROSS THE UNITED STATES

by

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**Abstract**

M.T. Jayasuriya. The Effects Of Riparian Management Zone Delineation On Timber Value And Ecosystem Services In Diverse Forest Biomes Across The United States. 213 pages + xv, 21 tables, 28 figures, 2020. Chicago Manual Style used.

Headwater streams are disproportionately affected by forest management activities in working forests of the United States (US) due to their high densities within watersheds. Thus, assigning the right buffer distance and buffer type to represent the ecology and topography of headwater streams is an important management decision. Focusing on headwater streams, this dissertation examines different riparian delineation techniques practiced within the US and proposes alternative approaches that balance ecological and economic factors. This primary objective was addressed using two datasets. The first dataset of stand data and understory vegetation was collected from forests distributed across New York and New Hampshire. The second dataset comprised of 1-meter digital terrain models and FIA data of 33 watersheds across 17 states within the contiguous US.

On a regional scale, an ecologically significant riparian buffer was mapped using understory plants along headwater streams in Northeastern forests. A threshold distance of 6-12 m from stream edge was identified using plant species richness. Although this is not the actual extent of a functional riparian area, this distance represents an important zone for increased plant species diversity.

A functional riparian area representing topography and forest structure developed by the US Forest Service was used as a variable width riparian buffer delineation technique in this study. The functional approach was compared with state-specific riparian delineation guidelines and a 30-meter fixed width riparian buffer across a broad range of forest regions in the US. From a regional context, when using the functional approach, 16–20 % of watersheds in the West and Pacific Northwestern regions were delineated as riparian. The functional method consistently delineated more land to the riparian area than other riparian delineating methods except for sampled watersheds in the Lake States where there was little to no topography along headwater streams.

Delineating valuable timber land as riparian areas is an opportunity cost for landowners given the density of headwater streams in working forests. Alternative riparian management options such as increasing carbon stocks within riparian management zones for carbon markets can not only offset riparian allocation costs but also serve as an investment opportunity for large-scale forest landowners.

Keywords: Riparian Management Zone, functional riparian buffer, plant species richness, carbon markets, improved forest management

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# Chapter 1 : Introduction

## Background

Riparian forests are widely studied components of forest ecosystems. Defined as the interface between terrestrial and aquatic ecosystems, the moist and often wet soils and high water tables make them one of the most important and diverse parts of a forest (Blinn and Kilgore 2004). In forested regions, riparian areas adjacent to streams provide valuable ecosystem functions, including regulating the flow of water, sediment, and nutrients across system boundaries (Lynch and Corbett 1990; Ward and Jackson 2004; Witt et al. 2013; Secoges et al. 2013); contributing organic matter to aquatic ecosystems (Jackson et al. 2001); carbon sequestration in living biomass and soils (Matzek et al. 2015); and increasing bank stability and reducing erosion (Keim and Schoenholtz 1999). In addition, forested riparian areas provide unique habitat with high species diversity that are used as dispersal corridors and refugia for birds and wildlife (Gregory et al. 1991; Jackson et al. 2007; Chizinski et al. 2010).

Removing and disturbing vegetation and coarse woody debris during logging operations were common practices in the US through the late 1960s (Richardson et al. 2012). Once the negative impacts (i.e. sedimentation, stream bank erosion) of these practices were realized, protective riparian buffer strips were gradually adopted by resource managers. This marked the beginning of the evolution of a riparian management zone (RMZ). A RMZ is a forestry best management practice (BMP) designed to reduce non-point source pollution during forest operations (Phillips and Blinn 2004). Numerous studies confirm that RMZs are effective in ameliorating the negative impacts of harvesting by trapping sediment in overland flow (Keim and Schoenholtz 1999; Ward and Jackson 2004; Lakel et al. 2010), regulating stream temperature (Keim and Schoenholtz 1999; Jackson et al. 2001), reducing total suspended solids, turbidity, dissolved oxygen, nitrate (Binkley and Brown 1993; Witt et al. 2013), and protecting wildlife and their habitat (Carroll et al. 2004; Jackson et al. 2007; Chizinski et al. 2010). Given the demand for forest products from commercial forests, it is imperative that timber harvesting practices maintain a high level of protection for forested riparian areas without unduly compromising the ability of forest landowners, either private or public, to pursue forest management and timber production.

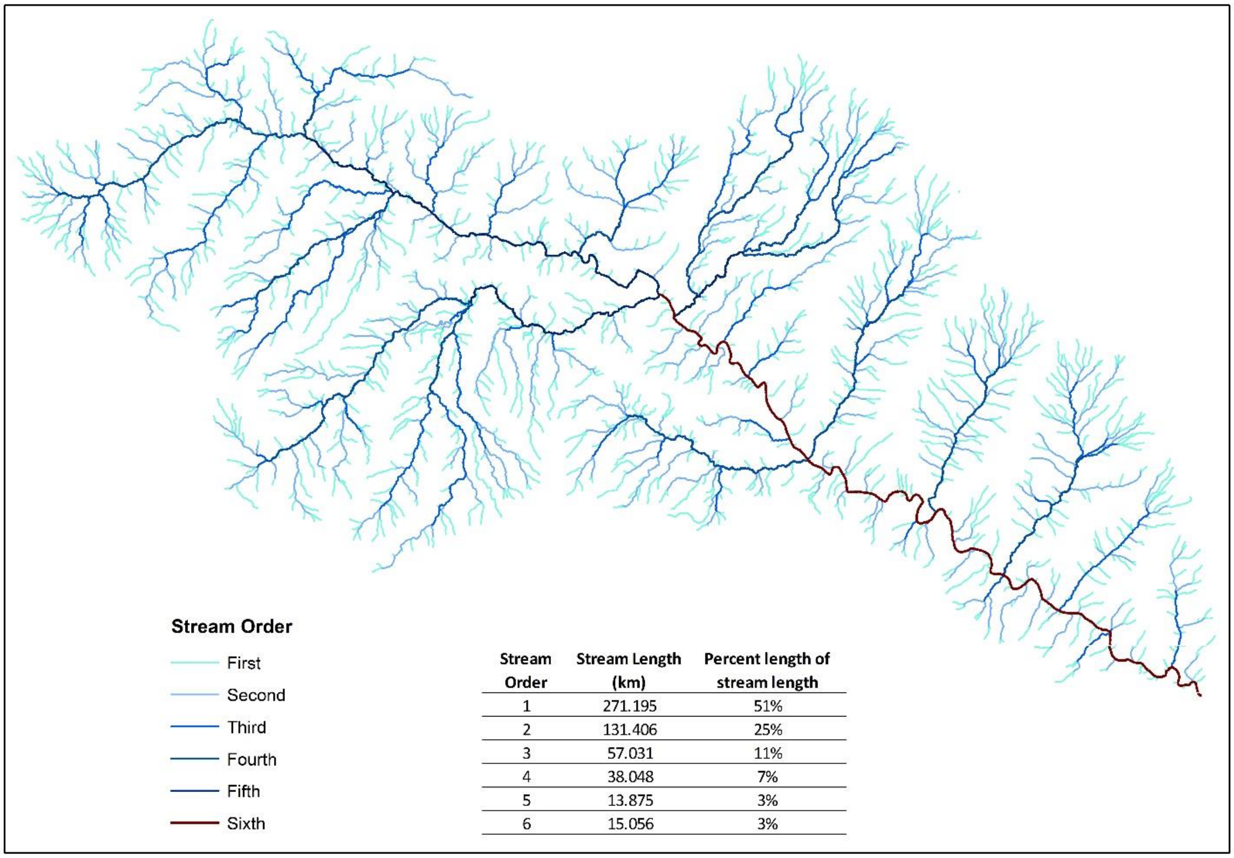
Many states formerly promote RMZs to regulate land disturbance activities, protect water quality, and comply with the Federal Clean Water Act (Ilhardt et al. 2000). BMP recommendations for operating within or adjacent to the RMZ are fairly consistent between states except for differences in riparian buffer distances and harvesting restrictions (Phillips et al. 2000; Jayasuriya 2016). These RMZ allocations are either regulatory, quasi-regulatory or non-regulatory (Cristan et al. 2018). The riparian buffer distance (one side of the stream) is greatly dependent on management objectives. Variances in riparian buffers reflect differences in the integration of ecological, economic, and social factors (Lee et al. 2004). Most of the potential contributions of riparian vegetation to the ecological functions within a stream are realized within the first 4.6 – 30.5 m (15 to 100 ft.) from the stream bank (Blinn and Kilgore 2001). This distance range of riparian buffers provide at least 50 % of potential effectiveness and often 75 % or greater effectiveness at protecting various stream functions (Castelle and Johnson 2000). State-specific riparian buffer guidelines are defined either as (1) fixed or standard width based on channel or waterbody type; or (2) variable width based on slope gradient (Phillips et al. 2000; Blinn and Kilgore 2004; Jayasuriya 2016). Approximately 35 % of state-specific riparian buffers are fixed width while the remaining states (65 %) promote the use of variable width buffers. There is a wide range of buffer widths, from as low as 6 m to 137 m (20 – 450 ft.) (National Research Council 2002; Jayasuriya 2016).

Variable width riparian buffers are customarily delineated based on one or more ecological functions of a riparian area. Bren (1998) used the concept of a constant buffer-strip loading design to allocate riparian buffers in a watershed draining into the West Tarago River in southeast Victoria, Australia. In his design, the buffer strip width along the stream varied from point to point along the stream bank depending on the upslope area contributing to each stream element. This variable-width buffer reflected the use of topography and gave priority to those areas along a stream where large loading from overland flow is expected. In the Tipton Creek watershed in north-central Iowa, Tomer et al. (2003) used terrain analysis to evaluate patterns of overland flow across the landscape and identified riparian locations with large wetness indices, where buffer vegetation could intercept sheet/rill flows from significant upslope areas. Their study shows that by accounting for the topographic variation in the landscape, variable-width buffers can be defined to intercept total maximum daily loads. Tiwari et al. (2016) used the cartographic depth-to-water (DTW) index to map the distribution of riparian soils along streams and lakes in the Krycklan catchment in Northern Sweden to predict a hydrologically adapted riparian buffer. The DTW-index was conceptually described as a measure of the depth down to a modeled groundwater surface. Groundwater discharge locations were discovered in these riparian areas with wet soils and found to be hotspots for plant biodiversity and biogeochemical cycling (Kuglerová et al. 2014a). Ilhardt et al. (2000) proposed a functional riparian area using the concept of a 50-year flood height and one-tree length to capture the flow of energy and material within the watershed.

**Rational**

Delineating riparian areas using either fixed or variable width buffers is a major management decision, especially along headwater streams. These streams require more attention because most forest operations are concentrated around headwater streams. Headwater streams form the majority of stream networks where their cumulative stream lengths can reach up to 80 % of total stream lengths within watersheds (Benda et al. 2005; Wipfli et al.2007). Unlike higher order streams, headwater streams (lower order streams) are often underrepresented, in other words, they are not mapped to the actual density seen within watersheds in many National Hydrographic Datasets (Baker et al. 2007; Brooks and Colburn 2011; Elmore et al. 2013). This could be a result of dense canopy cover and/or low-resolution digital terrain models used to map stream networks. As a result, headwater streams tend to receive less priority in forest management and harvest plans and thus, less protection which can lead to negative environmental impacts. The high density of headwater streams within working forests (Figure 1.1) provide challenges to forest managers seeking to conduct financially viable timber harvesting operations while simultaneously protecting the wide array of forest riparian values. Therefore, allocating the appropriate buffer width around headwater streams to ensure the protection and the integrity of the ecosystem within and around a stream is a critical question facing land managers.

Ideally, RMZs should capture the boundaries of sensitive areas adjacent to streams and be tailored to suit the environmental conditions of the forested landscape. However, identifying variable widths defined by the physical and biogeochemical characteristics of riparian areas during management is a concern faced by resource managers. Management or harvesting plans should outline identifiable environmental cues or threshold distances representing sensitive areas. The response of understory riparian vegetation shown by their distribution, structure and abundance to these biophysical and biochemical parameters within riparian areas has been used as a proxy to identify boundaries of riparian areas (Richardson et al. 2005; Hagan et al. 2006; Quinby et al. 2000; Goebel et al. 2003). Understory or groundcover vascular plant species are known to better represent changes in micro-environmental conditions to detect lateral gradients between riparian areas and upland forests (Decocq 2002; Richardson et al. 2005; Dieterich et al. 2006). Obligate or facultative hydrophytes growing in saturated or flooded soils in riparian areas can thus serve as useful indicators for riparian delineation that represents the ecology of this ecotone and its variable lateral boundaries. Only a handful of state BMP guidelines or state specific environmental conservation websites encourage or promote the use of indicator plants for identifying riparian areas. Hence the need for more research in this area, specifically for integration into forest management guidelines.




| Stream Order | Stream Length (km) | Percent length of stream length |
|---|---|---|
| 1 | 271.195 | 51% |
| 2 | 131.406 | 25% |
| 3 | 57.031 | 11% |
| 4 | 38.048 | 7% |
| 5 | 13.875 | 3% |
| 6 | 15.056 | 3% |

**Figure 1.1: A stream network. This diagram shows the representation of stream orders as defined by the Strahler classification system.**

However, given the variation involved in defining variable width buffers based on slope gradient across states, and a third of states across the US adopting fixed width buffers, it is useful to assess whether state-specific riparian buffer guidelines represent the ecology of riparian areas. The fixed-width approach practiced in many states is easy to implement and monitor for compliance, but may not address site-specific ecological characteristics and functions of the riparian area. Some contend that fixed-width buffers may result in errors when determining riparian area extent and characteristics (MacNally et al. 2008; Kuglerová et al. 2014b) and could, in some instances, have greater opportunity costs for the landowner (Tiwari et al. 2016) and/or the ecosystem. For example, areas that are clearly considered riparian, such as floodplains, may extend beyond a fixed-width buffer. This situation is quite prevalent in higher order streams. Alternatively, lands that are arguably non-riparian might be included in a fixed-width buffer, especially along headwater or lower order streams (Holmes and Goebel 2011; Jayasuriya et al. 2019).

Riparian buffer distances can have a significant impact on the overall costs of forest management within a harvesting area. As RMZs increase in area, the opportunity costs could increase, causing a negative economic impact on the landowner. Both the area of the RMZ and the associated harvesting restrictions within the RMZ can reduce potential stumpage revenues. Areas of high stream or drainage density further exacerbate the economic impacts. Jayasuriya et al. (2019) reported that the area dedicated to RMZs along first- and second-order streams on timberlands within the Catskill mountains in New York can range from 5 – 11 % while using the ~30 m (100 ft.) fixed or variable-width riparian buffer approach. Previous studies documented the percentage of area designated as RMZ from as low as 2.5 % to nearly 15 % (Kluender et al. 2000; Lippke et al. 2002; Lakel et al. 2015). Whatever the final determination, the RMZ can hold a substantial amount of valuable timber.

Given the potential for valuable forest stocking within RMZs, the economic stakes are significant for landowners. Timber value within RMZs can range as low as $136 per hectare ($55 per acre) to as high as $3,707.50 per hectare ($1,500 per acre) depending on the forest composition and merchantable log volume (Lakel et al. 2015; Jayasuriya et al. 2019). Thus, depending on the harvesting restrictions within RMZs, opportunity costs can represent 7-12 % of stumpage revenues for the landowner (Jayasuriya et al. 2019). At the same time, the growing emergence and demand for carbon offsets under current and future carbon trading mechanisms (e.g., California carbon markets) make it incumbent on forest managers to quantify the value and co-benefits of forest carbon stocks under alternative management scenarios (Matzek et al. 2015). Carbon trading within

RMZs could be a potential option for balancing opportunity costs of stumpage value and ecosystem services.

Riparian areas can carry higher stocking densities (Jayasuriya et al. 2019) and have greater productive capacities due to favorable growing conditions than their upland forest counterparts (Naiman et al. 2005; Dybala et al. 2019). However, there may be exceptions with changes in biome types and landuse history. The aboveground biomass of mature riparian forests range between 100 and 300 Mg[1]/ha, with few exceptions (Balian and Naiman 2005). Carbon stored within trees is assumed to be half of that of biomass (FVS 2014). Maraseni and Mitchell (2016) estimated the biomass carbon of riparian vegetation (trees and shrubs) and coarse-woody debris (CWD) along the Condamine River, in Queensland, Australia. Trees, shrubs and CWD inventoried from 17 sample plots classified into three categories of 'excellent', 'good' and 'poor' recorded average total carbon values of 291.7 t/ha, 134.8 t/ha and 4.3 t/ha respectively. Although studies of carbon sequestration within riparian forests in the Northeastern US are scarce, Nunery and Keeton (2010) quantified and projected carbon storage, using the USDA Forest Service's Forest Vegetation Simulator (FVS), in forests in the Northeast under nine different management scenarios. They observed a clear gradient of increasing C sequestration as forest management intensity ranged from high (clearcut) to low, and mean carbon sequestration was significantly greater for ''no management'' compared to any of the active management scenarios. RMZs could perform better in existing carbon markets in the US as separate management units of the larger working forest landscape as these areas only undergo partial harvesting and thus sequester significantly more carbon. However, to date, research is lacking on how riparian management guidelines and practices within forested headwater regions affect these multiple ecosystem services, and how these relationships may vary biogeographically in different forest types and topographic landscapes.

[1] 1 Megagram (Mg) is equivalent to 1 metric ton.

**Research Objectives**

The general goal of this dissertation is to examine how headwater streams in working forests of the US are delineated, and propose alternative approaches that balance ecological and economic factors.

Objectives supporting this primary goal are as follows:

1. Detecting riparian threshold distances defined by floristic distributions in headwater streams in Northeastern forests,
2. Comparing and contrasting the differences in riparian forest area delineated by a functional riparian buffer, a 30-m (100-ft) fixed width riparian buffer, and state-specific riparian buffer guidelines, and
3. Comparing long-term net revenue earning potential of riparian areas between timber value and the carbon markets under the California Compliance Offset Protocol.

**Organization**

This dissertation is composed of five Chapters that includes this introduction (Chapter 1), three manuscripts (Chapters 2 – 4), and a synthesis (Chapter 5).

The following is a brief description of the manuscripts:

**Chapter 2**, titled "*Detecting riparian zones using understory plant diversity and composition patterns in mesic headwater forests of the Northeastern U.S*." was completed using data sampled from forests in Central New York (Heiberg Memorial Forest and Cuyler Hill State Forest), the Adirondack Mountains (Huntington Wildlife Forest), and White Mountains in New Hampshire (Hubbard Brook Experimental Forest). This study identifies a lateral riparian threshold distance identified by plant species richness across three geographic locations across the Northeastern region and develops an empirical model for predicting species richness. This manuscript is prepared for submission to the Journal of Forestry.

**Chapter 3,** titled "*Assessing riparian area protection strategies along headwater streams in forested regions of the U.S.*" focuses on five timber producing regions of the US. Opportunity costs of land area allocation, identified as lost timber revenue, is compared between commonly used buffer types and functional riparian buffers across various forest cover types and topographic relief across the contiguous US. This will help land managers make informed decisions for allocating the suitable riparian buffer type to protect headwater streams that ultimately represents the forested watershed. The methods section of this paper introduces a new GIS tool for delineating functional riparian areas using high resolution LiDAR derived digital terrain models (DTM). High resolution DTMs of 33 watersheds distributed between 17 states were used in this analysis. A target journal has not been selected for this manuscript.

**Chapter 4,** titled "*Protecting timberland RMZs through carbon markets: A protocol for riparian carbon offsets*" provides an economic assessment of timber and carbon markets using three riparian management scenarios. This study was completed using data sampled from forests in the Adirondack Mountains in New York (Huntington Wildlife Forest) and White Mountain Forest in New Hampshire (Hubbard Brook Experimental Forest). This study investigates an investment opportunity for selling riparian carbon offsets to the California carbon markets. This could potentially be adapted as an alternative management option for riparian management that could offset opportunity costs of allocating RMZs in working forests and thereby delineating economically and ecologically efficient riparian buffers.

This manuscript was published in Forest Policy and Economics in 2020 (Jayasuriya et al. 2020).

Within each manuscript chapter, several additional resources are provided in the appendices. Each chapter includes a list of references for literature cited within the chapter and a complete list of references is included at the end of Chapter 5 for convenience. The last page contains my resume as per request by the Office of Instruction and Graduate Studies, SUNY ESF.

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## Chapter 2 : Detecting riparian zones using understory plant diversity and composition patterns in mesic headwater forests of the Northeastern U.S.


### Abstract

Riparian buffers allocated to minimize sedimentation during forest operations are rarely based on ecological criteria. Since most forest operations are concentrated around headwater streams, the primary research objective of this study was to identify a floristically significant riparian boundary for first- and second-order streams using plant species composition and indicator species to signify riparian environments distinct from the surrounding upland forest. Within three sampling locations of the Northeast U.S., understory vegetation plots were sampled along perpendicular transects extending from the stream bank into the upland forest. At all sites species richness was highest adjacent to the stream, decreasing exponentially within 6 -12m from the channel. Species composition closest to the stream was significantly different from all other lateral distances, but identified riparian indicator species were of limited use across all sites. However, changes in species richness can serve to identify a riparian area extent up to 6 – 12 m from headwater streams.



**Key words**: functional riparian buffer, understory vegetation, headwater streams, generalized linear mixed models, canonical discriminant analysis


### Introduction

Stream riparian areas are three-dimensional ecotones that encapsulate the spatial gradients and interactions between the aquatic, terrestrial and atmospheric environments within a stream corridor (Gregory et al. 1991). These transition areas provide numerous ecosystem services and benefits such as regulating flow of water, sediment, and nutrients across watersheds; maintaining stream bank stability; and provisioning shade, coarse-woody debris and refugia for wildlife (Naiman et al. 2005; Opperman et al. 2017). Designated riparian buffers are also known as riparian management zones (RMZs) and they may or may not extend to the functional extent of a riparian area. In forested regions, RMZs are designed to minimize or mitigate potential disturbances stemming from timber harvesting activities such as sedimentation, and also function as sources of shade and protection against streambank erosion (Richardson et al. 2012).

In the United States (US), Best Management Practice (BMP) guidelines established by individual states dictate riparian buffer distances and management guidelines. These prescriptions vary based on jurisdictions and natural resource management objectives (Blinn and Kilgore 2001). RMZ buffers are customarily applied using a fixed width (e.g., 30 m), or occasionally variable-width buffers based on site specific conditions and/or management objectives (Blinn and Kilgore 2001; Phillips et al. 2000). However, state-wide riparian buffer delineations are rarely supported by ecological targets and empirical data specific to their regions (Castelle et al. 1994). Arbitrary riparian buffers could underestimate the actual functional extent of a riparian area or else neglect to preserve sensitive riparian habitat critical for ecological processes and protected species.

Riparian areas along headwater streams require adequate guidelines for operating around and within them to protect the integrity and sustainability of downstream uses (Wipfli et al. 2007). First and second-order streams make up the majority of the stream density within a watershed (Shreve 1969) where most forest management activity is concentrated. Due to their smaller channel widths and high density within watersheds, they commonly receive less riparian buffer protection as compared to larger order streams (Blinn and Kilgore 2001). In contrast to higher order streams with defined floodplains distinct from upslope areas, riparian communities along headwater streams can be indistinguishable from the rest of the forest due to their closed canopy cover, narrow ravines with steep terrain, and the lack of alluvial benches (Richardson et al. 2005). Higher order streams with floodplains generally have identifiable plant zones associated with the local hydrogeomorphic conditions (Lite et al. 2005; Bendix and Stella 2013; Opperman et al. 2017). Furthermore, higher order streams in temperate forests are generally lined by obligate riparian tree species such as willow (*Salix sp*.) and cottonwood (*Populus sp*.) (Richardson et al. 2005; Opperman et al. 2017), and conifer species such as eastern hemlock (*Tsuga canadensis*) and spruce (*Picea sp*.) in the Northeast, thus making it relatively easier to delineate identifiable riparian areas both in the field and from remotely-sensed data. Such is not the case with headwater streams. However, given the importance of headwater streams in providing clean water and other ecosystem services, it is important that we investigate ecological features that can be used to delineate an ecologically relevant riparian boundary in these environments.

Identifiable vegetation characteristics or environmental cues that differentiate riparian from upland forests can assist forest managers in delineating buffers and preserving sensitive areas around streams to restrict access or limit uses within them. These can be based on field observations and/or remotely-sensed data (Goetz 2006; Dufour et al. 2012; Kui et al. 2017). In most instances riparian areas in arid and semi-arid regions are readily distinguishable from the surrounding landscape because of the strong gradients in hydrogeomorphic and/or biogeochemical interactions in these regions, as well as their distinct fauna and flora communities (Lewis et al. 2009; Stella et al. 2013). These gradients are more distinct on higher order streams when compared to lower order or headwater streams (Salinas et al. 2000; Lite et al. 2005). However, riparian areas in mesic environments may or may not be distinct from their upland counterparts, especially along headwater streams due to the lack of sharp moisture gradients (Richardson et al. 2005). Thus, lateral gradients of moisture in mesic forests with closed canopy cover may not be an ideal measure for distinguishing riparian forests from upland forests. Understory species are more sensitive to small-scale changes within landscapes than overstory species and may better represent changes in the micro-environment (Decocq 2002; Dieterich et al. 2006). Also, understory vegetation generally contributes more to plant diversity in temperate northern forests than does the overstory (Echiverri and Macdonald 2019), and riparian vegetation, in particular, typically displays a high degree of compositional diversity relative to the surrounding vegetation mosaic (Gregory et al.

1991). Thus riparian understory plants are good candidates for assessing the biodiversity effects of forested streams (Quinby et al. 2000; Hagan et al. 2006; Dieterich et al. 2006).

In this study, I addressed the question of whether riparian areas are floristically distinct in mesic forest biomes along headwater streams, using several measures of understory vegetation as ecological indicators. In addition, I investigated whether a floristically distinct species composition can be used to map a distance-based riparian boundary in mesic forest environments. These questions were addressed by examining lateral gradients of species composition and richness along headwater streams distributed among three locations within the Northeastern US. Primary research objectives were to: 1) model lateral distance thresholds for species richness within riparian areas, 2) identify differences in species composition between riparian areas and the surrounding upland forests and 3) identify indicator species that signify riparian environments distinct from the upland forest. By encompassing a broad geographic scope and variation in forest composition within the study, the aim was to derive useful ecological information for forest management across the region.

**Methods**

Study area

Field sampling was carried out at three locations in the Northeastern US: Hubbard Brook Experimental Forest in New Hampshire (Site 1), Huntington Wildlife Forest in the Adirondack Mountains of New York (Site 2), and Heiberg Memorial Forest and Cuyler Hill State Forest (Site 3) within the Great Lakes plain of Central New York (Figure 2.1). Hubbard Brook Experimental Forest (Site 1) is a 3,138 ha (7,754 ac) long-term experimental forest located within the White Mountain National Forest in central New Hampshire. Ranging in elevation from 222 to 1,015 m (728 – 3,330 ft.), it experiences an annual precipitation of 140 cm (55 in.) (NOAA 2018). January and July average temperatures are -6.3 °C (20.7 F) and 18.5 °C (65.3 F), respectively (NOAA 2018). The forest cover is primarily northern hardwoods (85 %), with the balance in spruce-fir (15 %) (Adams et al. 2004). Huntington Wildlife Forest (Site 2) is a 6,000 ha (14,826 ac) experimental forest located in Newcomb, NY and lies near the geographic center of the Adirondack State Park. Ranging in elevation from 457 to 823 m (1,500 – 2700 ft.), HWF has a mean annual precipitation of 102 cm (40 in.) (NOAA 2018). Temperature averages are -9.4 °C (15 F) for January and 18.5 °C (65.3 F) for July (NOAA 2018). The forest cover is dominated by northern hardwoods (72 %), followed by mixed hardwood-conifer (18 %), and spruce-fir (10 %) (SUNY-ESF n.d.). Situated within Central New York (Site 3), Heiberg Memorial Forest, is a 1,538 ha (3,800 ac) research forest located in Cortland County in the towns of Tully, Fabius, Pompey and Truxton. The elevation ranges from 382 to 625 m (1,253 – 2,051 ft.). Within the same general region and spanning an elevation range of 380 to 634 m (1,247 – 2,080 ft.), Cuyler Hill State Forest is a 2,229 ha (5,508 ac) state forest located on the northeastern border of Cortland and Chenango Counties in the town of Truxton. For both forests, monthly temperature averages are -6.9 °C (19.6 F) for January and 19.4 °C (66.9 F) for July (NOAA 2018). Both forests carry diverse forest cover types. Heiberg Memorial Forest is managed to produce a diverse representation of forest ecosystems in the Northeastern US. The forest cover comprises of an uneven distribution of both northern hardwoods and conifer species that includes red pine (*Pinus resinosa* Aiton), eastern white pine (*Pinus strobus* L.), Norway spruce (*Picea abies* (L.) Karst.), scots pine (*Pinus sylvestris* L.), eastern hemlock, northern white cedar (*Thuja occidentalis* L.), and tamarack/ eastern larch (*Larix laricina* (Du Roi) K. Koch). Cuyler Hill State Forest contains cover types of northern hardwood, northern hardwood-hemlock, and conifer species such as European larch (*Larix decidua* Mill.) and Japanese larch (*Larix kaempferi* (Lam.) Carrière), Norway spruce, red pine, northern white cedar, and white spruce (*Picea glauca* (Moench) Voss) (NYS DEC n.d.) (Table 2.1).

The upland forests at Huntington Wildlife Forest, Heiberg Memorial Forest, and Cuyler Hill State Forest have been subjected to forest management activities within the last 50 - 100 years. Hubbard Brook Experimental Forest has been subjected to long-term experiments investigating the impact of forestland management on water yield, water quality and flood flow since it was established in 1955 by the US Department of Agriculture Forest Service (Fahey n.d.).

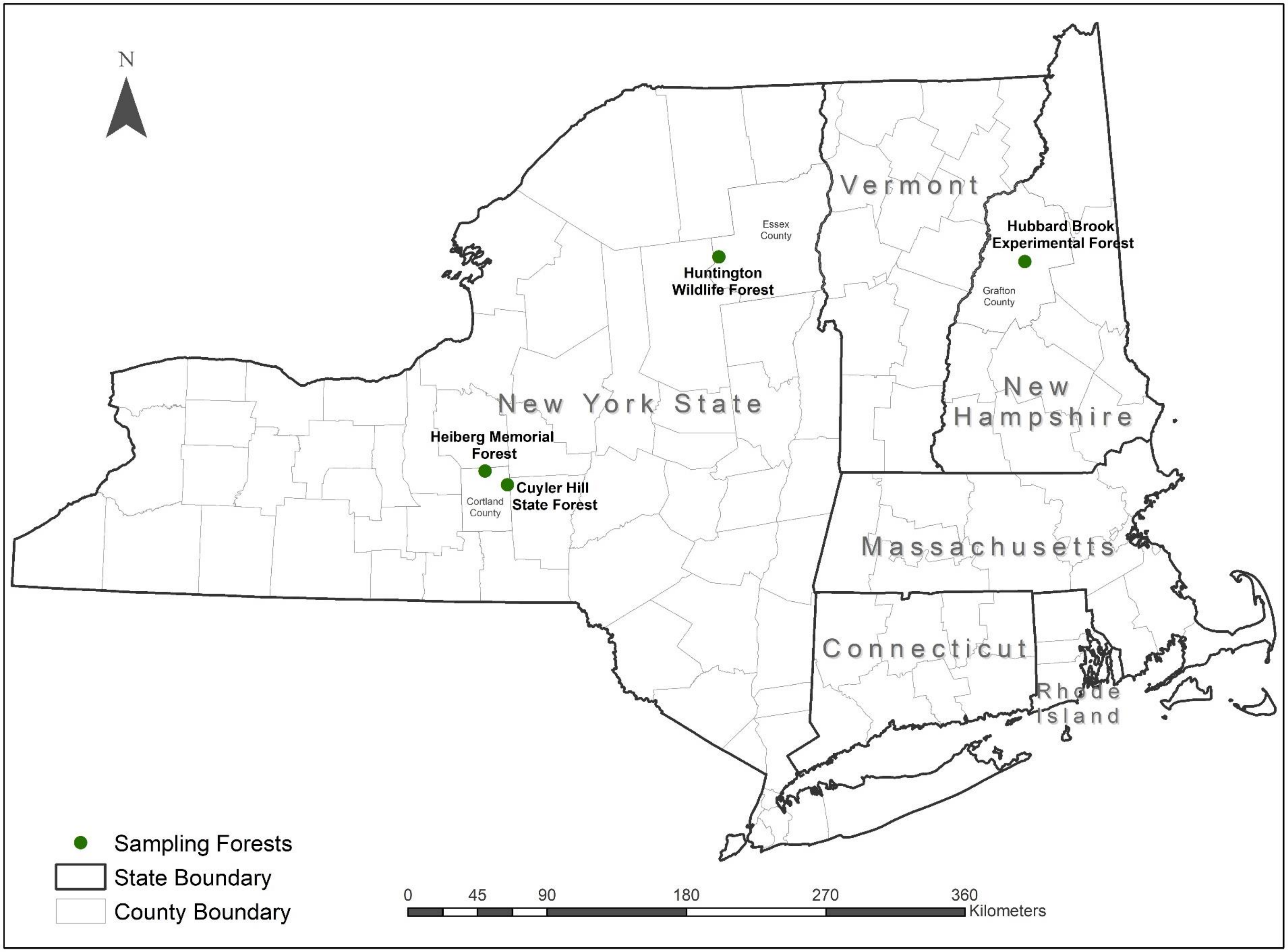


**Figure 2.1: Sampling locations across the Northeastern United States. (Site 1: Hubbard Brook Experimental Forest; Site 2: Huntington Wildlife Forest; Site 3: Heiberg Memorial Forest and Cuyler Hill State Forest.).**

**Table 2.1: Summary of sampling site descriptions and tally of plots at each location.**

| Sampling Sites | Hubbard Brook Experimental Forest (Site 1) | Huntington Wildlife Forest (Site 2) | Central New York Forests (Site 3) |
|---|---|---|---|
| **Coordinates** | 43° 56' 21.43" N 71° 41' 22.90" W | 43° 58' 18.31" N 74° 11' 05.27" W | 42° 45' 38.08" N 76° 05' 00.20" W |
| **Forest Cover** | Northern hardwoods (85 %), spruce-fir (15 %) | Northern hardwoods (72 %), mixed (18 %), spruce-fir (10 %) | Northern hardwoods, northern hardwood-hemlock, and conifer |
| **Sampled streams** | 11 | 8 | 11 |
| **Basal area ($m^2 \cdot ha^{-1}$)** | 32.8 | 30.3 | 40 |
| **QMD (cm)** | 12.9 | 15.7 | 19.6 |
| **Relative density (%)** | 76 | 89 | 100 |
| **Stem density ($stems \cdot ha^{-1}$)** | 77 | 94 | 87 |
| **Number of Transects** | 31 | 24 | 33 |
| **Average length of transects (m)** | 44 | 41 | 36 |
| **Number of 1 $m^2$ groundcover plots** | 224 | 161 | 232 |

Field data collection

I followed the Strahler method (Strahler 1952) for stream classification. Using a combination of maps generated through the National Hydrography Dataset (NHD) stream layers and digital terrain models of between 1 – 10 m resolution, and field verification, headwater streams that included first- and second-order streams were randomly selected for sampling within each site. Sampling took place along a reach of a selected stream which ranged between 122 – 305 m (400 - 1000 ft.). The length of a stream reach depended on the stream length of that stream. Sampling in headwater stream reaches was carried out across 11 streams at Site 1 and eight at Site 2 during July and August 2017 and 11 headwater streams at Site 3 during July 2019. In total, 30 headwater stream reaches were sampled across the Northeastern region (Table 2.1).

*Overstory sampling:*

I collected data on riparian forest composition and structure using circular overstory plots (Figure 2.2) of 7.32 m (24 ft. or 1/24 ac plot) radius for live trees with a diameter-at-breast-height (dbh) of ≥ 12.7 cm (≥5 in.) and subplots plots of 2.07 m radius (6.8 ft. or 1/300 ac plot) for live trees with a dbh between 2.54 – 12.7 cm (1 – 5 in.). Both types of overstory plots were located 12.80 m (42 ft.) away from the edge of the stream, perpendicular to direction of flow (Jayasuriya et al. 2018), where the subplot was nested within the larger radius overstory plot thereby sharing plot center. The number of overstory plots completed on each stream reach was limited to 10 plots if the standard error per plot was 20 % or less around the mean basal area at $\alpha = 0.05$. If not, plots were added until reaching this threshold (Munsell and Germain 2007). Plots were placed on either side of the stream in either an alternate or opposite configuration depending on the length of the reach being sampled. Thus, distance between two plots on the same side of the stream was either 20 m (66 ft. or 1 chain) or 40 m (132 ft. or 2 chains) apart based on the plot configuration along the streams.

*Understory sampling:*

For each stream reach sampled, three line transects (Figure 2.2) were placed perpendicular to streamflow direction on either side of the stream extending from stream bank to the upland forest. The transects were located at three of the overstory plot positions selected at random among all those within the stream reach. The length of each transect varied with topographic barriers such as rocks and boulders encountered in the field and also due to overlapping micro catchments of adjacent streams. Transect lengths varied from 24.4 m (80 ft.) to 48.8 m (160 ft.), and average transect length was 40 m (130 ft.). Understory vegetation was measured in 1 $m^2$ circular plots that were set along the line transect beginning at the stream bank (0 m) and spaced at 6.1 m (20 ft.) intervals (Figure 2.2). Due to the transects' variable length, the number of plots per transect ranged from 4 to 9, with an average of 7 plots per transect. Species and percent ground cover of herbaceous vascular plants (non-woody, ferns, and grasses), woody shrubs and tree seedlings were recorded within these plots. Mosses and other non-vascular plants were not surveyed. Grasses and sedges were identified to their genus level. Unidentified species in the field were collected and documented using digital photos before transporting them to the lab where they were identified. Species were identified using multiple flora guides, including the Peterson Field Guide for Wildflowers (Peterson and McKenny 1998), USDA plant database (USDA 2019) and a mobile application named PlantNet ("Pl@ntNet Identify" 2019). Percent cover was estimated in 1 % increments from 0 – 10 %, and in 5 % increments up to 100 %.

Percent canopy closure was calculated for every understory plot using a point transect sampling design along the understory transect using a GRS densitometer (Geographic Resource Solutions 2008; Adikari and MacDicken 2015). Measurements of canopy presence/absence was taken at every 3.05 m (10 ft.) along the transect. Therefore, three points were used to calculate the canopy closure for each understory plot except for the first understory plot at stream edge. Only two points were used to calculate percent canopy closure for the first understory plot.

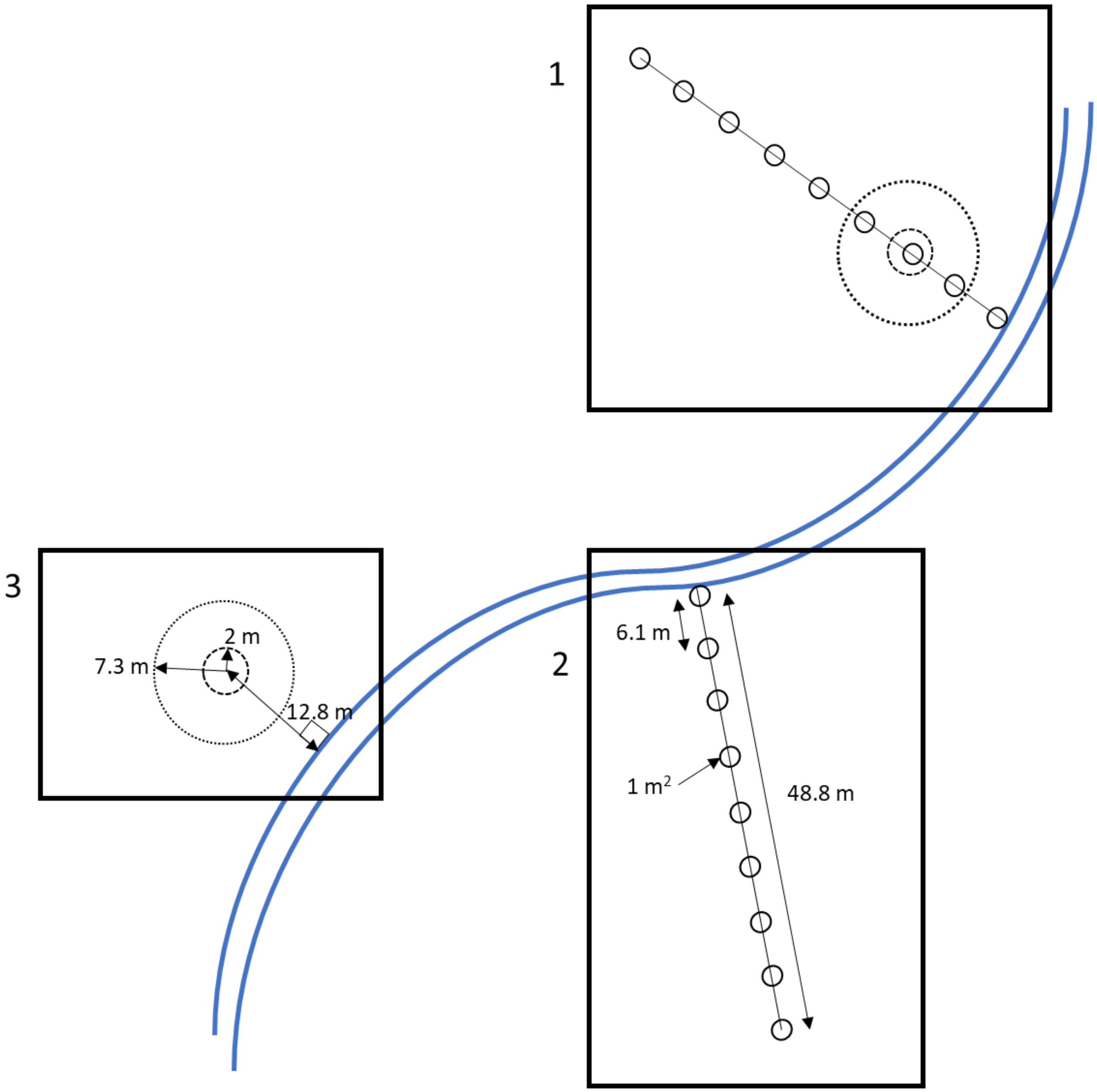


**Figure 2.2: Plot configuration of overstory, understory, and groundcover plots. Box 1 represents the plot layout of all three types of plots overlaid on each other as done in the field. Box 2 represents a descriptive diagram of the understory plot layout along a transect with a maximum length of 48.8 m (160 ft.) lying perpendicular to streamflow. Groundcover plots of 1 m$^2$ are placed at a 6.1 m (20 ft.) spacing along the transect. Box 3 represents a descriptive diagram of the plot layout of an overstory and understory plot sharing the same plot center. The plot center lies 12.8 m (42 ft) from the stream edge, perpendicular to streamflow. The overstory plot radius is 7.3 m (24 ft.) while the understory plot radius is 2 m (6.8 ft.).**

Data Analysis

*Generalized linear mixed models for species richness*

Generalized linear mixed models were used to predict plot level understory species richness (counts) across all sites using a Poisson error distribution with natural logarithm link function (Equation 2.1). The fixed predictors were distance from the stream as either raw scale or natural log transformed, percent canopy closure, and their interaction. Random effects included an intercept term for site and the site-specific slope effect of distance on richness. The model form for the fixed effects was:

$$ln(\hat{\mu}) = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_3 \qquad [2.1]$$

where $\hat{\mu}$ is the predicted count of species richness on the outcome variable given the specific values on the predictors $X_1$, $X_2$ and $X_3$.

A set of eight candidate models were compared with different combinations of fixed effects for canopy closure, distance from stream, and their interaction; these included an intercept only, or null model (Table 2.2). All models included a random intercept random effect for site. Six of the eight models were run with a random slope effect for Distance or ln(Distance) nested within site. A random slope for the model Sp.Rich ~ Perc CC was not included as it was assumed that the canopy closure gradient did not vary by site. This model and the intercept-only null model included only a random intercept.

**Table 2.2: Candidate mixed effects models for the fixed effects for distance or ln(distance) from stream and canopy closure, and random effects for site.**

| Fixed Effects Model | Random intercept | Random slope effect |
|---|---|---|
| Sp.Rich ~ Intercept (Null model) | Site | N/A |
| Sp.Rich ~ Distance | Site | Distance |
| Sp.Rich ~ Perc_CC | Site | N/A |
| Sp.Rich ~ Distance + Perc_CC | Site | Distance |
| Sp.Rich ~ Distance + Perc_CC + (Distance : Perc_CC) | Site | Distance |
| Sp.Rich ~ ln(Distance) | Site | ln(Distance) |
| Sp.Rich ~ ln(Distance) + Perc_CC | Site | ln(Distance) |
| Sp.Rich ~ ln(Distance) + Perc_CC + ln(Distance) : Perc_CC) | Site | ln(Distance) |

The best model was selected using a Likelihood-based approach by comparing Akaike Information Criteria (AIC) values between all candidate models (Burnham and Anderson 2001). A threshold criterion of delta AIC $\geq 2$ was used to distinguish between alternative models. Delta AIC is computed as the difference between the smallest AIC value and other AIC values. The *glmer* package (Bates et al. 2015) developed for the R statistical software (R Core Team 2019) was used for all model development and evaluation.

*Species composition analysis with distance from stream*

A Canonical Discriminant Analysis (CDA) and an indicator species analysis was used to understand how understory composition varied with distance from the stream. For the CDA, rare species were excluded to avoid masking vegetation data patterns: these were classified as those occurring in less than 5 % of sampling plots within a given site (Goebel et al. 2003). After rare species were excluded, a total of 32 species variables were tallied for all sites. A stepwise discriminant analysis was run to further reduce the species dimension. This allowed us to identify significant species for the CDA to compare understory compositional differences or similarities at each lateral sampling distance from the stream. For the CDA, SAS/STAT software, Version 9.4 for Windows (Copyright © [2013] SAS Institute Inc.) was used.

The Dufrene-Legendre Indicator Species Analysis (Dufrêne and Legendre 1997) was used to identify indicator species or groups of indicator species for each plot distance to the stream. This analysis calculates the indicator value (fidelity and relative abundance) of species in clusters or types (De C´aceres 2013). Presence/absence information of all species were tallied in every plot for each site. Within the analysis, clusters were defined by plot distance groups (i.e., distance from stream). The R statistical software (R Core Team 2019) was used for the indicator species analysis.

**Results**

<u>Species composition</u>

*Overstory Composition*

The riparian area at Site 1 was characterized by a BA of 32.8 $m^2$/ha (143 $ft.^2$/ac.), relative density of 76 % and a QMD of 12.9 cm (5.1 in.) (Table 2.1). The overstory was dominated by yellow birch (*Betula alleghaniensis* Britton) and the understory was dominated by red spruce (*Picea rubens* Sarg.). At Site 2, riparian areas had a BA of 30.3 $m^2$/ha (132 $ft.^2$/ac.), relative density of 89 % and QMD of 15.7 cm (6.2 in.). The overstory was dominated by American beech (*Fagus grandifolia* Ehrh.), sugar maple (*Acer saccharum* Marsh.), and yellow birch while the understory was dominated by American beech. Riparian areas at Site 3 had a BA of 40 $m^2$/ha (174 $ft.^2$/ac.), relative density of 100 % and a QMD of 19.6 cm (7.7 in.). Overstory species were mainly composed of eastern hemlock (*Tsuga canadensis* (L.) Carrière), sugar maple, and white ash (*Fraxinus americana* L.) while the understory was dominated by American beech and sugar maple.

*Understory Plant Species Composition*

Spinulose woodfern (*Dryopteris carthusiana* (Vill.) H.P. Fuchs) was the most common understory herbaceous plant, with percent occurrence in plots ranging between 35 – 57 % across the three sites (Table 2.3). Other herbaceous plants common to all three forested sites were Canada mayflower (*Maianthemum canadense* Desf.) (5 – 16 %), mountain wood sorrel (*Oxalis montana* Raf.) (9 – 32 %), and red trillium (*Trillium erectum* L.) (5–12 %). Of the woody plants in the understory, the most abundant were seedlings of maple (*Acer* sp.) (43 – 83 % across all sites), American beech (*Fagus grandifolia* Ehrh.) (13 – 41 %, at two sites) and white ash (*Fraxinus americana* L.) seedlings (46 %, at one site). Hobblebush (*Viburnum lantanoides* Michx.) was common at Sites 1 and 2 (35 – 50 %).

**Table 2.3: Common names of species and their percentage occurrence (%) observed at each site. Rare species are not included. See Appendix 2A for a full list of observed species for all sites.**

| Common name | Site 1 | Site 2 | Site 3 |
|---|---|---|---|
| American beech seedlings | | 41 | 13 |
| balsam fir seedlings | 11 | | |
| black cherry seedling | | 5 | 14 |
| bluebead | 20 | | |
| Canada mayflower | 13 | 5 | 16 |
| dewberry sp. | | | 13 |
| grass sp. | | 8 | 9 |
| heartleaf foam flower | | 11 | |
| hobblebush | 50 | 35 | |
| Jack in the pulpit | | | 18 |
| long beechfern | | 5 | |
| maple sp. seedlings | 43 | 83 | 45 |
| mountain maple | 5 | | |
| mountain wood sorrel | 32 | 18 | 9 |
| New York fern | | | 10 |
| partridgeberry | | | 28 |
| red spruce seedlings | 13 | 6 | |
| red trillium | 5 | 9 | 12 |
| shining club moss | 24 | 35 | 7 |
| small enchanter's nightshade | | | 5 |
| Smooth solomon's seal | | 12 | |
| spinulose woodfern | 57 | 51 | 35 |
| spotted jewelweed | | | 6 |
| starflower | 5 | 5 | |
| violet sp. | | 12 | 9 |
| white ash seedlings | | | 46 |
| white wood aster | | | 11 |
| wild sarsaparilla | 9 | | 9 |
| yellow birch seedlings | 10 | | |

Generalized Linear Mixed Models

The fitted generalized linear mixed models were compared using their AIC values, where ΔAIC differences ranged from 1 – 38.3 (Table 2.4). The three top models do not show substantial differences among each other as shown in the low AIC difference values (<2 ΔAIC).

**Table 2.4: Model diagnostics for random intercept and slope models for the generalized linear mixed models run on all sites.**

| Model (fixed effects only) | AIC | ΔAIC | Akaike weight | Cumulative weight |
|---|---|---|---|---|
| Sp.Rich ~ ln(Distance) + Perc_CC + ln(Distance):Perc_CC | 2704.6 | 0.0 | 0.458 | 0.458 |
| Sp.Rich ~ ln(Distance) | 2705.6 | 1.0 | 0.278 | 0.736 |
| Sp.Rich ~ ln(Distance) + Perc_CC | 2705.7 | 1.1 | 0.264 | 1 |
| Sp.Rich ~ Distance + Perc_CC + Distance:Perc_CC | 2724.1 | 19.5 | <0.001 | 1 |
| Sp.Rich ~ Distance | 2725.1 | 20.5 | <0.001 | 1 |
| Sp.Rich ~ Distance + Perc_CC | 2725.7 | 21.1 | <0.001 | 1 |
| Sp.Rich ~ Intercept | 2741.2 | 36.6 | <0.001 | 1 |
| Sp.Rich ~ Perc_CC | 2742.9 | 38.3 | <0.001 | 1 |

Given their equivalency, the model with ln(Distance) as the only fixed effect was selected to be the most parsimonious of the candidate model set. Both the intercept (1.635 ± 0.067 SE) and slope parameter (-0.080 ± 0.025 SE) for this top model were significant at $\alpha = 0.05$ with the negative parameter estimate for ln(Distance), indicating a decrease in species richness with increasing distance from the streambank. An untransformed equation (Equation 2) for the preferred model was generated using its parameter estimates of the fixed effects.

$$Species\ Richness = e^{1.635} \times Distance^{-0.08} \qquad [2]$$

Random effects for the three sites ranged from 1.53 to 1.74 for the intercept and -0.04 to -0.12 for the random slope effect of ln(distance) on species richness (Appendix 2B). These indicate that individual richness models for all sites varied by <2 species at the stream bank (i.e., the intercept values) and that richness decreased with greater distance (i.e., all slopes were negative).

Empirical species richness was highest closest to the stream with a range of species count from 1–11 species/m$^2$ (Figure 2.3). The generalized linear mixed model for species richness across all the sites predicts an exponentially decreasing relationship with distance from the stream, with a predicted richness value of 5.4 species/m$^2$ at the edge of the streambank. Richness decreases the most within the first 10 – 15 m to approximately 4.2 species/m$^2$ at the 12.2 m (40 ft.) plots, and a more gradual decrease to approximately 3.7 species/m$^2$ at greater distances (Figure 2.3).

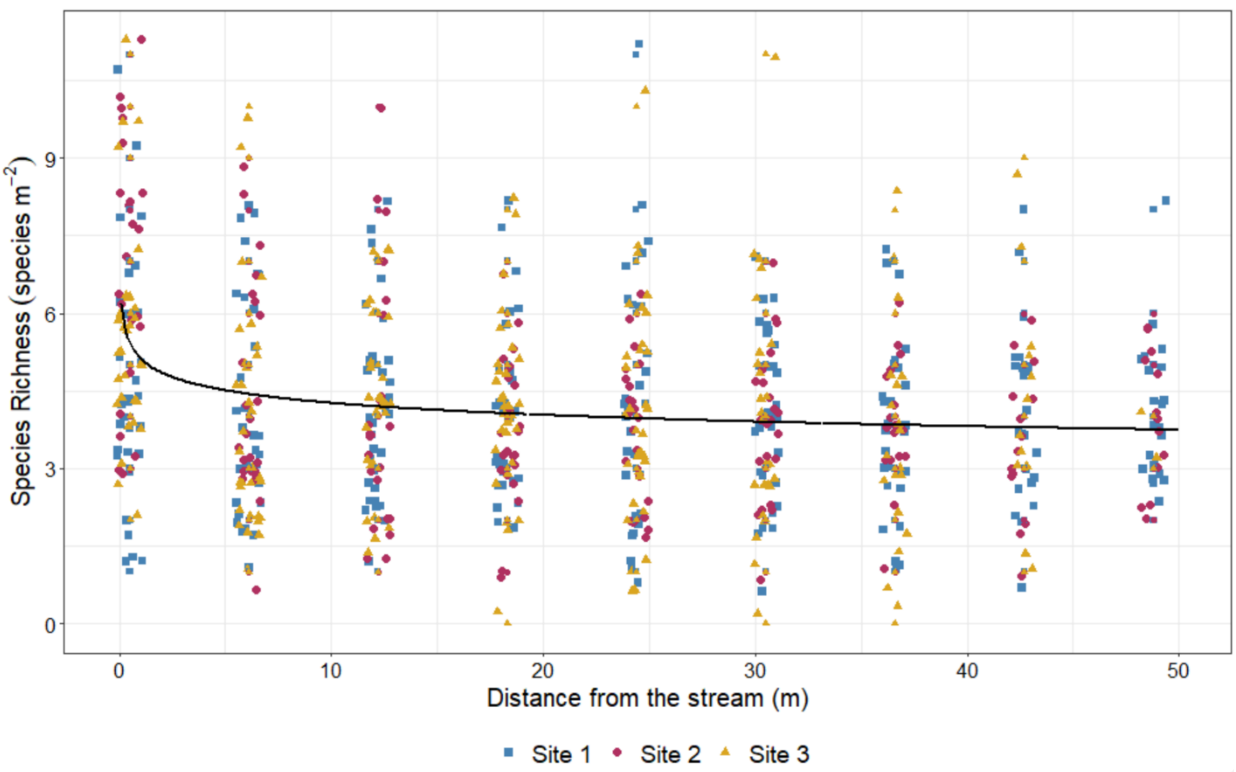


**Figure 2.3: Jittered scatter plot of species richness (counts) at regular lateral distances moving perpendicularly away from the stream. The predicted values (represented by the curved) are generated from the preferred model selected for predicting species richness at all four sites. The fitted model formula and parameter values were: Species Richness $= e^{1.635} \times \text{Distance}^{-0.08}$**

Canonical Discriminant Analysis

The stepwise discriminant analysis revealed four significant species out of the 32 species: spotted jewelweed (*Impatiens capensis* Meerb.), Jack in the pulpit (*Arisaema triphyllum* (L.) Schott), heartleaf foamflower (*Tiarella cordifolia* L.), and red spruce seedlings (*Picea rubens* Sarg.). The first canonical function of the CDA for the four species was significant for separating the difference between lateral distances from the stream (Wilks' Lambda p-value < 0.0001). The first two canonical functions accounted for 96 % of the total variation (the first 81 % and second 15 %). According to the probability values of the Mahalanobis Distance for squared distance, there was a significant separation between the lateral distance closest to the stream (0 m) and all other lateral distances. This means that species composition observed at the stream's edge was significantly different from the rest of the lateral distances from the stream. There were no significant differences between the remaining lateral distances.

Indicator Species Analysis

The analysis identified heartleaf foamflower (*Tiarella cordifolia* L.), grass sp., spotted jewelweed (*Impatiens capensis* Meerb.), Jack in the pulpit (*Arisaema triphyllum* (L.) Schott), sensitive fern (*Onoclea sensibilis*), and longbeech fern (*Phegopteris connectilis*) as indicator species in plots closest to the stream (0 m) (Table 2.5). Spinulose woodfern (*Dryopteris Carthusian*) was identified as an indicator furthest from the stream. From the six identified indicators, only spotted jewelweed and sensitive fern are commonly known facultative wetland indicator species (USDA 2019). However, spotted jewelweed and sensitive fern have very low indicator values (at a scale between 0 – 1) at the 0 m distance (Table 2.5). Of the total 617 plots surveyed across the three sites, spotted jewelweed and sensitive fern occurred in only 14 and 6 plots, respectively. Spinulose woodfern is classified as a facultative wetland species within the Northeastern US. (USDA 2019), even though it was identified as an indicator for the most distant plots along the transects in this study.

**Table 2.5: Indicator species significant at α=0.05 for various distances from stream at each study site.**

| Common name | Scientific name | Distance (m) | Indicator value | p value |
|---|---|---|---|---|
| heartleaf foamflower | *Tiarella cordifolia* L. | 0 | 0.104 | 0.0002 |
| Grass sp. | | 0 | 0.055 | 0.0051 |
| spotted jewelweed | *Impatiens capensis* Meerb. | 0 | 0.044 | 0.0202 |
| Jack in the pulpit | *Arisaema triphyllum* (L.) Schott | 0 | 0.039 | 0.0478 |
| longbeech fern | *Phegopteris connectilis* | 0 | 0.031 | 0.0494 |
| sensitive fern | *Onoclea sensibilis* | 0 | 0.028 | 0.0464 |
| spinulose woodfern | *Dryopteris Carthusian* | 48.8 | 0.104 | 0.0432 |

**Discussion**

Observing Lateral Distance Thresholds

Despite the complex and subtle nature of understory plant community patterns in mesic forests, this study documented that local species richness is highest at the riparian streambank and declines significantly with distance from the stream. This relationship, though subtle, is consistent across a diversity of forest types. The greatest reduction in species richness occurred within the first three plot distances (0 – 12 m), and given the constraints of plot spacing, it can be estimated that a threshold distance for plant species richness lies between 6 – 12 m (20 – 40 ft.) away from the stream. Species richness closest to the stream averaged 5 - 6 species/$m^2$ across all sites with an observed maximum of 11 species/$m^2$, and the most parsimonious model predicting species richness indicates an exponential decrease with distance. Within 12 m (40 ft.) of the streambank, mean species richness decreased by approximately 70 %, and remained in the range of 3.7 – 4.2 species/$m^2$ up to 50 m (~160 ft.) from the stream.

A potential reason for the locally higher riparian community biodiversity is the strong influence of hydrologic and biogeochemical gradients adjacent to streams (Naiman et al. 1993; Naiman et al. 2005). These ecotones also lie at the most downslope positions within local hydraulic gradients and thus receive fluxes of nutrients and organic matter (Burt et al. 2002) which favor vegetation growth. Also, these land strips along headwater streams endure infrequent but intense disturbances such as landslides and debris flow, particularly in steeper landscapes during peak flow events (Swanson et al. 1998). These infrequent disturbances may also be a factor for higher levels of species richness within riparian areas compared to the upland forests (Richardson et al. 2005; Naiman et al. 2005).

Compared to dryland regions, mesic forests generally have a well-balanced distribution of moisture throughout the landscape outside of the immediate riparian zone. When in combination with a closed canopy structure and well-distributed summer rainfall conditions, forest overstory communities in this biome rarely show distinction between riparian and upland areas (Richardson et al. 2005). These conditions likely contribute to why a pronounced drop in species counts was not evident with distance from the stream. From their study in the Nantahala National Forest in the Southern Appalachians, Clinton et al. (2010) suggests that understory vegetation is not a good parameter for defining riparian zone widths along first-order streams. They did not detect any significant difference in diversity, species richness or percent canopy cover in the understory comprising of perennial forbs, tree seedlings, woody shrubs, woody vines, ferns, graminoids. Thus, the mesic forest conditions along with the relatively high (between 60 - 100 %) canopy closure at sites may be masking ecological drivers that influence species richness patterns along lateral gradients from the stream.

The greatest reduction in species richness occurred within the first 12 m of the streambank, and further reductions at greater distances were more subtle. These results are consistent with other research findings in the region. In a similar study conducted in the Adirondack region of New

York, Dieterich et al. (2006) reported species richness at the stream edge to be significantly higher than in upland plots, and richness values gradually decreased until 12 m from the stream. Hagan et al. (2006) observed a narrow but detectable riparian plant community associated with 15 first-order streams in the western mountainous region of Maine. They reported a higher species richness and a significant species compositional difference within 5 m of the stream when compared to the rest of the plots along transects extending to 45 m. Quinby et al. (2000) suggests that a 16.5 m average width for a riparian zone is justified based on the decreasing abundance of two indicator species in the Cassels-Rabbit Lakes area of the Temagami region of Ontario. A width of between 10 – 30 m beyond the high-water mark captured 90 % of the streamside plant species along third- and fourth-order (mid-order) streams in a study by Spackman and Hughes (1995) in Vermont.

<u>Riparian Understory Species Composition</u>

Understory species diversity is highest in riparian areas (Naiman et al. 2005). A total of 95 understory species was observed across all sampling sites. Species composition can serve as an important variable in identifying riparian areas (Quinby et al. 2000; Spackman and Hughes 1995), because wetland indicator species would be associated with these areas. The distance based CDA revealed that species composition at 0 m distance (closest to the stream) was significantly different from species composition at other lateral distances. Even though six indicator species were identified for all sites, only *Onoclea sensibilis* L. (sensitive fern) and *Impatiens capensis* Meerb. (spotted jewelweed) are known facultative wetland species (USDA 2019). Due to the low indicator values reported through the assessment of these two species at plots closest to the stream, they are of limited use as indicator species.

The use of indicator species to identify unique riparian areas has mixed results in other regions. Pabst and Spies (1998) reported that general vegetation patterns within riparian areas along the Coastal Range of Oregon are highly variable and sometimes indistinct. Other studies based on riparian plant communities, however, did observe several riparian indicator species (Quinby et al. 2000; Hagan et al. 2006). Quinby et al. (2000) identified Joe-Pye weed (*Eupatorium maculatum*) and northern beech fern (*Phegopteris connectilis*) as riparian indicators based on their decrease in abundance from the streambank in their study at Cassels-Rabbit Lakes area of the Temagami region of Ontario, Canada. In a study conducted in western Maine, Hagan et al. (2006) observed that the proportion of wetland herbaceous species decreased with distance from the stream, whereas the proportion of forest specialist herbaceous plants increased. They also identified 23 riparian indicator species within a proximity of 5 m from the stream.

If higher species richness is associated with the abundant availability of moisture and nutrients closer to the stream, observations of this study suggest that a zone of 6 - 12 m (20 – 40 ft.) proximity to the stream — as circumscribed by the particular plot spacing in this study — can be categorized as a functional riparian area where a concentration of plant-nutrient and plant-hydrologic interactions take place. In a study investigating the structural and functional characteristics of riparian areas within the Southern Appalachians in North Carolina, Clinton et al. (2010) found that the majority of the parameters they tested transitioned between 10 and 20 m from the stream. However, they did not observe a significant difference in species diversity, richness or percent cover in the ground layer with distance from the stream. Other authors have suggested that the riparian boundary ranges from 5 – 16.5 m based on herbaceous plant species distributions (Quinby et al. 2000; Hagan et al. 2006; Dieterich et al. 2006). It is important to understand that these predictions are only based on the micro-environmental changes observed through groundcover or understory vegetation. Other riparian provisioning services such as shade for temperature regulation, the input of coarse woody debris into the stream, and in-stream habitat for fish and macroinvertebrates may require widths extending to up to 30 m and beyond (Sweeney and Newbold 2014b). Based on the plot spacing in this study design, results indicate that 12 m represents an important threshold for headwater streams. However, if taken into account the uncertainties involved in data collection, fixed lateral spacing between plots and also to be consistent with studies in similar mesic biomes, a distance ranging from 6 – 12 m (20 – 40 ft.) from the stream can be considered as an appropriate margin for high plant species richness. Within this zone, any timber harvesting that may be allowed, and other riparian management activities should be carried out with the primary emphasis on riparian area protection.

**Conclusion**

This study identified a threshold riparian distance of locally high plant species richness extending up to 6 – 12 m (20 – 40 ft.) from streambanks of headwater streams within Northeastern forests. Understory species composition closest to the stream differed significantly from that of all positions at greater lateral distances. Six taxa were identified as floral indicators of streamside positions, and of these, two (*Impatiens capensis* and *Onoclea sensibilis*) were categorized as facultative wetland indicators by US Federal agencies. However, due to their relatively low indicator values, their utility may be limited. Because headwater streams are disproportionately affected by forest management activities, and riparian protection guidelines are rarely based on locally available data, evidence-based studies such as the current research should guide regional riparian management to ensure that these areas continue to provide ecosystem services now and into the future.

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**Appendix 2A**

Plant species list observed in groundcover plots at Site 1 (Hubbard Brook Experimental Forest), Site 2 (Huntington Wildlife Forest), Site 3 (Heiberg Memorial Forest and Cuyler Hill State Forest).

| **Common name** | **Scientific name** | **Symbol** | **Site 1** | **Site 2** | **Site 3** |
|---|---|---|---|---|---|
| Allegheny monkeyflower | *Mimulus ringens* L. | MIRI | | | • |
| American basswood | *Tilia americana* L. | TIAM | | | • |
| American beech | *Fagus grandifolia* Ehrh. | FAGR | • | • | • |
| American witchhazel | *Hamamelis virginiana* L. | HAVI4 | | | • |
| anemone sp. | *Anemone* L. | ANEMO | | • | |
| Appalachian barren strawberry | *Waldsteinia fragarioides (Michx.) Tratt.* | WAFR | | | • |
| ashleaf maple | *Acer negundo* L. | ACNE2 | • | | |
| avens sp. | *Geum* | GEUM | | | • |
| balsam fir | *Abies balsamea* (L.) Mill. | ABBA | • | • | |
| black cherry | *Prunus serotina* Ehrh. | PRSE2 | | • | • |
| blackberry sp. | *Rubus* | RUBUS | | | • |
| blisterwort | *Ranunculus recurvatus Poir.* | RARE2 | | | • |
| bluebead | *Clintonia borealis* (Aiton) Raf. | CLBO3 | • | • | • |
| broadleaf enchanter's nightshade | *Circaea lutetiana L.* | CILU | | | • |
| broadleaf helleborine | *Epipactis helleborine (L.) Crantz* | EPHE | | | • |
| bunchberry dogwood | *Cornus canadensis* L. | COCA13 | • | • | |

| Common name | Scientific name | Symbol | Site 1 | Site 2 | Site 3 |
|---|---|---|---|---|---|
| Canada goldenrod | *Solidago canadensis* L. | SOCA6 | | | • |
| Canada mayflower | *Maianthemum canadense* Desf. | MACA4 | • | • | • |
| chokecherry | *Prunus virginiana* L. | PRVI | | | • |
| Christmas fern | *Polystichum acrostichoides* (Michx.) Schott | POAC4 | | | • |
| cinnamon fern | *Osmunda cinnamomea* L. | OSCI | | | • |
| climbing nightshade | *Solanum dulcamara L.* | SODU | | | • |
| common blackberry | *Rubus allegheniensis* | RUAL | | | • |
| common buckthorn | *Rhamnus cathartica* L. | RHCA3 | | | • |
| common gypsyweed/ Common Speedwell | *Veronica officinalis* L. | VEOF2 | | | • |
| common hepatica | *Anemone hepatica* L. | HENOO | | | • |
| common lady fern | *Athyrium filix-femina* (L.) Roth | ATFI | | | • |
| common nipplewort | *Lapsana communis L.* | LACO3 | | | • |
| creeping buttercup | *Ranunculus repens* L. | RARE3 | | | • |
| dew berry | *Rubus sp.* | RUBUS | • | • | |
| dogwood sp. | *Cornus* L. | CORNU | | | • |
| dwarf cinquefoil | *Potentilla recta* L. | PORE5 | | | • |
| Eastern hemlock | *Tsuga canadensis* (L.) Carrière | TSCA | • | • | • |
| Eastern waterleaf | *Hydrophyllum virginianum* L. | HYVI | | | • |
| European lily of the valley | *Convallaria majalis* L. | COMA7 | • | | |

| Common name | Scientific name | Symbol | Site 1 | Site 2 | Site 3 |
|---|---|---|---|---|---|
| fragrant bedstraw | *Galium triflorum* Michx. | GATR3 | | • | |
| garlic mustard | *Alliaria petiolata* M. Bieb. | ALPE4 | | • | • |
| grass sp. | | GRASS | • | • | • |
| hawthorn sp. | *Crataegus* sp. | CRATA | | | • |
| hay-scented fern | *Dennstaedtia punctilobula* (Michx.) T. Moore | DEPU2 | | | • |
| heartleaf foamflower | *Tiarella cordifolia* L. | TICO | | • | • |
| herb bennet | *Geum urbanum* L. | GEUR | | | • |
| honeysuckle sp. | *Lonicera L.* | LONIC | | • | |
| hophornbeam | *Ostrya virginiana* (Mill.) K. Koch | OSVI | | | • |
| Indian cucumber | *Medeola virginiana* L. | MEVI | | • | • |
| Jack in the pulpit | *Arisaema triphyllum* (L.) Schott | ARTR | | • | • |
| long beechfern | *Phegopteris connectilis* (Michx.) Watt | PHCO24 | • | • | |
| mapleleaf viburnum | *Viburnum acerifolium* L. | VIAC | • | | • |
| marsh violet | *Viola palustris* L. | VIPA4 | | | • |
| mountain maple | *Acer spicatum* Lam. | ACSP2 | • | | • |
| mountain wood sorrel | *Oxalis montana* Raf. | OXMO | • | • | • |
| multiflora rose | *Rosa multiflora* Thunb. | ROMU | | | • |
| New York fern | *Thelypteris noveboracensis* (L.) Nieuwl. | THNO | | • | • |
| northern wild rasin | *Viburnum nudum* L. | VINU | • | | |

| **Common name** | **Scientific name** | **Symbol** | **Site 1** | **Site 2** | **Site 3** |
|---|---|---|---|---|---|
| Norway spruce | *Picea abies* (L.) Karst. | PIAB | | | • |
| partridge berry | *Mitchella repens* L. | MIRE | • | • | • |
| prickly current | *Ribes lacustre* (Pers.) Poir. | RILA | • | | |
| red maple | *Acer rubrum* L. | ACRU | • | • | • |
| red raspberry | *Rubus idaeus* L. | RUID | | | • |
| red spruce | *Picea rubens* Sarg. | PIRU | • | • | |
| red trillium | *Trillium erectum* L. | TRER3 | • | • | • |
| Robert geranium | *Geranium robertianum* L. | GERO | | | • |
| rock polypody | *Polypodium virginianum* L. | POVI7 | • | • | |
| roundleaf geranium | *Geranium rotundifolium L.* | GERO3 | | | • |
| roundleaf yellow violet | *Viola rotundifolia* Michx. | VIRO3 | | | • |
| rue anemone | *Thalictrum thalictroides* (L.) Eames & B. Boivin | THTH2 | | • | |
| sassafras | *Sassafras albidum* (Nutt.) Nees | SAAL5 | | | • |
| sedge sp. | *Carex* | SEDGE | • | | • |
| sensitive fern | *Onoclea sensibilis* L. | ONSE | | | • |
| shining club moss | *Huperzia lucidula* (Michx.) Trevis. | HULU2 | • | • | • |
| small enchanter's nightshade | *Circaea alpina* L. | CIAL | | | • |
| smooth Solomon's seal | *Polygonatum biflorum* (Walter) Elliott | POBI2 | • | • | • |
| speckled alder | *Alnus incana* (L.) Moench | ALINR | | | • |

| Common name | Scientific name | Symbol | Site 1 | Site 2 | Site 3 |
|---|---|---|---|---|---|
| spinulose wood fern | *Dryopteris carthusiana* (Vill.) H.P. Fuchs | DRCA11 | • | • | • |
| spotted jewelweed | *Impatiens capensis* Meerb. | IMCA | | • | • |
| staghorn club moss | *Lycopodiella cernua* (L.) Pic. Serm. | LYCE2 | | | • |
| starflower | *Trientalis borealis* Raf. | TRBO2 | • | • | • |
| stinging nettle | *Urtica dioica* L. | URDI | | | • |
| striped maple | *Acer pensylvanicum* L. | ACPE | • | | • |
| sugar maple | *Acer saccharum* Marsh. | ACSA3 | • | • | • |
| sweetscented bedstraw | *Galium odoratum* (L.) Scop. | GAOD3 | | | • |
| threeleaf goldthread | *Coptis trifolia* (L.) Salisb. | COTR2 | • | • | |
| violet sp. | *Viola* sp. | VIOLA | | • | • |
| Virginia strawberry | *Fragaria virginiana* Duchesne | FRVI | | | • |
| waxflower shinleaf | *Pyrola elliptica* Nutt. | PYEL | | | • |
| white ash | *Fraxinus americana* L. | FRAM2 | | • | • |
| white rattlesnakeroot/white lettuce | *Prenanthes alba* L. | PRAL2 | • | | • |
| whorled wood aster | *Oclemena acuminata* (Michx.) Greene | OCAC | • | • | • |
| wild basil | *Clinopodium vulgare L.* | CLVU | | | • |
| wild sarsparilla | *Aralia nudicaulis* L. | ARNU2 | • | • | • |
| witch hobble/hobble bush | *Viburnum lantanoides* Michx. | VILA11 | • | • | • |
| wood strawberry | *Fragaria vesca* L. | FRVE | | | • |

| Common name | Scientific name | Symbol | Site 1 | Site 2 | Site 3 |
|---|---|---|---|---|---|
| yellow birch | *Betula alleghaniensis* Britton | BEAL2 | • | • | • |
| zigzag goldenrod | *Solidago flexicaulis* L. | SOFL2 | | • | • |

**Appendix 2B**

Parameter estimates for Intercept and ln(Distance) of the random effects (Sites) for the preferred generalized linear mixed model.

| Groups | Intercept | ln(Distance) |
|---|---|---|
| Site 1 | 1.528 | -0.040 |
| Site 2 | 1.740 | -0.120 |
| Site 3 | 1.639 | -0.082 |

**Appendix 2C**

Canonical Correlation Analysis

A canonical correlation analysis (CCA) was performed to assess the correlation of environmental and species variables for the four forests. For the analysis, each site included the following six environmental variables: lateral distance from the stream, percent canopy closure, total BA, percent conifer BA, percent hemlock BA, and overstory QMD. The multivariate analysis was performed using SAS/STAT software, Version 9.4 for Windows (Copyright © [2013] SAS Institute Inc.) and R statistical software to draw the plots (R Core Team 2019).

Based on the redundancy analysis, the CCA was not successful in predicting variations observed in species variables using environmental variables within the riparian areas of the sampling sites.

**Table 2C-1: Canonical redundancy analysis showing the percentage variance explained for each set of environmental and species variables by six linear canonical functions of the opposite variable. The number of linear canonical functions is determined by the smallest number of variables included in either set of variables.**

| Site | Standardized variance of environmental variables explained by canonical species variables | Standardized variance of species variables explained by canonical environmental variables |
|---|---|---|
| Site 1 | 10.4 % | 6.1 % |
| Site 2 | 14.7 % | 5.5 % |
| Site 3 | 45.5 % | 11.3 % |
| Site 4 | 34.2 % | 8.5 % |

The CCA revealed that the first two canonical correlations of Sites 2 (Huntington Wildlife Forest), 3 (Heiberg Memorial Forest) and 4 (Cuyler Hill State Forest) were statistically significant (Wilk's Lambda p-value < 0.001) and together explained 64%, 55%, and 62% of the total variation between the two sets of variables, respectively.

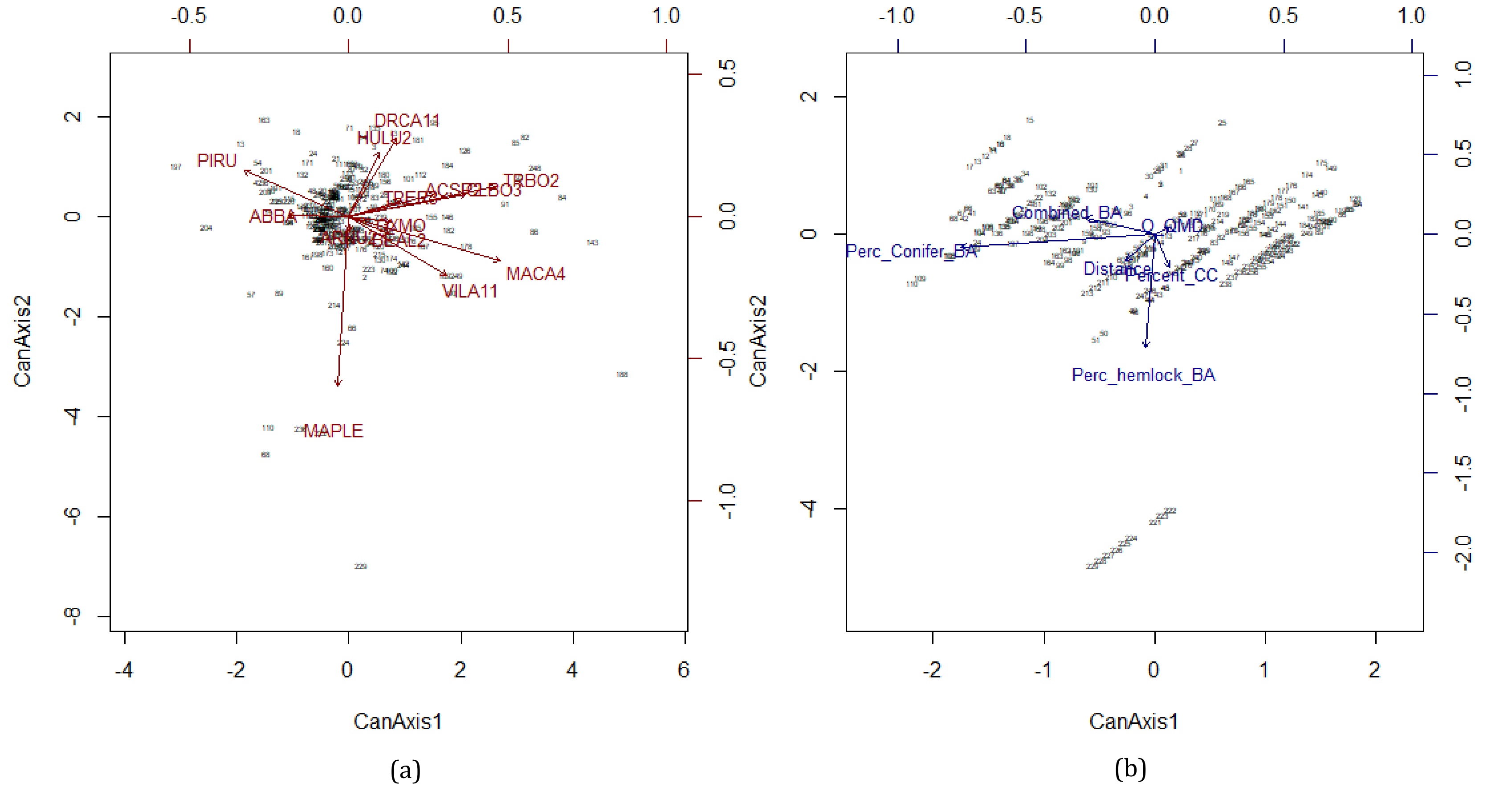
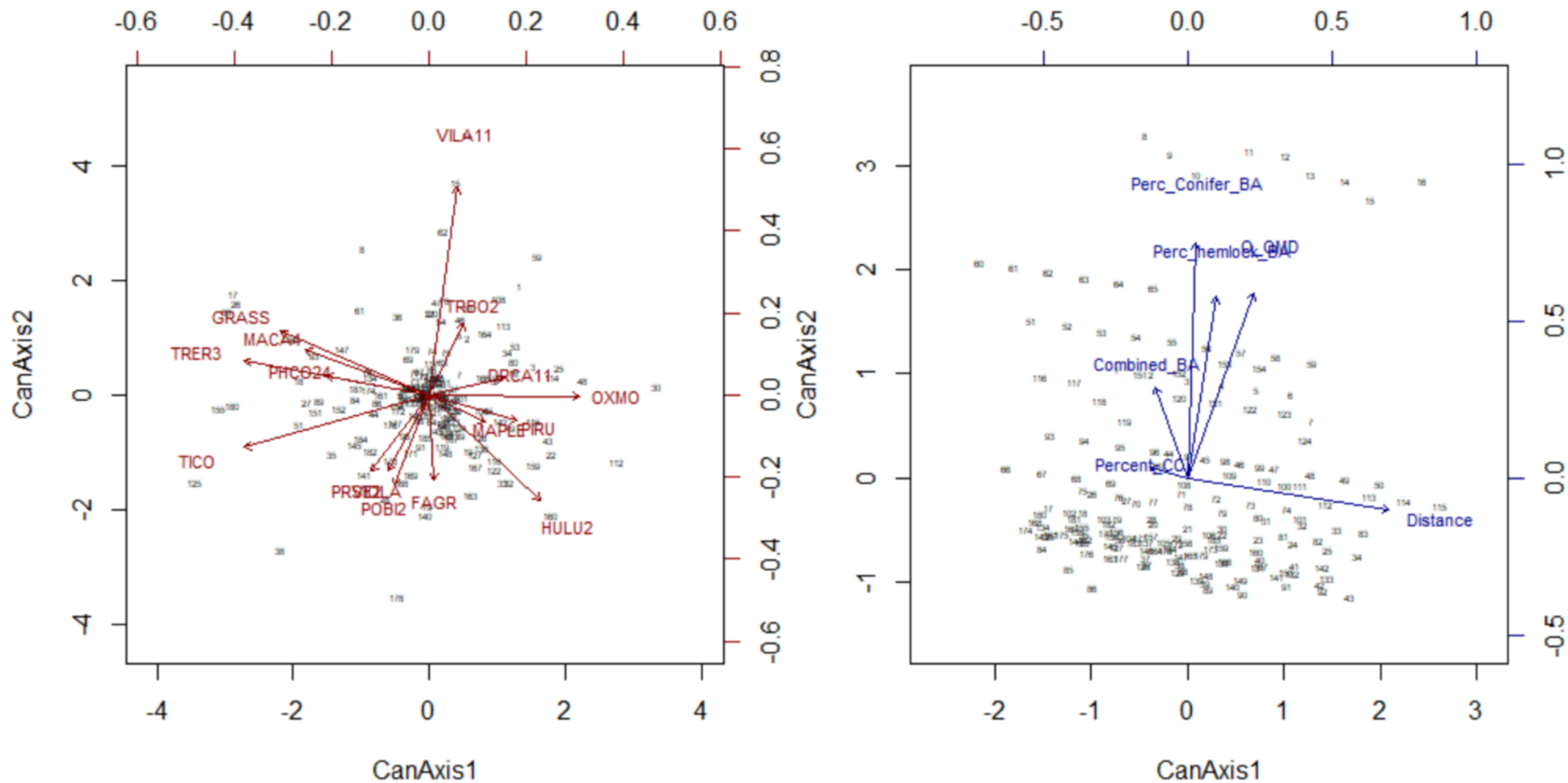
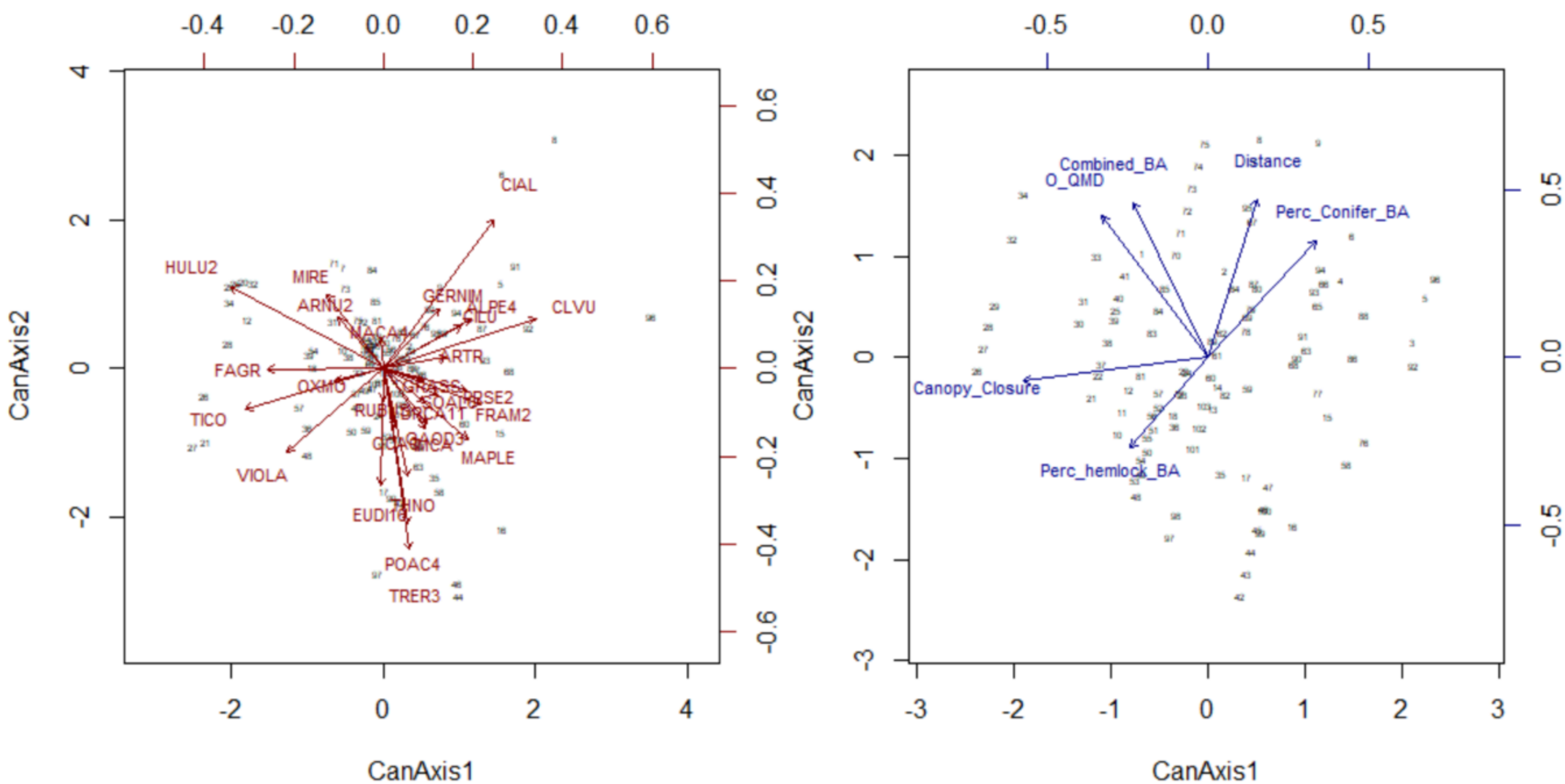



**Figure 2C-1**: **Canonical correlation biplots of objects and variables. The arrows (variable) in each plot displays correlations between (a) species variables and their canonical variables, and (b) environmental variables and their canonical variables at Hubbard Brook Experimental Forest (Site 1). The numbers within the graphs represents the (groundcover) plot number for the object scores of the canonical variate. For both graphs, the top axis and right axis correspond to the variable plot while the bottom axis and left axis correspond to the object plot for each biplot.**




(a) (b)

**Figure 2C-2: Canonical correlation biplots of objects and variables. The arrows (variable) in each plot displays correlations between (a) species variables and their canonical variables, and (b) environmental variables and their canonical variables at Huntington Wildlife Forest (Site 2). The numbers within the graphs represents the (groundcover) plot number for the object scores of the canonical variate. For both graphs, the top axis and right axis correspond to the variable plot while the bottom axis and left axis correspond to the object plot for each biplot.**




(a) (b)

**Figure 2C-3**: **Canonical correlation biplots of objects and variables. The arrows (variable) in each plot displays correlations between (a) species variables and their canonical variables, and (b) environmental variables and their canonical variables at Heiberg Memorial Forest (Site 3). The numbers within the graphs represents the (groundcover) plot number for the object scores of the canonical variate. For both graphs, the top axis and right axis correspond to the variable plot while the bottom axis and left axis correspond to the object plot for each biplot.**

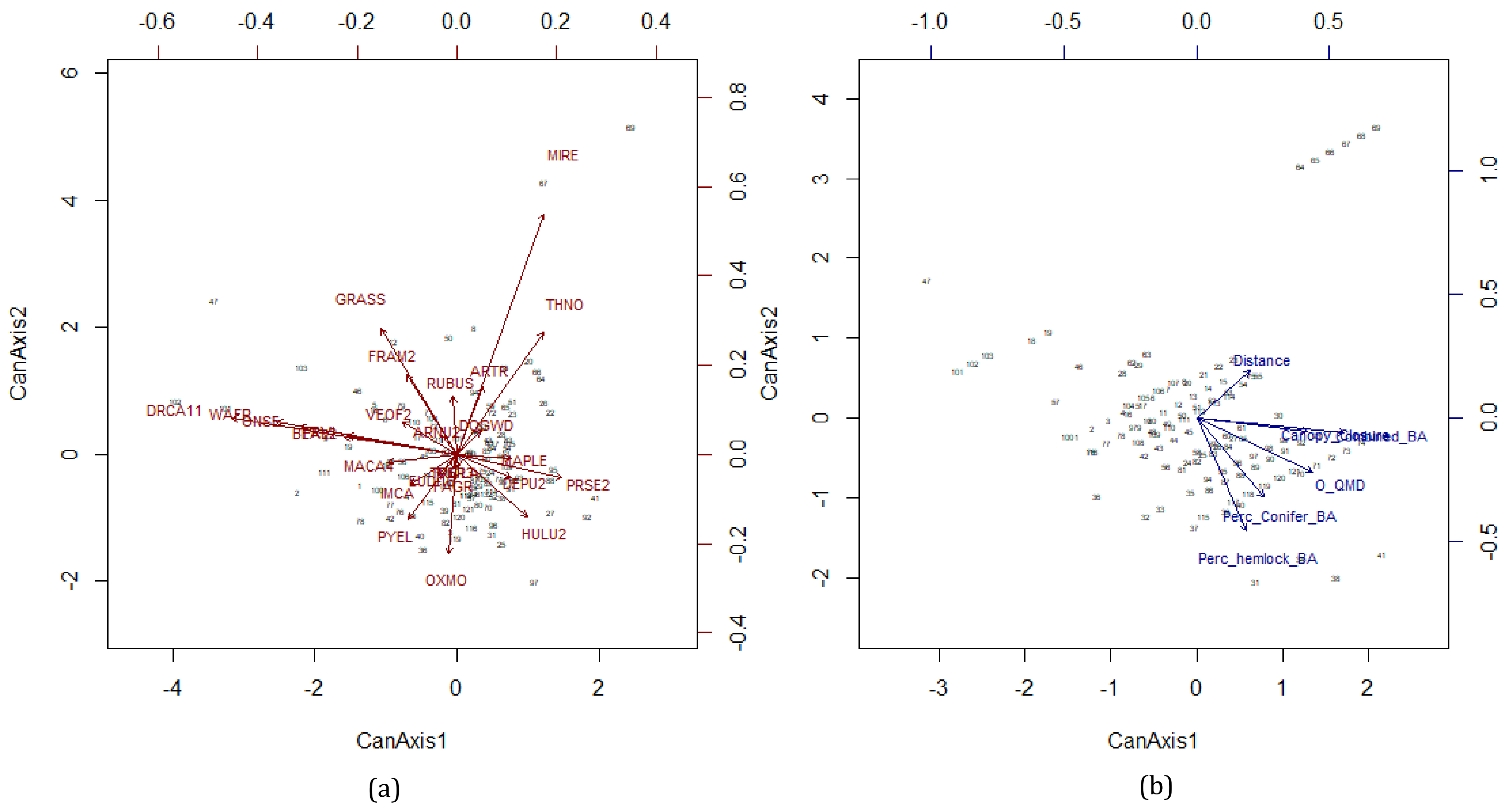



**Figure 2C-4**: **Canonical correlation biplots of objects and variables. The arrows (variable) in each plot displays correlations between (a) species variables and their canonical variables, and (b) environmental variables and their canonical variables at Cuyler Hill State Forest (Site 4). The numbers within the graphs represents the (groundcover) plot number for the object scores of the canonical variate. For both graphs, the top axis and right axis correspond to the variable plot while the bottom axis and left axis correspond to the object plot for each biplot.**

## Chapter 3 : Assessing riparian area protection strategies along headwater streams in forested regions of the U.S.


### Abstract

Riparian buffers assigned to protect streams can either be fixed or variable width buffers. Variable width buffers are designed to protect one or more ecological functions of a riparian area. Allocating fixed or variable width riparian buffers along streams depends on the complexity of buffer allocation and the opportunity cost of buffer areas. With a focus on headwater streams in five timber producing regions of the contiguous US, this study assessed land area differences between three buffer allocation strategies: functional based riparian buffer, state-specific riparian buffers, and a 30-m fixed width riparian buffer. This study also developed a GIS tool for delineating a functional riparian buffer using high resolution (1 m) digital elevation models.

Headwater streams dominate channel networks, comprising between 70 – 80 % of entire stream networks in all watersheds. Of the three buffer delineation types used in this study, the functional approach delineated the highest percentages of watershed area around headwater streams in most watersheds, even exceeding 20 % of forestland in some cases. State-specific riparian guidelines displaying differences between jurisdictions, delineated between 3.4 - 7.5 % of forestlands around headwater streams within watersheds. Although many state guidelines failed to identify the variable widths of functional riparian areas, some watersheds in the Lakes States over-delineated forestland as riparian when compared to the functional riparian areas. The topographic and forest compositional differences observed across timber producing regions of the contiguous US is not often represented by their respective state-specific RMZ guidelines. This study recommends employing a variable width buffer allocation such as the functional riparian buffer around headwater streams to ensure stream protection in working forests of the US.



**Key words**: functional riparian area, variable width riparian buffer, drainage density, riparian management zone, best management practice

## Introduction

Headwater streams dominate channel networks through cumulative stream length and can reach up to 80 % of an entire stream network within a watershed (Shreve 1969). Riparian areas around these headwater streams provide numerous ecosystem services and benefits such as regulating the flow of sediments and nutrients into waterways, increasing bank stability and preventing erosion, regulating stream water temperatures by provisioning shade, contributing organic material, and providing habitat and refugia for wildlife (Naiman et al. 2005; Opperman et al. 2017). Headwater streams are often under-represented, i.e. they are not being mapped to the actual density seen within watersheds in many National Hydrographic Datasets (Baker et al. 2007; Brooks and Colburn 2011; Elmore et al. 2013) and thus receive less attention when compared to larger order streams. This can have a negative impact, not only within and around headwater stream ecosystems, but also on downstream users within the watershed.

Forest managers encounter high densities of headwater streams within working forest landscapes and generally assign riparian buffers to protect the ecological integrity of the riparian area around them. Sediment, which washes into streams as a non-point source pollutant, is generally considered to be the most important type of water pollutant associated with forest operations in the United States (US) (Binkley and Brown 1993). Riparian management zones (RMZs) are a forestry Best Management Practice (BMP) designed to reduce non-point source pollution during forest operations (Phillips and Blinn 2004). Several decades of BMP studies have confirmed the effectiveness of RMZs against non-point source pollution (Sweeney and Newbold 2014). Sediment trapping efficiencies have reached beyond 95 % along headwater streams with riparian buffer allocations of between 7.6 – 30.5 m (25 – 100 ft.), even under intense silvicultural systems in upland forests (Lakel et al. 2010, Ward and Jackson 2004). In addition to effectively ameliorating the negative impacts of harvesting, RMZs can also protect wildlife habitat (Chizinski et al. 2010, Jackson et al. 2007).

Variances in buffer widths reflect differences in the integration of ecological, economic, and social factors (Lee et al. 2004). Most of the potential contributions of riparian vegetation to the ecological functions within a stream are realized within the first 4.6 to 30.5 m (15 to 100 ft.) from the stream bank (Blinn and Kilgore 2001). This range of riparian buffers provide at least 50 % of potential effectiveness and often 75 % or greater effectiveness at protecting various stream functions (Castelle and Johnson 2000). Forty-six states within the US have BMP guidelines or regulations, including recommendations for operating within or adjacent to RMZs (Appendix 3A). Ten of these 46 states regulate the implementation of their BMPs while 18 are quasi-regulatory and 17 states have voluntary BMP guidelines (Cristan et al. 2018). Quasi-regulatory states generally have voluntary BMP guidelines, but water quality infractions may result in fines. The single-sided width of State RMZs or state-specific riparian buffers are defined either as, (i) a fixed or standard width that may vary based on a water body/channel type or (ii) a variable width that is based on slope gradient of the terrace surrounding the stream. Sixteen states approach RMZ guidelines with fixed width buffers while 30 states have RMZ guidelines for variable width buffers based on slope gradient (Appendix 3A).

Since the 1960s, riparian protection measures consisted of allocating fixed width buffers of ~30 m or 100 ft. coupled with silvicultural guidelines (Richardson et al. 2012). In recent decades, due to continued research on stream protection and riparian areas, researchers are recommending the adoption of variable width buffers. Variable width buffers are delineated with a focus on one particular riparian function or a group of functions. Most commonly they have been modeled based on slope gradient for regulating sediment flow. Variable width riparian buffers have also been delineated using other topographic features such as loading areas of streams (Bren 1998), terrain analysis (Tomer et al. 2003), and hydrology (Tiwari et al. 2016; Kuglerová et al. 2014). The US Forest Service follows a "functional" approach method to delineate a variable-width buffer (Ilhardt et al. 2000) (Figure 3.1). This variable-width riparian buffer seeks to capture the functions of a riparian area by considering (1) the stream, (2) the floodplain, which is (if present) seasonally inundated, (3) the terrace slope, which is (if present) either partially or fully inundated during a 50-year flood, and finally (4) one tree length from the top of each terrace (Ilhardt et al. 2000), under the assumption that coarse woody debris input to the stream is collected at the average distance of a tree length (Sweeney and Newbold 2014). The variable terrace slope widths of functional riparian buffers hold riparian vegetation that provides increased bank stability, over-hanging bank cover, and nutrient uptake from groundwater and stream water (Swanson et al. 1982; Gregory et al. 1991; Naiman et al. 2005). The average canopy tree length represents the distance of natural recruitment of large woody debris by trees directly falling into the stream (Sweeney and Newbold 2014; Richardson et al. 2012; Ilhardt et al. 2000). Large woody debris provides allochthonous nutrient inputs into streams that serve as food for aquatic organisms, creates and increases instream habitat diversity and helps dissipate energy during high flows to reduce sediment movement to downstream reaches (Flores et al. 2011; Diez et al. 2001; Harmon et al. 1986). Since this variable width buffer encompasses numerous riparian ecosystem functions, throughout this study, the "functional" buffer is considered as the ideal approach for delineating riparian buffers.

However, the assignment of a variable width buffer that either considers topography or ecological functions will depend upon the complexity of guidelines as opposed to a simple and easily applied fixed width riparian buffer. Fixed width riparian buffers are conveniently applied in the field and on mapping software such as geographic information systems (GIS). Variable width buffer delineation in the field may require practical experience and knowledge of riparian ecology or tedious measurements of terrace slopes along all streams within a working forested boundary. Current developed methods and tools require multiple resources or data layers to define these variable width buffers (Abood et al. 2012; Williams et al. 2003). These methods have not been simplified for forest management and thus may discourage land managers and foresters from implementing them despite their ecological importance.

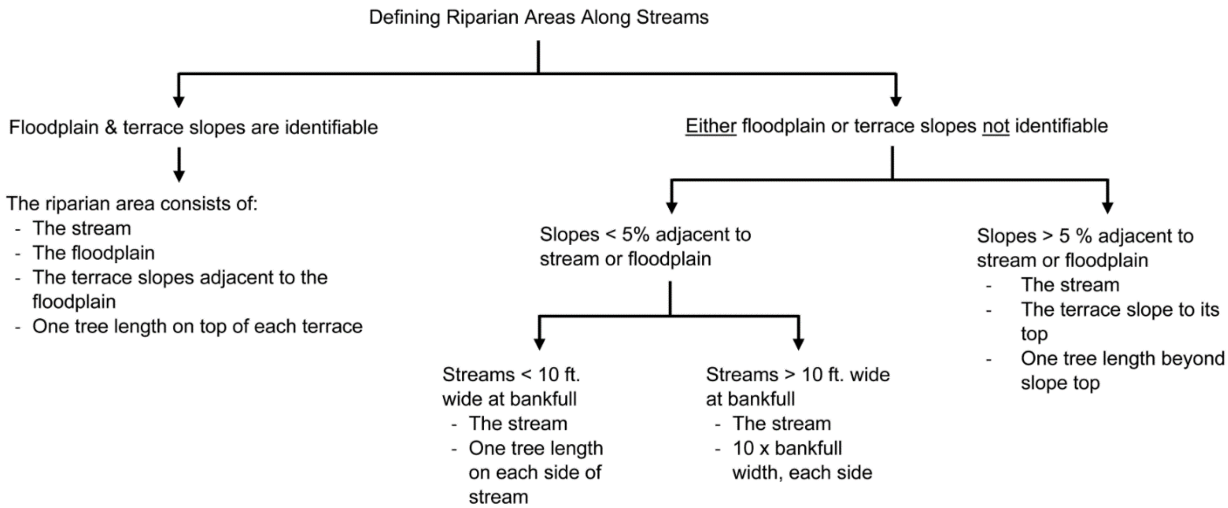



**Figure 3.1:Field key for identifying a "functional" based riparian area (based on Ilhardt et al. 2000)**

Deciding on the type of riparian buffer delineation method along headwater streams is an important management decision due to the high density of first- and second-order streams in working forest landscapes. Assigning a fixed width riparian buffer may underestimate or overestimate the functional role represented by the riparian area that can be described by a variable width buffer. Therefore, it is important to assess if individual state-specific riparian buffer guidelines, whether fixed- or variable width, or "functional" riparian buffer delineations as proposed by the USFS are comparable with each other. Over-estimations of riparian areas may have opportunity costs for the landowner and may discourage them from complying with BMP guidelines while an underestimation can lead to negative environmental consequences. Depending on the stream/drainage densities of watersheds, these opportunity costs could be exacerbated.

The objectives of this study were as follows: 1) to calculate drainage density of watersheds across various timber producing regions of the country, 2) to compare and contrast the differences in RMZ forest areas delineated between a "functional" based riparian area (defined by Ilhardt et al. (2000)), the RMZ based on State BMP guidelines, and a 100 ft. (30 m) fixed-width riparian buffer, 3) to estimate the percent land area of riparian buffer types occupied by first- and second-order streams (headwater streams) within sampled watersheds and, 4) to develop a GIS tool for mapping variable width buffers based on topography.

## Methods

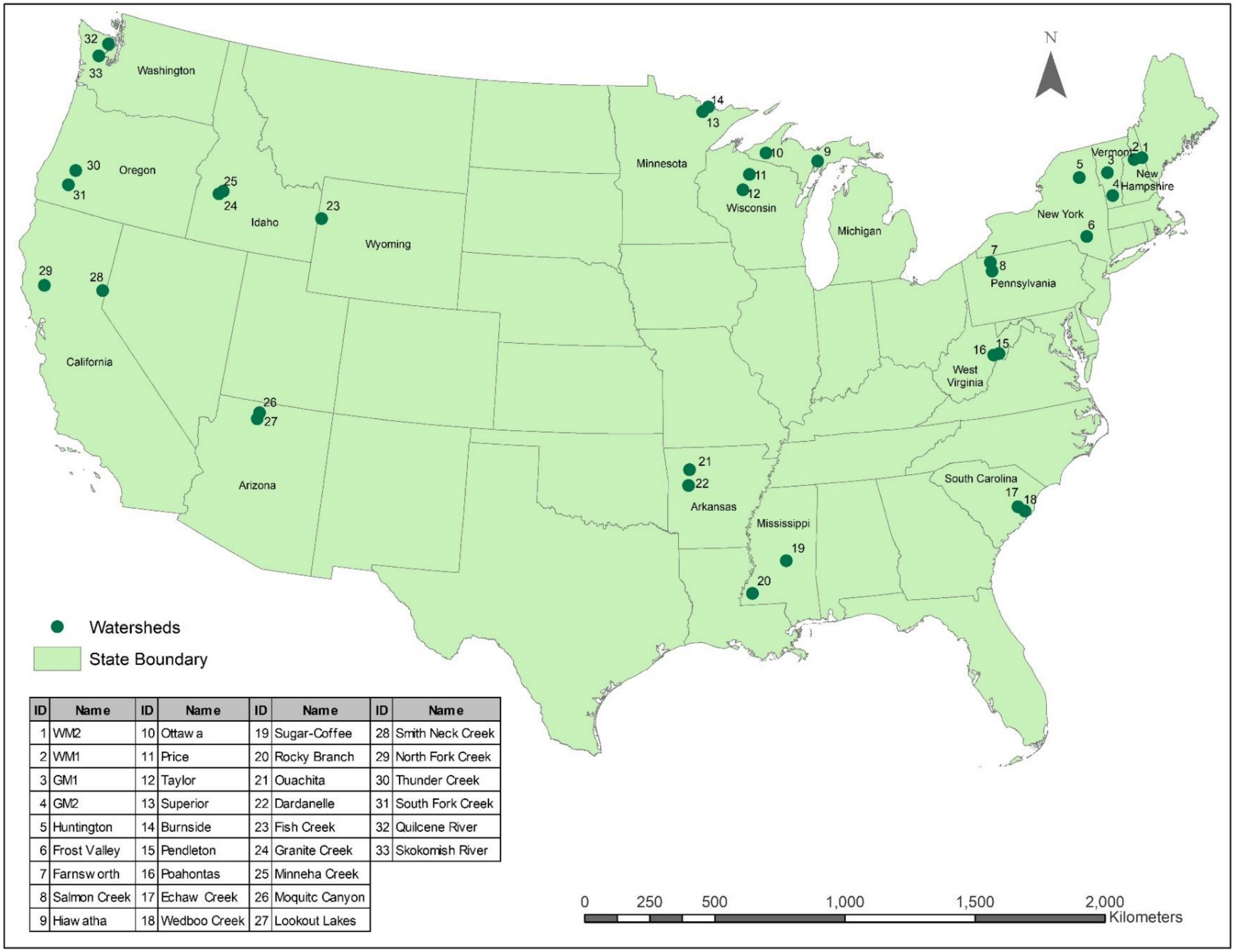
Study design

The contiguous US was divided into five geographically identified regions of the Northeast (NE), Lake States (LS), Southeast (SE), West (W), and Pacific Northwest (PNW) (Table 3.1). Sample states for each region were selected based on several forestry criteria. Within each region, states with 15 % or more forest cover and a significant forest-based economy based on state's GDP industry share (https://www.bea.gov/) were selected. Areas that were designated as timberlands or managed land within a state were overlaid with land cover data. From this selection, forested watersheds were highlighted. These forests were further filtered by the availability of LiDAR derived digital terrain model (DTM) data of 1 m spatial resolution or higher. This resulted in a sample of 17 states across five regions of the country, totaling 33 watersheds (Table 3.1 and Figure 3.2). Two independent watersheds were selected for each state with the exception of Wyoming due to limited DTM data availability.

**Table 3.1: Sample locations for the study distributed among the Northeast (NE), Lake States (LS), Southeast (SE), West (W), and Pacific Northwest (PNW) regions across the contiguous United States.**

| Region name | Region code | State | Watershed name/ID | Watershed location |
|---|---|---|---|---|
| Northeast | NE | New Hampshire | WM1 | White Mountains National Forest |
| | | | WM2 | White Mountains National Forest |
| | | Vermont | GM1 | Green Mountains National Forest |
| | | | GM2 | Green Mountains National Forest |

| Region name | Region code | State | Watershed name/ID | Watershed location |
|---|---|---|---|---|
| | | New York | Huntington Wildlife Forest | Adirondacks |
| | | | Frost Valley | Catskills |
| | | Pennsylvania | Farnsworth | Alleghany National Forest |
| | | | Salmon Creek | Alleghany National Forest |
| Lake States | LS | Michigan | Hiawatha | Hiawatha National Forest |
| | | | Ottawa | Ottawa National Forest |
| | | Wisconsin | Taylor County WS | Chequamegon-Nicolet National Forest |
| | | | Price County WS | Chequamegon-Nicolet National Forest |
| | | Minnesota | Burnside | Burnside State Forest |
| | | | Superior | Superior State Forest |
| Southeast | SE | West Virginia | Pocahontas | Monongahela National Forest |
| | | | Pendleton | Monongahela National Forest |
| | | South Carolina | Echaw Creek | Marion National Forest |
| | | | Wedboo Creek | Marion National Forest |
| | | Mississippi | Sugar-Coffee Bogue | Bienville National Forest |
| | | | Rocky Branch | Homochitto National Forest |
| | | Arkansas | Dardanelle | Mount Magazine State Park |
| | | | Ouachita | Ozark National Forest |
| West | W | Wyoming | Fish Creek | Teton National Forest |
| | | Arizona | Lookout Lakes | Kaibab National Forest |
| | | | Moquitch Canyon | Kaibab National Forest |
| | | Idaho | Granite Creek | Boise National Forest |
| | | | Minneha Creek | Boise National Forest |
| | | California | North Fork Creek | Mendocino National Forest |
| | | | Smith Neck Creek | Tahoe National Forest |
| Pacific Northwest | PNW | Washington | Quilcene River | Olympic National Forest |
| | | | Skokomish River | Olympic National Forest |
| | | Oregon | South Fork Cow Creek | Rouge River National Forest |
| | | | Thunder Creek | Umpqua National Forest |




| ID | Name | ID | Name | ID | Name | ID | Name |
|---|---|---|---|---|---|---|---|
| 1 | WM2 | 10 | Ottaw a | 19 | Sugar-Coffee | 28 | Smith Neck Creek |
| 2 | WM1 | 11 | Price | 20 | Rocky Branch | 29 | North Fork Creek |
| 3 | GM1 | 12 | Taylor | 21 | Ouachita | 30 | Thunder Creek |
| 4 | GM2 | 13 | Superior | 22 | Dardanelle | 31 | South Fork Creek |
| 5 | Huntington | 14 | Burnside | 23 | Fish Creek | 32 | Quilcene River |
| 6 | Frost Valley | 15 | Pendleton | 24 | Granite Creek | 33 | Skokomish River |
| 7 | Farnsw orth | 16 | Poahontas | 25 | Minneha Creek | | |
| 8 | Salmon Creek | 17 | Echaw Creek | 26 | Moquitc Canyon | | |
| 9 | Hiaw atha | 18 | Wedboo Creek | 27 | Lookout Lakes | | |

**Figure 3.2: Sampled watersheds distributed across the contiguous United States.**

*Forest composition*

Forest Inventory Analysis (FIA) data was used to calculate forest stand statistic information for each watershed. FIA plots within a 6 – 15 km (3.7 – 9.3 miles) radius of watersheds in the inventory years after 2015 (2016, 2017, 2018, 2019) were selected. The radii were based on capturing data coverage for 20 - 30 overstory subplots (1/24$^{th}$ ac). This was done under the assumption that the standard error around the mean basal area was less than 20 % at $\alpha = 0.05$ (Munsell and Germain 2007). The precision of FIA subplot locations was of concern only to the extent of their watershed. Therefore, this analysis was not affected by the errors in 'fuzzy' plot cluster locations provided by the US Forest Service on their public domain. Overstory data for trees with a dbh > = 12.7 cm (5 in.) were processed using NED-3 software to quantify basic stand statistics such as basal area (BA), relative density, quadratic mean diameter (QMD), and stem density. The average canopy tree height was calculated for the dominant and co-dominant trees within a watershed.

The Northeastern region watersheds are comprised of northern hardwoods and spruce-northern hardwood forests. Basal area ranged between 19.5 – 33.8 $m^2$/ha (85 – 147 ft.$^2$/ac.) with overstory relative stand densities ranging from 55 % in NH to 94 % in NY. The lowest average canopy tree height of 13.1 m (43 ft.) was recorded in one watershed in NH while the canopy tree heights ranged between 16.2 – 25.6 m (53 – 84 ft.) in the other watersheds within the Northeast (Table 3.2).

Forest types in the Lake States region ranged from pine forests in MN to hardwood and spruce-northern hardwood forests in WI and MI. The lowest BA of 14 $m^2$/ha (61 ft.$^2$/ac.) was recorded in the Burnside State Forest watershed in MN while the BA ranged from 18.4 – 24.1 $m^2$/ha (80 – 105 ft.$^2$/ac.) for the remaining watersheds of the region. Relative stand densities in the Lake States region ranged from 38 % in MN to 63 % in WI. Average canopy tree heights ranged between 16.2 – 21.6 m (53 – 71 ft.) in the watersheds of the region.

Oak-southern pine forests dominated the Southeastern States of AR and MS while southern bottomland hardwoods and hardwood forests dominated SC and WV, respectively. BA in the southern pine stands was as low as 13.1 $m^2$/ha (57 $ft.^2$/ac.) and as high as 35.1 $m^2$/ha (153 ft. $^2$/ac.) while BA in the southern bottomland hardwood stands averaged 26.2 $m^2$/ha (116 $ft.^2$/ac.) and 32.6 $m^2$/ha (142 $ft.^2$/ac.) in the northern hardwood stands in WV. Relative stand density in the southern pine stands ranged between 41 – 95 % while relative density ranged between 75 – 89 % in hardwood stands in the Southeastern region. Average canopy tree heights in the oak-southern pine stands ranged from 18.3 – 30.5 m (60 – 100 ft.) and 19.2 – 23.7 m (63 – 78 ft.) in the hardwood stands.

In the Western region, a mix of Douglas-fir and ponderosa pine forest stands comprised watersheds in CA, ID, and AZ while watersheds in WY had an Engelmann spruce - subalpine fir forest type in its watershed. The Douglas-fir and ponderosa pine forest stands had BAs ranging from 24.3 – 31.2 $m^2$/ha (106 – 136 $ft.^2$/ac.) and relative densities of 53 – 64 %, while the BA recorded in the Engelmann spruce - subalpine fir forest was 14.7 $m^2$/ha (64 $ft.^2$/ac.) with a relative density of 36 %. The Douglas-fir and ponderosa pine forest stands recorded average canopy tree heights of 16.8 – 21.6 m (55 – 71 ft.) while the Engelmann spruce - subalpine fir forest stands recorded an average canopy tree height of 16.5 m (54 ft.).

The Pacific Northwest watersheds were comprised of Douglas-fir forest stands where BAs ranged from 48 $m^2$/ha (209 $ft.^2$/ac.) up to 138.4 $m^2$/ha (603 $ft.^2$/ac.) and relative stand densities over 100 %. The highest average canopy tree height of 32 m (105 ft.) was recorded in one of the watersheds in OR while the average canopy tree height ranged between 21.9 – 28 m (72 – 92 ft.) in the remaining watersheds of the region. It is noteworthy that some individual stems were over 60m tall, but this only occurred on a few selected plots.

**Table 3.2: Summary of forest composition in study watersheds across the United States.**

| Region | State | Watershed name/ID | Dominant species | Landcover type | Forest type | BA (ft.$^2$/ac.) | stems/ac. | QMD (in.) | Relative Density (%/ac.) | Average Dominant/Co-dominant tree height (ft.) |
|---|---|---|---|---|---|---|---|---|---|---|
| NE | NH | WM1 | red spruce (*Picea rubens*)<br>black spruce (*Picea mariana*)<br>paper birch (*Betula papyrifera*)<br>balsam fir (*Abies balsamea*) | Mixed Coniferous/ Broadleaf forest | spruce-northern hardwoods | 85 | 275 | 8 | 55 | 43 |
| NE | NH | WM2 | yellow birch (*Betula alleghaniensis*)<br>paper birch (*Betula papyrifera*)<br>American beech (*Fagus grandifolia*)<br>sugar maple (*Acer saccharum*)<br>red maple (*Acer rubrum*) | Mixed Coniferous /Broadleaf forest | spruce-northern hardwoods | 126 | 225 | 10 | 92 | 62 |
| NE | VT | GM1 | sugar maple (*Acer saccharum*)<br>balsam fir (*Abies balsamea*) | Mixed Coniferous/ Broadleaf forest | spruce-northern hardwoods | 100 | 172 | 11 | 70 | 53 |
| NE | VT | GM2 | sugar maple (*Acer saccharum*)<br>red maple (*Acer rubrum*)<br>American beech (*Fagus grandifolia*)<br>yellow birch (*Betula alleghaniensis*) | Mixed Coniferous/ Broadleaf forest | spruce-northern hardwoods | 127 | 195 | 11 | 85 | 57 |
| NE | NY | Huntington | American beech (*Fagus grandifolia*)<br>sugar maple (*Acer saccharum*)<br>yellow birch (*Betula alleghaniensis*)<br>eastern hemlock (*Tsuga canadensis*) | Broadleaf forest | northern hardwoods | 120 | 232 | 10 | 89 | 77 |
| NE | NY | Frost Valley | yellow birch (*Betula alleghaniensis*)<br>red maple (*Acer rubrum*) | Broadleaf forest | northern hardwoods | 131 | 126 | 14 | 94 | 77 |

| Region | State | Watershed name/ID | Dominant species | Landcover type | Forest type | BA (ft.$^2$/ac.) | stems/ac. | QMD (in.) | Relative Density (%/ac.) | Average Dominant/Co-dominant tree height (ft.) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | sugar maple (*Acer saccharum*)<br>eastern hemlock (*Tsuga canadensis*)<br>American beech (*Fagus grandifolia*) | | | | | | | |
| | PA | Farnsworth | black cherry (*Prunus serotina*)<br>red maple (*Acer rubrum*)<br>eastern hemlock (*Tsuga canadensis*)<br>sweet birch (*Betula lenta*) | Broadleaf forest | Allegheny hardwoods | 147 | 139 | 14 | 75 | 79 |
| | PA | Salmon Creek | red maple (*Acer rubrum*)<br>black cherry (*Prunus serotina*)<br>eastern hemlock (*Tsuga canadensis*) | Broadleaf forest | Allegheny hardwoods | 123 | 146 | 13 | 70 | 84 |
| LS | MI | Hiawatha | sugar maple (*Acer saccharum*)<br>eastern hemlock (*Tsuga canadensis*)<br>white spruce (*Picea glauca*)<br>red maple (*Acer rubrum*) | Mixed Coniferous/ Broadleaf forest | spruce-northern hardwoods | 80 | 149 | 10 | 51 | 66 |
| LS | MI | Ottawa | American basswood (*Tilia americana*)<br>quaking aspen (*Populus tremuloides*)<br>eastern white pine (*Pinus strobus*)<br>sugar maple (*Acer saccharum*) | Broadleaf forest | other hardwoods | 105 | 189 | 10 | 62 | 71 |
| LS | WI | Price | sugar maple (*Acer saccharum*)<br>arborvitae (*Thuja occidentalis*)<br>American basswood (*Tilia americana*)<br>quaking aspen (*Populus tremuloides*) | Mixed Coniferous/ Broadleaf forest | other mixedwoods | 100 | 187 | 10 | 64 | 62 |
| LS | WI | Taylor | red maple (*Acer rubrum*)<br>sugar maple (*Acer saccharum*) | Broadleaf forest | maple | 91 | 178 | 10 | 62 | 68 |

| Region | State | Watershed name/ID | Dominant species | Landcover type | Forest type | BA (ft.$^2$/ac.) | stems/ac. | QMD (in.) | Relative Density (%/ac.) | Average Dominant/Co-dominant tree height (ft.) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | quaking aspen (*Populus tremuloides*) | | | | | | | |
| | MN | Superior | red pine (*Pinus resinosa*)<br>paper birch (*Betula papyrifera*)<br>black spruce (*Picea mariana*) | Coniferous forest | pine | 100 | 239 | 9 | 60 | 56 |
| | MN | Burnside | eastern white pine (*Pinus strobus*)<br>paper birch (*Betula papyrifera*)<br>red pine (*Pinus resinosa*)<br>quaking aspen (*Populus tremuloides*) | Coniferous forest | pine | 61 | 152 | 9 | 38 | 54 |
| SE | WV | Pocahontas | black cherry (*Prunus serotina*)<br>red maple (*Acer rubrum*)<br>sugar maple (*Acer saccharum*) | Broadleaf forest | Allegheny hardwoods | 143 | 212 | 11 | 83 | 78 |
| SE | WV | Pendleton | red spruce (*Picea rubens*)<br>red maple (*Acer rubrum*)<br>northern red oak (*Quercus rubra*)<br>chestnut oak (*Quercus montana*)<br>black cherry (*Prunus serotina*)<br>sugar maple (*Acer saccharum*) | Broadleaf forest | oak-northern hardwoods | 140 | 198 | 11 | 89 | 63 |
| SE | SC | Echaw | loblolly pine (*Pinus taeda*)<br>swamp tupelo (*Nyssa biflora*)<br>red maple (*Acer rubrum*) | Forested wetlands | southern bottomland hardwoods | 114 | 187 | 11 | 75 | 65 |
| SE | SC | Wedboo | loblolly pine (*Pinus taeda*)<br>sweetgum (*Liquidambar styraciflua*) | Forested wetlands | southern bottomland hardwoods | 117 | 206 | 10 | 79 | 69 |
| SE | MS | | loblolly pine (*Pinus taeda*) | | | 153 | 132 | 14 | 95 | 100 |

| Region | State | Watershed name/ID | Dominant species | Landcover type | Forest type | BA (ft.²/ac.) | stems/ac. | QMD (in.) | Relative Density (%/ac.) | Average Dominant/Co-dominant tree height (ft.) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Sugar-Coffee | willow oak (*Quercus phellos*) | Mixed Coniferous/ Broadleaf forest | oak southern pine | | | | | |
| | | Rocky Branch | loblolly pine (*Pinus taeda*) | Mixed Coniferous/ Broadleaf forest | oak southern pine | 105 | 138.1 | 12 | 67.9 | 78.41 |
| | AR | Dardanelle | shortleaf pine (*Pinus echinata*)<br>northern red oak (*Quercus rubra*)<br>post oak (*Quercus stellata*) | Mixed Coniferous/ Broadleaf forest | oak southern pine | 57 | 95 | 11 | 41 | 60 |
| | AR | Ouachita | white oak (*Quercus alba*)<br>shortleaf pine (*Pinus echinata*)<br>mockernut hickory (*Carya alba*) | Mixed Coniferous/ Broadleaf forest | oak southern pine | 91 | 149 | 11 | 69 | 71 |
| W | WY | Fish Creek | subalpine fir (*Abies lasiocarpa*)<br>Engelmann spruce (*Picea engelmannii*)<br>Douglas-fir (*Pseudotsuga menziesii*)<br>quaking aspen (*Populus tremuloides*) | Coniferous forest | Engelmann spruce - subalpine fir | 64 | 102 | 11 | 36 | 54 |
| W | AZ | Lookout Lakes, Moquitch Canyon | ponderosa pine (*Pinus ponderosa*)<br>blue spruce (*Picea pungens*)<br>white fir (*Abies concolor*)<br>twoneedle pinyon (*Pinus edulis*) | Coniferous forest | ponderosa pine | 106 | 106 | 14 | 53 | 57 |
| W | ID | | Douglas-fir (*Pseudotsuga menziesii*) | | Douglas-fir | 107 | 93 | 14 | 54 | 65 |

| Region | State | Watershed name/ID | Dominant species | Landcover type | Forest type | BA (ft.$^2$/ac.) | stems/ac. | QMD (in.) | Relative Density (%/ac.) | Average Dominant/Co-dominant tree height (ft.) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Granite Creek, Minneha Creek | ponderosa pine (*Pinus ponderosa*) | Coniferous forest | | | | | | |
| | CA | North Fork | Douglas-fir (*Pseudotsuga menziesii*)<br>ponderosa pine (*Pinus ponderosa*)<br>canyon live oak (*Quercus chrysolepis*)<br>sugar pine (*Pinus lambertiana*) | Coniferous forest | Douglas-fir | 136 | 65 | 20 | 64 | 71 |
| | CA | Smith Neck | ponderosa pine (*Pinus ponderosa*)<br>Jeffrey pine (*Pinus jeffreyi*) | Coniferous forest | ponderosa pine | 115 | 92 | 15 | 58 | 54 |
| PNW | WA | Quilcene | Douglas-fir (*Pseudotsuga menziesii*)<br>western hemlock (*Tsuga heterophylla*)<br>western red cedar (*Thuja plicata*) | Coniferous forest | Douglas-fir | 603 | 357 | 17 | 253 | 83 |
| PNW | WA | Skokomish | Douglas-fir (*Pseudotsuga menziesii*)<br>western hemlock (*Tsuga heterophylla*)<br>bigleaf maple (*Acer macrophyllum*) | Coniferous forest | Douglas-fir | 209 | 201 | 14 | 107 | 71 |
| PNW | OR | South Fork | Douglas-fir (*Pseudotsuga menziesii*) | Coniferous forest | Douglas-fir (pure) | 530 | 205 | 22 | 202 | 105 |
| PNW | OR | Thunder Creek | Douglas-fir (*Pseudotsuga menziesii*) | Coniferous forest | Douglas-fir | 372 | 181 | 19 | 152 | 92 |

Data sources

Watershed boundaries for the analysis were downloaded to the 12-digit hydrologic unit (HU) through the National Hydrography Dataset (NHD) published by the United States Geological Survey (USGS) (Table 3.3). Hydrographic data of HU-8 subbasin extent was downloaded to the target areas of the watersheds. The NHD layer was mapped to 1:24,000 map scale.

A raster layer of the USGS National Land Cover Database (NLCD) was used to identify all forest cover categories of deciduous, evergreen, and mixed forests. Selected watersheds had over 90 % forest cover.

DTMs with a spatial resolution of 1-m or higher resolution were obtained from either State GIS Clearinghouses/GIS databases or The National Map of the USGS, or Open Topography (https://opentopography.org/). Data for each watershed area was downloaded as tiles, which were used to create a mosaic of continuous coverage to the extent of the watershed.

Information on silvicultural treatments and managed forestlands was obtained from the data published by the USDA Forest Service (https://data.fs.usda.gov/geodata/webapps/EDW_DataExtract/) as file geodatabases.

Forest Inventory Analysis (FIA) data were obtained through the FIA data mart published by the USDA Forest Service. The plot coordinate file (PLOT.csv) for each state was downloaded and converted to a point feature class using the 'XY Table to Point Tool' on ArcGIS Pro. Once the plots within areas of interest in and around watersheds were identified by creating a separate shapefile, the unique plot identity numbers were matched with the TREE.csv files for each respective state. The TREE.csv file included inventory information of plot clusters at a given coordinate position. A plot cluster generally contains 4 subplots that are 1/24 ac. (24 ft. or 7.32 m radius) in size.

**Table 3.3: Data inputs and their sources used for the study.**

| **Data** | **Source** |
|---|---|
| Land cover | National Land Cover Database (NLCD), USGS |
| Stream networks | National Hydrography Dataset (NHD), USGS<br>State GIS clearinghouses |
| DTM | State GIS clearinghouses<br>USGS<br>Open Topography |
| Timberlands | USDA Forest Service |
| FIA | FIA Data Mart, USDA Forest Service |

Data Analysis

*Drainage density*

Stream networks within each watershed were generated with 1 m DTMs using the Hydrology tool set in ArcGIS. The NHD layer was used as a reference tool for the delineated network. Stream orders were defined for the delineated raster stream networks using the Strahler method (Strahler 1952) via the Stream Order tool in ArcGIS. Total drainage density, which is the total length of all streams and rivers in a watershed or drainage basin as a proportion of the total area of the watershed, was calculated. The following equation (Equation 3.1) was used to calculate drainage density (Zävoianu 1985).

$$Drainage\ density = \frac{Length\ of\ stream\ network\ (km)}{Watershed\ area\ (km^2)} \quad [3.1]$$

Headwater drainage density was also calculated by considering the summed lengths of first- and second-order streams within the watershed. Additionally, headwater stream percentage for a watershed was calculated as a proportion of total length of streams and rivers within the watershed.

The differences between the average drainage density of regions was assessed using a robust one-way ANOVA based on trimmed means (20% trimming level). Pairwise comparisons between regions were tested for trimmed means using Hochberg's multiple comparison adjustment (Wilcox 1986). The WRS2 package (Mair and Wilcox 2020) on R Studio (R Core Team 2019) was used for the analysis.

*Fixed width riparian buffer allocation*

Within each watershed, three smaller scaled stream networks comprising of headwater streams (first- and second order) were selected for the buffer allocations. I first applied a 30.5 m (100 ft.) buffer around the selected stream networks. Then the fixed-width buffer areas for first- and second-order streams within a network were isolated and their respective land area allocations were recorded.

*"Functional" riparian buffer delineations and GIS tool development ("Ridge Finder")*

In the field, the "functional" riparian buffer is delineated based on topographic features and specific parameters of forest composition. In other words, the "functional" buffer is defined as the stream, floodplain, terrace slope and one-tree length from the top of the terrace slope (Ilhardt et al. 2000). The hybrid GIS tool, "Ridge Finder", identifies the floodplain (if present) and terrace slope tops using a high-resolution (1 m) DTM. I used ArcGIS Pro 2.5.1 and R Studio to develop the 'Ridge Finder'. Perpendicular transects were generated at 1-m intervals along a stream using 'Generate transects along lines' tool in ArcGIS Pro. The length of a transect was based on the topography of a watershed to reduce the overall run-time of the tool. Based on visual observation of the hillshade layer of watersheds, streams with wider ravines had a maximum transect length of 400 m while watersheds with narrower ravines had a minimum transect length of 40 m. Next, points were generated along transects at 1 m intervals using the 'Generate points along lines' tool in ArcGIS Pro. Elevation values for each point were extracted using the 'Extract multi-values to point' tool and the attribute table for this layer was exported as a .csv file.

Using R Studio (R Core Team 2019), a program was developed to detect the terrace slope tops on both sides of a stream using the .csv file exported from ArcGIS Pro. The program first splits the transects in two and identifies the points on each transect that intersects the stream. The slope is calculated using elevation values for each point along the transect sequentially moving away from the stream. The program is then set on a loop for each point to check a set of conditions compatible with the field key for identifying the terrace slope top of the functional buffer. The criteria for identifying the terrace slope are as follows:

- If the slope of the first point is 0 %,
- If the slope is greater than 5 %,
- If the previous slope is 0 % and this point slope is 0 % (floodplain),
- If the slope is less than 5 %.

The conditional loop flags the ridge or terrace slope top identified along each split transect based on the above set conditions.

Once the split transects are merged back to its original full transect, this file is then exported as a .csv file into ArcGIS pro. It is then merged with the point layer with elevation values using the 'Join' function. The flagged points are then selected and converted to a separate point feature class. These points indicate the boundary, or the terrace slope tops around a stream. Next, a buffer distance that represents the average canopy tree height for each watershed is allocated around the *terrace slope top* point layer. This buffer is then dissolved using the 'Dissolve' tool to represent the continuous “functional” riparian area. (The complete code for this program is available in Appendix 3B.)

The “Ridge Finder” tool was used to define the “functional” riparian buffer around the three selected stream networks within a watershed. The “functional” buffer areas for first- and second-order streams within a network were isolated and their respective land area allocations were recorded.

*State-specific riparian buffer delineation and development of GIS methods*

State RMZ guidelines are unique to each state and therefore buffer allocation approaches were tailored for each state (Table 3.4). Washington, Oregon, Idaho, California, and West Virginia have regulatory state RMZ guidelines, while Wisconsin, Michigan, Pennsylvania, New York, Vermont, New Hampshire, and South Carolina have quasi-regulatory state RMZ guidelines. Minnesota, Mississippi, Arkansas, Arizona, and Wyoming have voluntary state RMZ guidelines (Cristan et al. 2018). Five of the 17 states have separate guidelines for fish-bearing streams and therefore state stream network GIS files were used to identify those fish bearing streams.

Ten of the 17 states selected defined buffer widths based on the gradient of the terrace slope. For these states, their high resolution DTMs were resampled to 10 m resolution DTMs. This was done to get the average slope of the terrace around the stream. The resampled DTMs were used to derive a slope layer using the 'Slope' tool in ArcGIS Pro. Next a 10 m transect was generated ('Generate transects along lines') along the selected streams at 1 m intervals and points were generated along the transect ('Generate points along lines') at 5 m intervals. All points intercepting the streamline feature class were removed to reduce processing time and this resulted in only two points on either end of a transect. Slope values were extracted to these points ('Extract multi-values to points'). For each state, slope classes were isolated on ArcGIS in the attribute tables of the Slope raster layer. The recommended buffer distances per state-specific riparian guidelines were allocated for all points on the transect. Since the points were positioned 5 m away from the stream, this distance was deducted when specifying the buffer distances for those respective states. Afterwards, all buffers for slope classes were first merged and then dissolved to create one continuous shape file.

Once state-specific riparian buffers were delineated for the three stream networks within watersheds, buffer areas for first- and second-order streams within a network were isolated and their respective land area allocations were recorded.

**Table 3.4: State-specific riparian buffer allocation guide for selected states in the study. The width indicates the distance of the riparian buffer allocation on one side of the stream.**

| Region name | Region code | State | Implementation | Channel Type/ Restrictions | Slope (%) | Width (ft.) | Width (m) |
|---|---|---|---|---|---|---|---|
| Northeast | NE | New Hampshire | Quasi-regulatory | | 0 - 10 | 50 | 15 |
| | | | | | 11 - 20 | 70 | 21 |
| | | | | | 21 - 30 | 90 | 27 |
| | | | | | 31 - 40 | 110 | 34 |
| | | | | | 41 - 50 | 130 | 40 |
| | | | | | 51 - 60 | 150 | 46 |
| | | | | | 61 - 70 | 170 | 52 |
| | | | | | 71 - 80 | 190 | 58 |
| | | | | | 81 - 90 | 210 | 64 |
| | | Vermont | Quasi-regulatory | | 0 - 10 | 50 | 15 |
| | | | | | 11 - 20 | 70 | 21 |
| | | | | | 21 - 30 | 90 | 27 |
| | | | | | 31 - 40 | 110 | 34 |
| | | | | | 41 - 50 | 130 | 40 |
| | | | | | 51 - 60 | 150 | 46 |
| | | | | | 61 - 70 | 170 | 52 |
| | | | | | 71 - 80 | 190 | 58 |
| | | | | | 81 - 90 | 210 | 64 |
| | | New York | Quasi-regulatory | | 0 - 10 | 35 | 11 |
| | | | | | 11 - 20 | 65 | 20 |
| | | | | | 21 - 40 | 75 | 23 |
| | | | | | >40 | 100 | 30 |
| | | Pennsylvania | Quasi-regulatory | | 0 - 10 | 25 | 8 |
| | | | | | 11 - 20 | 45 | 14 |
| | | | | | 21 - 30 | 65 | 20 |
| | | | | | 31 - 40 | 85 | 26 |
| | | | | | 41 - 50 | 105 | 32 |
| | | | | | 51 - 60 | 125 | 38 |
| | | | | | 61 - 70 | 145 | 44 |
| | | | | | 71 - 80 | 165 | 50 |
| | LS | Michigan | | | 0 - 10 | 100 | 30 |

| Region name | Region code | State | Implementation | Channel Type/ Restrictions | Slope (%) | Width (ft.) | Width (m) |
|---|---|---|---|---|---|---|---|
| Lake States | | | Quasi-regulatory | | 11 - 20 | 115 | 35 |
| | | | | | 21 - 30 | 135 | 41 |
| | | | | | 31 - 40 | 155 | 47 |
| | | | | | > 40 | 175 | 53 |
| | | Wisconsin | Quasi-regulatory | Trout stream | | 100 | 30 |
| | | | | >3ft. Wide channel | | 100 | 30 |
| | | | | <3ft. Wide channel | | 35 | 11 |
| | | Minnesota | Non-regulatory | Trout stream | | 165 | 50 |
| | | | | >3ft. Wide channel | | 120 | 37 |
| | | | | <3ft. Wide channel | | 50 | 15 |
| Southeast | SE | West Virginia | Regulatory | | | 100 | 30 |
| | | South Carolina | Quasi-regulatory | Trout stream | < 5 | 40 | 12 |
| | | | | | 5 - 20 | 120 | 37 |
| | | | | | 21 - 40 | 160 | 49 |
| | | | | | > 40 | 200 | 61 |
| | | | | non-trout stream | < 5 | 40 | 12 |
| | | | | | 5 - 20 | 80 | 24 |
| | | | | | 21 - 40 | 120 | 37 |
| | | | | | > 40 | 160 | 49 |
| | | Mississippi | Non-regulatory | | 0 - 5 | 30 | 9 |
| | | | | | 6 - 20 | 40 | 12 |
| | | | | | 21 - 40 | 50 | 15 |
| | | | | | > 40 | 60 | 18 |
| | | Arkansas | Non-regulatory | | 0 - 20 | 35 | 11 |
| | | | | | > 20 | 50 | 15 |
| West | W | Wyoming | Non-regulatory | | < 35 | 50 | 15 |
| | | | | | > 35 | 100 | 30 |
| | | Arizona | Non-regulatory | Sediment control | | 33 | 10 |

| Region name | Region code | State | Implementati-on | Channel Type/ Restrictions | Slope (%) | Width (ft.) | Width (m) |
|---|---|---|---|---|---|---|---|
| | | Idaho | Regulatory | Class II (headwater) | | 30 | 9 |
| | | California | Regulatory | | < 30 | 75 | 23 |
| | | | | | 30 -50 | 100 | 30 |
| | | | | | > 50 | 150 | 46 |
| Pacific Northwest | PNW | Washington | Regulatory | Type F | Site Class I | 200 | 61 |
| | | | | | Site Class II | 170 | 52 |
| | | | | | Site Class III | 140 | 43 |
| | | | | | Site Class IV | 110 | 34 |
| | | | | | Site Class V | 90 | 27 |
| | | | | Type Np | | 50 | 15 |
| | | Oregon | Regulatory | Small Type F | | 50 | 15 |
| | | | | Medium Type F | | 70 | 21 |
| | | | | Small Type D | | 20 | 6 |
| | | | | Medium Type D | | 50 | 15 |
| | | | | Small Type N | | 20 | 6 |
| | | | | Medium Type N | | 50 | 15 |

*Standardizing riparian areas*

Buffer areas for the three riparian buffer types: 30 m (100 ft) fixed-width riparian buffer, functional riparian buffer, and state-specific riparian buffers, were standardized to their respective stream lengths according to the following equation (Equation 3.2).

$$Standardized\ riparian\ area = \frac{Riparian\ buffer\ area\ (m^2)}{Stream\ length\ (m)} \qquad [3.2]$$

If assumed that a stream is a straight line (for calculation purposes), the standardized riparian area represents twice the length/distance of an average riparian buffer (Figure 3.3). For example, a stream length of 10 m having a state-specific riparian area of 500 $m^2$ would have a standardized state-specific riparian area of 50 m. Therefore, the average state-specific riparian buffer for that stream is 25 m.

The above assumption was used when calculating the horizontal distance of the terrace slope around streams that were identified through the standardized “functional” riparian area.

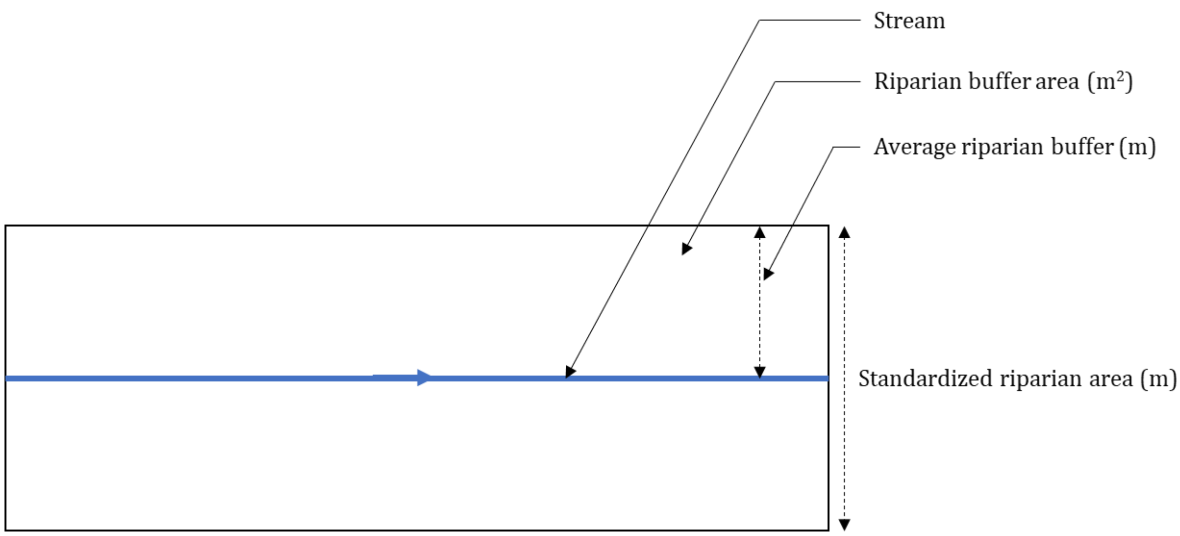



**Figure 3.3: Descriptive diagram of riparian buffer allocation nomenclature. The standardized riparian area represents twice the distance of an average riparian buffer.**

Calculating horizontal terrace slope distance for “functional” riparian buffers:

The standardized riparian area calculated for a “functional” riparian buffer was first divided by two to yield the average “functional” riparian buffer. Then the average canopy tree height for that watershed was deducted from the average “functional” riparian buffer to calculate the average horizontal terrace slope distance for a stream (Figure 3.4).

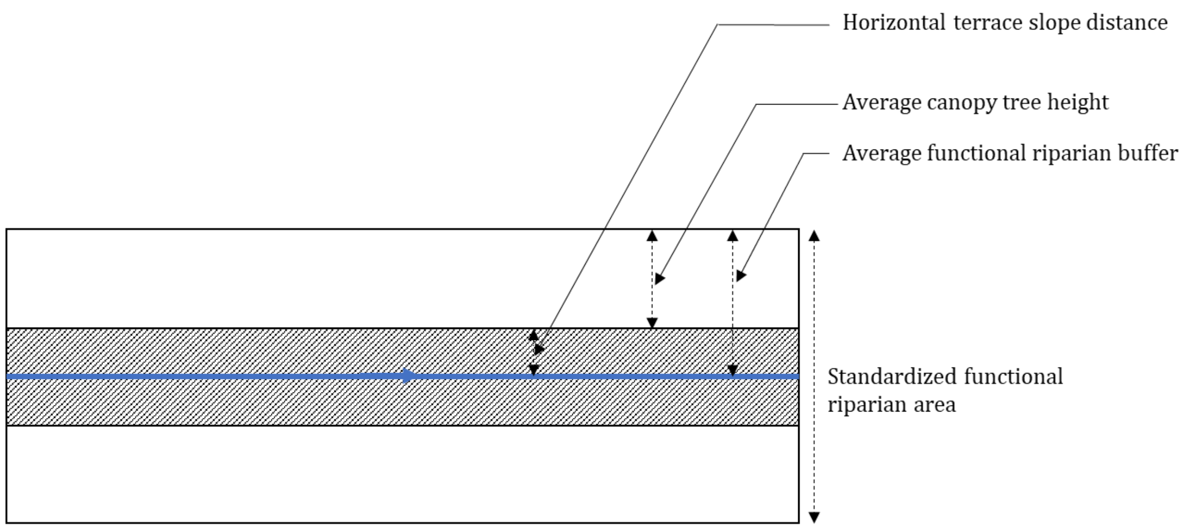



**Figure 3.4: Descriptive diagram of nomenclature for a "functional" riparian buffer allocation around a stream.**

*Aligned rank transform for nonparametric factorial ANOVAs*

Since the residuals of these datasets did not conform to normality, a nonparametric two-way ANOVA was performed on the dataset (Wobbrock et al. 2011; Kay and Wobbrock 2020). The aligned rank test was performed on factors of states and stream orders, and regions and stream orders. This allowed me to investigate the land area differences between a "functional" riparian buffer allocation and state-specific riparian buffer allocation across states, regions, and stream orders. In addition, the horizontal terrace slope distance that represents the topography around streams was compared across states, regions, and stream orders. I also assessed the difference in land area allocation between: "functional" riparian buffers vs. state-specific riparian buffers, "functional" riparian buffers vs. 30 m fixed-width riparian buffers, and 30-m fixed-width vs. state-specific riparian buffers. I used the ARTool package in R Studio (Kay and Wobbrock 2020) for the analysis.

*Proportion of riparian areas within a watershed*

As stated above, I assumed that the standardized riparian area represents twice the length/distance of an average riparian buffer. I also assumed that the standardized buffers calculated for the selected stream networks within a watershed is representative of all headwater streams within that watershed. This allowed me to calculate the percentage of watershed area that riparian buffers occupied through extrapolation.

The average standardized riparian area for each order of headwater streams within a watershed was multiplied by its stream length. This gives the total area occupied by the riparian buffers along an order of the headwater streams within that watershed. After calculating the areas occupied by both orders, the percentage of watershed area for headwater riparian buffers was calculated as a proportion of the watershed area following this equation (Equation 3.3).

$$Percentage\ of\ watershed\ area\ (\%) = \frac{Headwater\ stream\ riparian\ area\ (km^2)}{Watershed\ area\ (km^2)} \times 100 \qquad [3.3]$$

For example, within a 40 km$^2$ watershed, 3.2 km$^2$ of headwater stream “functional” riparian area would account 8 % of the watershed area. This calculation was performed on each riparian buffer type and regional averages of percent watershed areas were calculated.

**Results**

<u>Drainage density</u>

The robust one-way ANOVA indicates the mean drainage densities between regions were significantly different between each other ($F (4, 9.54) = 7.458$, p-value = 0.005). The one-way trimmed means comparison revealed that this significant difference lies between the Lake States and the Pacific Northwest region. The drainage densities of other regions were not significantly different from each other.

With a mean drainage density of 2.52 km/km$^2$, the PNW had the highest watershed drainage density of the five regions (Table 3.5 and Figure 3.5). The average headwater streams represented between 70 – 80 % of entire stream networks within watersheds across the regions (Table 3.5).

**Table 3.5: Descriptive statistics of watershed drainage density and headwater stream network percent area within watersheds in the Lake States, Northeast, Pacific Northwest, Southeast, and Western regions of the United States.**

| Region | n | Watershed drainage density[2] (km/km$^2$) | | | Headwater stream network percentage[3] | | |
|---|---|---|---|---|---|---|---|
| | | Mean | Median | Std. Dev | Mean | Median | Std. Dev |
| LS | 6 | 1.23 | 1.07 | 0.717 | 72% | 75% | 6% |
| NE | 9 | 1.68 | 1.63 | 0.393 | 76% | 77% | 4% |
| PNW | 4 | 2.52 | 2.48 | 0.489 | 75% | 75% | 2% |
| SE | 8 | 1.56 | 1.48 | 0.878 | 79% | 79% | 9% |
| W | 7 | 2.26 | 1.66 | 1.33 | 75% | 73% | 6% |

[2] Watershed drainage density was calculated using Equation 3.1.
[3] Headwater stream network percentage was calculated using Equation 3.3.

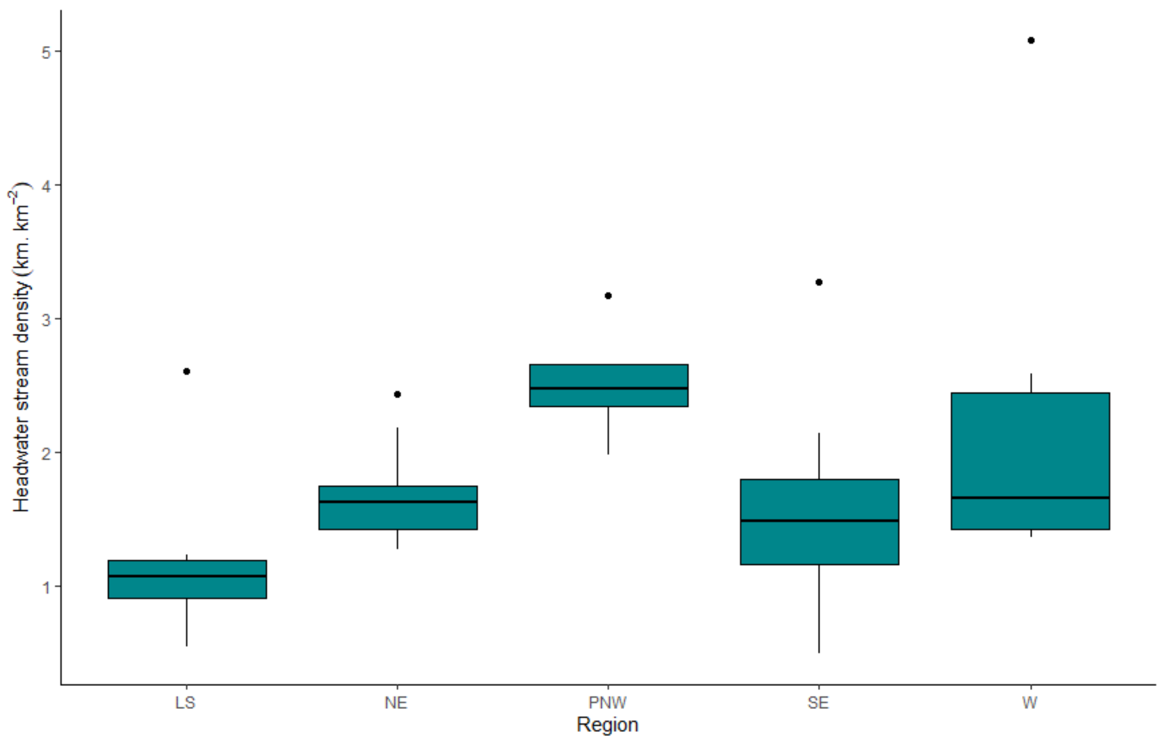



**Figure 3.5: Boxplot of stream densities of headwater streams across the Lake States, Northeast, Pacific Northwest, Southeast, and Western regions within the United States.**

The highest drainage density of 5.08 km/km$^2$ was recorded in a watershed within the Mendocino National Forest in California while the lowest drainage densities between 0.49 - 0.61 km/km$^2$ was recorded in woody wetland watersheds within the Hiawatha National Forest in Michigan and Marion National Forest in South Carolina (Table 3.6).

**Table 3.6: Drainage densities of watersheds in sampled states.**

| Region name | Region code | State | Watershed name/ID | Drainage density (km/km$^2$) |
|---|---|---|---|---|
| Northeast | NE | New Hampshire | WM1 | 1.35 |
| | | | WM2 | 1.42 |
| | | Vermont | GM1 | 1.75 |
| | | | GM2 | 1.63 |
| | | New York | Huntington Wildlife Forest | 1.64 |
| | | | Frost Valley | 2.43 |
| | | Pennsylvania | Farnsworth | 1.27 |
| | | | Salmon Creek | 1.42 |
| Lake States | LS | Michigan | Hiawatha | 0.54 |
| | | | Ottawa | 2.61 |
| | | Wisconsin | Taylor County WS | 1.09 |
| | | | Price County WS | 1.05 |
| | | Minnesota | Burnside | 1.23 |
| | | | Superior | 0.86 |
| Southeast | SE | West Virginia | Pocahontas | 1.55 |
| | | | Pendleton | 1.34 |
| | | South Carolina | Echaw Creek | 0.49 |
| | | | Wedboo Creek | 0.61 |
| | | Mississippi | Sugar-Coffee Bogue | 2.14 |
| | | | Rocky Branch | 3.27 |
| | | Arkansas | Dardanelle | 1.68 |
| | | | Ouachita | 1.41 |
| West | W | Wyoming | Fish Creek | 1.39 |
| | | Arizona | Lookout Lakes | 1.66 |
| | | | Moquitch Canyon | 2.31 |
| | | Idaho (lower) | Granite Creek | 1.36 |
| | | | Minneha Creek | 1.45 |
| | | California | North Fork Creek | 5.08 |
| | | | Smith Neck Creek | 2.59 |

| Region name | Region code | State | Watershed name/ID | Drainage density (km/km²) |
|---|---|---|---|---|
| Pacific Northwest | PNW | Washington | Quilcene River | 2.46 |
| | | | Skokomish River | 3.17 |
| | | Oregon | South Fork Cow Creek | 2.49 |
| | | | Thunder Creek | 1.98 |

<u>“Functional” riparian buffer differences across regions and states</u>

The nonparametric factorial ANOVA test revealed that the land area allocated for “functional” riparian buffers across regions were significantly different (p-value < 0.0001). Similarly, there was a significant difference in “functional” riparian areas between states (p-value < 0.0001) and stream orders (p-value = 0.014) across states. The Pacific Northwest and Western regions were significantly different from the Northeast, Southeast, and Lake States regions while the Northeast and Southeastern regions were significantly different from the Lake States region (Figure 3.6). In states such as AZ and WA, standardized “functional” riparian areas were over 100 m (328 ft.) along both stream orders with median “functional” riparian buffers greater than 50 m (164 ft.) (Figure 3.7). States such as MN and WI had standardized “functional” riparian areas extending less than 50 m (162 ft.), with median “functional” riparian buffers less than 25 m (82 ft.) (Figure 3.7).

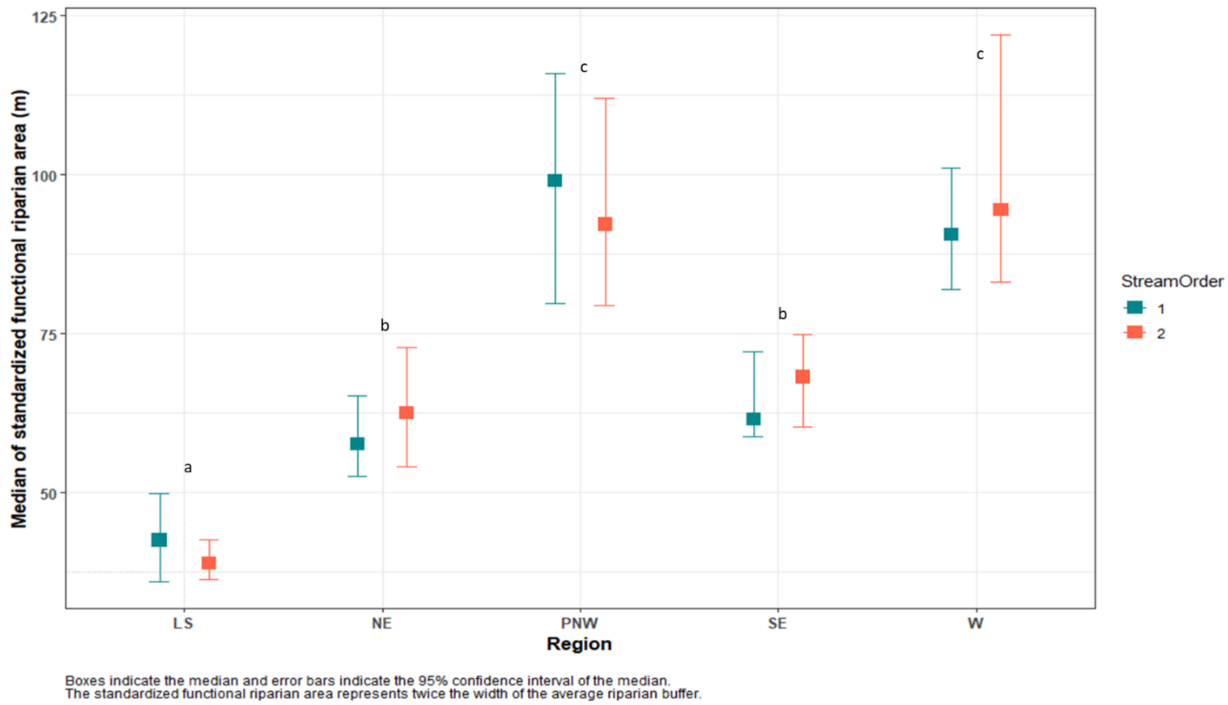



**Figure 3.6: Standardized "functional" riparian area differences of headwater streams across the Lake States, Northeast, Pacific Northwest, Southeast, and Western regions within the United States. Regions not sharing the same letters are significantly different from each other.**

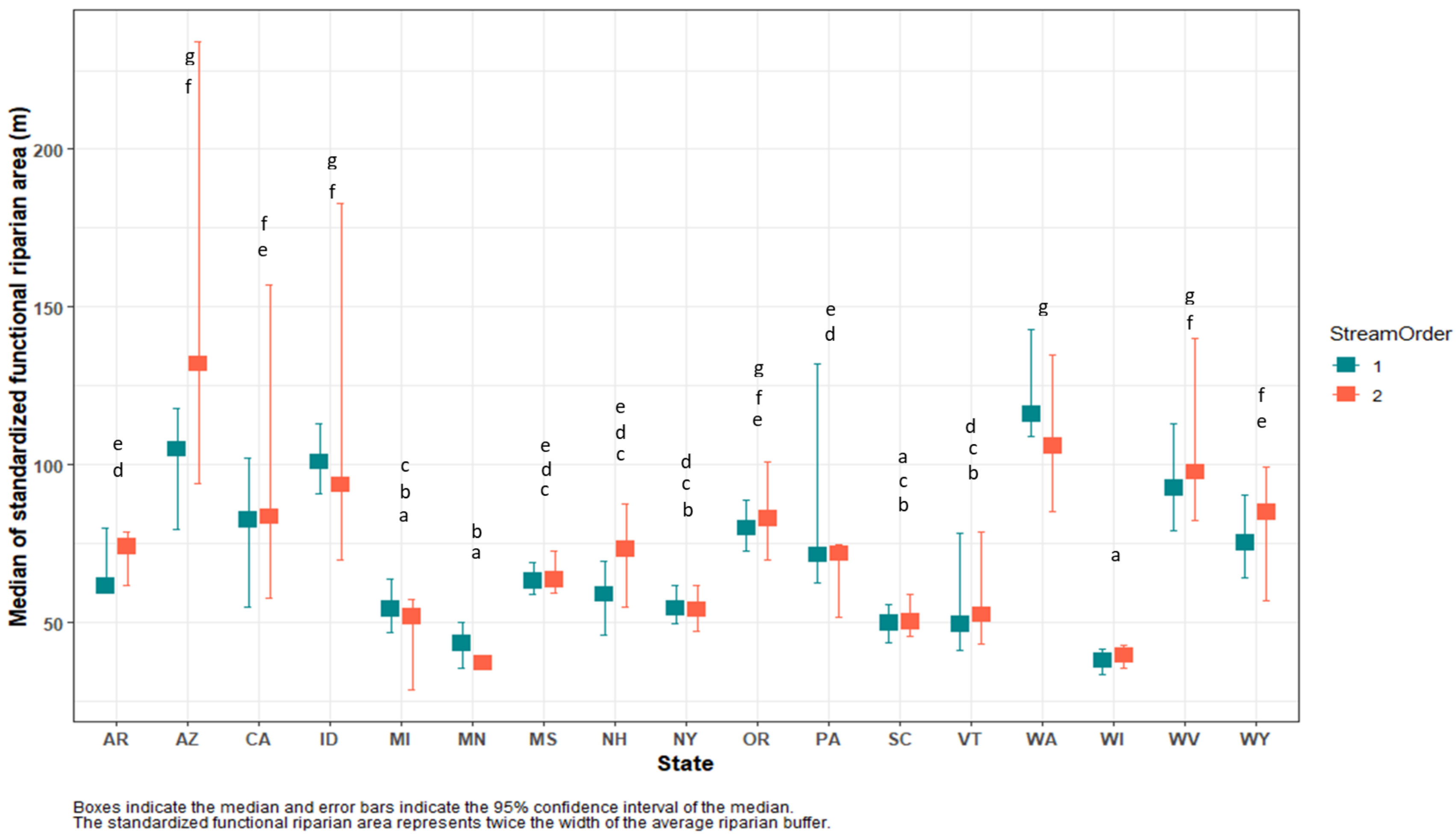



**Figure 3.7: Standardized "functional" riparian area differences of headwater streams across states in the United States. States not sharing the same letter groupings are significantly different from each other.**

The ANOVA test for the assessment of horizontal terrace slope distance provided a topographic comparison of stream terraces across regions and states. Significant difference in slope distances were observed across regions (p-value <0.0001) and states (p-value <0.0001). Similar to the functional buffer analysis, the Pacific Northwest and Western regions were significantly different (p-value <0.0001) from the Northeast, Southeast, and Lake States regions while the Northeast and Southeastern regions were significantly different from the Lake States region (Figure 3.8). Horizontal terrace slope distances were greatest along streams in the Western region where the median terrace slope distance along second-order streams in AZ was 50 m and extended up to 100 m (Figure 3.9). Headwater streams in WI showed no terrace development indicating very low topographic relief within sampled watersheds which is observed by the 0 m median horizontal terrace slope distance (Figure 3.9). The median horizontal terrace slope distance ranged between 0 – 12 m (0 – 40 ft.) in MI, MN, MS, SC, NY, OR, PA and VT.

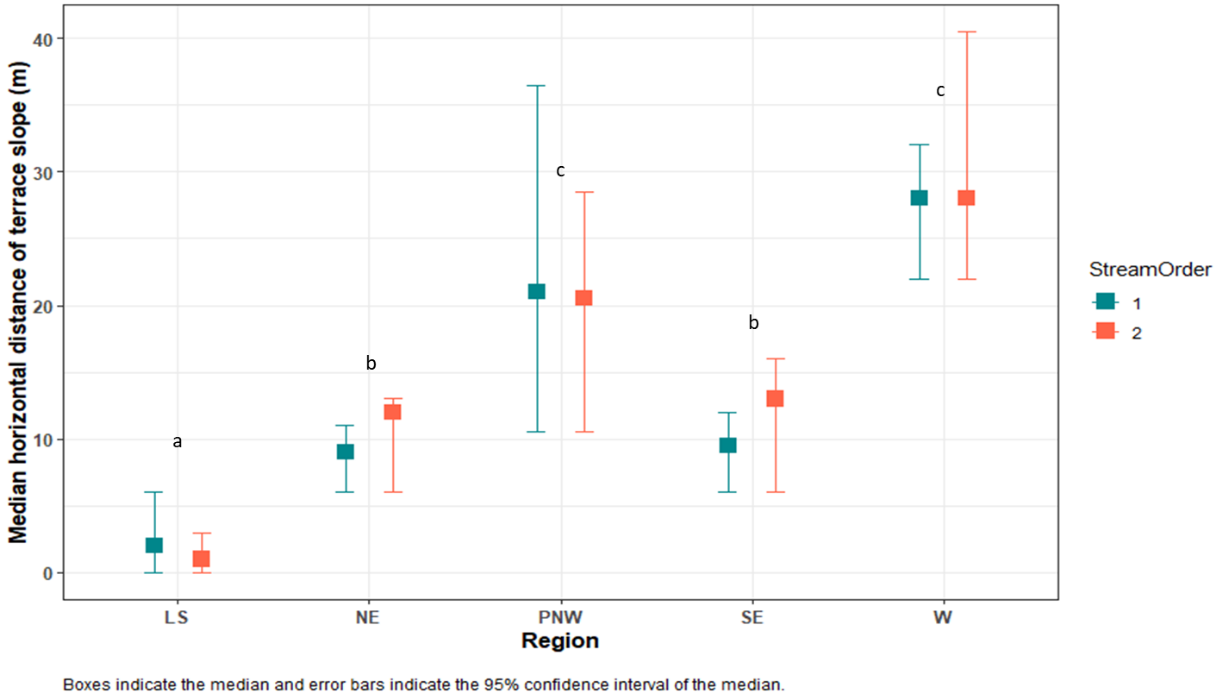



**Figure 3.8: Horizontal terrace slope distances of headwater streams across the Lake States, Northeast, Pacific Northwest, Southeast, and Western regions within the United States. Regions not sharing the same letters are significantly different from each other.**

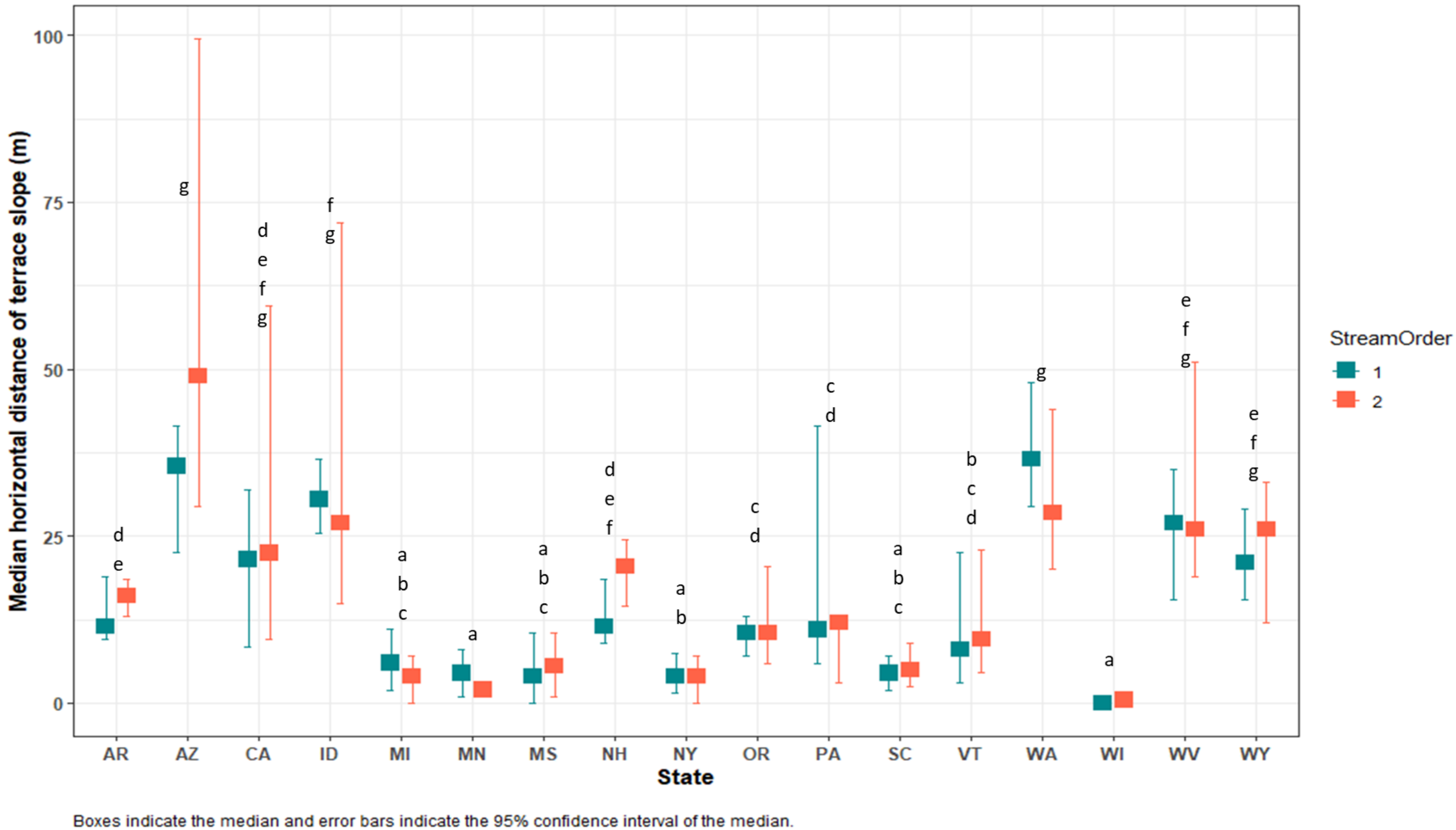



**Figure 3.9: Horizontal terrace slope distance of headwater streams across States in the United States. States not sharing the same letter groupings are significantly different from each other.**

State-specific riparian buffer differences across regions and states

Land area dedicated to state-specific riparian buffers along headwater streams showed significant differences (p-value <0.0001) between regions and states. There was also a significant difference between land area as delineated by state-specific riparian buffers between first- and second-order streams across states (p-value = 0.029). State-specific riparian buffers allocated more land area along headwater streams in the Lake States, resulting in a significant difference from other regions (Figure 3.10). This is exemplified by MN which recorded the widest state-specific riparian buffer allocation, reaching up to 90m across the stream (standardized riparian area) (295ft) or approximately a 45 m (148 ft.) buffer (Figure 3.11). Significant differences in state-specific riparian land area allocations were also observed between first- and second-order streams in OR and WI where second-order streams delineated more riparian area than their first-order streams, and in WV where first-order stream riparian areas were greater than their second-order streams (Figure 3.11)

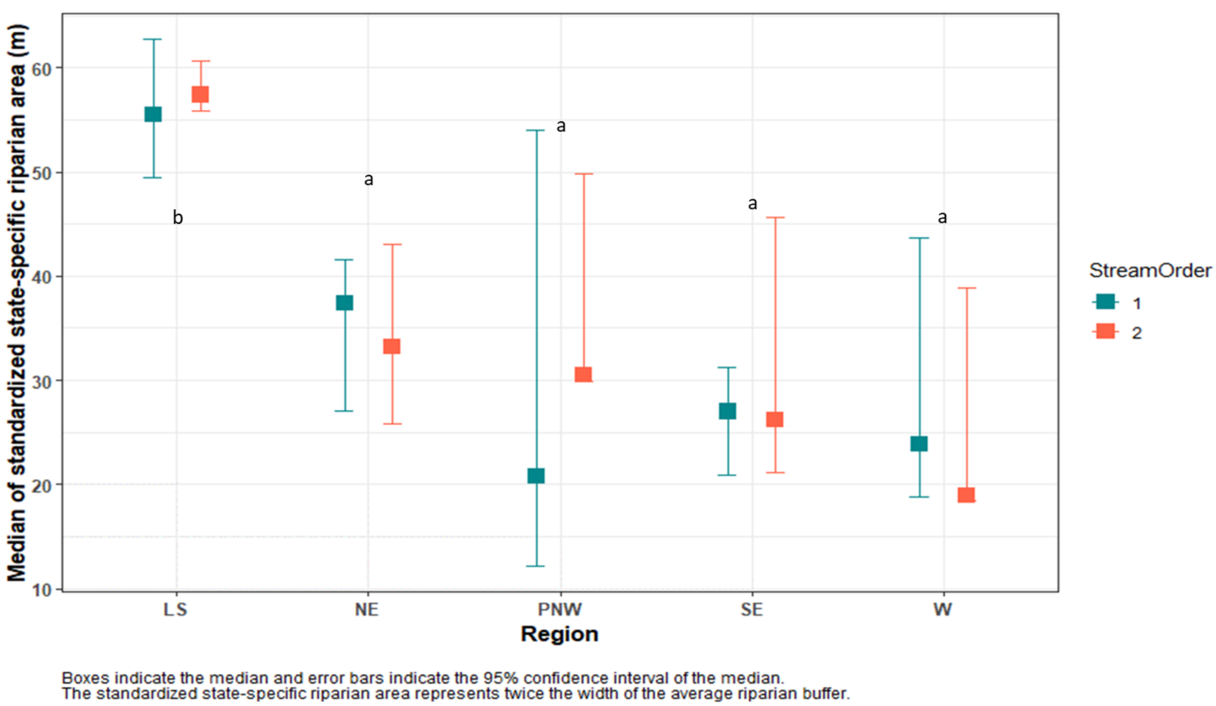



**Figure 3.10: Standardized state-specific riparian area differences of headwater streams across the Lake States, Northeast, Pacific Northwest, Southeast, and Western regions within the United States. Regions not sharing the same letters are significantly different from each other.**

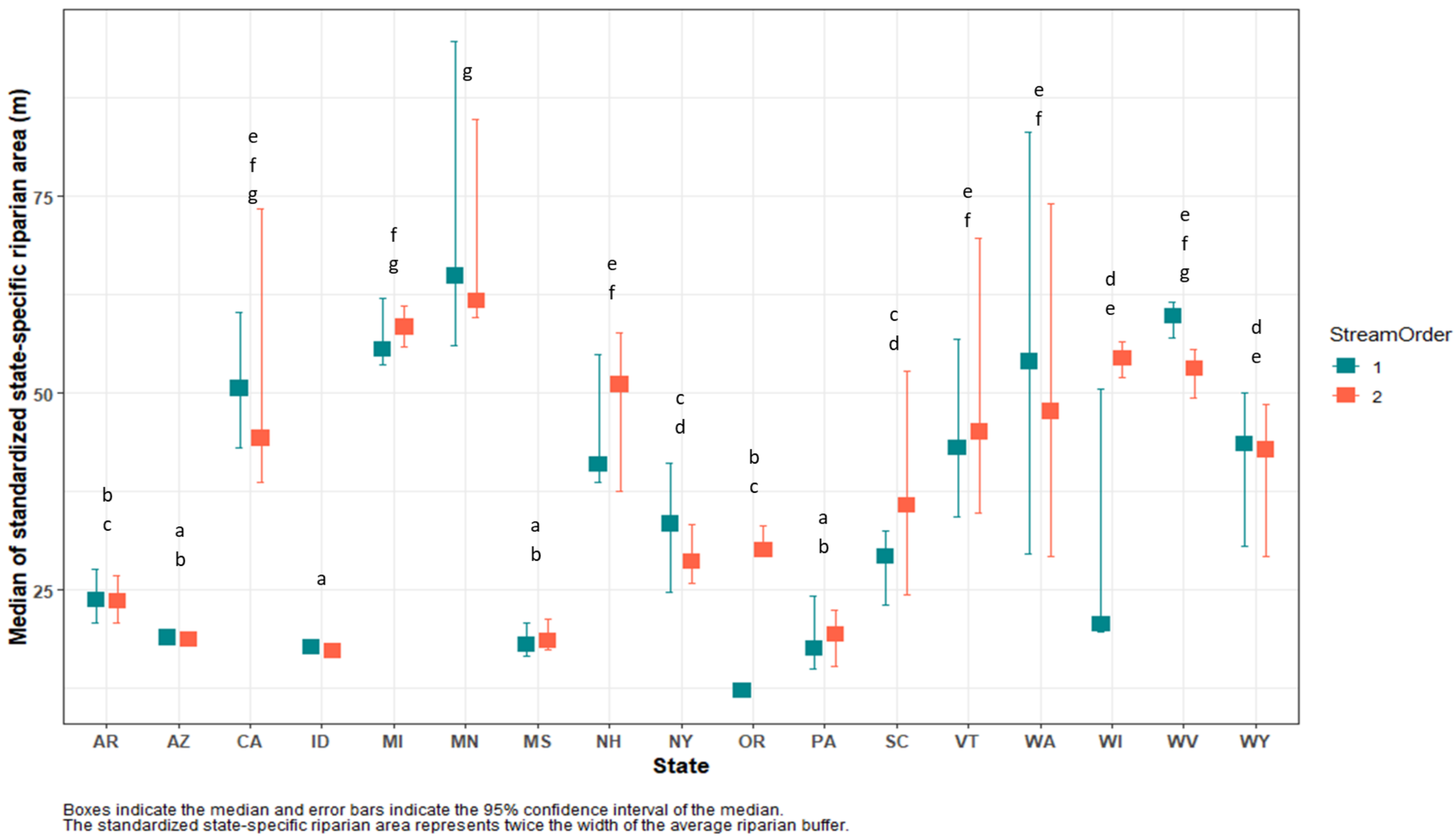



**Figure 3.11: Standardized state-specific riparian area differences of headwater streams across States in the United States. States not sharing the same letter groupings are significantly different from each other.**

Comparison of a “functional” riparian buffer with a state-specific riparian buffer

There was a significant difference in land area allocations between a “functional” riparian buffer and state-specific riparian buffer between regions (p-value <0.0001) and between states (p-value <0.0001). The Pacific Northwest and Western regions dedicated more “functional” riparian area than their state-specific riparian area when compared to the Lake States, Northeast, and Southeast (Figure 3.12). The Lake States, except for first order-streams in WI, allocated more land area using state-specific riparian buffers as riparian when compared to a “functional” buffer allocation (Figure 3.13). The median “functional” riparian area and state-specific riparian area delineated along headwater streams in MI and VT were comparable to each other based on the slight deviation observed from the 0 m difference level (Figure 3.13). Standardized “functional” riparian areas in western region states of AZ and ID allocated over a 75 m (246 ft.) distance across headwater streams than their respective standardized state-specific riparian areas. This is approximately an average 37.5 m (123 ft.) buffer difference (Figure 3.13).

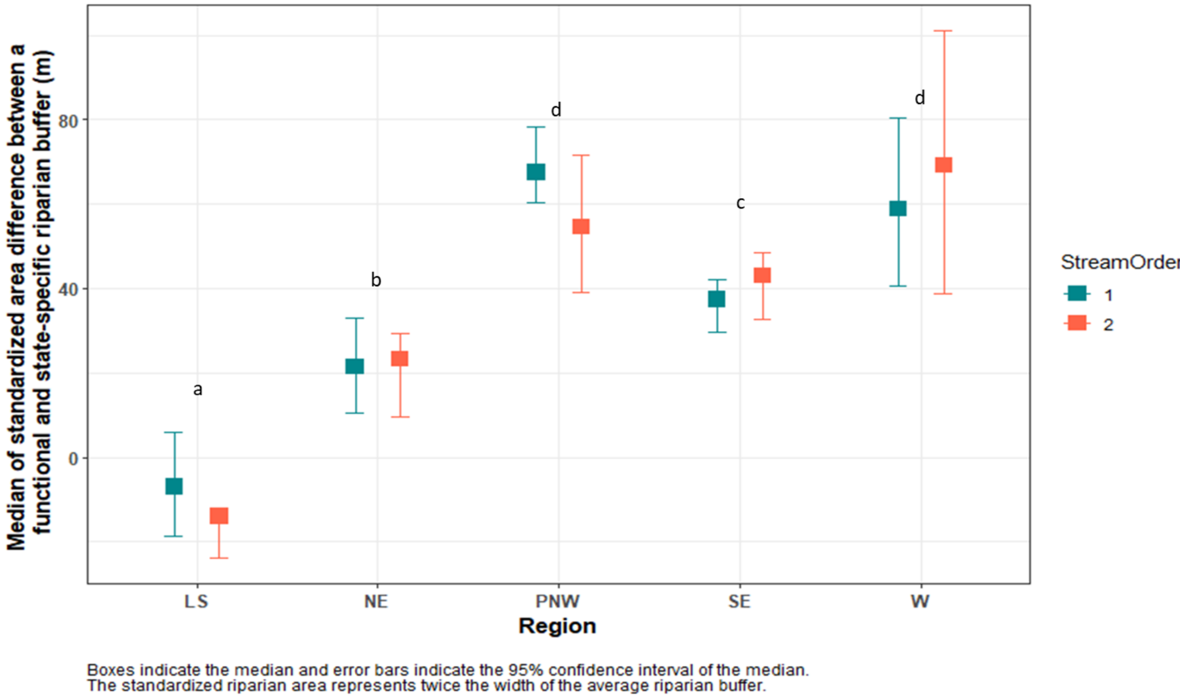



**Figure 3.12: Standardized area difference between a "functional" riparian buffer and state-specific riparian buffer along headwater streams across the Lake States, Northeast, Pacific Northwest, Southeast, and Western regions within the United States. Regions not sharing the same letters are significantly different from each other.**

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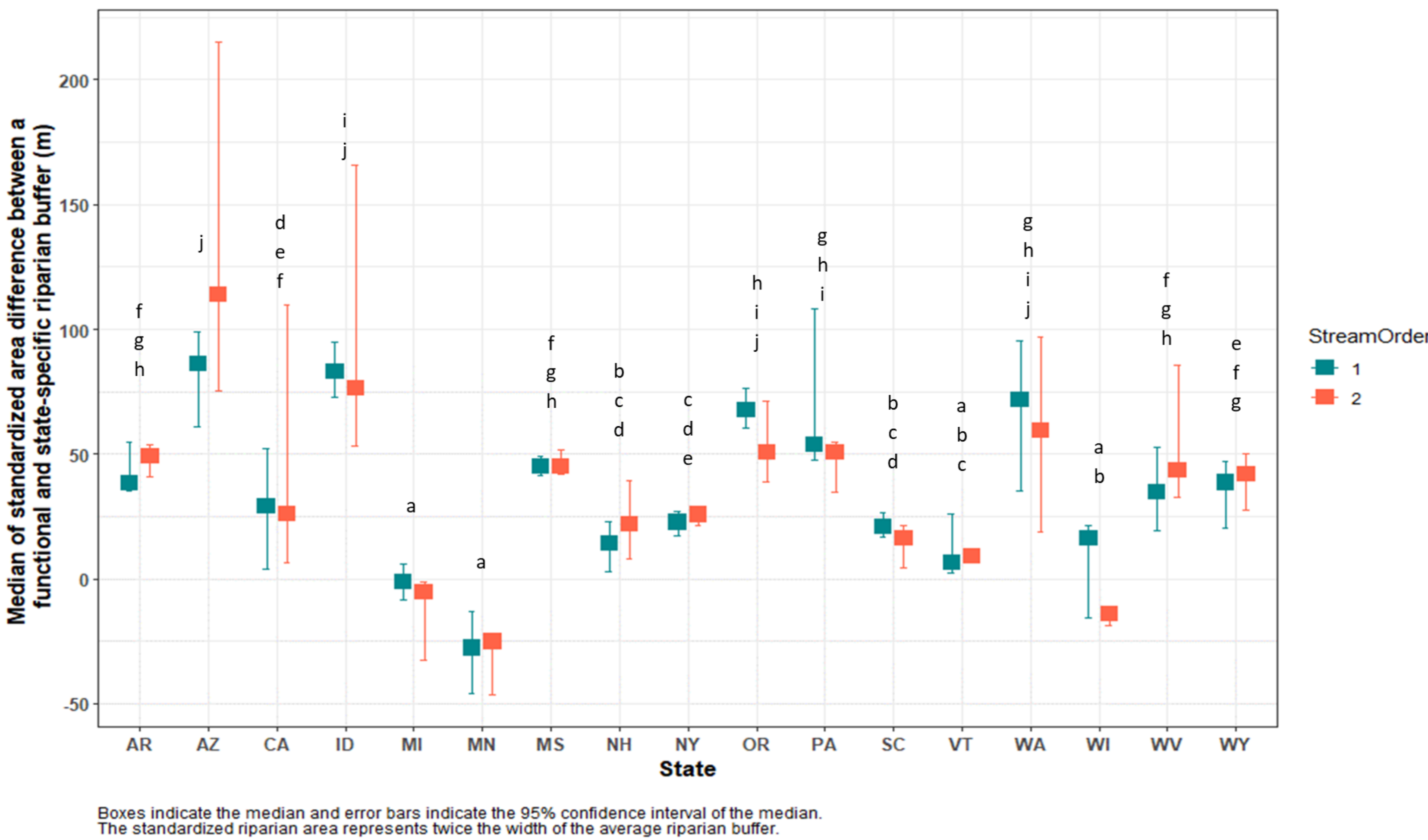


**Figure 3.13: Standardized area difference between a "functional" riparian buffer and state-specific riparian buffer along headwater across States in the United States. States not sharing the same letter groupings are significantly different from each other.**

Comparison of a “functional” riparian buffer with a 30 m (100-ft) fixed-width riparian buffer

The difference in land area allocation between a “functional” riparian buffer and 30 m fixed-width riparian buffer was significantly different between regions (p-value <0.0001) and between states (p-value <0.0001). A significant difference between stream orders was also observed across states (p-value = 0.001). The Pacific Northwest and Western regions significantly allocated more “functional” buffer land area than a 30 m fixed-width riparian buffer when compared to the Lake States, Northeast, and Southeast (Figure 3.14). The Lake States region was significantly different from all other regions as more land area was delineated using a 30 m fixed-width riparian buffer than a “functional” riparian buffer. The 30 m fixed-width riparian buffer allocated more land area than the “functional” riparian buffer along headwater streams in MN, SC, VT, WI, and along second-order streams in MI (Figure 3.15). The “functional” riparian buffer was comparable to the 30 m fixed-width riparian buffer in NY, with a 0m difference between the areas of the two buffer types (Figure 3.15).

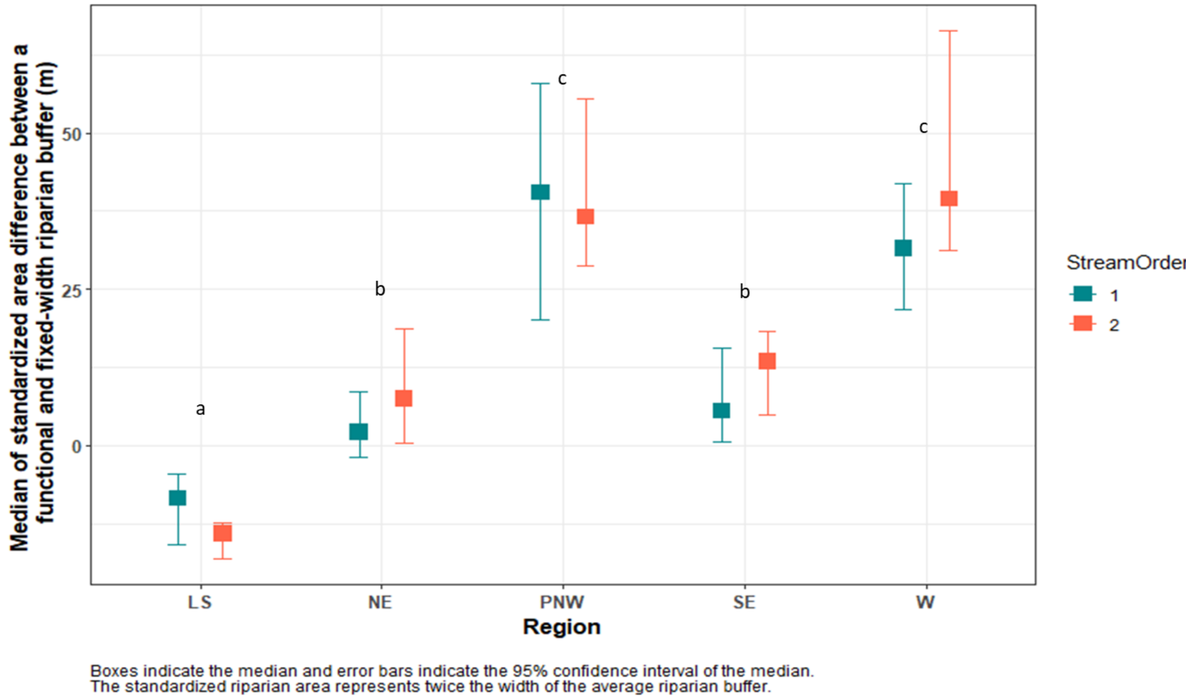


**Figure 3.14: Standardized area difference between a "functional" riparian buffer and 30 m (100 ft.) fixed-width riparian buffer along headwater streams across the Lake States, Northeast, Pacific Northwest, Southeast, and Western regions within the United States. Regions not sharing the same letters are significantly different from each other.**

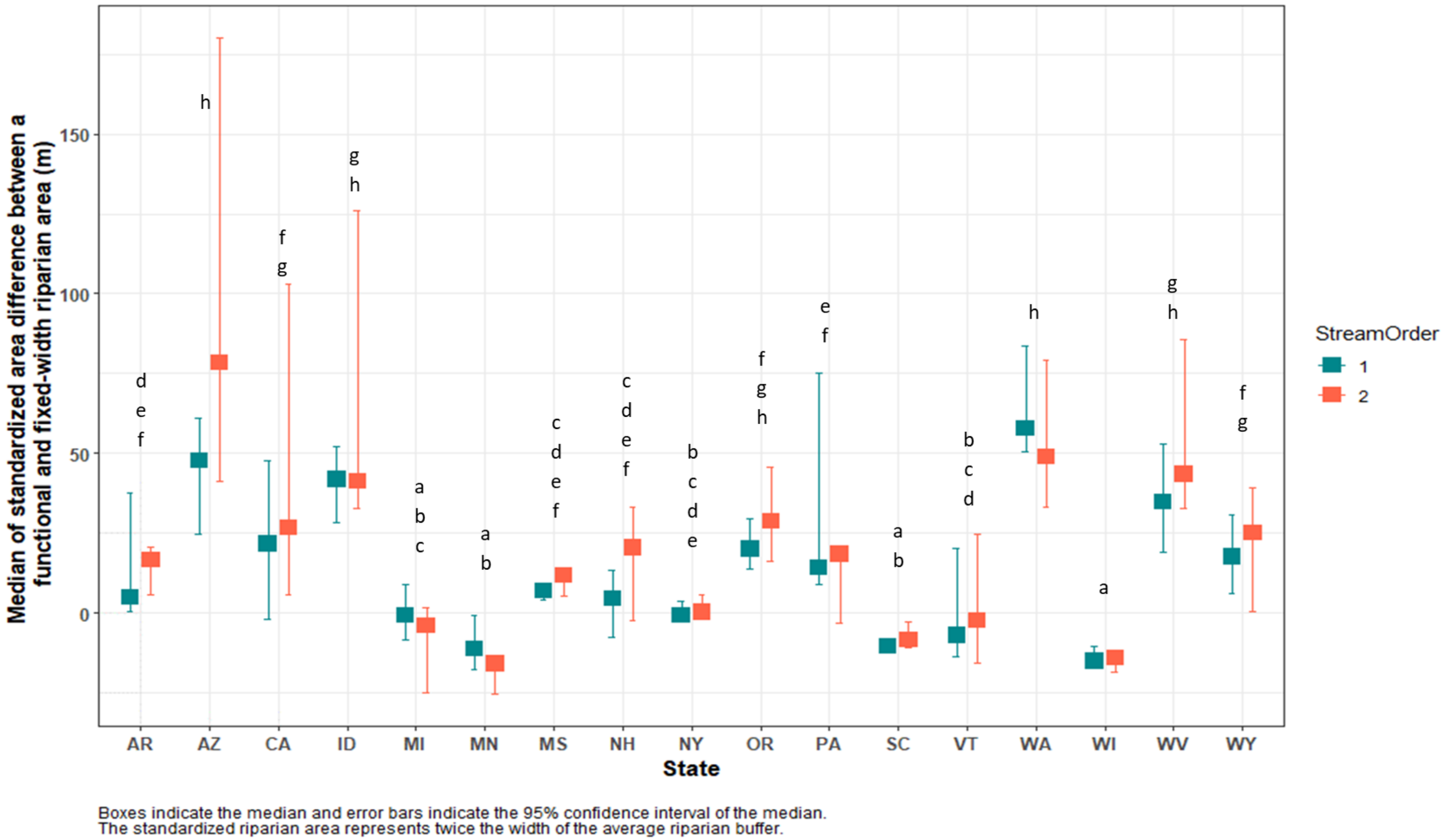


**Figure 3.15: Standardized area difference between a "functional" riparian buffer and 30 m (100 ft.) fixed-width riparian buffer along headwater streams across states in the United States. States not sharing the same letter groupings are significantly different from each other.**

Comparison of a 30 m (100 ft.) fixed-width riparian buffer with state-specific riparian buffers

There was a significant difference in land area allocation between 30 m fixed-width riparian buffers and state-specific riparian buffers between regions (p-value <0.0001) and between states (p-value <0.0001). The buffer differences were also significantly different between stream orders across regions (p-value <0.0001), and across states (p-value = 0.043). The Lake States region was significantly different from all other regions, allocating more land area for a 30 m fixed-width riparian buffer than the state-specific riparian buffer (Figure 3.16). Second-order streams in WI and headwater streams in WV have a 30 m state-specific riparian buffer allocation, and thus showed no difference with the 30 m fixed-width riparian buffer. The median standardized state-specific riparian area for MI was comparable to the standardized 30 m fixed-width riparian area, with a 0 m area difference in the two buffer types (Figure 3.17). There was a significant difference between first- and second- order streams in OR and ID where a 30 m fixed-width riparian buffer allocated more land area as riparian along first-orders in comparison to the state-specific riparian buffer (Figure 3.17).

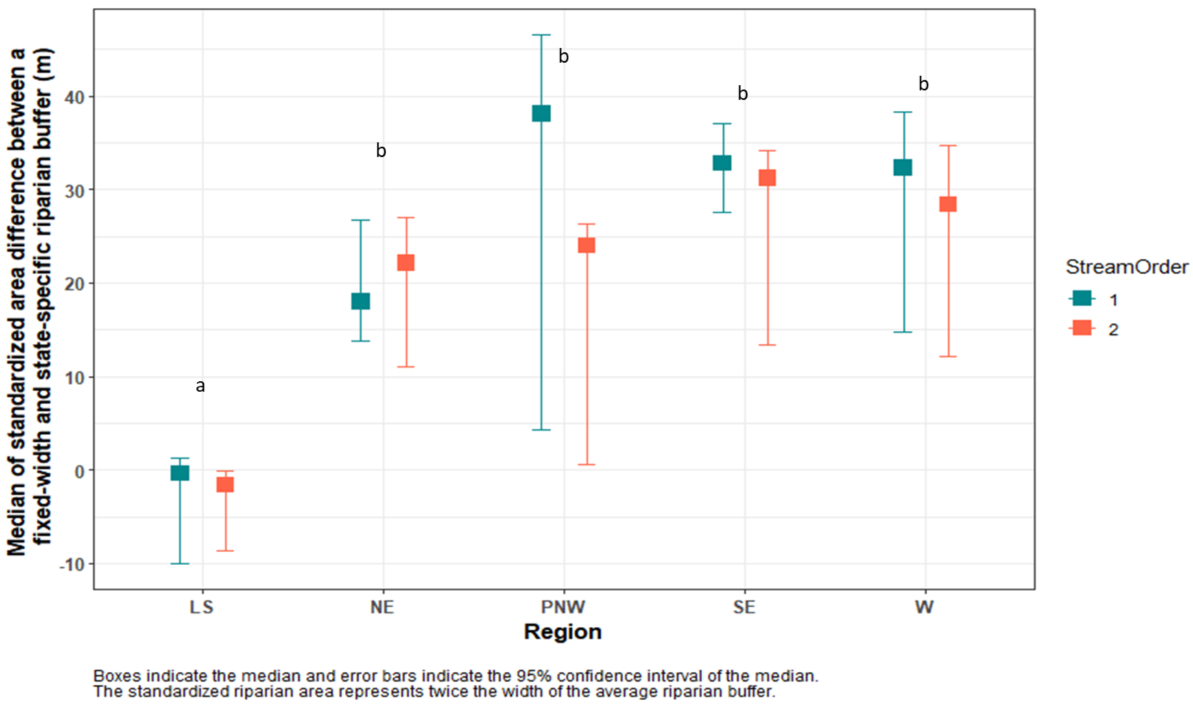


**Figure 3.16: Standardized area difference between a 30 m fixed-width riparian buffer and state-specific riparian buffer along headwater streams across the Lake States, Northeast, Pacific Northwest, Southeast, and Western regions within the United States. Regions not sharing the same letters are significantly different from each other.**

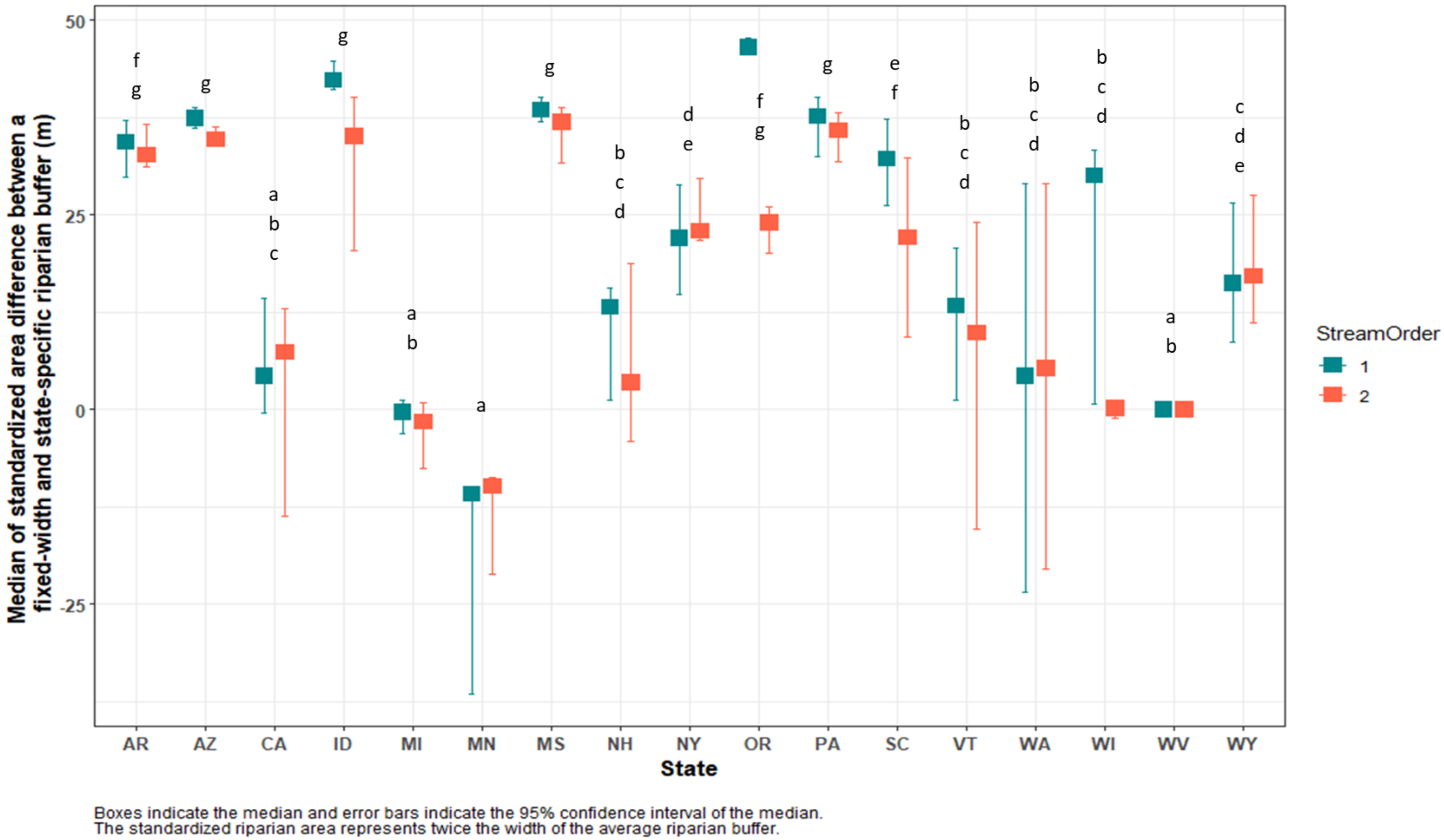


**Figure 3.17: Standardized area difference between a 30 m (100 ft.) fixed-width riparian buffer and state-specific riparian buffer along headwater streams across states in the United States. States not sharing the same letter groupings are significantly different from each other.**

Riparian buffer areas in watersheds

Of the three types of riparian buffers, the “functional” riparian buffer allocated the most area within sampled watersheds in all regions except for the Lake States region (Table 3.7 and Figure 3.18). On average, “functional” riparian buffers made up 4.2 % of watershed area in the Lake States region to 19.5 % of watershed area in the Pacific Northwest. The 100 ft. fixed-width RMZ ranged between 4.8% of watershed area in the Lake States region to an average of 11 % in the Pacific Northwest. State-specific RMZs occupied between 3.4 % of watershed area in the Southeast region to 7.5 % in the Pacific Northwest. The average percentage of state-specific riparian buffer areas within watersheds were the lowest of the three RMZ buffer types in the Northeast, Pacific Northwest, Southeast, and Western regions.

**Table 3.7: Percentages of watershed area occupied by headwater stream “functional” riparian buffers, 30-m (100-ft) fixed-width riparian buffers, and state-specific riparian buffers distributed across the Lake States (LS), Northeast (NE), Pacific Northwest (PNW), Southeast (SE), and the Western (W) region of the United States.**

| Region | n | Average buffer area percentage | | |
|---|---|---|---|---|
| | | “Functional” | 100 ft. Fixed width | State-specific |
| LS | 6 | 4.2 % ± 1.4 | 4.8 % ± 1.3 | 5.2 % ± 1.3 |
| NE | 8 | 7.6 % ± 0.6 | 6.9 % ± 0.5 | 4.6 % ± 0.7 |
| PNW | 4 | 19.5 % ± 3.5 | 11.0 % ± 0.8 | 7.5 % ± 3.6 |
| SE | 8 | 8.5 % ± 1.5 | 6.8 % ± 1.2 | 3.4 % ± 0.6 |
| W | 7 | 16.4 % ± 2.9 | 9.6 % ± 2.3 | 6.4 % ± 2.9 |

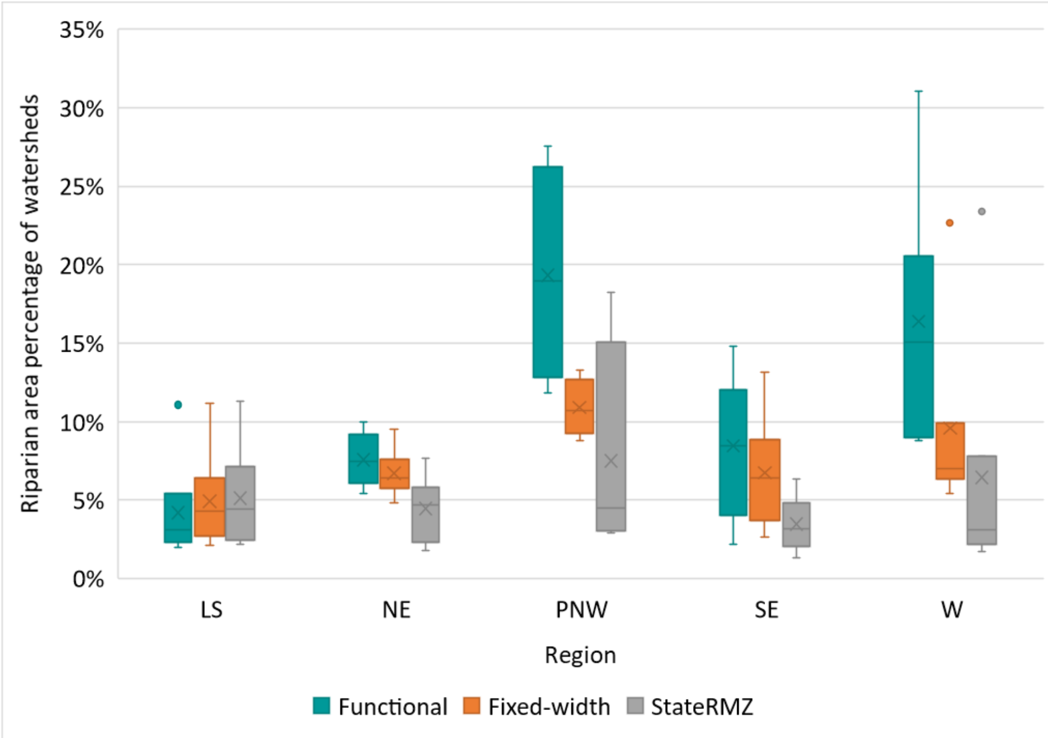


**Figure 3.18: Boxplot of percent watershed area of headwater stream functional riparian buffers, 30 m (100 ft.) fixed width riparian buffers, and state-specific riparian buffers distributed across the Lake States (LS), Northeast (NE), Pacific Northwest (PNW), Southeast (SE), and Western (W) region of the United States.**

The highest percent watershed area delineated by all three buffer types was recorded in the North Fork Creek watershed within the Mendocino National Forest in California (Table 3.8). The lowest percent watershed area delineated by all three buffer types was recorded in the woody wetland watersheds of South Carolina and woody wetland watershed in the Hiawatha National Forest in Michigan (Table 3.8).

**Table 3.8: Percent watershed area occupied by "functional" riparian buffers, 30-m (100-ft) fixed width riparian buffers, and state-specific riparian buffers around headwater streams.**

| Region name | Region code | State | Watershed name/ID | Percent watershed area | | |
|---|---|---|---|---|---|---|
| | | | | "Functional" | Fixed width | State-specific |
| Northeast | NE | New Hampshire | WM1 | 6 % | 6 % | 5 % |
| | | | WM2 | 8 % | 6 % | 5 % |
| | | Vermont | GM1 | 10 % | 8 % | 8 % |
| | | | GM2 | 5 % | 7 % | 5 % |
| | | New York | Huntington Wildlife Forest | 7 % | 7 % | 4 % |
| | | | Frost Valley | 10 % | 10 % | 6 % |
| | | Pennsylvania | Farnsworth | 8 % | 5 % | 2 % |
| | | | Salmon Creek | 7 % | 6 % | 2 % |
| Lake States | LS | Michigan | Hiawatha | 2 % | 2 % | 2 % |
| | | | Ottawa | 11 % | 11 % | 11 % |
| | | Wisconsin | Taylor County WS | 3 % | 4 % | 4 % |
| | | | Price County WS | 3 % | 4 % | 3 % |
| | | Minnesota | Burnside | 4 % | 5 % | 6 % |
| | | | Superior | 2 % | 3 % | 5 % |
| Southeast | SE | West Virginia | Pocahontas | 12 % | 6 % | 6 % |
| | | | Pendleton | 8 % | 5 % | 5 % |
| | | South Carolina | Echaw Creek | 2 % | 3 % | 1 % |
| | | | Wedboo Creek | 3 % | 3 % | 2 % |
| | | Mississippi | Sugar-Coffee Bogue | 11 % | 9 % | 3 % |
| | | | Rocky Branch | 15 % | 13 % | 4 % |
| | | Arkansas | Dardanelle | 8 % | 8 % | 3 % |
| | | | Ouachita | 9 % | 7 % | 3 % |
| West | W | Wyoming | Fish Creek | 9 % | 7 % | 5 % |
| | | Arizona | Lookout Lakes | 13 % | 6 % | 2 % |
| | | | Moquitch Canyon | 21 % | 9 % | 3 % |
| | | Idaho (lower) | Granite Creek | 9 % | 5 % | 2 % |
| | | | Minneha Creek | 15 % | 7 % | 2 % |
| | | California | North Fork Creek | 31 % | 23 % | 23 % |
| | | | Smith Neck Creek | 17 % | 10 % | 8 % |
| Pacific Northwest | PNW | Washington | Quilcene River | 22 % | 11 % | 6 % |
| | | | Skokomish River | 28 % | 13 % | 18 % |

| Region name | Region code | State | Watershed name/ID | Percent watershed area | | |
|---|---|---|---|---|---|---|
| | | | | "Functional" | Fixed width | State-specific |
| | | Oregon | South Fork Cow Creek | 16 % | 11 % | 3 % |
| | | | Thunder Creek | 12 % | 9 % | 3 % |

## Discussion

This study investigated and contrasted three riparian buffer delineation techniques: a "functional" riparian buffer, state-specific riparian buffer, and a 30 m (100 ft.) fixed width riparian buffer, on headwater streams across five timber producing regions of the contiguous United States. The "functional" riparian buffer was delineated based on the field key defined by Ilhardt et al. (2000). This delineation key is based on the topography and forest composition around the stream. The majority of riparian functions are realized within the limits identified by this boundary (Swanson et al. 1982; Gregory et al. 1991; Naiman et al. 2005) and thus, is considered a close representation of a functional riparian area. The aim of this study was to evaluate the land area differences between the three buffer types and to assess the compatibility of state-specific riparian buffers with a variable width buffer recommended as a "functional" riparian buffer by the U.S Forest Service. Assigning the appropriate riparian buffer width is an important management decision due to the high density of first- and second-order streams in working forest landscapes. Delineating riparian buffers as management zones should not undermine riparian protection and neither should it create an economic burden to the landowner.

The "functional" riparian buffer delineation identifies terrace slope distances along the lateral widths of a stream. Thus, this assessment revealed terrain characteristics of sampled watersheds in each state. It should also be noted that these sampled watersheds are not representations of the entire topography of those states and that terrain characteristics revealed through the assessment maybe unique to that watershed. The median horizontal terrace slope distance was greater or close to 25 m in watersheds in the Western region and this was reflected by the wider standardized "functional" riparian areas for these states. Both states in the Pacific Northwest delineated wider standardized "functional" riparian areas, but only Washington state displayed wider horizontal terrace slope distances. The average canopy tree heights of Douglas-fir of approximately 30 m (100 ft.) in Oregon contributed to the wider "functional" riparian buffers within its sampled watersheds. Therefore, the topography around headwater streams in sampled watersheds of Western region states and watersheds in the Olympic Peninsula in Washington have wider ravines compared to watersheds in other states. Based on the terrace slope assessment, the sampled watersheds in the Northeastern and Southeastern region states were comparable to each other with moderate horizontal slope distances (0 – 12 m) except in West Virginia where wider terrace slope development was observed in its watersheds in the more mountainous Monongahela National Forest. The Lake States of Minnesota and Michigan also displayed moderate slope distances along first- and second-order streams. Headwater streams in Wisconsin displayed no terrace development, indicating that the "functional" riparian buffer within its watersheds was determined only by its average canopy tree height of roughly 20 m (65 ft). Thus, this variable width riparian buffer depicts the topography around the streams through the terrace slope distance, and forest structural characteristics through the average canopy tree height.

Comparisons between the state-specific riparian buffers and "functional" riparian buffers revealed that in states such as Michigan and Vermont, the combined topography and forest structure represented by the "functional" riparian buffer is reflected in their state-specific riparian buffer. According to state-specific riparian buffers, the Lake States region allocated the widest buffers around headwater streams within their watersheds while the Western region states on average delineated narrower state-specific riparian buffers compared to other regions. However, the opposite was observed for buffers delineated using the functional riparian buffer for these two regions. When comparing the two types of buffer allocations, the Lake States (except MI) delineated more land area as riparian using state-specific riparian buffers compared to the “functional” riparian buffer while the Western states delineated significantly less land area using state-specific riparian buffers when compared to the “functional” riparian buffer. In a meta-data analysis of a riparian efficacy study by (Sweeney and Newbold 2014), sediment trapping efficiencies of between 65 – 85 % were calculated for riparian buffers between 10 - 30 m. The compiled studies were conducted either on streamside buffers receiving flow from an unconfined upslope (undefined slope) area, or on plots which the peak hydraulic loading onto the buffer exceeded 1.0 l/s/m measured traverse to flow. The state-specific riparian buffers for Arizona and Idaho are less than or equal to 10 m which may not properly buffer sediment runoff during forest operations within these watersheds. Given that these watersheds within Arizona and Idaho displayed high topographic relief with wide ravines, riparian buffers required to regulate sedimentation may exceed 30 m in watersheds with similar topography. Similarly, in Washington and Oregon, state-specific guidelines define fixed width riparian buffers for non-fish bearing streams that are less than or equal to 15 m. Groom et al. (2011) reported that 31 - 52 m riparian buffers along first- to third-order streams within the Coastal Range Forests of Oregon did not show changes in maximum stream temperature post-harvest. However, they recorded an average increase of 0.7 °C in stream temperature on streams with 15 - 21 m buffers. Changes in stream temperature can have significant impacts on trout habitat (Beschta et al. 1987) and increases in sedimentation can have adverse impacts on aquatic habitat for both macro and micro invertebrate communities (Newbold et al. 1980; Davies and Nelson 1994). Having displayed wide ravines in the terrace slope analysis, particularly in watersheds of Washington, the state-specific riparian buffers for the Pacific Northwest fail to represent the actual topography and functional riparian area for headwater streams with similar characteristics in that region.

Buffer comparisons of a “functional” riparian buffer and a 30 m (100 ft.) fixed width riparian buffer revealed that these two buffers were comparable to each other in sampled watersheds of New York and Michigan. This means that within these states, a 30 m fixed width riparian buffer could be used in place of a variable width buffer such as the "functional" riparian buffer along headwater streams with similar topography and forest structure. When comparing the "functional" riparian buffer with the 30 m (100 ft.) fixed width riparian buffer, Jayasuriya et al. (2019) reported that there was no significant difference between the two buffer types from a case study based at the Frost Valley Model Forest in the Catskill region of NY. However, in a broader study across the entire region, they discovered that the 100 ft. fixed width buffer over-delineated land as riparian along first-order streams while failing to capture the full extent of a riparian area along second-order streams. Allocating a 100 ft. fixed width riparian buffer around headwater streams in Minnesota, South Carolina, Vermont, and Wisconsin, in place of a "functional" riparian buffer dedicates more land area as riparian. If the variable width functional approach represents the actual extent of a riparian area, a 30 m fixed width buffer may create an opportunity cost for the landowner by over-delineating forest land as riparian area within these watersheds. Minnesota was the single state that delineated more land area as riparian using its state-specific riparian buffer when compared to a 30m fixed width riparian buffer. This is due to the allocation of a 50 m (165 ft.) riparian buffer around trout streams in state BMP guidelines. Michigan was the only state where all three buffer types delineated approximately the same land and therefore, they could be used interchangeably within watersheds comparable to the topography and forest composition of the sampled watersheds in Michigan.

The "functional" approach captures the various widths of ravines along a stream that are easily erodible, thereby ensuring bank stability. With an additional one-tree length distance it captures functions of, but not limited to shade, stream temperature regulation, allochthonous inputs of fine and coarse woody debris, and wildlife habitat (Sweeney and Newbold 2014; Gregory et al. 1991; Swanson et al. 1982). Depending on the topography, a fixed-width buffer may fail to capture most of these functions and stream protection would be compromised, especially with buffers that don't fully capture the extent of wider ravines. “Functional” riparian buffer areas usually result in RMZs that exceed buffers commonly used around the US, whether defined as fixed or variable width buffers in their State BMP guides. This is the case in this study for most states except for sampled watersheds in the Lake States. Forested watersheds with similar topography and forest structure as represented by the sampled watersheds in the Lake States will likely experience opportunity costs due to state specific guidelines that define riparian buffers greater than those identified by a “functional” riparian buffer. However, in watersheds where state-specific riparian buffers fail to capture broad ravines on steeper landscapes (West and Pacific Northwest watersheds), they should ensure that riparian buffers incorporate full terrace slope distances where soil is more susceptible to erodibility.

Drainage Density

Headwater streams dominated channel networks by their cumulative stream lengths in all watersheds. They represented between 70 – 80 % of entire stream networks across all regions within the contiguous US. The mean watershed stream densities in the Lake States, Northeast and Southeastern regions ranged between 1.23 – 1.68 km/km$^2$ while watersheds in the Pacific Northwest and Western regions recorded higher values ranging between 2.26 – 2.56 km/km$^2$. Wemple et al. (2001) recorded drainage densities of 3.0 and 2.9 km.km$^{-2}$ in two watersheds west of the Cascade Range in Oregon. The current study recorded similar values in the Pacific Northwest watersheds with values ranging from 1.98 km/km$^2$ in the Cascade Range of Oregon to 3.17 km/km$^2$ in the Olympic Peninsula of Washington. Kuska and Arra (1973) recorded drainage densities of between 0.087 – 1.119 km/km$^2$ within watersheds of the St. Croix river drainage basin that spans over Minnesota and Wisconsin. Their values were however calculated using USGS map scales of 1:62,500 as opposed to the higher resolution 1:24,000 map scale of the NHD layer used in this study. Despite the difference in map resolutions used, the mean and median drainage density of 1.23 km/km$^2$ and 1.07 km/km$^2$ for the Lake States region is consistent with Kuska and Arra (1973).

The watersheds west of the Cascade Mountains in Northern California within the Mendocino National Forest, yielded 5.08 km/km$^2$, the highest drainage density of the sampled Western region states. In previous studies, a higher range of 7.5 – 8.2 km/km$^2$ was recorded in watersheds within Tennessee Valley, California by Montgomery and Dietrich (1989). Watersheds within the San Dimas Experimental Forest in Southern California have recorded a large variation in drainage densities ranging from 8.39 – 20.47 km/km$^2$ (Patton and Baker 1976). Drainage densities within the Northeast ranged between 1.27 km/km$^2$ in the Allegheny Mountains in Pennsylvania to 2.43 km/km$^2$ in the Catskills Mountains in New York. Jayasuriya et al. (2019) recorded drainage densities for headwater streams in the Catskill region of New York ranging from 0.85 – 1.43 km/km$^2$. This amounts to between 1.12 – 1.88 km/km$^2$ of the total drainage network when extrapolated. Patton et al. (1976) estimated values between 1.60 – 3.57 km/km$^2$ in the Appalachian Plateau from watersheds across the Allegheny Mountains, Allegheny Plateau, and the Cumberland Plateau.

High drainage densities within watersheds is indicative of higher rates of surface flow and high topographic relief areas (Patton and Baker 1976). The watersheds in the Western region and Washington of the Pacific Northwest displayed wider 'canyon-like' ravines in the horizontal terrace slope distance assessment. Watersheds with relatively lower drainage densities like in the Northeast and Southeast had intermediate or moderate topographic relief which was observed by the horizontal terrace slope assessment. These watersheds also had high relative stand densities of >70 % (except 1 watershed in NH) and > 65 % in the Northeast and Southwest, respectively. Higher infiltration capacities as a result of high canopy cover and moderate topographic relief could have resulted in the lower drainage densities in these watersheds (Patton and Baker 1976). High surface flow rates in high drainage density watersheds with steep slopes could make land adjacent to streams more susceptible to erosion (EnviroAtlas 2015). Wider riparian buffers may allow for more infiltration time in these areas and reduce the risk of erosion. Therefore, a "functional" riparian buffer that identified wide lateral slope distances of headwater streams may be more appropriate for streams in these watersheds, especially in the Western region and the Pacific Northwest.

Proportion of riparian areas in watersheds

Headwater riparian areas in the Pacific Northwestern region reserved the highest proportions of watersheds. This is due to the high drainage densities, high topographic relief, and wider ravines observed within these watersheds. "Functional" riparian buffers delineated the largest proportion of land as riparian in comparison to the 30-m fixed width riparian buffer and state-specific riparian buffers in all regions except in the Lake States. This was due to the low drainage densities, low topographic relief, and little to no terrace slope development observed within the Lake States watersheds. In a study conducted across six watersheds in the Catskill region of New York, Jayasuriya et al. (2019) estimated that a 30 m (100 ft.) fixed width riparian buffer around headwater streams occupies between 5.20 – 8.68 % of forestland while a "functional" riparian buffer occupies between 5.21 – 9.88 %. This study estimates an average area of 6.9 % for a 30 m (100 ft.) fixed width riparian buffer and an average area of 7.6 % for a "functional" riparian buffer within watersheds of the Northeast. Additionally, this study recorded an average area of 10 % dedicated to headwater streams with a 30 m (100 ft.) fixed width RMZ and a "functional" RMZ for the Frost Valley Model Forest watershed within the Catskills area in New York. Lippke et al. (2002) estimated that 14.8 % of commercial forest land in western Washington would fall within riparian buffer of 45.7m (150 ft) for fish-bearing streams (classes I to III), 30 m (100 ft.) for class IV streams, and 15.2 m (50 ft.) for class V streams. These buffer parameters reflected state-specific riparian buffers during the time of this research. When extrapolated to represent headwater streams, the percent acreage of riparian areas represented approximately 11 % of the commercial forest. This study recorded an average area of 12 % delineated for state-specific riparian areas along headwaters for the watersheds of the Olympic Peninsula in Washington. In a study done across the USDA Crossett Experimental Forest, University of Arkansas Forest, Ouachita National Forest, and

Ozark National Forest, Kluender et al. (2000) reported an average 6.3 % of forestland dedicated to all streams with a 20 m (66 ft.) fixed width riparian buffer. If extrapolated to percent headwater streams represented by Southeastern watersheds, the average forestland dedicated to riparian areas along headwater streams would be approximately 5 %.

Drainage density plays a key role in determining the proportion of the forestland delineated as riparian areas in addition to the buffer type used. As drainage densities within a watershed increase, the percent acreage of riparian areas increases proportionately. This is evident within the Pacific Northwest and Western watersheds of this study. Headwater streams tend to be under-represented by current NHD layers within forested watersheds where their actual densities may be higher than those represented by hydrographic datasets (Baker et al. 2007; Brooks and Colburn 2011; Elmore et al. 2013). Field verification is therefore required and recommended when mapping headwater streams within working forests for management as this can have a significant impact on costs for allocating riparian buffers regardless of buffer type used.

Delineating variable width buffers

Many studies support the application of a variable width riparian buffer over a fixed width buffer (Ilhardt et al. 2000; Tiwari et al. 2016; Tomer et al. 2003) because variable width riparian buffers are more likely to capture one or more ecological functions and/or is a representation of the topography of the landscape. Many of these variable width buffers have not been designed to be easily adopted for forest managers. Furthermore, the costs for delineating variable width buffers will be determined by the technical expertise of natural resource professionals. Based on the definition of a "functional" riparian area provided by Ilhardt et al. (2000), Abood et al. (2012) developed a Riparian Buffer Delineation Model for mapping "functional" riparian buffers. This tool has now been developed to be adopted in a national context and is published as a riparian delineation tool (RBDM v3.5) by the US Forest Service (Abood et al. 2019). For better representation and an accurate prediction, this model utilizes several data inputs such as hydrology data (streams, lakes, watersheds), 50 year flood height values per stream type and order, wetlands, soils, elevation, and land cover (Abood et al. 2012). The Ridge-finder tool developed in this study only requires three data layers of streams, elevation (preferably LiDAR derived high-resolution), and average canopy tree height of the forested landscape. USGS Stream gauges for calculating 50 year flood height may not always fall within a managed watershed and data may not be readily available for use due to computational requirements of the RBDM v3.5 tool. The Ridge-finder tool is more easily adoptable for practitioners, requiring only readily available data from The National Map (TNM) powered by the USGS and inventory from a timber cruise for canopy tree heights. However, the Ridge-finder tool is not fully developed to the scale of the RBDM v3.5 and remains a work-in-progress. Ultimately, it has the potential to be developed as an ArcGIS tool through python scripting to increase its user-friendly features. Currently, the model utilizes a combination of tools existing on the ArcGIS Pro toolbox and R programming.

Williams et al. (2003) developed a GIS procedure to delineate variable width riparian buffers using state RMZ guidelines in the Southeastern states. Their procedure calculates the average slope within a predefined fixed width buffer to generate the average side slope ("Focal mean") of the stream to which buffer widths are assigned to slope intervals specified in state BMP manuals. Lemoine et al. (2006) developed a GIS-based analytical tool in ArcGIS to delineate streamside management zones (or RMZs) along streams in the Cumberland Plateau of Tennessee. They first digitized the streamside management zones using aerial photos and derived 50-m GIS sections of the polygon. After drawing "width lines" to a straightened stream path within the polygon, they obtained slope percentage values on the “width lines”. Using a framework (unspecified) they categorized and analyzed the width and slope data onto streams using Tennessee State streamside management zones standards. The method developed in this study (section 2.3.5) for delineating state defined riparian buffers can be adopted to any guideline/regulation that uses slope as a function of the variable width. This method only uses available tools in ArcGIS Pro and is reproducible.

Implications for Management

RMZs are an integral part of forestry BMPs for controlling sediment runoff and protecting other riparian values during and after forest operations. Regardless of whether RMZs are regulated or voluntary, it is important to define riparian buffers as RMZs around headwater streams to protect and preserve the integrity of forested landscapes. As this study reveals, the percentage of area designated as riparian areas along headwater streams can range from as low as 3.4 % to nearly 20 %. Jayasuriya et al. (2019) estimated that riparian areas in the Catskill region of NY represented a stumpage value for northern hardwoods of over $3,707 /ha ($1,500 /ac.). They reported that if RMZ harvesting restrictions limited removals to 1MBF/ac., the opportunity cost of allocating RMZs along headwater streams accounted for 7 % of the total timber revenue for that timberland. Lakel et al. (2015) recorded values from as little as $135 /ha ($55 /ac.) to $3,128 /ha ($1,266 /ac.) for stands that were mainly composed of loblolly pine and white oak in an efficacy study estimating the minimum riparian width along first order streams in watersheds of the Piedmont Plateau in Virginia. Considering differences in forest cover types across the five regions in this study, the riparian area or RMZ can hold a substantial amount of valuable timber based on studies estimating opportunity costs for RMZs (Ice et al. 2006; Jayasuriya et al. 2019; Jayasuriya et al. 2020; Lakel et al. 2015). Thus, partial harvesting management options such as selection harvesting, without compromising riparian functions, should be allowed within RMZs to decrease the cost of allocating riparian buffers along headwater streams.

RMZs should represent the topography and structure of the forested landscape. Adopting a 'one-size fits all' buffer or in other words, a fixed width buffer, could under-represent or over-represent the extent of a 'functional' riparian area except in instances which there is no significant difference between a 30 m (100 ft.) fixed width riparian buffer and a “functional” riparian buffer as seen in this study. Given the availability of published GIS tools (Abood et al. 2012) and the new GIS tools developed in this study, the assignment of a variable width buffer, such as a "functional" riparian buffer or other variable width riparian buffers (per state BMP manuals) is recommended.

**Conclusion**

Headwater streams dominate channel networks, comprising between 70 – 80 % of entire stream networks in all watersheds. The high densities of headwater streams within working forests provide challenges to forest managers seeking to conduct financially viable timber operations while simultaneously protecting riparian ecosystem functions. With increased stream densities, delineating the appropriate riparian buffer around headwater streams has become an important management decision due to concerns of overestimating or underestimating riparian areas using different buffer types.

Fixed width riparian buffers customarily applied due to convenience may fail to capture the topography of landscapes and characteristics of forests around headwater streams. The "functional" riparian buffer used in this study captures the variable widths of ravines that are easily erodible and additional distances ensuring allochthonous inputs of fine and coarse woody debris. Thus, variable width buffers such as the "functional" buffers are more likely to provide an actual representation of an ecologically meaningful riparian area. Given the topographic variation observed across the contiguous US, states should ensure that the topography and forest structure/composition around headwater streams are properly represented in state-specific RMZ guidelines.

Sampled watersheds in this study only provide a fraction of topography seen within a state and do not represent the topographic range seen across working forests within those states. Therefore, further research on terrain analysis that includes more watersheds distributed across a state is required to facilitate revising or amending state-specific RMZ guidelines. The increasing attention received for ‘tailored riparian area protection’ highlights the importance of other riparian functions in addition to water quality protection, such as (but not limited to) biogeochemical cycling, groundwater recharge, biomass accumulation, and wildlife habitat. Based on the results of this study, I recommend that forest managers adopt ecologically significant RMZ allocations.

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## Appendix 3A

Summary of forestry best management practice (BMP) guidelines for riparian management zones/streamside protection zones across the United States. Guidelines were referred from their respective States using the most recent BMP Field Guides available online.

| State | Waterbody | Width | Specification | Harvesting restriction | Implementation |
|---|---|---|---|---|---|
| Alabama | intermittent and perennial streams | minimum of 35 ft from a definable bank | Width should be extended to account for erodibility of soil, steepness of slopes and activities to be performed outside. If wildlife is a major objective, a minimum SMZ of 50 feet is recommended. | perennial- Partial cut only within minimum of 35 ft; partial cut or regeneration cut beyond 35 ft. Minimum Residual Cover: 50%<br>intermittent- Partial cut or regeneration cut when water quality degradation can be avoided. Minimum Residual Cover: vegetative<br>No mechanical site preparation allowed within SMZs. | Quasi-regulatory |
| Arizona | Not specified | Suggestions made based on the water quality goals of the land manager.<br>Nitrogen: 20 to >40m<br>Sediment: 3 to >10m<br>Phosphorous: >20m<br>Pathogens: 3 to >6m<br>Pesticides: >9m | Geared more towards agricultural BMPs | | Non-regulatory |
| Arkansas | Non-Ephemeral Streams (perennial or intermittent), Ephemeral Streams, Braided Streams, Lakes and Ponds | Non-Ephemeral: 7% slope min. 35 ft., 7- 20% slope min. 50 ft., >20 slope min. 80 ft.<br>Lakes and Ponds: min 35 ft. | | Non-Ephemeral: residual min 50 sq.ft. of BA.<br>Ephemeral streams: no restrictions. | Non-regulatory |

| State | Waterbody | Width | Specification | Harvesting restriction | Implementation |
|---|---|---|---|---|---|
| California | Class I:Domestic water supply on site and/or fish present always/seasonally, Class II: Fish present always/seasonally, Class III: No aquatic life present | Class I: 23-45m, Class II: 15-30m, Class III: 7.5-15m | Based on slope gradient (<30%, 30-50%, >50%) | Class I: Residual 50% overstory and 50% understory, Class II: Residual 50% total canopy and 25% overstory conifers, Class III: Residual 50% understory canopy | Regulatory |
| Colorado | Streams (not classified), lakes and other water bodies | 50-foot-wide strip on both sides of the stream | 50 feet wide on each side of a stream measured from the ordinary (yearly average) high-water mark of a definable bank. | The SMZ is not a "keep out" zone<br>Leave the following adjacent to streams: hardwoods and unmerchantable conifers and shrubs.<br>Leave merchantable trees where there is insufficient vegetation to adequately stabilize stream banks.<br>Do not "clearcut" to the stream edge. | Non-regulatory |
| Connecticut | Streams near Truck roads | 25 - 165ft | determined by slope ranging from 0 - 70% | Timber harvesting is permitted in filter strips, Limited logging machinery.<br>Limit harvesting to 50% crown cover. | Quasi-regulatory |
| Delaware | intermittent and perennial streams, | 50 ft, 75ft, 100ft | based on slope % (0-10%, 11-20%, 21-45% respectively). | Min. residual BA/ac 60 sq.ft.<br>The SMZ shall contain a 25-ft wide NO-HARVEST zone from the top edge of the stream.<br>At the perimeter of the 25 ft NO-HARVEST zone, the SMZ should be a minimum of a 50-ft wide active management area for slopes of 0 to 10% . | Quasi-regulatory |
| Florida | perennial (stream width: < 20 ft., 20-40 ft., > 40 ft.) and intermittent streams, lakes, sinkholes and special waters | Perennial: 35 - 200 ft.(primary zone), 60 ft. (secondary zone for stream widths <20 ft.) | dependent on whether perennial or intermittent and stream width. Divided into 2 zones: primary and secondary. | Primary zone: Clearcut prohibited within 35 ft of all perennial waters and within 50 ft of all waterbodies designated as OFW, ONRW or Class I Waters.<br>Selective harvesting may be conducted to maintain 50% of a fully stocked stand.<br>Secondary zone: Unrestricted selective harvesting and clearcut harvesting | Quasi-regulatory |

| State | Waterbody | Width | Specification | Harvesting restriction | Implementation |
|---|---|---|---|---|---|
| Georgia | Perennial and intermittent streams and other water bodies (lakes, ponds, reservoirs) | Perennial: 40, 70, 100 ft. Intermittent: 20, 35, 50 ft. Perennial trout stream: 100 ft. Intermittent trout stream: 35, 35, 50 ft. | dependent on slope class (Slope class: <20%, 21-40%, >40%) | Perennial streams: Residual of 50 sq.ft. BA/ac or 50% canopy cover. Trout streams: no-harvest in first 25 ft.<br>Intermittent streams: Residual of 25 sq.ft. BA/ac or 25% canopy cover. Trout streams: Residual of 50 sq.ft. BA/ac | Non-regulatory |
| Idaho | Class I and II streams (based on watershed area) | Class I: 75 ft., Class II: 30 ft. | | Operation of ground-based equipment is not allowed<br>Option 1: Within 25 ft from the ordinary high water mark on each side of the stream, live conifers and hardwoods will be retained to maintain a min. 60 sq.ft./ac BA. Between 25 - 75 ft. min. 25 sq.ft./ac BA.<br>Option 2: Within 50 ft from the ordinary high water<br>mark on each side of a stream, live conifers and hardwoods will be retained to maintain a min. 60 sq.ft./ac BA. Between 50 - 75 ft. min.10 sq.ft/ac BA. | Regulatory |
| Illinois | Perennial and intermittent streams, | Intermittent - 25 - 145 ft., Perennial - 50 - 290 ft., Lakes-50 ft. | Based on slopes ranging from 0 - 60%. For every 10% increment of slope, Intermittent buffer increases by 20 ft. and Perennial buffer increases by 40 ft. | No restrictions. Must maintain adequate vegetation cover. | Quasi-regulatory |
| Indiana | perennial and intermittent streams, sinkholes, reservoirs, lakes and ponds | Perennial: 50 - 200 ft., Intermittent: 25 - 105 ft. | Based on perennial stream width (20', 20-40', >40') and slope gradient. | Cut few if any trees within 15 feet of permanent watercourses. Retain at least 50% canopy cover in the primary RMZ on perennial water courses. | |

| State | Waterbody | Width | Specification | Harvesting restriction | Implementation |
|---|---|---|---|---|---|
| Iowa | perennial and intermittent streams | 50ft, 75ft, 150ft | Based on stream width (20', 20 - 40', >40') | Minimize harvesting in and around the RMZ | Non-regulatory |
| Kentucky | Perennial streams, lakes and ponds | warm water aquatic habitats: 25 or 55 ft., cold water aquatic habitats: 60 ft. | WAH based on slope gradient <15% and >15% | WAH: retain 50% tree cover, CAH: 75% tree cover | Regulatory |
| Louisiana | perennial and intermittent streams | intermittent- 35ft perennial 50ft - 100ft | Based on perennial stream width (<20ft., >20ft.) | Permanent residual tree cover is not required along intermittent and ephemeral streams if vegetation and organic debris are left to protect the forest floor during regeneration. | Non-regulatory |
| Maine | streams, lakes, ponds, and non-forested wetlands | 25 - 165 ft. | Based on slope gradient. Width increases by 20ft with a 10% increase in slope (0 - 70%) | As determined by logger, no legal restriction | Quasi-regulatory |
| Maryland | Ephemeral, intermittent and perennial streams | 50 - 90ft | Based on slope gradient. Width increases by 10 ft. with a 5% increase in slope. | Harvesting or machine operation not allowed without SMZ plan. | Regulatory |
| Massachusetts | Ponds, lakes, regulated streams, and certified vernal pools. | 50ft - 450ft | Based on slope gradient. Width increases by 40 ft with a 10% increase in slope. | No logging equipment may operate in a filter strip unless it is included in an approved forest cutting plan.<br>No more than 50%/ac of BA may be cut at any one time in a buffer strip.<br>Sensitive streams: 15ft no-cut buffer | Regulatory |
| Michigan | perennial and intermittent streams, lakes, ponds, or other open water bodies (e.g. open water wetlands) where | 100 - 175 ft. | Based on slope gradient. | Residual BA of 60-80 ft2/ac | Quasi-regulatory |

| State | Waterbody | Width | Specification | Harvesting restriction | Implementation |
|---|---|---|---|---|---|
| Minnesota | Streams (trout bearing and non trout bearing), lakes, open water wetlands | 50 - 165 ft. | Based on stream width and trout bearing/non-trout bearing stream | Residual BA of 60 sq.ft/ac | Non-regulatory |
| Mississippi | Perennial, Intermittent and drains | Perennial: 30 - 60 ft., Intermittent: 30 ft. | Based on slope gradient | Perennial: must leave 50% crown cover | Non-regulatory |
| Missouri | Perennial and intermittent streams | 50ft - 145ft | Based on slope gradient | Residual BA of > 40 sq.ft/ac | Non-regulatory |
| Montana | Class 1-3 streams | 50ft, 100ft | Based on slope gradient. Class 1 and 2 streams and lakes: 50 ft. for slopes <35%. 100 ft. for slopes > 35%. Class 3 streams: 50 ft. | Retain at least 50% of the trees ≥ 8 inches DBH on each side of stream or 10 trees per 100-foot segment, whichever is greater. | Quasi-regulatory |
| Nebraska | Perennial and intermittent streams | 50, 75, 200ft. | Based on stream width (20ft, 20-40ft, > 40ft.) | Selective harvesting | Non-regulatory |
| Nevada | lake, reservoir, stream | 200ft | | No harvest and machine operation zone | Quasi-regulatory |
| New Hampshire | wetland, intermittent and perennial streams | 50ft - 110ft | Based on slope gradient. Width increases 20 ft. with a 10% increase in slope. | Limited harvesting | Quasi-regulatory |
| New Jersey | pond, lake, stream, marsh | 25ft-200ft | Based on slope gradient and soil erodibility | Harvesting that limits soil disturbances | Regulatory |
| New Mexico | perennial and intermittent streams, lakes and wetlands | 50ft | from a stream, lake, or any wetland area | Timber harvesting or thinning in the SMA should only be done to remove invasive species or otherwise restore the health of the ecosystem | Regulatory |

| State | Waterbody | Width | Specification | Harvesting restriction | Implementation |
|---|---|---|---|---|---|
| New York | intermittent or perennial streams, lakes, ponds, regulated wetlands | 35 ft - 100 ft. | Based on slope gradient. Zone 1: 15ft. Zone 2: 20 - 85ft. | Maintain forest cover in Zone 1. Residual 60 sq.ft./ac of BA or residual 50% canopy cover within Zone 2. | Quasi-regulatory |
| North Carolina | Perennial and intermittent streams, and perennial waterbodies | 50 ft. min. Ranges from 30 - 300 ft. | Can be adapted based on management objectives (sediment control, nutrient management, streambank stabilization, aquatic and wildlife habitat | Limit heavy equipment usage within 10 ft. of the stream bank, and maintain half of pre-harvest canopy cover. | Quasi-regulatory |
| North Dakota | intermittent and perennial streams | 60ft - 150ft. | Based on stream width and slope gradient | Limit harvesting within 15 ft of the ordinary high-water mark, targeting only problem trees. Retain trees necessary for bank stabilization. Do not remove all trees from the riparian area. | Non-regulatory |
| Ohio | perennial, intermittent, and ephemeral streams, ponds, or lakes | 25 - 225 ft. for common logging areas. 50 - 450 ft. for municipal watersheds and critical areas. | Based on slope gradient | The filter strips along perennial streams may be selectively harvested only. All trees casting shade on the stream should be left. | Quasi-regulatory |
| Oklahoma | perennial, intermittent or ephemeral streams | perennial: 50ft minimum width intermittent: 35ft minimum width | Minimum width recommendations for slopes < 20%. | Residual BA of 50 sq.ft/ac | Non-regulatory |
| Oregon | Streams types F,D, and N | Type F: 50 - 100 ft., Type D: 20-70 ft., Type N: 50 - 70 ft. | Based on stream size of small, medium and large | Various prescriptions based on geographic regions of Coast Range and South Coast, Interior and Western Cascade, Siskiyou, Eastern Cascade and Blue Mountain | Regulatory |
| Pennsylvania | Streams, ponds and spring seeps | Temporary ponds and spring seeps: 50 ft., Streams: 25 - 165 ft. | Based on slope gradient for streams. Increase width by 20 ft. with slope increase of 10%. | Maintain at least 50% crown cover as a residual stand to prevent an increase in water and ground surface temperature. | Quasi-regulatory |

| State | Waterbody | Width | Specification | Harvesting restriction | Implementation |
|---|---|---|---|---|---|
| South Carolina | perennial, intermittent, and ephemeral streams and ponds or lakes | Primary zone: 40 ft. (non-trout), 80 ft. (trout streams), Secondary zone: 0 - 120 ft. | Secondary zone width based on slope gradient | Primary SMZ: On perennial streams, residual of 50 sq.ft./ac of BA. On intermittent streams, permanent residual tree cover is not required as long as other vegetation and organic debris are left.<br>Secondary SMZ: Use all types of silvicultural harvest systems. | Quasi-regulatory |
| South Dakota | Perennial streams | 50ft. | width of the SMZ should extend beyond the 50 foot minimum to include wetlands in the stream bottom. | Retain trees necessary for bank stabilization and to provide a future source of large woody debris for the stream channel. | Non-regulatory |
| Tennessee | Intermittent and perennial streams | 25ft-145ft | Based on slope gradient. Increase width by 20ft. With slope increase by 10%. | No heavy machine operation within RMZ. Residual of 50% canopy cover. | Non-regulatory |
| Texas | Perennial and Intermittent streams, Municipal water supplies | Perennial and intermittent: 50ft., Municipal water supplies: 100-200 ft. | Recommends to adjust width beyond min. to account for slope, soils and cover type along streams. | Residual of 50 sq.ft/ac BA | Non-regulatory |
| Utah | Class I and Class II streams | Class I: 75ft. - 100ft., Class II: 35ft. - 50ft. | Based on slope gradient (<35% and >35% gradient) | no harvest 15ft buffer from streams.<br>Class I: 50sqft/ac residual basal area, 50% canopy coverage<br>Class II: 25sqft/ac residual basal area, 25% canopy coverage | Non-regulatory |
| Vermont | perennial and intermittent streams | 50ft-110ft | Based on slope gradient. Width increases by 20ft for a 10% increase in slope. | Only partial cutting can occur such that openings in the forest canopy are minimal and continuous forest cover is maintained. Not specific. | Quasi-regulatory |

| State | Waterbody | Width | Specification | Harvesting restriction | Implementation |
|---|---|---|---|---|---|
| Virginia | perennial and intermittent streams, lakes, ponds, natural springs, municipal water supplies | Warm water fisheries: 50ft., cold water fisheries: 66-125ft., municipal water supplies: 100-200ft. | Based on slope gradient | Residual up to 50% of BA or up to 50% of the forest canopy can be harvested in the SMZ | Quasi-regulatory |
| Washington | Type S and F streams (fish bearing), Type Np (perennial), Type Ns (intermittent) | Western Washington-Types S or F: 90-200ft. Eastern Washington-Types S or F: 75-130ft. (stream width <=15ft.), 100-130 ft. (stream width> 15ft.) | Based on stream type, bankfull width, and site class (I-V) | Western Washington: 50ft. No harvest zone, Eastern Washington: 30ft. No harvest zone. Harvesting guidelines within the rest of the RMZ depends on residual tree diameter, residual number of conifer trees per ac. | Regulatory |
| West Virginia | ephemeral, intermittent and perennial streams | perennial and intermittent: 100ft minimum ephemeral: 25ft minimum | minimum can be adjusted to be wider if conditions call for it | none | Regulatory |
| Wisconsin | intermittent and perennial streams | 100ft. (trout streams and streams that are >3ft. wide, 35ft (streams < 3ft. wide) | Based on stream width. Wider RMZs recommended for steep slopes, high erodible soils, long continuous slopes, etc. | Trout streams, and streams >3ft. wide: no wheeled machine operation within 15 ft of ordinary high water mark. All streams: residual of 60 sq.ft./ac with a min. harvesting interval of 10 years. Timber harvesting next to lakes and streams must be consistent with local zoning ordinances. | Quasi-regulatory |
| Wyoming | Perennial and intermittent streams | 50ft - 100ft. | Width extended to 100ft. when slope gradient >35% | Some larger trees should be retained in the SMZ, to provide shade | Non-regulatory |

**Appendix 3B**

Ridge Finder R Code

```
setwd("C:/Users/8378gunawac/Desktop/Manee Code/New York")

#install.packages("fpp2")

library(tidyverse)

library(fpp2)

library(lubridate)

#______________________________________________________________________________

# Arrange TransectID in ascending order before importing

# reading data in

All_Data = read_csv("StNet3_Elevation.csv")



# add new column with New_ID for data manupilation

All_Data = add_column(All_Data, New_ID = 1:length(All_Data[[1]]))

#add empty column to save ridge locations

All_Data = add_column(All_Data, Ridge = "NO", Slope = 0)

#______________________________________________________________________________

# Deleting the 42nd data entry for each transect

Initial_Data = All_Data

pb = txtProgressBar(min = 1, max = length(All_Data[[1]]), initial = 0, style = 3) # progress bar
```

```
for (i in 1:(length(All_Data[[1]])-1)) {
  setTxtProgressBar(pb,i)
  if (length(All_Data[which(All_Data[2] == All_Data[[i,2]]),][[1]]) == 42) {
    All_Data = All_Data[-c(i+41),]
  }
}
# error only indicates that the number of iterations don't match the data set - not an issue
length(All_Data[[1]]) %% 41 == 0  # if TRUE, the deleting process is properly done (if each transect has 41 points)
#___________________________________________________________________________________________
## Creating odd and even tables for analysis
# creating the tables to store the data
odd_data = data.frame(matrix(ncol = length(All_Data)))[-1,]
colnames(odd_data) = colnames(All_Data)
even_data = data.frame(matrix(ncol = length(All_Data)))[-1,]
colnames(even_data) = colnames(All_Data)
# data extraction
pb = txtProgressBar(min = 1, max = length(All_Data[[1]]), initial = 0, style = 3) # progress bar
for (i in 1:(length(All_Data[[1]]))) {
  setTxtProgressBar(pb,i)
```

```
  if (i %% 41 == 1) {
    odd_data = bind_rows(odd_data, All_Data[which(All_Data[2] == All_Data[[i,2]]),][1:21,])
    even_data = bind_rows(even_data, All_Data[which(All_Data[2] == All_Data[[i,2]]),][21:41,])
  }
}
#____________________________________________________________________________
# changing order of the odd data, this is the starting side of the transect
odd_data = arrange(odd_data, desc(odd_data$New_ID))
#____________________________________________________________________________
# adding slope data (the last point of the transect is 0 slope)
pb = txtProgressBar(min = 1, max = length(even_data[[1]]), initial = 0, style = 3) # progress bar
for (i in 1:(length(even_data[[1]])-1)) {
  setTxtProgressBar(pb,i)
  if (even_data[[i,2]]==even_data[[(i+1),2]]) {
    even_data[[i,7]] = (even_data[[(i+1),3]]-even_data[[i,3]])*100
  }
}
pb = txtProgressBar(min = 1, max = length(odd_data[[1]]), initial = 0, style = 3) # progress bar
for (i in 1:(length(odd_data[[1]])-1)) {
```

```
  setTxtProgressBar(pb,i)
  if (odd_data[[i,2]]==odd_data[[(i+1),2]]) {
    odd_data[[i,7]] = (odd_data[[(i+1),3]]-odd_data[[i,3]])*100
  }
}
#________________________________________________________________________________
# ODD DATA ANALYSIS
pb = txtProgressBar(min = 2, max = length(odd_data[[1]])-1, initial = 0, style = 3) # progress bar
# check for first point
if (odd_data[[1,7]] < 5) {
  odd_data[[1,6]] = "YES"
}
# Loop for odd data (loop does not analyze the first and last points of the table)
for (i in 1:(length(odd_data[[1]])-2)) {
  setTxtProgressBar(pb,i)
  # checking whether the ridge is found previously - skip next iterations
  if (sum(sum(odd_data[which(odd_data[2] == odd_data[[i+1,2]]),] == "YES")) == 1) {
    next
  }
```

```
 # If the ridge is not found throughout the transect, flag the last point
 if ((sum(sum(odd_data[which(odd_data[2] == odd_data[[i+1,2]]),] == "YES")) == 0) &
(odd_data[[i+1,2]] != odd_data[[i+2,2]])) {
  odd_data[[i+1,6]] = "YES"
 }
 # if the slope of the first point is 0, skip to next
 if ((odd_data[[i+1,2]] != odd_data[[i,2]]) & (odd_data[[i+1,7]] == 0)) {
  next
 }
 # if slope is more than 5%, skip
 else if (odd_data[[i+1,7]] > 5) {
  next
 }
 # if previous slope is 0 and this is 0, skip (flood plain)
 else if ((odd_data[[i+1,7]] <= 1) & (odd_data[[i,7]] <= 1)) {
  next
 }
 # if slope is below 5%, ridge found
 else if (odd_data[[i+1,7]] <= 5) {
  odd_data[[i+1,6]] = "YES"
```

```
 }
}
# check for last point
if (sum(sum(odd_data[which(odd_data[2] == odd_data[[nrow(odd_data),2]]),] == "YES")) == 0) {
 odd_data[[nrow(odd_data),6]] = "YES"
}
#________________________________________________________________________________
# EVEN DATA ANALYSIS
pb = txtProgressBar(min = 2, max = length(odd_data[[1]])-1, initial = 0, style = 3) # progress bar
# check for first point
if (even_data[[1,7]] < 5) {
 even_data[[1,6]] = "YES"
}
# loop for even data (loop does not analyze the first and last points of the table)
for (i in 1:(length(even_data[[1]])-2)) {
 setTxtProgressBar(pb,i)
 if (sum(sum(even_data[which(even_data[2] == even_data[[i+1,2]]),] == "YES")) == 1) {
  next
 }
```

```
  # If the ridge is not found throughout the transect, flag the last point
  if ((sum(sum(even_data[which(even_data[2] == even_data[[i+1,2]]),] == "YES")) == 0) &
(even_data[[i+1,2]] != even_data[[i+2,2]])) {
    even_data[[i+1,6]] = "YES"
  }
  # if the slope of the first point is 0, skip to next
  if ((even_data[[i+1,2]] != even_data[[i,2]]) & (even_data[[i+1,7]] == 0)) {
    next
  }
  else if (even_data[[i+1,7]] > 5) {
    next
  }
  else if ((even_data[[i+1,7]] <= 1) & (even_data[[i,7]] <= 1)) {
    next
  }
  else if (even_data[[i+1,7]] <= 5) {
    even_data[[i+1,6]] = "YES"
  }
}
```

```
# check for last point

if (sum(sum(even_data[which(even_data[2] == even_data[[nrow(even_data),2]]),] == "YES")) == 0)

{

  even_data[[nrow(even_data),6]] = "YES"

}

#____________________________________________________________________________

# combine odd and even

All_Final = bind_rows(odd_data,even_data)

# write the files to folder

write_csv(All_Final, path = "C:/Users/8378gunawac/Desktop/Manee Code/New
York/StNet3_Ridge.csv")

#____________________________________________________________________________
```

# Chapter 4 : Protecting Timberland RMZs through Carbon Markets: A Protocol for Riparian Carbon Offsets


## Abstract

Riparian Management Zone (RMZ) allocations can place a burden on landowners due to restrictions (sometimes prohibitions) on harvesting. The opportunity cost for the landowner may be minimized by shifting the primary management objective in RMZs from timber production to compensation for above-ground carbon. The primary objective was to compare long-term net revenue generating potential of RMZs under three scenarios: (I) compensation for carbon credits without harvesting; (II) partial harvesting using Best Management Practices (BMP) guidelines without carbon credits; (III) partial harvesting combined with carbon credits as per the California Compliance Offset Protocol. Basic stand data on trees of 2.5 cm and higher were collected in riparian forest plots along headwater streams within two experimental forests in the northeast US. The USFS Forest Vegetation Simulator was used to simulate growth and yield and schedule management activities over 20-year cutting cycles. Timber volumes and registry offset credits along with their market values were calculated for the respective scenarios and a Net Present Value (NPV) and Equal Annual Equivalent (EAE) analysis was performed under assumptions of constant prices and costs. The initial aboveground carbon stocks at both locations were 32 % and 140 % higher than the average value for their assessment areas. Having above average carbon stocks and basal areas between 30 – 33 $m^2$/ha, all scenarios returned positive NPVs and EAEs. The hardwood riparian forest had a higher NPV and EAE by not participating in the carbon markets and pursuing partial harvesting as per BMP guidelines (Scenario II) at lower discount rates but had higher NPVs and EAEs under carbon markets at higher discount rates (Scenario I and III). The conifer/mixed species riparian forest provided greater positive net revenue flows by participating in the carbon markets either in a no harvesting scenario or under partial harvesting as per guidelines in the Protocol (Scenarios I and III). Results indicate that a protocol for compensating landowners with

large forest holdings for riparian carbon offsets provides an opportunity to generate positive net revenues in scenarios in which state BMP guidelines may restrict harvesting in RMZs. Given the high density of ecologically critical headwater streams in the Northeast and potential RMZ restrictions, the carbon offset option provides landowners with the opportunity to remain economically viable.

Keywords: Riparian management zones, California Compliance Offset Protocol, Forest carbon offsets, Forest management, Headwater streams

## Introduction

Riparian areas are three dimensional ecotones (Gregory et al. 1991) which regulate the flow of water (Opperman et al. 2017), sediment (Ward and Jackson 2004; Lakel et al. 2010), and nutrients (Secoges et al. 2013; Witt et al. 2013) across system boundaries; contribute organic matter to aquatic ecosystems (Jackson et al. 2001; Gonçalves and Callisto 2013; Opperman et al. 2017); sequester carbon in living biomass and soil (Matzek et al. 2015; Dybala et al. 2019); and increase bank stability and reduce bank erosion (Keim and Schoenholtz 1999). Riparian areas are also considered to be biodiversity hotspots providing unique and critical habitat for fish and wildlife (Naiman et al. 2005; Jackson et al. 2007; Chizinski et al. 2010). In an attempt to conserve and ensure a continuous flow of these functions, natural resource managers create buffers, commonly referred to as riparian management zones (RMZs), around streams to minimize and mitigate potential disturbances stemming from forest management activities.

Nationwide, states follow best management practice (BMP) regulations or guidelines dictating riparian buffer widths as well as silvicultural limitations. RMZ habitat protection (terrestrial or aquatic) is largely driven by the buffer width allocation which can sometimes encompass significant land area (Young 2000; National Research Council 2002; Hawes and Smith 2005), creating a burden on landowners and forest managers due to constraints on harvesting. For example, riparian widths can extend up to 300 m along Class II streams with fish in California under state RMZ guidelines (Young 2000). The percentage of forested land area designated under RMZs can range between 6 – 12 % of the total harvest area (Kluender et al. 2000; Lakel et al. 2015; Jayasuriya et al. 2019) and can hold a substantial amount of valuable timber (stumpage), particularly when the RMZ is regulated as a no-cut zone (Lakel et al. 2015; Jayasuriya et al. 2019). Although partial harvesting is often allowed in RMZs, the long-term financial sacrifice from foregone stumpage net revenues can have a significant impact on the economic viability of forest management. A potential option that may minimize the opportunity costs for the landowner could be shifting the primary management objective in RMZs from timber to above-ground carbon offsets.

Carbon Markets – A Brief Overview

The greenhouse gas (GHG) cap-and-trade program is an emission trading program that allows for emitters to trade access allowances under their emissions cap with other emitters who have exceeded their emission cap. It is a market-based approach for controlling pollution. Launched in 2012 under the purview of the California Air Resource Board (CARB), the California cap-and-trade program allows for carbon offsets from forest management as a compliance mechanism for achieving GHG reduction goals (CA 2006). A "forest carbon offset" is a metric ton of carbon dioxide equivalent which is newly stored and can be purchased by GHG emitters to compensate for their emissions. There are three types of forest management activities that may produce carbon offsets under the California Compliance Offset Protocol, hereafter referred to as "the Protocol". Namely they are, 1) afforestation/reforestation projects, 2) avoided conversion projects and 3) improved forest management (IFM) projects – the latter being the most relevant to RMZs (CARB 2015).

IFM projects were designed to accommodate working forests in which management activities maintain or increase carbon stocks relative to baseline levels (CARB 2015). Eligible activities listed under the Protocol may include, but are not limited to: (1) increasing the overall age of the forest by increasing rotation ages; (2) increasing forest productivity by thinning diseased and suppressed trees; (3) managing competing brush and short-lived forest species; (4) increasing the stocking of trees in understocked areas; and/or (5) maintaining stocks at a high level (CARB 2015). These eligible activities may not attract many standard timber focused silvicultural treatments that are practiced regularly within forests of the US (Ruseva et al. 2017) as rules and guidelines under the Protocol may discourage removals of carbon stocks exceeding certain limits bound by individual projects. Consequently, forest managers may find it difficult to financially remain in carbon projects due to constraints dictated by carbon stocks of "common practice" within their assessment area (Kerchner and Keeton 2015). The Protocol defines common practice as "the average carbon stocks (metric tons) of the above-ground portion of standing live trees from within the forest project's assessment area, derived from FIA (Forest Inventory Analysis) plots on all private lands within the defined assessment area" (CARB 2015). Projects entering the carbon markets are rewarded for the amount of carbon stocks they have above the common practice value and this is where the majority of the net revenue is generated from these projects (Kerchner and Keeton 2015). Landowners can choose to earn revenue for the newly stored carbon within the boundary of their management unit. However, this newly stored carbon, regardless of being credited for value, must be above the levels of carbon stored from the 'business-as-usual' scenario for that management unit, thereby ensuring 'additionality' requirements of carbon markets (Ruseva et al. 2017). For carbon projects to remain financially viable throughout the long-term commitment period (a minimum of 125 years (25 year crediting period + 100 year monitoring period)), a minimum land area of 2,500 ac (1,011 ha) (White 2015) is recommended by project developers. This, however, depends on the amount of aboveground carbon stocks available and the proposed management activities within the project boundaries. This study proposes that these projects be exclusively applied to RMZs within larger forested tracts (> 4,000 ha). This allows standard timber focused silvicultural prescriptions to take place in the remaining larger forested area while reserving RMZs for preserving trees or restricted harvesting due to constraints from state BMP regulations or by the Protocol, thus ultimately resulting in a higher amount of aboveground carbon storage.

Since many RMZ state guidelines or regulations require a specific residual basal area threshold or limit the percent canopy removal (Jayasuriya 2016), there is some symmetry with the Protocol. Also, given that riparian areas are often richer in carbon pools when compared to upland forests (Sutfin et al. 2016; Matzek et al. 2018), RMZs could store significantly higher carbon stocks than "common practice" values, resulting in potentially higher financial returns for certain projects when compared to upland forests. Given this context, can participation in the carbon markets incentivize landowners to protect RMZs and all their associated benefits over the long-term more effectively without foregoing the opportunity to implement forest management? This study sought to address this question by combining a Net Present Value (NPV) and Equal Annual Equivalent (EAE) analysis for two RMZ projects within the Northeastern US, specifically a hardwood dominated riparian forest in the Adirondacks of New York State (NYS) and a mixed-wood riparian forest in the White Mountains of New Hampshire (NH). The primary objective was to compare long-term net revenue potential under three mutually exclusive (and repeatable) scenarios using three discount rates:

- Scenario I: Net revenue from carbon markets with no harvesting within RMZs using the California compliance offset protocol for improved forest management (IFM) (carbon markets without harvests)
- Scenario II: Stumpage revenue from harvesting within RMZs under state BMP guidelines/regulations (harvests without carbon markets),

- Scenario III: Net revenue from carbon markets coupled with stumpage harvesting revenues within RMZs using the California compliance offset protocol for improved forest management (IFM) (harvests with carbon markets).

The "additionality" criterion of the Protocol is fulfilled by both the 'no harvest' option (Scenario I) and the reduced harvesting in compliance with the carbon markets option (Scenario III) compared to "business as usual" (partial harvesting based on BMP guidelines). Consequently, carbon stocks will increase under Scenarios I and III when compared to Scenario II.

**Methods**

Study area

Field sampling was carried out at two experimental forests: Huntington Wildlife Forest (HWF) in NYS and Hubbard Brook Experimental Forest (HBEF) in NH (Figure 4.1). HWF is a 6,000 ha. experimental forest located in Essex County, in the Town of Newcomb, and lies near the geographic center of the Adirondack Park in NYS. Ranging in elevation from 457 to 823 m, HWF has a mean annual precipitation of 102 cm. The month of January averages about -9.4 °C while the month of July averages about 18.5 °C (NOAA 2018). The forest cover is dominated by northern hardwoods (72 %), followed by mixed hardwood-conifer (18 %), and spruce-fir (10 %) (SUNY-ESF n.d.). HBEF is a 3,138 ha. long-term experimental forest located in Grafton County, Towns of Woodstock and Thornton, within the White Mountain National Forest in central NH. Ranging in elevation from 222 to 1,015 m, HBEF experiences an annual precipitation of about 140 cm. The month of January averages about -6.3 °C and the average July temperature is 18.5 °C (NOAA 2018). Similar to HWF, the forest cover is primarily northern hardwoods (85%), with the balance in spruce-fir (15 %) (Adams et al. 2004).

Sampling was carried out across 9 headwater streams at HWF and 11 headwater streams at HBEF during the months of July and August in 2017. In this analysis, a 30.48 m (100 ft.) fixed-width buffer was allocated along these streams which were considered to be separate riparian forest stands.

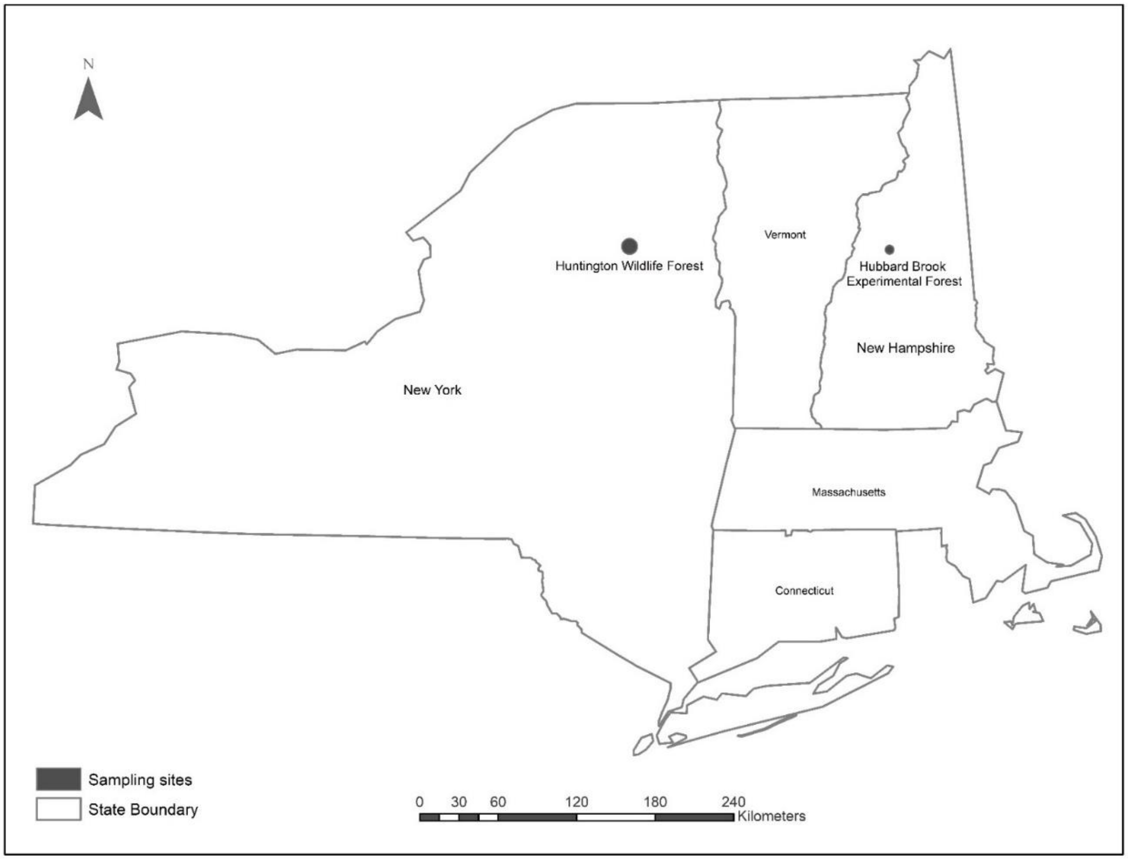



**Figure 4.1: Map of study sites of Huntington Wildlife Forest located in New York State and Hubbard Brook Experimental Forest located in New Hampshire.**

Sampling methods

Fixed radius overstory plots with a 7.32 m radius (1/24th ac. or 24 ft. radius) and understory plots with a 2.07 m radius (1/300th ac. or 6.8 ft. radius) were used to collect primary data on riparian forest stands. Overstory plots were located 12.80 m (42 ft.) away from the edge of the stream, perpendicular to stream flow (Jayasuriya et al. 2019) and understory plots were located at plot centers of every overstory plot. The number of overstory plots completed on each stream segment was based on meeting a confidence interval of ≤ 20 % margin of error around the mean basal area at $\alpha = 0.05$ (Munsell et al. 2008). Plots were placed on either side of the stream in either an alternated or opposite configuration depending on the length of the reach being sampled. Thus, distance between two plots on the same side of the stream was either one or two chains apart (1 chain = 20.12 m = 66 ft.) based on the plot configuration along the streams.

I sampled live trees with a diameter-at-breast-height (dbh) of 12.7 cm (5 in.) and greater within overstory plots and live trees between ≥ 2.54 – 12.7 cm (1 – 5 in.) within understory plots. Within overstory plots I collected information on species, dbh, timber quality (acceptable/unacceptable growing stock), product type (sawtimber/pulpwood) and number of merchantable logs. One canopy tree in each plot was measured for total height. Within the understory plots information on species and dbh was recorded.

To complete the analysis on carbon accounting, the Protocol required information on both live and dead trees within plots. As I only collected stand data on live trees, standing dead trees within plots were simulated using FIA data in their respective regions. Twenty six FIA overstory plots (2016 inventory) around HWF and 34 overstory plots (2016 inventory) around HBEF were used to estimate standing dead trees per acre. This information was randomly distributed into the sampled plots using R (https://www.rstudio.com/).

Data Management

Inventory data of overstory and understory plots were entered in a Microsoft Access Database provided by the US Forest Service Forest Vegetation Simulator (https://www.fs.fed.us/fvs/software/complete.php). Each sampled stream was considered as a separate stand and data on stands were entered into the FVS_StandInit form in the database. All stands were recorded under the Northeastern variant with location codes 920 (Green Mountain – Finger Lakes) and 922 (White Mountains, NH) for HWF and HBEF, respectively. Information on aspect, slope, and elevation for each stand was also entered. Next, tree level data was entered into the FVS_TreeInit form in the Access database. In this form, a Tree Value Code (TVC) of 1 was assigned to all Acceptable Growing Stock (AGS) trees that were marked for sawlog and pulp. This allows FVS to calculate sawlog volume and merchantable volume. A TVC of 2, along with Damage code 27 and Severity code 99, was assigned to all trees marked as Unacceptable Growing Stock (UGS) to avoid calculating sawlog and merchantable volume. A TVC 3, along with Damage code 27 and Severity code 99, was assigned to all cull trees.

Simulation

USFS Forest Vegetation Simulator (FVS) was used to simulate growth and yield for each stand for 100 years. FVS is an individual-tree, distance-independent, growth and yield model that is also an approved growth and yield model by the California ARB. Under management actions, a regeneration subroutine for background regeneration was initiated as the Northeastern variant is a partial establishment model. Background regeneration was scheduled every 20 years for species observed in the understory plots. Based on inventory, a total of 2020 trees/ha (818 trees/ac.) (red spruce - 70.8 %, American beech - 16.2 %, balsam fir - 5.8 %, striped maple - 5.4 %, yellow birch - 1.2 %, paper birch - 0.4%, and sugar maple - 0.4%) at HBEF and a total of 1008 trees/ha (408 stems/ac.) (American beech - 62.1 %, red spruce - 19.3%, sugar maple – 15 %, balsam fir - 2.1 %, yellow birch - 0.7 %, and striped maple - 0.7 %) at HWF were regenerated naturally. In the regeneration model, the 5-year average height of a tree was assigned as 3 m (10 ft.) for hardwood species and conifer species. All species were assigned an 80% survival rate (Kerchner and Keeton 2015).

*Cutting cycles within RMZs*

A cutting cycle of 20 years for a period of 100 years was initiated within the RMZs beginning in 2018 (cutting cycles were at 2018, 2038, 2058, 2078, 2098 and 2118). Silvicultural treatments of thinning from above and below were performed alternatively depending on stand parameters and stocking guidelines (Leak et al. 1987). The objective over the set of treatments was to maintain continuous forest cover and to convert even-aged stands to uneven-aged riparian forests (Buongiorno 2001). Treatments were limited to tending because the study was operating within the constraints of harvesting restrictions within RMZs. Therefore, not more than 40% of basal area (BA), and 40% of canopy cover was removed at any given entry.

NED-3 Forest Ecosystem Decision Support Software (Twery and Thomasma 2018) was used in conjunction with FVS to run silvicultural prescriptions on the stands. Both programs were utilized simultaneously for two reasons. Firstly, FVS did not allow us to enter information on the number of logs counted on AGS trees and thus resulted in an overestimation in sawlog volumes in its simulations. Secondly, FVS failed to produce detailed separate information on overstory statistics such as relative stand density, number of stems per acre, and Quadratic Mean Diameter (QMD) that were essential to determine harvesting parameters. However, the growth rates (merchantable volume accretion) in FVS were comparable to the growth rates recorded for the region while NED-3 was not (except basal area accretion rates). Therefore, I used the accretion rates from the FVS simulations to project sawlog and merchantable volume accretion on NED-3 calculated sawlog volumes for the 2017 inventory. Merchantable volumes were used to estimate the pulp and firewood volumes after deductions on sawlog volumes.

This study acknowledges that this cascading stand manipulation and regeneration could introduce errors into the growth and yield modeling. However, NED and FVS do not provide any statistics to test or correct for the potential error propagation. Given the deterministic nature of these models, using them in this manner is common practice. However, to be conservative in the growth and yield modeling, five 20-year cutting cycles were used.

*Biomass and carbon projection*

Using FVS Fire and Fuel Extension (FFE), I estimated and simulated carbon pools of aboveground live, belowground live and dead, standing dead, forest dead down wood, forest floor, and shrubs and herbs, for 100 years. The CARB requires that carbon stocks be calculated using species specific equations. However, Kerchner and Keeton (2015) estimated that carbon stocks calculated using CARB's method were within a range of ± 10 % of total carbon from FVS FFE results. Therefore, I used the FVS FFE biomass prediction model to build stand carbon for the financial analysis. Carbon within harvested wood products were calculated as per the guidelines in the Protocol.

Data analysis

*Quantifying carbon stocks for offset credits*

Net GHG Reductions and Removals were quantified by using the following equation as per the Protocol:

$$QR_y = [(\Delta AC_{onsite} - \Delta BC_{onsite}) + \left(AC_{wp,y} - BC_{wp,y}\right) * 0.80 + SE_y] \quad [1]$$

As described in the Protocol, $QR_y$ is the quantified GHG emission reductions and GHG removal enhancements for reporting period y, $\Delta AC_{onsite}$ is the change in actual onsite carbon since the last reporting period, $\Delta BC_{onsite}$ is the change in baseline onsite carbon since the last reporting period, $AC_{wp,y}$ is the actual carbon in harvested wood products produced in reporting period $y$ that is projected to remain stored for at least 100 years, and $BC_{wp,y}$ is the average annual baseline carbon in harvested wood products that would have remained stored for at least 100 years (CARB 2015). The $SE_y$ represents the secondary effect GHG emissions caused by the project activity in the reporting period $y$ and 0.80 is the market response to changes in wood product production (CARB 2015). Common practice above ground carbon stocks were referenced from the Adirondacks and Green Mountains – Northern hardwood assessment area for HWF and White Mountains – Northeast spruce-fir assessment area for HBEF. Finally, annual Registry Offset Credits (ROCs) issued to the landowner were accounted for after deductions of 17.6 % (as per Appendix D of CARB (2015)) of Net GHG Reductions and Removals ($QR_y$) to buffer pool. ROCs were calculated for the first reporting period (first year of project). Net GHG Reductions and Removals ($QR_y$) were calculated for the next 24 consecutive years.

*Financial analysis*

Scenarios I (carbon markets without harvests) and III (harvests with carbon markets) have a planning horizon of 125 years while Scenario II (harvests without carbon markets) has a planning horizon of 100 years. As these are mutually exclusive and repeatable projects with different planning horizons, the standard approach is to combine an NPV and EAE analysis to account for these differences (Newman et al. 2014; Fehr 2017; Brigham and Huston 2019). NPV is calculated initially, then used to calculate the EAE. As per the literature cited, the EAE is used to compare projects financially. Scenario I required calculating the NPV and EAE of the carbon project without any harvesting, Scenario II required calculating stumpage net revenues derived from silvicultural treatments in accordance to RMZ guidelines. A Faustmann (1849) approach was used to calculate the Land Expectation Value (LEV) instead of a NPV due to the assumption that this land was allocated for perpetual timber production.  The LEV was used to calculate the EAE for Scenario II. Scenario III required both the NPV and EAE of the carbon project in conjunction with the stumpage net revenues generated from silvicultural treatments in alignment with the Protocol.

Stumpage prices –

Initially, for Scenario II and III I estimated real stumpage price increases by species using the standard approach described by Sendak (1991, 1994); Howard and Chase (1995); Irland et al. (2001); Wagner and Sendak (2005); Smith et al. (2012); Sendak and McEvoy (2013). Based on historic real stumpage price data, the estimated annual real rates of increase for NYS ranged from -1.941 % for spruce species to 4.185% for sugar maple and for NH, 1.086% for spruce species to 3.345 % for sugar maple. The real rates projected stumpage prices for sugar maple at $ 21,200 per MBF in NYS and up to $ 8,400 per MBF in NH at the end of 100 years (2118). I concluded that these projections would lead to unrealistic value estimates for the NPV. Therefore, I decided to proceed with the analysis using the assumption of constant price for the life of the project. The winter 2018 stumpage price report for NYS and the price report for the period October 2018 – March 2019 for NH was used (Table 4.1).

Carbon credit price and cost projection –

The price of a forest carbon offset (carbon credit) in the California compliance offset market was \$14.05 /$MTCO_2e$ in September 2018. I used this price to calculate the value of the ROC issued for each riparian project for the first reporting period.

The cost structure shown in Table 4.2 was used for the financial analysis. I solicited and received the cost structure from a developer of forest carbon offsets. It is important to note that the costs and deal structures to develop and service offset projects are highly variable and depend on many different factors that continue to change and evolve.

Calculating NPV and EAE –

The cash flow of each project for the three separate scenarios (Figure 4.2) were set up for a project life of a 125 years for Scenarios I and II, and a 100 years for Scenario II. For all three scenarios, cash flows were set up under the assumption of prices remaining constant throughout the project lifetimes. Finally, using real discount rates of 4, 6, and 8 %, a sensitivity analysis for each scenario to assess the financially viability of the projects was completed. Since stumpage prices are considered to be a residual price, the cumulative present value for all stumpage projects were calculated.

The financial analysis did not consider other costs associated with land such as property taxes. It was assumed that these costs were being paid by (private) landowners regardless of implementing either one of these projects and thus would be considered as a sunk cost. This is consistent with a similar study done by Kerchner and Keeton (2015). Management costs would, however, have to be considered as a percentage (based on the percent distribution of RMZs) of the larger timberland management area.

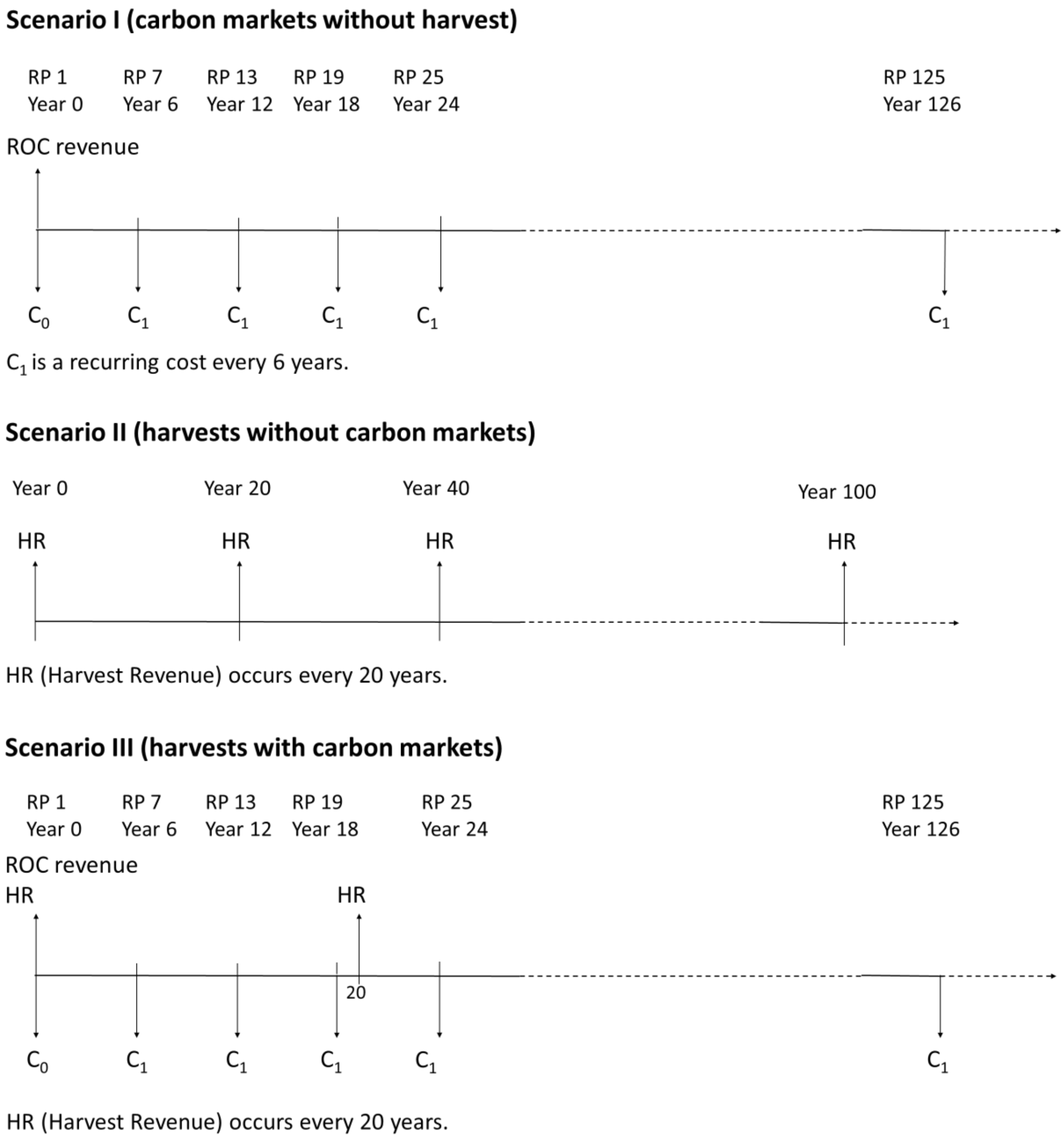
**Table 4.1: Stumpage prices for species observed within New York State (NYS) and New Hampshire (NH). Prices for NYS are based on the winter 2018 price report and prices for NH are based on the price report for the period 0ctober 2018 - March 2019.**

| | Stumpage price (per MBF) | |
|---|---|---|
| **Species** | **NYS** | **NH** |
| red maple (*Acer rubrum* L.) | $175 | $130 |
| sugar maple (*Acer saccharum* Marsh.) | $350 | $342 |
| black cherry (*Prunus serotina* Ehrh.) | $350 | NA |
| white ash (*Fraxinus Americana* L.) | $200 | $160 |
| American beech (*Fagus grandifolia* Ehrh.) | $45 | $55 |
| paper birch (*Betula papyrifera* Marsh.) | $90 | $120 |
| yellow birch (*Betula alleghaniensis* Britton) | $235 | $187 |
| spruce (*Picea rubens* Sarg.) | $90 | $140 |
| pulp | $10 | $5 |
| firewood | $10 | $5 |

NA = not available

**Table 4.2: Cost structure for the carbon project (2018).**

| Item | Cost | Frequency of cost |
|---|---|---|
| Inventory (initial reporting period) | $ 20/ac | Once |
| Inventory in subsequent years (90% of initial) | $ 18/ac | Every 6 years |
| GIS | $ 1/ac | Once |
| Full verification | $60,000 | Every 6 years |
| Offset project registry fee | $500 | Annual |
| Registration | $0.21/tonne | When ROCs are issued |
| Developer fees | 20% of offsets | When ROCs are issued |




$C_0$ includes costs of inventory, GIS, full verification, registration and developer fees.
*$C_1$ includes costs of inventory and full verification.*
*RP represents the reporting period in carbon markets.*

**Figure 4.2: Cash flow diagrams of Scenario I (carbon markets without harvests), Scenario II (harvests without carbon markets), and Scenario III (harvests with carbon markets).**

## Results

<u>Species Composition within Riparian Areas</u>

The riparian areas at HWF were characterized by a BA of 30.3 $m^2$/ha (132 $ft^2$/ac.), relative density of 89 % and QMD of 15.7 cm (6.2 in.). Overstory species consisted mainly of American beech *(Fagus grandifolia* Ehrh.*)* (27 % of BA)*, sugar maple (Acer saccharum* Marsh.*)* (27 % of BA)*,* and yellow birch *(Betula alleghaniensis* Britton*)* (20% of BA). The balance of the overstory BA was composed of eastern hemlock *(Tsuga canadensis* (L.) Carrière*)*, red maple *(Acer rubrum* L.*)*, red spruce *(Picea rubens* Sarg.*)*, white ash *(Fraxinus Americana* L.*)*, black cherry *(Prunus serotina* Ehrh.*)*, striped maple *(Acer pensylvanicum* L.*)*, and balsam fir *(Abies balsamea* (L.) Mill.*)*. The understory was dominated by American beech, sugar maple, and red spruce. At the HBEF site, the forest descriptive statistics were as follows: BA of 32.8 $m^2$/ha (143 $ft^2$/ac.), relative density of 76 % and a QMD of 12.9 cm (5.1 in.). Over 60 % of the overstory BA composition was comprised of yellow birch (31 %) and red spruce (30 %). Other species in order of abundance included sugar maple, balsam fir, American beech, paper birch *(Betula papyrifera* Marsh.*),* red maple, eastern hemlock, and striped maple. The understory was dominated by red spruce.

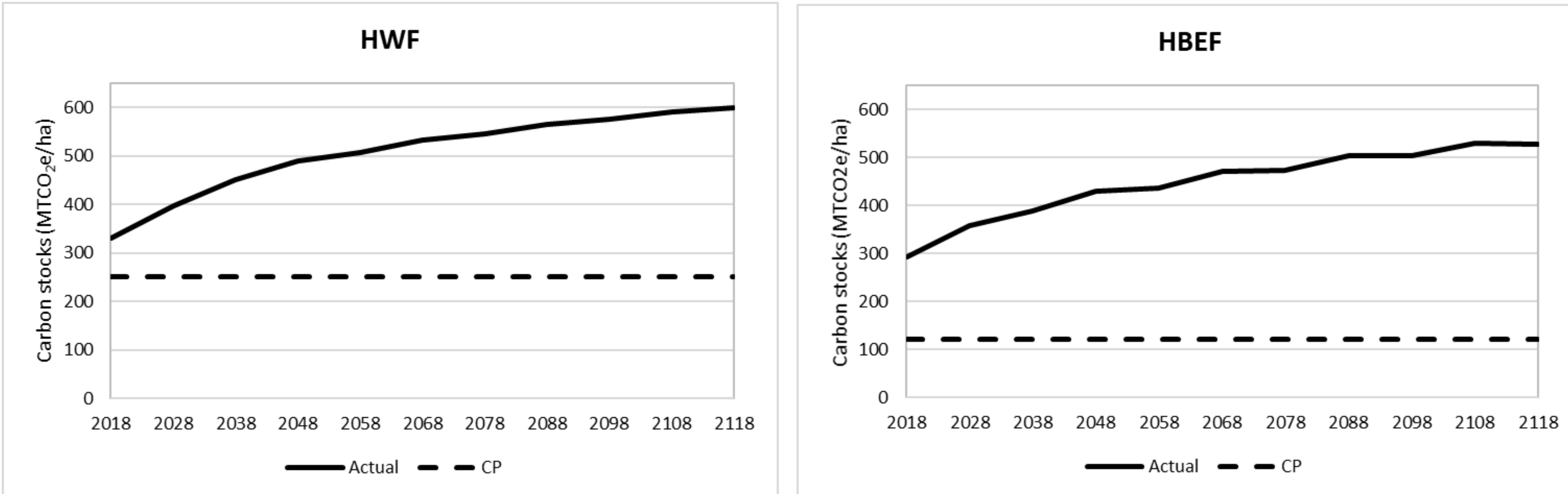
<u>Scenario I (Carbon markets without harvesting)</u>

*Carbon Stocks*

The common practice aboveground live carbon stocks for the HWF assessment area is 252 $MTCO_2$/ha (102 $MTCO_2$/ac.) (CARB 2015). As this is a no harvest scenario, the initial carbon stocks during the time of inventory will be considered as the actual carbon stocks for each site. Accordingly, the actual aboveground live carbon stocks at HWF were established at 331 $MTCO_2$/ha (134 $MTCO_2$/ac.) which is 32 % above the common practice value (Figure 4.3 (a)). At HBEF, the actual aboveground live carbon stocks of 292 $MTCO_2$/ha (118 $MTCO_2$/ac.) were 140 % above the common practice value of 121 $MTCO_2$/ha (49 $MTCO_2$/ac.) (COP 2015) for its respective assessment area (Figure 4.3 (b)).

*Registry Offset Credits*

Per the California Compliance Offset Protocol, at HWF, 69,330 Registry Offset Credits (ROCs) will be issued in 2018 (first reporting period) with a value of $974,091 while at HBEF, 76,714 ROCs will be issued with a value of $1,077,827. According to the cost structure (Table 2), costs during the first reporting period are estimated at $304,528 for HWF and $ 313,175 for HBEF. When prices remain constant throughout the project duration, the NPV of the carbon project at HWF will range from $321,484 (4 % discount rate) to $510,478 (8 % discount rate), i.e. $481 /ha (4 %) to $764 /ha (8 %) (Table 3). At HBEF, when prices remain constant, the NPV will range from $460,355 (4 %) to $625,502 (8 %), i.e. $1,138/ha (4 %) to $1,546 /ha (8 %), which is a higher price range when compared to HWF (HWF RMZ area = 668 ha. and HBEF RMZ area = 405 ha). The NPV increases as the discount rate increases due to the cash inflows occurring at the beginning of the project, but cash outflows continue for a 125 years or the lifetime of the project. Here the cash inflow, which is the very large revenue from the sale of ROCs at year 0 is not impacted by the increasing discount rates while the cash out flows, which are the costs, are impacted heavily (Figure 4.2).




(a) (b)

**Figure 4.3: Aboveground live carbon stocks in Huntingtin Wildlife Forest (HWF) (a) and Hubbard Brook Experimental Forest (HBEF) (b).The Actual represents the aboveground live carbon stocks at each sampling site, and CP represents the common practice aboveground live carbon stocks for the corresponding assessment areas.**

**Table 4.3: Scenario I (carbon market without harvests) Net Present Value of carbon project in Huntington Wildlife Forest (HWF)and Hubbard Brook Experimental Forest (HBEF) over 125 years for real discount rates of 4, 6, and 8%.**

| | Discount Rate | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Location** | **4 %** | | | **6 %** | | | **8 %** | | |
| | **NPV ($)** | **NPV ($/ac)** | **NPV ($/ha)** | **NPV ($)** | **NPV ($/ac)** | **NPV ($/ha)** | **NPV ($)** | **NPV ($/ac)** | **NPV ($/ha)** |
| **HWF** | $ 321,484 | $ 195 | $ 481 | $ 447,047 | $ 271 | $ 669 | $ 510,480 | $ 309 | $ 764 |
| **HBEF** | $ 460,355 | $ 460 | $ 1,138 | $ 570,073 | $ 570 | $ 1,409 | $ 625,502 | $ 626 | $ 1,546 |

Scenario II (harvest without carbon markets)

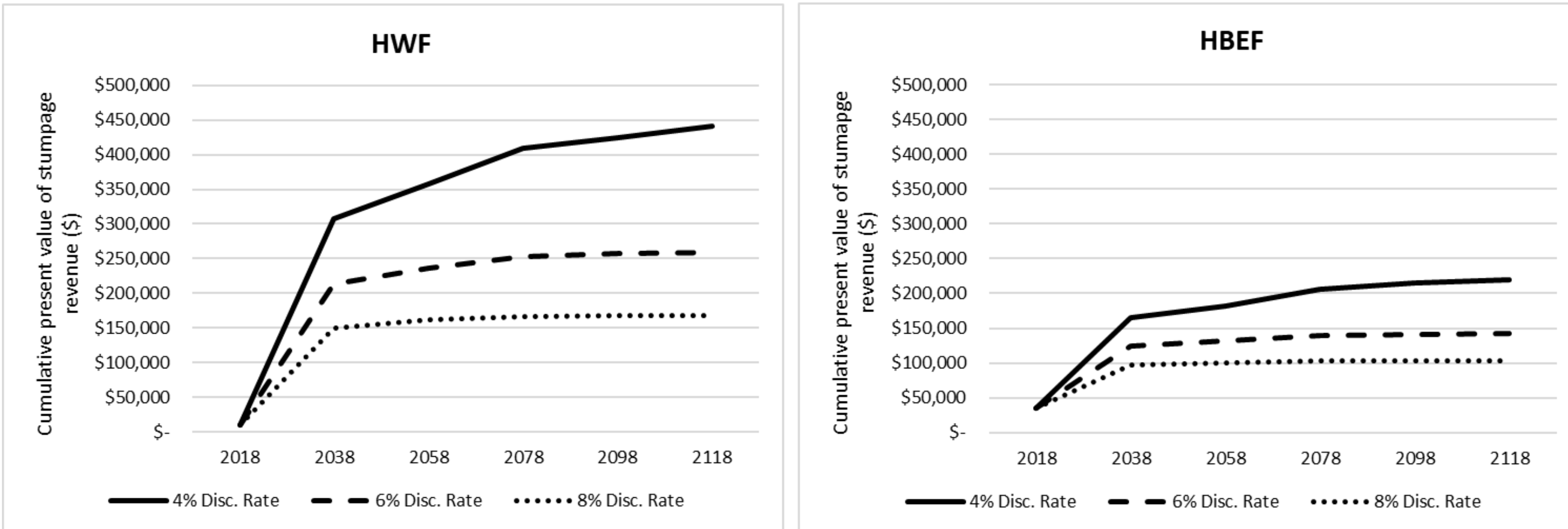
*Harvested Volumes and Stumpage Revenues*

The initial harvest (2018) on both sites consisted of a thinning from below, generating little sawtimber (Table 4.4). The next entry in 2038 is a crown thinning which will generate 9 $m^3$/ha (1.5 MBF/ac.) of mostly sugar maple, yellow birch, and white ash at HWF and 10 $m^3$/ha (1.8 MBF/ac.) of red spruce, and yellow birch at HBEF. When stumpage prices remain constant, HWF will average $ 786 /ha ($318 /ac.) through the 20-year cutting cycles (average calculated for harvests from 2038 - 2118) (Table 4.4). At HBEF, stumpage values under constant prices will average $436 /ha ($205 /ac.) over the cycles (average calculated for harvests from 2038 - 2118).

The LEV for HWF will range from $167,580 (8 % discount rate) to $ 448,522 (4 % discount rate), i.e. $251 /ha (8 %) to $672 /ha (4 %), while at HBEF, the range is $103,245 (8 %) to $222,474 (4 %), i.e. $255 /ha (8 %) to $550 /ha (4 %) (Figure 4.4a & 4.4b).

**Table 4.4: Stumpage volumes and prices for trees harvested within Riparian Management Zones in Huntington Wildlife Forest (HWF) and Hubbard Brook Experimental Forest (HBEF) for Scenario II (harvests without carbon markets).**

| Study site | Year | Sawlog volume (bd.ft./ac.) | Sawlog volume ($m^3$/ha) | Wood products net revenue ($/ac.) | Wood products net revenue ($/ha) | Total Wood products net revenue ($) |
|---|---|---|---|---|---|---|
| HWF | 2018 | 0 | 0.00 | $ 6 | $ 15 | $ 9,964 |
| | 2038 | 1471 | 8.58 | $ 395 | $ 977 | $ 652,110 |
| | 2058 | 514 | 3.00 | $ 147 | $ 363 | $ 242,244 |
| | 2078 | 1469 | 8.57 | $ 331 | $ 817 | $ 545,785 |
| | 2098 | 806 | 4.70 | $ 212 | $ 525 | $ 350,289 |
| | 2118 | 1848 | 10.78 | $ 505 | $ 1,248 | $ 833,603 |
| | | | | | | |
| HBEF | 2018 | 99 | 0.58 | $ 36 | $ 88 | $ 35,745 |
| | 2038 | 1778 | 10.37 | $ 283 | $ 699 | $ 282,890 |
| | 2058 | 308 | 1.80 | $ 80 | $ 197 | $ 79,674 |
| | 2078 | 1571 | 9.16 | $ 263 | $ 650 | $ 263,038 |
| | 2098 | 1201 | 7.00 | $ 201 | $ 497 | $ 201,123 |
| | 2118 | 1378 | 8.04 | $ 197 | $ 486 | $ 196,700 |




(a) (b)

**Figure 4.4: Graphs showing the LEV of stumpage revenue for Scenario II (harvests without carbon markets), for Huntington Wildlife Forest (HWF) (a) and Hubbard Brook Experimental Forest (HBEF) (b).The three lines in each graph represents the cumulation of revenue at 4, 6, and 8 % real discount rates.**

Scenario III (harvests with carbon markets)

*Harvested Volumes and Stumpage Revenues*

Similar to Scenario II, due to the silvicultural prescription of thinning from below, both HWF and HBEF will generate minimal sawtimber in 2018 (Table 4.5). The next entry in 2038 results in more sawtimber, with 4 $m^3$/ha (0.7 MBF/ac.) from HWF and less than 3 $m^3$/ha (0.5 MBF/ac.) from HBEF. These volumes are significantly lower than what was observed in Scenario II. HWF will average $281 /ha ($114 /ac.) through the 20-year cutting cycles (average calculated for harvests from 2038 - 2118) (Table 4.5). At HBEF, stumpage values will average $185 /ha ($75 /ac.) over the cycles.

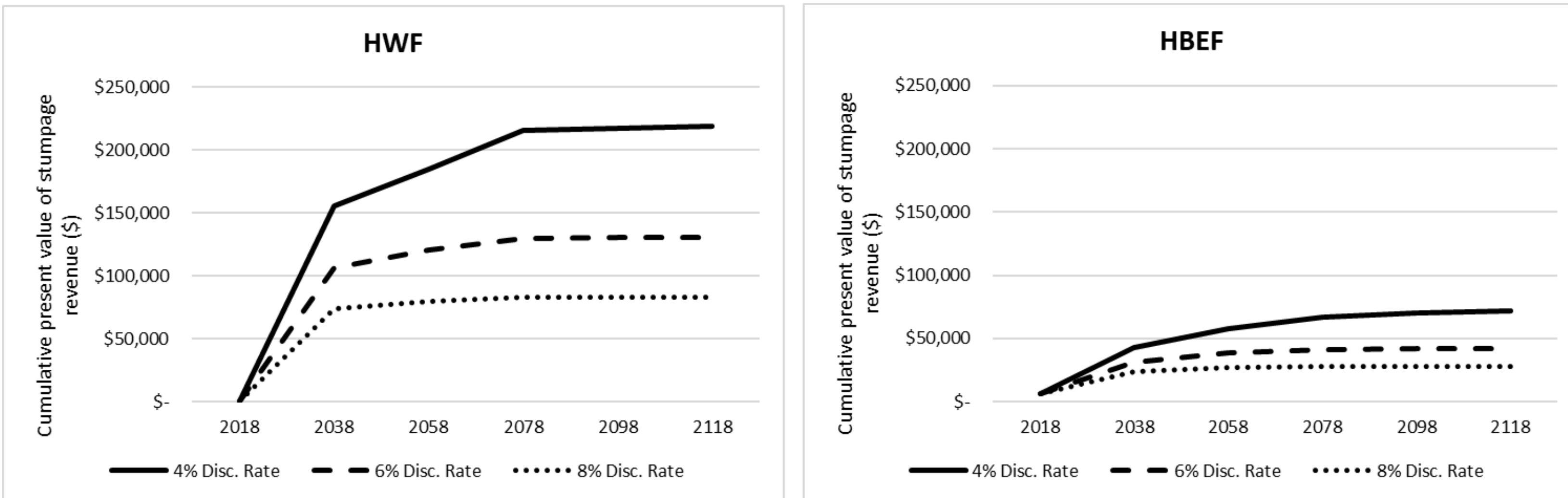
The NPV for HWF will range from $83,328 (8% discount rate) to $221,487 (4 % discount rate), i.e. $125 /ha (8 %) to $332 /ha (4 %), while at HBEF, the range is $27,922 (8 %) to $ 72,067(4 %), i.e. $69 /ha (8%) to $178 /ha (4 %) (Figure 4.5).

*Carbon stocks*

The initial (pre-harvest) aboveground live carbon stocks at HWF were established at 331 $MTCO_2$/ha (134 $MTCO_2$/ac.). After the first thinning, the actual aboveground live carbon stocks will decline to 300 $MTCO_2$/ha (121 $MTCO_2$/ac.) which is 19 % above the common practice value of 252 $MTCO_2$/ha (102 $MTCO_2$/ac.) (COP 2015) (Figure 4.6a). At HBEF, the initial (pre-harvest) aboveground live carbon stock of 293 $MTCO_2$/ha (118 $MTCO_2$/ac.) declined to 263 $MTCO_2$/ha (106 $MTCO_2$/ac.) after the first thinning. The actual aboveground live carbon stocks at HBEF were recorded to be 116 % higher than the common practice value of 121 $MTCO_2$/ha (49 $MTCO_2$/ac.) (COP 2015) (Figure 4.6b).

**Table 4.5: Stumpage volumes and prices for trees harvested within Riparian Management Zones in Huntington Wildlife Forest (HWF) and Hubbard Brook Experimental Forest (HBEF) for Scenario III (harvests with carbon markets).**

| Study site | Year | Sawlog volume (bd.ft./ac) | Sawlog volume ($m^3$/ha) | Wood products net revenue ($/ac) | Wood products net revenue ($/ha) | Total wood products net revenue ($) |
|---|---|---|---|---|---|---|
| HWF | 2018 | 0 | 0.00 | $ 0 | $ 1 | $ 605 |
| | 2038 | 708 | 4.13 | $ 206 | $ 509 | $ 340,005 |
| | 2058 | 267 | 1.56 | $ 85 | $ 209 | $ 139,524 |
| | 2078 | 700 | 4.08 | $ 197 | $ 487 | $ 325,028 |
| | 2098 | 48 | 0.28 | $ 21 | $ 52 | $ 34,755 |
| | 2118 | 308 | 1.79 | $ 59 | $ 147 | $ 98,080 |
| | | | | | | |
| HBEF | 2018 | 5 | 0.03 | $ 6 | $ 15 | $ 6,149 |
| | 2038 | 458 | 2.67 | $ 81 | $ 200 | $ 80,987 |
| | 2058 | 427 | 2.49 | $ 71 | $ 174 | $ 70,549 |
| | 2078 | 511 | 2.98 | $ 96 | $ 237 | $ 96,071 |
| | 2098 | 570 | 3.32 | $ 82 | $ 202 | $ 81,590 |
| | 2118 | 232 | 1.35 | $ 46 | $ 114 | $ 46,022 |




(a) (b)

**Figure 4.5: Graphs showing the cumulative present value (PV) of stumpage revenue for Scenario III (harvests with carbon markets), for Huntington Wildlife Forest (HWF) (a) and Hubbard Brook Experimental Forest (HBEF) (b).The three lines in each graph represents the cumulation of revenue at 4, 6, and 8 % real discount rates.**

*Registry Offset Credits*

At HWF, 50,303 ROCs will be issued in 2018 (first reporting period) with a value of $706,757 while at HBEF, 66,218 ROCs will be issued with a value of $930,356. Costs during the first reporting period are estimated at $ 247,065 for HWF and $281,477 for HBEF (Table 4.1). The NPV of the carbon project at HWF will range from $ 111,612 (4% discount rate) to $300,608 (8 % discount rate), i.e. $167 /ha (4 %) to $450 /ha (8 %) (Table 4.6). At HBEF, the NPV will range between $344,583 (4 % discount rate) to $509,731 (8 % discount rate), i.e. $851 /ha (4 %) to $1,260 /ha (8 %). Net present values for carbon projects are significantly higher at HBEF than HWF (Table 4.6). This result is again due to cash inflows occurring at the beginning of the project, but cash outflows continue for a 125 years or the lifetime of the project. Similar to Scenario I, the cash inflow in the carbon project, which is the very large revenue from the sale of ROCs at year 0 is not impacted by the increasing discount rates while the cash out flows, which are the costs in the carbon markets, are impacted heavily (Figure 4.2).

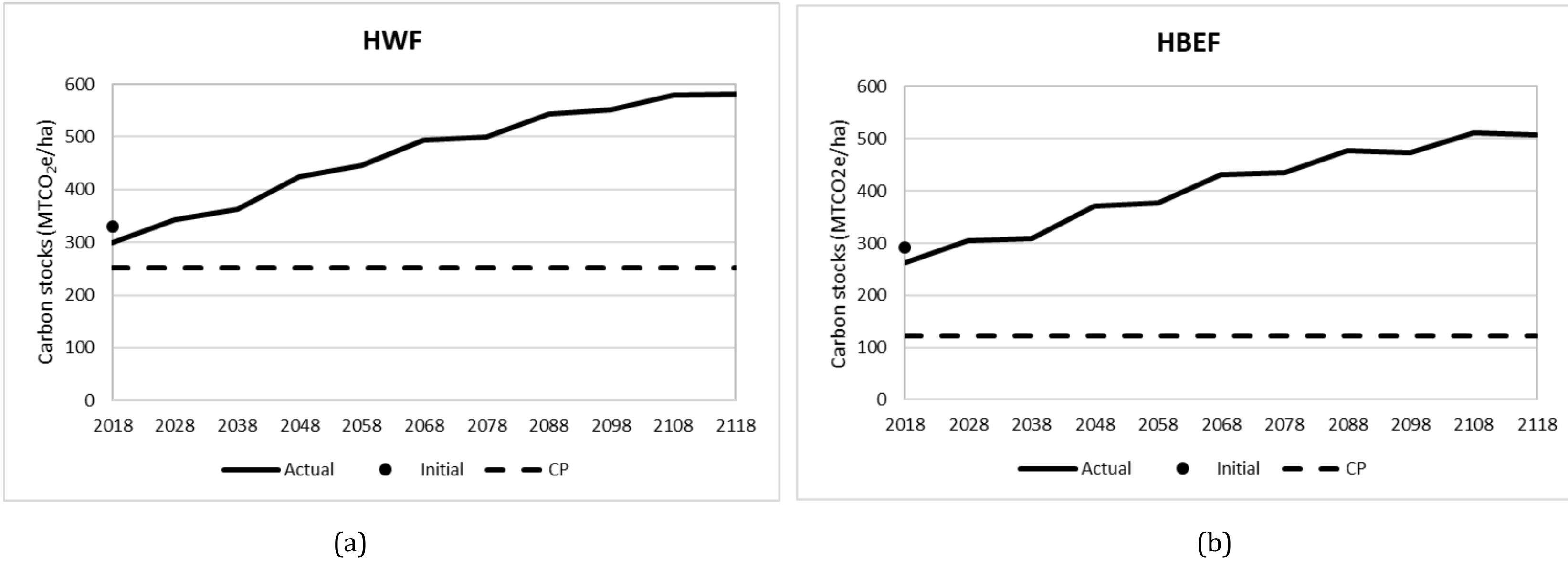



**Figure 4.6: Aboveground live carbon stocks in Huntington Wildlife Forest (HWF) (a) and Hubbard Brook Experimental Forest (HBEF) (b).The Actual represent the aboveground live carbon stocks before harvesting, the Initial represents the aboveground live carbon stocks after harvesting, and CP represents the common practice aboveground live carbon stocks for the assessment areas**

**Table 4.6: Scenario III (harvests with carbon markets) Net Present Value (NPV) of carbon project in Huntington Wildlife Forest (HWF) and Hubbard Brook Experimental Forest (HBEF) over 125 years for real discount rates of 4, 6, and 8 %.**

| | **Discount Rate** | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Location** | **4 %** | | | **6 %** | | | **8 %** | | |
| | **NPV ($)** | **NPV ($/ac)** | **NPV ($/ha)** | **NPV ($)** | **NPV ($/ac)** | **NPV ($/ha)** | **NPV ($)** | **NPV ($/ac)** | **NPV ($/ha)** |
| **HWF** | $ 111,612 | $ 68 | $ 167 | $ 237,176 | $ 144 | $ 355 | $ 300,608 | $ 182 | $ 450 |
| **HBEF** | $ 344,583 | $ 345 | $ 851 | $ 454,301 | $ 454 | $ 1,123 | $ 509,731 | $ 510 | $ 1,260 |

*Combined projects: Stumpage and Carbon*

Net present values are positive at both HWF and HBEF when stumpage revenues and ROC net revenues for carbon projects are combined (Table 4.7). The combined project NPV will range between $ 333,099 (4% discount rate) to $383,937 (8% discount rate), i.e. $499 /ha (4 %) to $575 /ha (8 %) at HWF (Table 4.7). At HBEF, the combined project NPV will range between $416,649 (4 % discount rate) to $537,652 (8 % discount rate), i.e. $1,030 /ha (4 %) to $1,329 /ha (8 %) (Table 4.7).

<u>Comparison of Equal Annual Equivalent for Scenarios I, II and III</u>

A comparison of EAEs for all scenarios reveals positive EAEs (Table 8). At HWF, the EAE of Scenario I (carbon markets without harvests) ranges from $20 /ha/yr (4 % discount rate) to $62 /ha/yr (8 % discount rate), for Scenario II (harvests without carbon markets) it ranges from $27 /ha/yr (4 %) to $20 /ha/yr (8 %), and for Scenario III (harvests with carbon markets) it ranges from $20 /ha/yr (4 %) to $47 /ha/yr (8 %). At a discount rate of 4 %, Scenario II yields the highest EAE. However, at higher discount rates (6 and 8 %), Scenario I yields the highest and Scenario II yields the lowest EAE. This means that when prices and costs are assumed to be constant, at lower discount rates the most financially viable management strategy for HWF is to forego the carbon markets and carry out partial harvesting following BMP guidelines within RMZs. However, at higher discount rates, HWF would be the most financially viable if carbon stocks within RMZs were accounted for in the carbon markets in a no harvesting scenario.

At HBEF, the EAE of Scenario I ranges from $47 /ha/yr (4 % discount rate) to $124 /ha/yr (8 % discount rate), for Scenario II it ranges from $22 /ha/yr (4 %) to $20 /ha/yr (8 %), and for Scenario III it ranges from $42 /ha/yr (4 %) to $106 /ha/yr (8 %). Scenario I yields the highest EAE for all discount rates while Scenario II yields significantly lower EAEs than either Scenario. Thus, at HBEF it would be more financially viable if carbon stocks within RMZs were accounted for in the carbon markets either in a no harvesting scenario or under a partial harvesting scenario following IFM guidelines in the Protocol.

Table 4.8 gives a summary of EAEs of all projects. The sensitivity analysis shows an increasing trend in Scenario I (carbon markets without harvests) and a decreasing trend in Scenario II (harvests without carbon markets) as discount rates increase from 4 to 8 % due to the nature of the net cash flows.

**Table 4.7: Scenario III (harvests with carbon markets) Net Present Value (NPV) of stumpage and carbon projects in Huntington Wildlife Forest (HWF) and Hubbard Brook Experimental Forest (HBEF) over 125 years for real discount rates of 4, 6, and 8 %.**

| Location | Discount Rate | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | 4 % | | | 6 % | | | 8 % | | |
| | NPV ($) | NPV ($/ac) | NPV ($/ha) | NPV ($) | NPV ($/ac) | NPV ($/ha) | NPV ($) | NPV ($/ac) | NPV ($/ha) |
| **HWF** | $ 333,099 | $ 202 | $ 499 | $ 368,066 | $ 223 | $ 551 | $ 383,937 | $ 233 | $ 575 |
| **HBEF** | $ 416,649 | $ 417 | $ 1,030 | $ 496,449 | $ 496 | $ 1,227 | $ 537,652 | $ 538 | $ 1,329 |

**Table 4.8: Comparison of Equal Annual Equivalent (EAE) of projects under Scenario I (carbon markets without harvests), Scenario II (harvests without carbon markets), and Scenario III (harvests with carbon markets) for Huntington Wildlife Forest (HWF) and Hubbard Brook Experimental Forest (HBEF).**

| Location | Scenario | Discount rate | | | | | |
|---|---|---|---|---|---|---|---|
| | | 4 % | | 6 % | | 8 % | |
| | | EAE ($/ac/yr) | EAE ($/ha/yr) | EAE ($/ac/yr) | EAE ($/ha/yr) | EAE ($/ac/yr) | EAE ($/ha/yr) |
| **HWF** | **I** | $ 8 | $20 | $16 | $40 | $25 | $ 62 |
| | **II** | $11 | $27 | $ 9 | $22 | $ 8 | $ 20 |
| | **III** | $ 8 | $20 | $13 | $32 | $19 | $ 47 |
| **HBEF** | **I** | $19 | $47 | $34 | $84 | $50 | $124 |
| | **II** | $ 9 | $22 | $ 9 | $22 | $ 8 | $ 20 |
| | **III** | $17 | $42 | $30 | $74 | $43 | $106 |

## Discussion

Selecting the Best Project

When comparing the highest EAEs from each scenario at HWF, the "harvests without carbon markets" scenario (II) is 35 % more than both the "carbon markets without harvests" scenario (I), and the "harvests with carbon markets" scenario (III) at lower discount rates (4 % or less). However, the "carbon markets without harvests" scenario (I) is 82 – 210 % more than the "harvests without carbon markets" scenario (II), and 24 – 45 % more than the "harvests with carbon markets" scenario (III) at discount rates between 6 – 8 %. Costs of carbon projects are being heavily discounted (8 %) along with stumpage net revenue under the assumption of constant prices throughout the project lifetime. In reality, stumpage prices of individual species and the price of a carbon credit would vary, and costs of carbon projects would appreciate. In the initial analysis where revenues and costs were increased at real rates, the "harvests without carbon markets" scenario (II) fetched a significantly higher NPV and EAE than the remaining scenarios due to valuable timber species such as sugar maple and yellow birch appreciating in value throughout the project lifetime. This still holds true within the current projections under a lower discount rate. In fact, the "carbon markets without harvests" scenario (I) has the lowest EAE at a 4 % discount rate for two reasons. The first being, the actual carbon stock at HWF is not much higher than the common practice value for its assessment region (19 %) (compared to the 116 % at HBEF). The higher the actual carbon stock from the common practice value, the higher the ROCs issued during the first reporting period, and thus higher the net revenue that can be earned. A study by Kerchner and Keeton (2015) investigating the viability of northeastern forests in the regulatory carbon markets in California, revealed that the most important predictor of an offset project was a property's initial above-ground carbon stocking above common practice. They suggest that projects with stocking levels greater than 39 % above common practice yield the second greatest cumulative cash flow at a 7 % discount rate in the short-term after properties with both carbon stocks >39 % above common practice and project land area greater than 417 ha, which leads us to the second reason. The project land area is smaller than the minimum recommended area by developers (White 2015). Even though the initial C stocks were only slightly higher than the common practice value, a larger land base could have compensated for the initial low above-ground stocking. In general, all three project scenarios at various discount rates have positive NPVs and EAEs at the hardwood dominated HWF.

When comparing the highest EAEs for each scenario at HBEF, the “carbon markets without harvests” scenario (I) is 114 – 520 % more than the “harvests without carbon markets” scenario (II), and 12 – 17 % more than the “harvests with carbon markets” scenario (III) between discount rates of 4 and 8 %. Shifting objectives from management for stumpage to management for carbon would be a better economic option for HBEF due to lower value timber species coupled with a high initial carbon base (Kelly and Schmitz 2016). The “harvests without carbon markets” scenario (II) at HBEF isn't very attractive because it doesn't fetch higher stumpage revenue despite having approximately the same average volumes as HWF for the stumpage projects in Scenarios II and III. This is because the stumpage volume is primarily made up of spruce and fir species which are not as valuable as the hardwoods. Therefore, encouraging more structural retention or a 'no harvesting' option within riparian conifer forests will ensure the highest financial viability between the three project options for HBEF.

When comparing carbon sequestration potential between projects (Scenarios I and III), the ”carbon markets without harvests” option yields slightly higher aboveground carbon stocks than the “carbon markets with harvests” option at both locations (3 % higher at HWF and 4 % higher at HBEF) at the end of the project lifetime. Nunery and Keeton (2010) reported that their “no management stands” in the northeastern forests of the US had 140 Mg C/ha of aboveground live carbon when compared to the 83 Mg C/ha with high structural retention, low harvesting frequency using the individual tree selection system. Their “no management option” was almost 60 % higher in aboveground live carbon stocks than their lowest intensity harvesting systems after 160 years. A study by Lippke and Perez-Garcia (2008) reported that with a “no management option”, carbon stocks reach up to 400 % higher than a managed forest scenario and almost 100 % higher than a managed forest which also accounts for stored carbon in wood products at the end of 165 years. As this study reveals, either management option for the carbon projects (Scenario I or III) will yield somewhat similar aboveground live carbon stocks at the end of the project lifetime (125 years), thus making either management regime attractive for achieving carbon sequestration goals.

Costs are a significant factor that should be considered when approaching a project. The costs associated with carbon projects are very high and can range from approximately $247,070 during the first reporting period (2018) to a total present value of $652,600 at the end of the project lifetime at HWF (2143). The range for HBEF can be from $281,480 during the first reporting period (2018) to a total present value of $617,470 (2143). The project development fee (onetime cost) is one of the two highest costs associated with these projects and it can reach up to 64% and 69% of the costs during the first reporting period for HWF and HBEF, respectively. Kerchner and Keeton (2015) reported that their average project development fees, which was informed by a third-party verifier of carbon projects, was approximately $105,000. They acknowledge that this value is lower than what it would be if the projects were undertaken by project developers and also associate it to the lower land area of the projects (600 ha) which is the lower range of their financially viable projects. The high upfront costs are a financial risk for landowners because it takes months for ROCs to be realized by the Air Resource Board Offset Credits (ARBOCs), the official instrument for cap-and-trade offset market compliance (Kelly and Schmitz 2016; Ruseva et al. 2017). As a result, revenues may not be realized during the first year. However, this is a calculated risk large-scale landowners should be willing to take when selecting between the scenarios.

RMZs as a Separate Protocol for Carbon Offsets

The uncertainty associated with carbon projects and the long-term commitment period for the project are two main concerns for landowners considering earning net revenue via carbon offset programs (Ruseva et al. 2017; Caldwell et al. 2014). The management regime planned for the carbon project has to be carried out despite developments in timber markets and recurring costs, including project verification, and inventory has to continue for a minimum of 125 years. This long-term commitment is one way of ensuring 'permanence' of storage of an issued credit under the Protocol. This could pose a significant opportunity cost for the landowner. However, RMZs are designated areas for either restricted use or conservation for wildlife habitat. Timber management, if allowed within these areas, would still be regulated by state-imposed BMP guidelines. The degree of BMP restrictions would vary from state to state. Lighter harvests (forest tending) within RMZs within the parameters of the Protocol as well as riparian management objectives will relieve the stress of a voluntary reversal and invalidation of the project due to over-harvesting. Also, long-term project duration is not a major concern in this case as RMZs are unlikely to be designated for other purposes except for conservation objectives (water and/or wildlife). Having carbon projects limited to RMZs allows large-scale landowners conducting forest management to proceed with regular harvesting practices (even-aged or uneven-aged management) within the larger upland forest areas without the governing restrictions of the Protocol and opportunity costs associated with carbon projects.

As reported by Kerchner and Keeton (2015), the financial viability of carbon projects relies on initial and actual carbon stocking. Landowners are rewarded for the difference between those carbon stocks and the common practice aboveground live carbon stocks for the assessment region. Riparian areas in the temperate forests of the Northeast are known to have more productivity due to the favorable conditions of moisture and nutrients available for trees resulting in more biomass accumulation (Sutfin et al. 2016; Matzek et al. 2018). Jayasuriya et al. (2019) recorded a basal area of 30 $m^2$/ha (131 $ft.^2$/ac) within RMZs and a basal area of 26 $m^2$/ha (112 $ft.^2$/ac.) in non-riparian upland forests in a northern hardwood timberland in the Catskill region of NYS. This suggests that forested riparian areas in temperate northern hardwood forests have a high probability of carrying more carbon stocks than their upland counterparts and thus are more likely to have higher stocking than common practice values. This is clear in this study as the initial carbon stocks in HWF and HBEF were 32 % and 140 % higher than the common practice value, respectively. However, the disparity seen in the percent increase above common practice values between assessment areas and forest types should be acknowledged. Hence, these percentages are subject to change based on the location, forest type, site class, and the current forest management regime practiced within RMZs. Considering riparian areas or RMZs within larger forested tracts as a separate land management unit in the Protocol for forest carbon offsets thereby minimizes opportunity costs associated with regular forest carbon offset projects.

Difficulties of Simulation

In concurrence to the growth and yield model in FVS, NED-3 had to be used to calculate sawlog volumes for stumpage revenue estimations. Unlike NED-3, FVS does not provide the flexibility to record number/height of sawlogs and pulpwood. This limitation leads FVS to either over predict or in some cases under predict sawlog volume. Also, harvesting using decision rules of silviculture is difficult in FVS as there is no distinction between overstory and understory statistics. NED-3 provides more detailed stand statistics to facilitate management activities within stands, but does not predict volume accretion rates in line with growth rates suggested in literature. Therefore, either software could not be used as a standalone simulation model for this study. Until either software is improved to facilitate management activities providing detailed reports, this study recommends using FVS (approved growth and yield simulation model by the California Air Resource Board) along with a forest ecosystem management decision support system for scheduling harvesting activities within forest carbon projects.

As a result of the Northeastern Variant in FVS being a partial establishment model, a background regeneration model unique to each site of this study area was created. Kerchner and Keeton (2015) scheduled background regeneration for every 10 year cycle with 494 sapling/ha (200 saplings/ac.) and 80% survival. They also scheduled pulse regeneration post-harvest that included 2465 saplings/ha (998 saplings/ac.) for clearcuts, 1971 saplings/ha (798 saplings/ac.) for shelterwoods, 1482 saplings/ha (600 saplings/ac.) for group selections, and 988 saplings/ha (400 saplings/ac.) for individual tree selection harvests. Based on the inventory, a background regeneration for every 20 year cycle with 1008 saplings/ha (408 saplings/ac.) at HWF and 2021 saplings/ha (818 saplings/ac.) at HBEF with an 80% survival rate was scheduled. A pulse regeneration was not scheduled as only tending activities (thin from below and above) were being scheduled and not regeneration harvests within the riparian stands. In a study assessing the best modeling approach for annualized ingrowth count data for mixed species and mixed cohort stands in the Acadian Forest Region of North America, Li et al. (2011) suggested that the average ingrowth was 22.8 ± 34.1 counts/ha/year (mean ± SD) (9.2 ± 13.8 counts/ac/yr), with a range between 1 and 299 counts/ha/year (0.4 and 121 counts/ac./yr). If regeneration numbers of this study were scaled to their resolution, stands of this study grow 50.4 stems/ha/yr (20.4 stems/ac./yr) at HWF and 101 stems/ha/yr (40.9 stems/ac./yr) at HBEF which is between the range predicted by Li et al. (2011). The distribution of merchantable products during management activities throughout the period of simulation will vary based on the regeneration models. Thus, careful consideration must be given when selecting the appropriate regeneration model for a site.

## Conclusion

Riparian Management Zones around headwater streams can represent a significant proportion of land area within working forests of the Northeast. This can lead to an economic burden for landowners especially if these areas carry valuable stocking. Although state-wide BMP guidelines customarily permit partial harvesting within RMZs, landowners with large forest holdings (> 4,000 ha) now have the option of further minimizing opportunity costs of timber harvests by participating in carbon markets. This study investigated the tradeoffs between participation in carbon markets versus partial harvesting and how both those scenarios compare against a compromise between the two options, specifically for RMZs. This study concludes that all project scenarios return positive NPVs and EAEs for their project durations. As per results, RMZs dominated by high value northern hardwood forests, as represented by HWF, will have the highest NPV and EAE by not participating in the carbon markets and pursuing partial harvesting as per BMP guidelines (Scenario II) at lower discount rates. However, with higher discount rates, the returns from carbon markets become more favorable (Scenario I and III). RMZs dominated by mixed wood forests, as represented by HBEF, will have the highest NPV and EAE by participating in the carbon markets either in a no harvesting scenario or under partial harvesting as per the guidelines in the Protocol. The relatively high-volume level of primarily spruce and yellow birch resulted in a higher percentage of aboveground live carbon stocks available from the level of common practice for that assessment region. That combination of high stocking volume and low stumpage value represents a situation in which the carbon markets are a financially viable option for landowners. Landowners should also be aware that timber market prices and carbon credit prices will fluctuate in the future and that the Protocol should be adapted to significantly increase carbon stocks above the respective "business-as-usual" management within RMZs in order to meet the “additionality” criterion of the Protocol. In the end, participation in the carbon markets can potentially incentivize landowners to more effectively protect RMZs and all their associated benefits over the long-term without foregoing the opportunity to implement forest management.

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# Chapter 5 : Synthesis

The evolution of a buffer strip around forest streams came about in response to growing concerns related to the effects of harvesting operations on water quality. Research over the past decades have highlighted the importance of riparian areas on ecological functions outside of water quality, to include wildlife habitat, biogeochemical cycling, nutrient inputs, and biomass accumulation. A great deal of research also has focused on determining the appropriate riparian buffer distance required to protect streams during forest management activities. These buffers are designed to minimize sediment runoff into waterways and to conserve wildlife habitat that is affected by forest operations. Given the abundance of headwater streams in working forest landscapes (i.e. 70 – 80 % of entire stream networks), delineating the right buffer distance and buffer type within working forests is an important decision for forest resource managers.

A riparian buffer that is too narrow may fail to protect critical ecological functions of a riparian area, thereby creating negative environmental impacts, whereas a riparian buffer that is too wide may over-estimate actual functional riparian boundaries, thereby creating undue burden/increased opportunity costs for landowners. Despite numerous research findings confirming the efficacy of ecologically based riparian buffers, forest managers are predisposed to using a 'one-size fits all' buffer, or what is commonly known as fixed width riparian buffers. One of the main reasons forest resource managers favor fixed width buffers is that they are easy to implement and monitor for compliance, as opposed to ecological or functional riparian buffers.

Ecological riparian buffers are generally variable width buffers that aim to protect one or more ecological functions of a riparian area. Many published studies advocate for measuring numerous environmental parameters that often require complex equipment or tedious procedures to delineate ecological or functional buffers. Most of these studies are site specific and therefore, may not offer general guidelines that are applicable in a regional context. Ecological buffer methods which are not user-friendly regarding implementation, or lack proper instructions/guidelines for field application, further discourage resource managers from using variable width buffers, opting instead for fixed width riparian buffers. Although some studies have evaluated the costs of allocating RMZs in terms of timber value, research on contrasting and comparing the allocation of different riparian buffer types across various forest cover and topography is lacking. This information can help forest managers make an informed decision on implementing the appropriate riparian buffer type for headwater streams within working forests.

Forest management within RMZs hasn't received much attention when compared to the larger forest matrix around them. Although RMZs along headwater streams can represent a significant portion of working forests, few studies have addressed the potential opportunity costs of riparian area protection, while none have addressed improved forest management for increasing riparian carbon stocks. Alternative management scenarios like carbon markets could decrease opportunity costs of allocating riparian areas for landowners, and perhaps become an additional source of income. Specifically, no study had examined the economic viability of protecting RMZs via participation in the existing carbon markets in the US.

This dissertation sought to fill these aforementioned gaps in riparian area research. The following section provides a brief synopsis of my research contributions.

**Summary of major results and conclusions**

<u>Detecting riparian zones using understory plant diversity and composition patterns in mesic headwater forests of the Northeastern US</u>

Riparian buffers around headwater streams, allocated to minimize sedimentation during forest operations are rarely based on ecological criteria. Thus, the primary research objective was to identify a floristically significant riparian boundary for first- and second-order streams using plant species composition and indicator species to signify riparian environments distinct from the surrounding upland forest.

Across three forest sites distributed in the Northeastern US, this study detected a threshold riparian distance extending up to 6 -12 m (20 – 40 ft.) from streambanks of headwater streams using plant species richness. Empirical species richness was highest closest to the stream with a range of species count from 1–11 species/$m^2$. Even though this is not the actual functional extent of the riparian area, this distance represents an important threshold distance for plant species richness.

The discriminant analysis revealed that understory species composition closest to the stream differed significantly from that of all positions at greater lateral distances. This finding was further supported by the indicator species analysis where it identified six taxa as floral indicators of streamside positions. Of these indicators, only two were categorized as facultative wetland species in the Northeastern region. However, the indicator species analysis failed to identify strong indicators to represent the threshold distance mapped by the species richness model. This could be due to riparian forests along headwater streams in mesic environments having a closed canopy structure and well distributed summer rainfall patterns.

From a management perspective this study suggests that regional RMZ guidelines in the Northeast should designate the zone extending up to 12 m (40 ft.) on each side of the stream as a particularly sensitive area of the RMZ. Because headwater streams are disproportionately affected by forest management activities, and riparian protection guidelines are rarely based on locally available data, evidence-based studies such as the current research should guide regional riparian management to ensure that these areas continue to provide ecosystem services now and into the future.

<u>Assessing riparian area protection strategies along headwater streams in forested regions of the US</u>

Allocating fixed or variable width riparian buffers along streams depends on the complexity of buffer allocation and the opportunity costs of buffer areas. With headwater stream densities reaching 80% of entire stream networks, there is a need to assess if existing state-specific riparian guidelines, whether fixed- or variable width, are comparable with a "functional" riparian buffer as proposed by the USFS. Therefore, with a focus on headwater streams in five timber producing regions of the contiguous US, this study assessed land area differences between three buffer allocation strategies: functional based riparian buffer, state-specific riparian buffers, and a 30-m fixed width riparian buffer.

The Pacific Northwest and Western region watersheds delineated the widest "functional" riparian buffers along headwater streams due to their wide ravines, and/or relatively tall average canopy tree heights. Delineating variable width riparian buffers guided by the topography around streams and forest structural characteristics such as canopy tree height, includes a wide range of riparian ecosystem services and benefits that ensures the protection of these ecotones. Based on BMP manuals, state-specific riparian buffer guidelines in all sampled watersheds except those in the Lake States region, failed to identify the full extent of their "functional" riparian areas. Forest structure within working forests can change with time due to natural causes such as growth and disturbances, and forest management with forest tending and/or regeneration harvests. However, topography around headwater streams have low probability for change and is likely to remain as the constant variable in defining the "functional" riparian buffer. The terrace slopes around streams foster vegetation that is critical for maintaining bank stability to prevent bank erosion among other numerous ecosystem services. It is advisable that state-specific riparian buffer guidelines define buffer distances that encompasses, but not limit to, the variable terrace slope distances observed in the topography of their states.

Riparian protection guidelines provided by state BMP manuals should adequately protect riparian functions and dedicate more attention towards headwater streams. Along with the support of numerous studies performed throughout the years comparing fixed width buffers with ecologically meaningful variable width buffers, this study recommends the use of variable width buffers such as the "functional" riparian buffer to be used during forest management. Given the availability of recently published GIS tools and new GIS tools developed in this study, the assignment of variable width riparian buffers can be realistically adopted by forest managers to ensure riparian protection along headwater streams in working forests.

Protecting Timberland RMZs through Carbon Markets: A Protocol for Riparian Carbon Offsets

Harvesting restrictions within RMZs can place a burden on landowners especially when they delineate significant portions of forest lands as riparian areas. The opportunity costs for the landowner may be minimized by shifting the primary management objective in RMZs from timber production to compensation for above-ground carbon. Therefore, the primary objective of this study was to compare long-term net revenue generating potential of RMZs under three scenarios: (I) compensation for carbon credits without harvesting; (II) partial harvesting using Best Management Practices (BMP) guidelines without carbon credits; (III) partial harvesting combined with carbon credits as per the California Compliance Offset Protocol.

Managing forest carbon in riparian areas of large forest tracts can not only offset buffer allocation costs, but also act as a potential investment opportunity for landowners. Of the management options investigated in this study, northern hardwood riparian forests performed best under a partial harvesting scenario as per BMP guidelines at lower discount rates of 4 %. However, at higher discount rates between 6 – 8 %, improved forest management scenarios as per guidelines of the California compliance offset protocol, were more favorable due to their higher returns. On the other hand, spruce-fir forests or mixed conifer riparian forests did not perform as well as high value northern hardwoods in timber markets under the partial harvesting scenario as per BMP guides. However, at all discount rates between 4 – 8 %, these riparian forests fetched the highest returns via improved forest management options in the California compliance offset protocol.

Due to their geographic positions within watersheds, riparian areas are some of the most productive environments within landscapes. Average biomass stocks per unit area within riparian areas have been found to be significantly higher than that of their upland forest counterparts. Given that not a lot of applied research has yet been carried out using the California compliance offset protocol, the carbon sequestration potential in riparian areas as a separate candidate for carbon markets under improved forest management options has not been explored till this study. Through net positive outcomes of the economic analysis, riparian carbon offsets have proven to be an economically feasible option to be explored as separate management units of large forest holdings (> 4,000 ha) in the California carbon markets.

However, landowners have to be advised that even though profit margins can be high in carbon markets when compared to timber management options, costs for entering and maintaining sold carbon credits can exceed half a million USD throughout the lifetime of projects. This is a calculated risk large-scale landowners should be willing to take when selecting between the forest management scenarios in riparian areas.

From a riparian protection standpoint, this protocol limits biomass removals to even lower limits than in most state-specific RMZ guidelines. These lighter harvests create less disturbance activities within riparian areas which align with riparian conservation strategies and objectives (water and/or wildlife). Depending on forest composition, this study shows that riparian carbon offsets can bring greater returns to landowners than timber markets. Thus, this riparian forest management strategy not only lowers opportunity costs for landowners but also serves as another source of income in addition to timber management in the larger forested landscape.

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**Resume**


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## EDUCATION

**SUNY College of Environmental Science and Forestry**, Syracuse, NY

*Ph.D., Natural Resources Management* Aug 2020

*M.S., Ecology and Ecosystems* May 2016

**University of Kelaniya**, Kelaniya, Sri Lanka

*B.S., Environmental Conservation and Management* March 2013

## RESEARCH

**Dissertation Researcher (Ph.D.),** SUNY ESF Sep. 2016 – Aug 2020

Thesis Title: *The effects of Riparian Management Zone delineation on timber value and ecosystem services in diverse forest biomes across the United States.*

**Dissertation Researcher (M.S.),** SUNY ESF Sep. 2014 - May 2016

Thesis Title: *Contrasting functional-based riparian management zones with the fixed-width buffer approach and how it relates to riparian management guidelines.*

**Dissertation Researcher (B.S.),** University of Kelaniya, Sri Lanka Aug. 2011 - March 2013

Thesis Title: *Analysis of climate change impacts on a cascade agricultural system in the intermediate zone and evaluation of the adaptive capacity of the paddy farming community.*

## TEACHING

- Guest Lecturer for Natural Resources Managerial Economics, SUNY ESF Spring 2019
- Teaching Assistant for Statistics, Economics and Principles of Management, SUNY ESF 2014 – 2018
- Teaching Assistant for Informational Technology, Environmental Management, University of Kelaniya 2013 – 2014

## OUTREACH

Co-author of Riparian Management Zone guidelines for the New York State BMP Field Manual 2018

## FELLOWSHIPS AND AWARDS

- C. Eugene Farnsworth Fellowship, SUNY ESF (Ph.D.) 2019
- Research Assistantship, Dept. of Sustainable Resources Management, SUNY ESF 2017-2020
- ESF Graduate Student Travel Grant (Dean's Call) 2016,2017,2018
- C. Eugene Farnsworth Fellowship, SUNY ESF (M.S.) 2016
- First prize winner in the poster competition, SAF National Convention, Baton Rouge, LA 2015
- Diversity Scholarship/Diversity Student Ambassador, SAF National Convention, Baton Rouge, LA 2015
- Graduate Assistantship, Dept. of Forest and Natural Resources Management, SUNY-ESF 2014-2017

## MEMBERSHIP IN PROFESSIONAL SOCIETIES:

- Student member of Society of American Foresters (SAF) 2015 - Present
- Student member of United States Society of Ecological Economics (USSEE) 2019 - Present
- Life member of the Sri Lanka Association for the Advancement of Science (SLAAS) 2013 - Present
- Member of the Young Scientists Forum, National Science and Technology Commission, Sri Lanka. 2013 - Present

## CAMPUS INVOLVMENT

<u>SUNY ESF</u>

| | | |
|---|---|---|
| • Forestry Club, SUNY ESF | Member | 2015 - Present |

<u>University of Kelaniya, Sri Lanka</u>

| | | |
|---|---|---|
| • Environmental Conservation & Management (ENCM) Society | President | 2011 |
| | Member | 2009 - 2012 |
| • Inspiring the Next Career Fair | Co-Chairperson | 2011 |
| | Organizing Committee | 2009 - 2012 |
| • Cynosure Magazine (3rd issue), ENCM Society | Co-Editor | 2010 |