EFFECTS OF SKI SLOPES ON APPALACHIAN HEADWATER STREAMS

A Thesis
by
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Abstract

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Mountain ecosystems are increasingly stressed by human activities and global climate change. Ski resorts and other winter recreation areas (WRAs) are popular in temperate montane regions including the Southern Appalachian Mountains in southeastern North America. Large-scale land clearing and water extraction associated with WRAs may alter stream physicochemical attributes, biota and ultimately function. I examined impacts of four WRAs on physicochemical parameters and invertebrates in eight streams in the mountains of western North Carolina. I measured physicochemical parameters monthly, sampled invertebrates seasonally in 2011 and 2012 and measured fall and spring nutrient concentrations. Streams draining WRA-impacted catchments exhibited significantly elevated specific conductance and NO_3^- relative to control streams during spring sampling. Invertebrate data revealed lower total densities at all but one of the impacted streams (F = 42.6, p = 0.03) and higher total diversity at control streams (F = 5.1, p = 0.03). EPT metrics varied considerably among streams and treatment. Total density and EPT richness were negatively correlated with several water chemistry and substrate parameters, whereas EPT H’ and total richness were positively correlated with open riparian land use. Water chemistry and

habitat parameters exhibited more consistent responses to WRAs than did invertebrates. Counter-intuitive increases in some invertebrate metrics suggest that headwater stream responses to land use changes may be buffered by adjoining forested reaches or catchments and that responses to habitat and water chemistry alteration may be highly taxon-specific. The results of my study indicate that altered land use associated with WRAs may alter headwater stream communities and possibly ecosystem function through land use changes.

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Foreword

This thesis is formatted with the journal “Freshwater Science”. Throughout my research I gained a broader understanding on the difficulties associated with developing recreational areas to maximize profits while minimizing environmental impacts. I hope this research provides valuable insight into conserving our freshwater ecosystems.

Introduction

Winter recreation areas (WRAs) are a growing industry and important to the economy of mountain communities around the world. However, climate change is increasingly altering annual snowfall and accumulation patterns, resulting in the need for increased mitigation such as artificial snow production. There is growing concern over the environmental impacts of WRAs like ski resorts (Rixen 2002, Rixen et al. 2003, Wemple et al. 2007). To date, few studies have empirically assessed ecological effects of WRAs on montane headwater ecosystems. Headwater streams contain up to 80% of the world's liquid fresh water and supply a large portion of the world's population with water (Viviroli et al. 2003). The mountains of Western North Carolina concentrate and buffer water supplies and provide recreation for the growing human population in the southeastern U.S. A more comprehensive and quantitative understanding of how land use affects headwaters is critical to conserving the region's freshwater resources.

Natural snowfall at WRAs in western North Carolina ranges from 40 cm in the southern mountains (e.g., Sapphire Valley) to nearly 250 cm annually at the highest elevations in the northern mountains (e.g., Beech Mountain 1678 m, Perry et al. 2007). On average, natural snowfall does not adequately support WRA operation in the southeastern U.S. To compensate, resorts produce machine-made snow to increase base depth as well as to lengthen the ski season. In 2001, 90% of North American WRAs produced machine-made snow (Rixen 2002). Snow production allows some North Carolina WRAs to have additional snow base layers >250 cm. Machine-made snow is made by adding ice-nucleating agents

(INAs) to water (usually pulled from retention ponds) to increase ice crystal formation rates. Natural snow formation requires air temperatures <−7 °C; however, with INAs, snow can be produced at wet-bulb temperatures as high as −3 °C (Rixen et al. 2003).

Environmental effects of machine-made snow are still relatively unknown, and the few prior studies focused primarily on terrestrial ecosystems. Rixen et al. (2003, 2004) demonstrated that snow augmentation, through snow production and grooming, may provide deeper snow cover, buffering against vegetation damage from snow grooming machines and soil frost. Under deep snow pack, soil temperatures remain similar to unplowed natural snow conditions; however, machine-made snow may postpone snowmelt for up to 4 weeks, which may delay plant development (Rixen 2002, Rixen et al. 2003, Keller et al. 2004, Rixen et al. 2008, David et al. 2009). Changes in vegetation communities surrounding streams can alter stream habitats, nutrient cycling, and energy inputs. Over longer periods of time, vegetation community structure is shifted towards woody, frost-tolerant species (Rixen et al. 2003, Wipf et al. 2005). During the off-season, resorts maintain slopes for golfing and mountain biking, which have been known to further exacerbate runoff problems.

Machine-made snow exhibits a different chemical composition from natural snow because it is pulled from streams or retention ponds. Studies have shown that machine-made snow has a higher pH, higher conductivity, and higher nitrate and ion concentrations, thus altering stream nutrient concentrations (Kammer 2002; Rixen et al. 2003; Wemple et al. 2007). WRAs alter stream catchments and ecosystems by increasing nutrient inputs from snow run-off and altering hydrologic cycles (Molles and Gosz 1980; Wemple et al. 2007). Such changes to stream nutrient concentrations may alter community structure, function, and productivity (Molles and Gosz 1980, Lemly 1982). These processes have the potential to directly and/or indirectly cause significant changes to stream permanence, geomorphology, water temperature and quality, and invertebrate assemblages (Molles and Gosz 1980, Wemple et al. 2007, David et al. 2009).

Machine-made snow requires 0.7-2x (100-600 mm) the water input and yields two times the snow-water equivalent (SWE) of natural snow (Mosimann 1998, Rixen et al. 2004). Due to the increased water input and SWE values, snowmelt run-off has the potential to cause more severe and frequent flooding in streams adjoining WRAs. Even though up to 39% of water used for snowmaking is lost during the process or through sublimation (Eisel et al. 1988; Eisel et al. 1990), Stoecki and Rixen (2000) found snowmelt runoff doubled in streams adjoining WRAs. Wemple et al. (2007) also noted WRA-impacted streams had 36% higher runoff rates compared to control streams, even though snow making only accounts for 3-4% of annual precipitation.

Little is known about the impact of WRAs on streams, especially in the Eastern United States. Mountain streams are sensitive ecosystems; they have short growing seasons and unique nutrient cycles based on seasonal allochthonous input from deciduous riparian vegetation and detrital processing (Gomi et al. 2002, Shanley and Wemple 2002). Headwater streams also provide downstream habitats with important nutrients and organic matter subsidies (Vannote et al. 1980). WRAs alter stream catchments and ecosystems in numerous ways, including increased nutrient inputs from snow run-off and altered hydrological cycles, which have the potential to directly or indirectly cause significant changes to stream permanence, geomorphology, water temperature, water quality, and invertebrate communities (Molles and Gosz 1980, Wemple et al. 2007, David et al. 2009).

WRA-mediated stream modifications may have adverse effects on invertebrate production, which may include eliminating pollution intolerant taxa such as stoneflies and caddisflies, creating community shifts towards tolerant taxa. Aquatic invertebrates occupy a broad range of functional niches and are important to stream ecosystem processes including nutrient cycling, regulating primary production, and organic matter decomposition (Wallace and Webster 1996, Covich et al. 1999; Gage et al. 2004). Invertebrates are commonly used as ecosystem health indicators because of their abundance and wide range of pollution tolerance (Lenat et al. 1980; Muralidharan et al. 2010). Larvae of the insect orders Ephemereoptera, Trichopera and Plecoptera (known collectively as EPT taxa) are sensitive to environmental stressors and may served as indicators of stream impairment. Highly-impacted streams tend to have invertebrate communities dominated by tolerant species (e.g., Diptera, Oligochaeta). Impaired invertebrate communities reduce stream function and energy flow to higher trophic levels resulting in bottom up trophic changes (Wallace and Webster 1996; Covich et al. 1999; O'Driscoll et al. 2010).

Invertebrate communities in streams adjoining WRAs are likely to be affected by stressors similar to those experienced by urban streams, albeit at much smaller scales. Land use plays an important role in stream communities by regulating hydrology, channel geomorphology, and nutrient dynamics and is a leading cause of aquatic invertebrate species loss (Rader and Belish 1999, Olsen and Townsend 2005, Roy et al. 2007, Walsh et al. 2005, O'Driscoll et al. 2010). WRA-impacted streams are likely to experience both excessive flooding during spring snowmelt and exacerbated drought conditions during warm or dry winters, through snowmaking, as often seen in urban streams (Rixen et al. 2003, O'Driscoll et al. 2010). WRAs may also increase erosion and decrease stream bank stability through the flattening of terrain (Ries 1996; David et al. 2009). Sediment loads have been shown to increase downstream from WRAs (Molles and Gosz 1980, Wemple et al. 2007; David et al. 2009). Sediments may retain pollutants and nutrients and facilitate their transport during snowmelt and disturbance events such as flooding. Increased sediment transport may fill interstitial spaces and reducing benthic invertebrate habitat availability, respiration and ultimately abundance and richness (Waters 1995, Gage et al. 2004, Wemple et al. 2007, David et al. 2009). In one of the few published studies of WRA effects on stream ecosystems, Molles and Gosz (1980) found that invertebrate abundance and biomass were reduced downstream from WRAs, a phenomenon they attributed to increased sedimentation. Despite the ecological importance of headwater stream ecosystems, the increasing popularity of WRAs, and the unknown consequences of climate change, no known studies have assessed their impacts in the southeastern U.S. This study examines effects of four WRAs on western NC headwater streams with the following two objectives: 1) Assess differences in physical habitat, riparian land use, and water chemistry among draining from WRA-affect (impacted) and nearby forested (control) watersheds and 2) Assess differences in invertebrate assemblages among streams draining from WRA-affected (impacted) and nearby forested (control) watersheds.

Methods

Study sites

My study focused on streams affected by four WRAs near Boone, NC: (1) Sugar Mountain (Banner Elk, Avery County, NC), (2) Beech Mountain (Beech Mountain, Watauga/Avery counties, NC), (3) Hawksnest Resort (Seven Devils, Watauga County, NC), and (4) Appalachian Ski Mountain (Blowing Rock, Watauga County NC; Fig. 1). All four WRAs have the capacity to cover 100% of their slopes with machine-made snow. Hawksnest Resort (HWK) was converted from a ski slope into a tubing hill in 2008 but still produces machine-made snow regularly. Appalachian Ski Mountain (ASM) upgraded snow blowing machines in 2008 and now has the highest snow-making capacity in the southeastern U.S. In an average season, ASM spends up to 600 hours blowing snow and can pull over 22.7 million liters of water a day from a retention pond at the base of the mountain (Appalachian Ski Mountain 2011). Beech Mountain (BCH) is the highest elevation resort (1678 m), and Sugar Mountain (SGR) has the largest skiable area in North Carolina (46.5 hectares; Sugar Mountain 2011).

At each WRA, I identified one ski slope-impacted stream and one control stream. Streams were 1^{st} order and were selected based on their proximity to the ski slopes and accessibility. Control streams were as similar, based on field evaluations, in terms of riparian cover, substrate, gradient, etc., to impacted streams as possible but were not affected by WRA management practices. SGR streams drain into the Upper Elk River, BCH and HWK into the Watauga River, and ASM streams flow into the South Fork of the New River. Stream habitat quality was qualitatively assessed using the EPA Rapid Bioassessment Protocols for streams and wadeable rivers (Environmental Protection Agency 2010).

Land Use Analysis

I obtained GPS data points for each site using a Garmin GPSmap76Cx handheld unit and mapped them using ArcGIS (Version 10, ESRI Inc., 2012). NLCD 2006 land use data were obtained from the Multi-Resolution Land Characteristics Consortium website (Fry et al. 2011). I used 11 land use categories for analysis: developed open space, developed low intensity, developed medium intensity, developed high intensity, barren, deciduous forest, evergreen forest, mixed forest, shrub/scrub, grassland/herbaceous, and pasture/hay. Watersheds for each sampling point were delineated using the AGREE method in ArcHydro tools using a 10-m National Elevation Dataset DEM (Gesch et al. 2002; Gesch 2007). Streams within the study watersheds were buffered (100 m) in ArcMap and used to extract land-cover data. I calculated the percentage of watershed sub-basin and riparian land use (100 m) associated with classifications for each stream's catchment.

Physical Habitat

I measured depth (nearest mm) and flow (Marsh-McBurney Flo-Mate Model 2000) in the center of each Surber sample point prior to invertebrate sampling. I categorized substrates within each riffle by measuring 30 random particles along three transects spaced 1 m apart (n = 90 particles per riffle) once during the year. I measured the maximum diameter of each lithic particle and categorized organic or unmeasurable particles (i.e., sand, silt, bedrock, wood, clay, and organic matter).

Water Quality

At each site I measured dissolved oxygen (DO), pH, conductivity, and percent saturation monthly using a YSI model 63 (pH-conductivity) and YSI model 55 (DO) meters. Temperature loggers placed at each sampling site recorded temperatures at four hour intervals for one year. Nutrient (NO_3^- and NH_4^+) concentrations were measured twice during the study, once in the fall before snow production occurred and once in the spring during spring runoff. Nitrate determinations were made by reducing nitrate to nitrite with vanadium (III) choride followed by subsequent colorimetric nitrite analysis with Griess' reagent (Doane and Horwath 2003). Ammonium concentrations were determined via the indophenol blue method modified by Mulvaney (1996) with sodium dichloroisocyanurate as a hypochlorite source and ammonium chloride standard solutions.

Invertebrate Sampling

Invertebrates were sampled from three riffle habitats (spaced 100 m apart) with two 0.09 m^2 Surber samples per riffle in each stream. Surber samples (n = 6) were pooled, preserved in 70% ethanol, and keyed to family or genus (EPT taxa) in the Appalachian State University Aquatic Conservation Research Lab. I assessed differences in invertebrate abundance, diversity, and community composition between control and reference streams. I computed total richness, density $(m^2)^{-1}$, Shannon diversity (total H'), EPT richness, EPT density, and Shannon diversity (EPT H') for all samples. Sampling was conducted seasonally during March, July, and September 2011.

Statistical Analysis

All water physiochemistry and invertebrate data were analyzed using SPSS 19 (v. 19.0; SPSS, Inc., Chicago, IL). Shannon-Wiener Diversity (H'), a measurement of diversity in two or more habitats, was used to calculate diversity and by accounting for number of species and evenness of species. Invertebrate and EPT density were calculated by dividing total abundance by 0.59 m^2 (the total area sampled in each riffle). I tested for among-site and among-treatment differences in invertebrate community metrics and habitat parameters using 2-way ANOVA to assess the degree to which invertebrate communities respond to WRA-related changes. Ammonium and EPT density failed to meet the assumptions of normality (Shapiro-Wilk test) and data were $\log_{10}$ transformed to improve normality. Conductivity and temperature data failed to meet the assumptions of normality so a Kruskal-Wallis test was used. If significant interactions between site and treatment were found, data were split by site and 1-way ANOVA was used to compare between treatments. An independent t-test was used to find differences in land use between control and impacted sites. I utilized Principle Components Analysis (PCA) to reduce independent (abiotic) parameters into orthogonal variables (i.e., PCs) prior to multivariate analyses. I then used multiple linear regression to examine the ability of habitat parameters (as PCs) to predict biotic (invertebrate) metrics. Finally, Spearman's rho correlations were used to find associations between riparian land use and in stream habitat and invertebrate communities.

Results

Physical Habitat

Rapid habitat assessments describe stream physical health by ranking substrate, habitat diversity, and riparian land use across sites and calculating a score between 0 (highly impaired) to 200 (optimal). Control streams had a mean score of 133, and impacted sites had a mean score of 124. Habitat score ranged from 92 (SGR control) to 177 (HWK impacted). While no stream received a perfect score (200), BCH control, ASM impacted, and both HWK streams received scores in the optimal range (150-200), which was influenced by the high percentage of forested riparian zone around the streams. T-tests revealed no significant difference in habitat score between control and impacted streams.

Water Chemistry

Specific conductance was significantly higher at impacted (mean 112.6 us/cm) versus control (mean 45.1 us/cm) sites (ASM $H(2) = 29.4$, $p < 0.0001$; BCH $H(2) = 41.4$, $p < 0.0001$; SGR $H(2) = 45.6$, $p < 0.0001$; HWK $H(2) = 40.2$, $p < 0.0001$; Fig. 2).

Temperature differed significantly between sites ($H(2) = 697.5$, $p = <0.001$) but was not affected by treatment. A Kruskal-Wallis test revealed a significant within-site treatment effect on temperature at BCH and HWK ($H(2) = 209.2$, $p = <0.001$; $H(2) = 37.1$, $p = <0.001$).

DO saturation averaged 82.1% and 81.5% at control and impacted streams respectively and ranged from 77.7% (ASM impacted) to 84.5% (SGR impacted). ASM control stream has significantly higher DO saturation compared to the ASM impacted stream ($F = 5.0$, $p = 0.026$). There was a significant within site treatment effect at BCH with the control stream having higher DO concentration than impacted ($F = 8.9$, $p = 0.003$). The HWK impacted stream had higher flow velocity compared to the control stream ($F = 11.363$, $p = 0.001$; Table 2).

Fall nitrate concentrations were significantly higher at HWK, SGR, and ASM ($F = 33.7$, $p = 0.01$; Fig. 3). Mean control stream concentrations ranged from 0.11 to 0.32 ug/mL (HWK and BCH, respectively), and impacted streams ranged from 0.27 to 0.41 ug/mL (SGR; ASM). The strongest treatment effect was observed at HWK, where impacted streams had 2 times the $NO3^{-}$ concentrations of the control stream ($F = 7.31$, $p = 0.001$); however, there was no significant impact of WRAs on BCH stream NO_3^{-} levels.

Detectible NH_4^+ concentrations were found in both control and impacted streams at all WRAs. The ASM control stream had the highest NH_4^+ concentrations (0.10 ug/mL), while BCH impacted had the lowest (0.07 ug/mL). Although not significant, NH_4^+ concentrations were slightly higher in all control streams compared to impacted streams.

Invertebrates

I sampled a total of 2804 invertebrates, representing 37 families. Of the 2804, 1709 were EPT taxa from 43 EPT genera. The most abundant invertebrate genera were *Elimia* (Gastropoda: Pleuroceridae, n = 881) and *Epeorus* (Ephemeroptera: Heptageniidae, n = 187). Total density ranged from 211.3 individuals $(m^2)^{-1}$ (HWK impacted) to 21.6 individuals

$(m^2)^{-1}$ (BCH impacted) and averaged 62.4 individuals $(m^2)^{-1}$ in control and 83.1 individuals $(m^2)^{-1}$ in impacted streams (Fig. 4).

All metrics, except Shannon H', showed significant interactions between site and treatment; therefore, 1-way ANOVAs were used to examine differences between control and impacted streams within individual WRAs, where each individual riffle is a unit of replication. Control sites had significantly higher invertebrate densities compared to impacted streams at all sites except at HWK, where the impacted stream had significantly higher total density compared to the control stream ($F = 42.6$, $p < 0.0001$; Fig. 4). The higher total density at HWK impacted was attributed to the high number of *Elimia* found at the site. The ASM control stream had significantly higher richness than did ASM impacted ($F = 13.8$, $p < 0.0001$; Fig. 4). Mean total H' was 1.55 at control sites and 1.27 at impacted sites and was highest in the ASM control stream (1.88) and lowest in the BCH impacted stream (1.15; Fig. 4). Treatment significantly affected family H' diversity, and control streams were more diverse than impacted streams ($F = 5.1$, $p = 0.028$). The ASM control stream was significantly more diverse compared to the impacted stream ($F = 10.8$, $p = 0.002$).

One-way ANOVA revealed ASM and SGR control streams had significantly higher EPT density compared to respective impacted streams ($F = 12.9$, $p = 0.001$; $F = 6.0$, $p = 0.017$; Fig. 5). Conversely, though not statistically different, the HWK impact stream had 2 times higher EPT densities compared to the control stream. Both EPT richness and H' were significantly higher in the ASM control compared to the ASM impacted stream (both $F > 14.2$, $p < 0.001$; Fig. 5).

Physicochemical Habitat

Principle components analysis (PCA) revealed that 6 principal components (PCs) described 83.4% of the variability in physicochemical habitat across study sites (Table 3). PC_1 explained 26.4% of variability in habitat conditions and pH, specific and relative conductivity, salinity, and DO concentration loaded most heavily on PC_1 (Table 3). PC_2 explained 14.6% of habitat variability among sites and was heavily influenced by pH, percentage of wood substrate, depth, and flow and is likely a proxy of stream size. PC_3 explained 14.1% of the habitat variability among sites and is influenced by temperature, percent bedrock, and mean substrate particle size. PC_4 explained 11.2% of habitat variability and was strongly influenced by percent clay and organic substrate. Percent sand substrate and DO saturation loaded heavily on PC_5 (Table 3). Percent bedrock substrate loaded heavily on PC_6.

Invertebrate and Habitat Associations

Correlations revealed associations between habitat parameters and invertebrate metrics. EPT richness was negatively correlated with PC_1 (pH, specific and relative conductivity, salinity, and DO concentration; $r_s = -0.408$, $p = 0.048$), and total invertebrate density was positively correlated with PC_4 (percent clay and organic substrate; $r_s = 0.410$, $p = 0.047$).

Several invertebrate and habitat parameters were associated with land use. Specific conductivity correlated with percent-developed riparian land ($r_s = 0.550$, $p = 0.015$; Fig. 6). Forested riparian land was negatively correlated with percentage sand substrate ($r_s = -0.405$, $p = 0.050$). PC3 was positively correlated with percentage opened riparian land ($r_s = 0.444$, p

= 0.30; Fig. 7). Total richness was positively correlated with open land use (r_s = 0.559, p = 0.004, Fig. 8). Open riparian land was negatively correlated with DO mg/L while being positively correlated with temperature, PC_1 (r_s = -0.427 , p = 0.038; r_s = 0.587, p = 0.003; r_s = 0.559 =, p = 0.005)

Discussion

The Southeastern U.S. is one of the fastest developing regions in the country (O'Driscoll et al. 2010). Land use change due to this population growth has caused decreased water quality, and many streams exhibit reduced invertebrate abundance, density, diversity and richness (O'Driscoll et al. 2010). Although studies have focused on human-mediated water quality impacts of this growth in urban areas (e.g., Lenat and Crawford 1994, Gage et al. 2004), few studies have addressed the effects of ex-urban development; and no studies of which I am aware have addressed impacts of WRAs on southeastern stream ecosystems (Wemple et al. 2007).

I found that impacted streams had significantly more developed land use within a 100 meter wide riparian zone. Riparian zone developed land use was correlated with conductivity, DO mg/L, temperature, and PC_3 (percent bedrock, mean particle size and temperature). All impacted streams had significantly higher conductivity compared to control streams. BCH, SGR, and HWK control streams all had low conductivity levels, despite having up to 45% developed riparian land. These streams were heavily forested (≥ 50% at the catchment scale), which may mitigate effects of riparian development. The highest conductivity was seen at BCH impacted (175.5 μS/cm), which had twice the development of any other impacted stream and was 60% less forested than the BCH control stream. The ASM, SGR, and HWK impacted sites had < 25% development, often less than their respective control streams, which could have resulted in elevated conductivity and could thus either be from increased anthropogenic inputs from road salting as Molles and Gosz (1980) suggested or from artificial snow runoff.

Although autumn NO_3^- levels were higher in most impacted streams, concentrations were still relatively low compared to highly urbanized streams. BCH control and impacted streams had similar NO_3^- concentrations. This is attributed to the fact that both the BCH impacted and control streams flow out of retention ponds, which have been shown to mediate stream NO_3^- concentrations (Bernot and Dodds 2005). Wemple et al. (2007) suggested elevated NO_3^- levels associated with WRAs were caused by anthropogenic N-loading and land disturbance, while Molles and Gosz (1980) attributed higher NO_3^- levels to road salting. In my study, NO_3^- was not correlated with land-use or any other data suggesting impairment may originate from nonpoint sources. Molles and Gosz (1980) found NO_3^- levels to be seasonal, with the highest concentration in the winter months and low in the summer months. My study only sampled during autumn and spring and may have overlooked seasonal inputs of machine-made snow runoff.

WRA-impacted streams exhibited significantly reduced EPT richness and total H', suggesting that WRAs affect sensitive taxa, resulting in more homogeneous invertebrate communities; however, I also found relatively low invertebrate metrics for headwater streams at all sites, suggesting all streams likely suffer from some degree of anthropogenic impact. Invertebrate metrics varied considerably between WRAs, which may be attributable to natural variation in geology and biogeography. In addition, the 2009/2010 winter was considered to be an extreme winter with higher than average snowfall. Harsh winters and delayed snowmelt could have decreased the number of invertebrate larvae surviving to reproduce during the 2011 sampling season. The 2011 sampling season was uncharacteristically warm, which limited snow production in this region and may have reduced WRA impacts on streams. Future studies should account for variations in winter conditions prior to the sampling season.

Responses to ski slopes varied across WRAs. Unlike Molles and Gosz (1980), I found streams draining WRA-impacted watersheds had significantly lower total H' compared to streams draining reference watersheds. Only at ASM did the control stream exhibit higher metrics compared to impacted streams. Although the HWK impacted stream had higher total density and richness compared to control streams, EPT density, richness and H' were all lower in the impacted stream. This may be attributable to the high number of *Elimia* found at the impacted site. Although highly forested algae was abundant in the HWK stream, suggesting nutrient enrichment. Similar patterns were found at SGR; however, impaired stream communities were dominated by dipterans, suggesting a community shift towards pollution-tolerant taxa. Invertebrate parameters were typically lowest at both BCH streams despite differences in land use and significant differences in conductivity, suggesting both streams may be impacted by factors not accounted for in my study.

In headwater streams, riparian cover contributes organic matter for invertebrate processing, regulates energy inputs for food webs and downstream catchments, and influences nutrient cycling (Vannote et al. 1980). Land alteration that removes or reduces riparian cover adversely affects in-stream communities and physical habitats. Correlations between stream invertebrate and habitat (PC) metrics revealed that total density and EPT richness were negatively associated with water chemistry and substrate (percent fines) parameters. Total richness, total density, EPT density, and EPT H' were all correlated with a reduced percentage of open land in riparian areas. It is likely that community shifts towards

EPT genera, typically grazers, may result from increased primary production and sunlight. Previous studies have also shown water chemistry and invertebrate communities are strongly linked to terrestrial land cover (Richards et al. 1997, Sponseller and Benfield 2001).

This study makes a contribution to the limited research on the impacts of WRAs on environmental variables and is the first explicitly focused on Western North Carolina. Results from this study provide evidence that streams draining WRAs may have a negative impact on water quality but have only marginal impacts on invertebrate communities in headwater stream ecosystems. A longer sampling period (sampling over multiple years and multiple times within a year) is likely needed to more accurately quantify the impacts of snow production on stream water quality and invertebrate communities. In addition, historic land use was not examined in this study and may be contributing to the overall low numbers of invertebrates (Cuffney et al. 2010, O'Driscoll et al. 2010). Finally, future studies should quantify the snow melt runoff and snow additives entering streams from ski resorts. Recent research has shown that headwater streams are important to both downstream water quality and ecosystem function; however, no environmental regulations are currently in place to address WRA impacts on aquatic systems. By understanding the impacts of WRAs on streams in this region, resource managers, WRA-operators, and community leaders can start developing measures to help protect and conserve local habitats. A better understanding of the true costs of WRAs to regional water quality and ecosystem health will help managers better assess the tradeoffs associated with human population growth and development in sensitive montane ecosystems.

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Figures

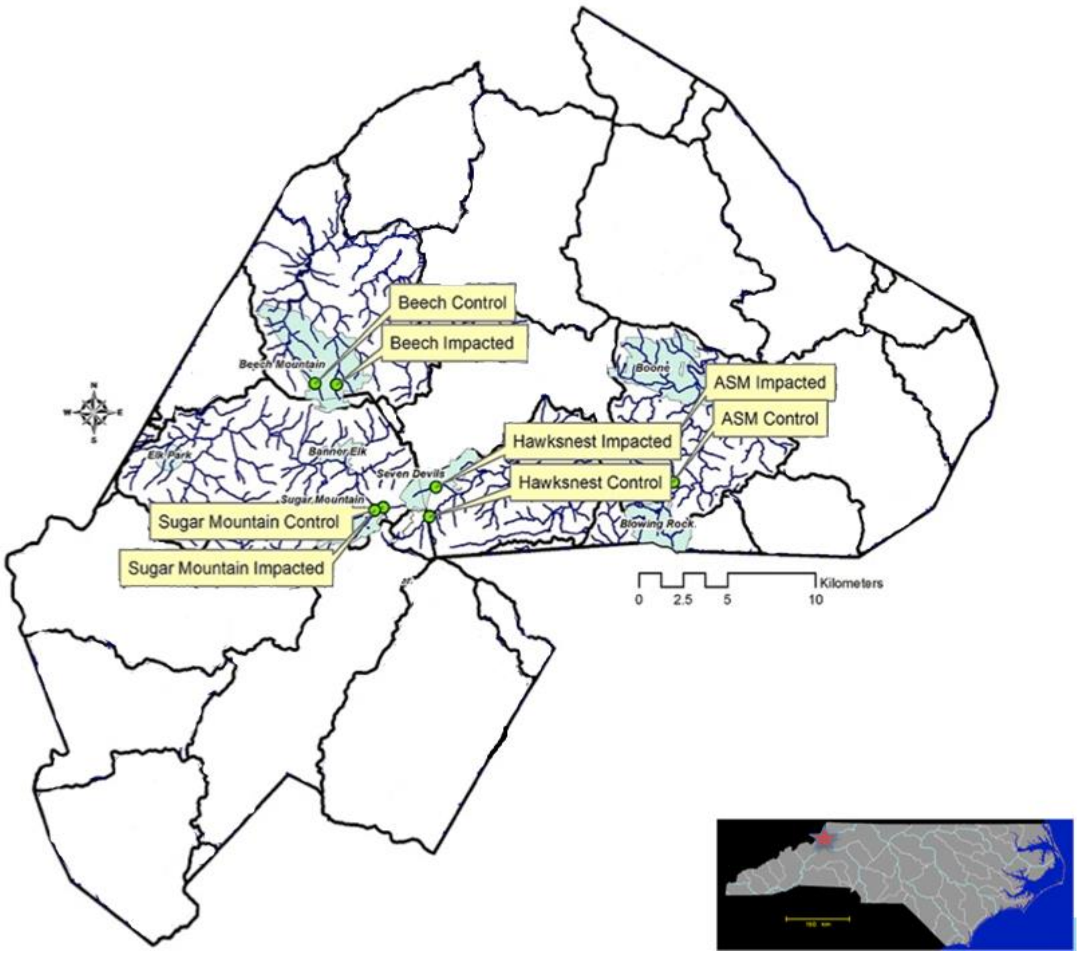

Figure 1. Control and impacted sites in Watauga and Avery County, North Carolina. Blue lines indicate streams, black lines delineate watersheds and blue shaded are populous areas.

Site	Developed	Forest	Open
ASM Control	18.5	56.4	25.2
ASM Impacted	12.5	79.9	7.7
BCH Control	31.2	68.8	0
BCH Impacted	71.5	18.5	10.1
HWK Control	2	98	0
HWK Impacted	23.3	60.3	16.3
SGR Control	45.7	52.2	2
SGR Impacted	19.9	65.9	14.2

Table 1. Percentage of riparian land use at Appalachian Ski Mountain, Beech, Hawksnest and Sugar resorts control and impacted streams.

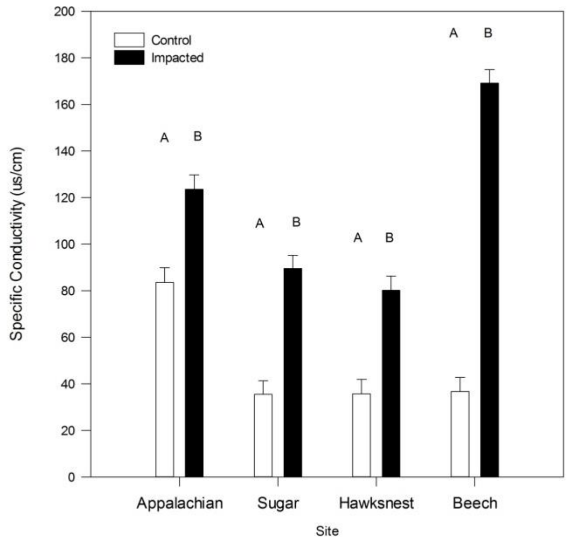

Figure 2. Mean conductivity at Appalachian Ski Mountain, Beech, Hawksnest and Sugar control (white) and impacted (black) streams. Sites with different letters above the bars indicate significant differences between control and impacted streams (ANOVA, p > 0.05).

	Temp	pH	D.O. %	D.O. (mg/L)	SC	RC	Sal	Flow	Depth
Control									
App	12.25	6.82	83.13	9.04	78.3	62.15	0.03	0.4	0.11
Beech	9.56	6.48	82.51	14.91	33.43	24.17	0.01	0.42	0.12
Hawksnest	9.63	6.67	80.88	11.3	33.9	24.85	0	0.27	0.18
Sugar	11.36	6.78	82.11	9.15	34.56	25.78	0.01	0.39	0.12
Impacted									
App	11.96	6.77	77.74	8.91	118.02	95.32	0.08	0.27	0.09
Beech	10.47	7.33	82.56	9.18	165.43	119.79	0.08	0.46	0.15
Hawksnest	11.37	7.14	81.26	9.04	78.61	60.13	0.01	0.52	0.16
Sugar	11.48	6.84	84.52	9.31	88.4	66.61	0.03	0.35	0.14

Table 2. Site water chemistry yearly averages. Abbreviations are as follows: SC = specific conductivity and RC = relative conductivity.

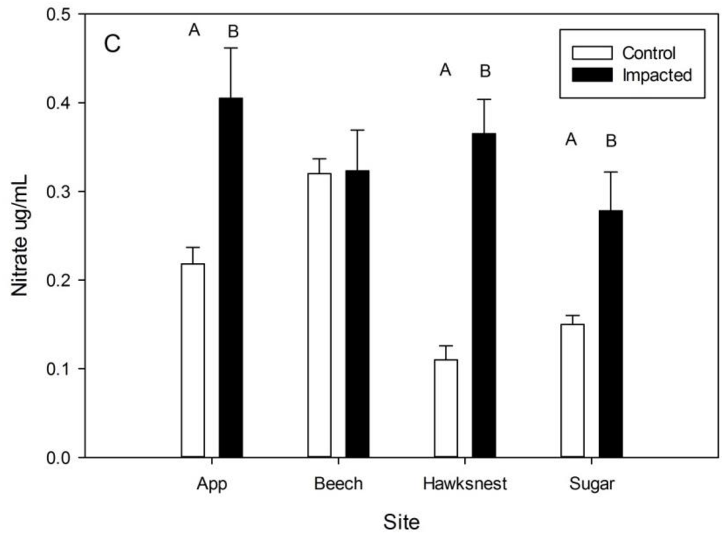

Figure 3. Fall NO_3^+ concentrations at ASM, BCH, SGR and HWK control (white) and impacted (black) streams. Sites with different letters above the bars indicate significant differences between control and impacted streams (ANOVA, $p > 0.05$).

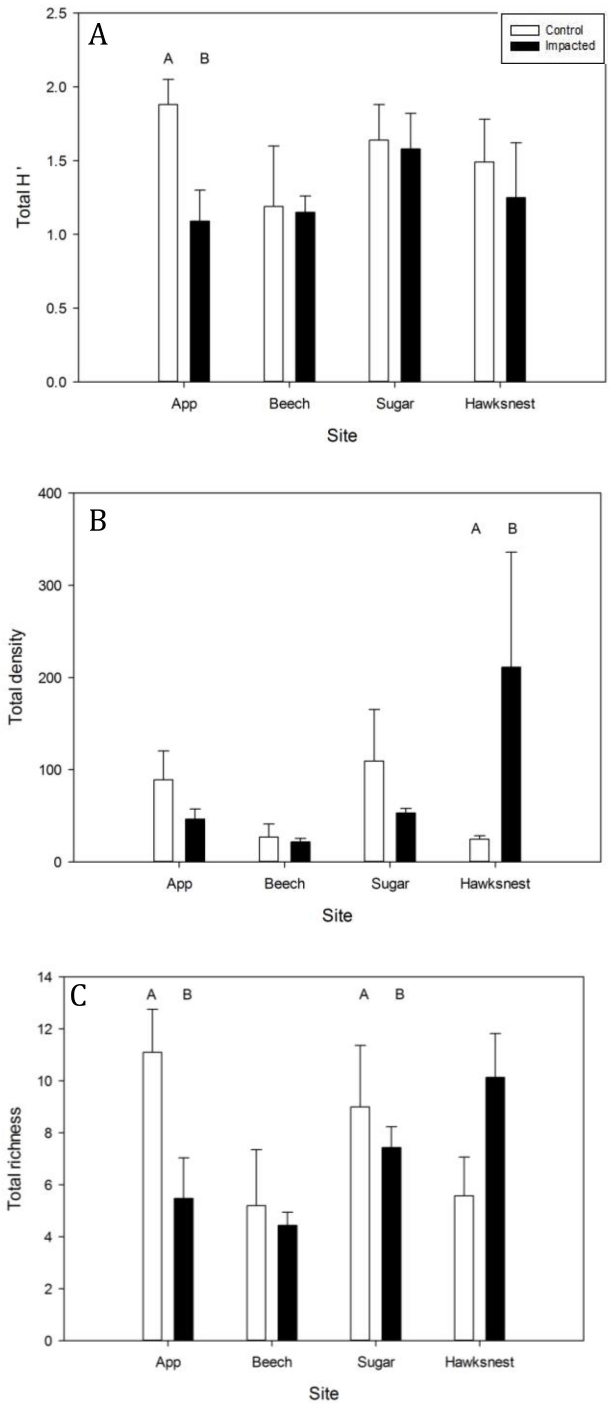

Figure 4. Benthic invertebrate diversity (H’, A), density (#/m^2, B), and richness (C) from streams at Appalachian SM, Beech, Sugar and Hawksnest. Sites with different letters above

the bars indicate significant differences between control and impacted streams (ANOVA, $p > 0.05$).

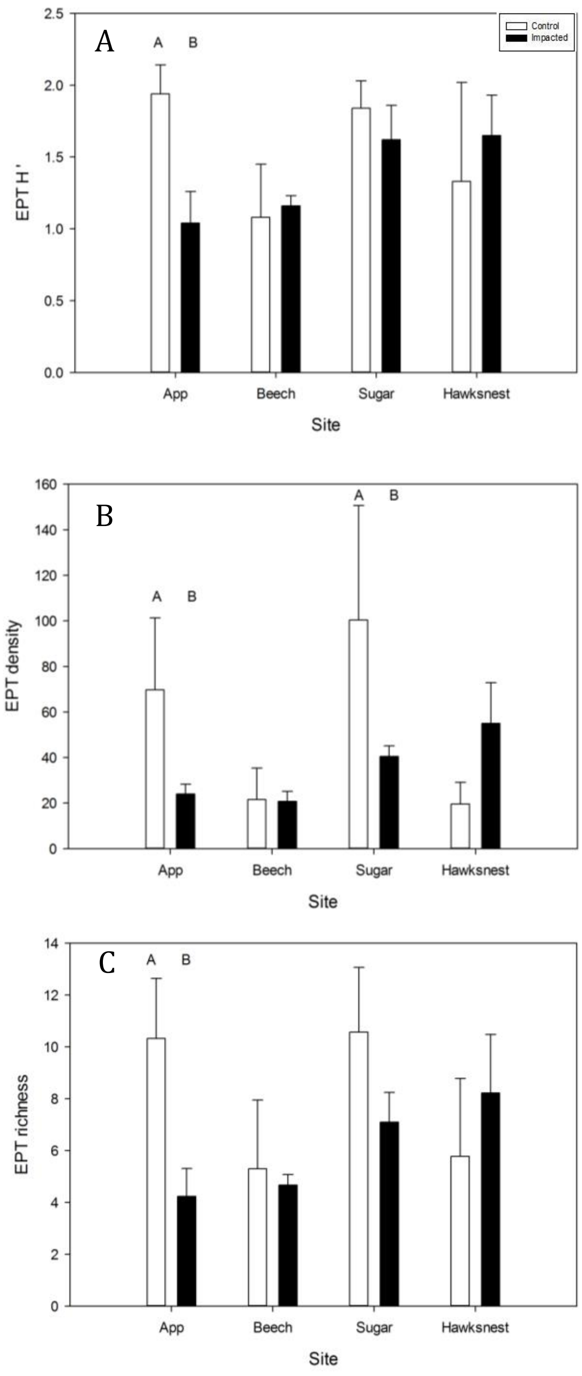

Figure 5. Mean Epheremoptera, Plecoptera and Trichoptera (EPT) diversity (H’, A), density (#/m^2, B), and richness (C) at Appalachian SM, Beech, Sugar and Hawksnest control (white) and impacted (black) streams. Sites with different letters above the bars indicate significant differences between control and impacted streams (ANOVA, p > 0.05).

Habitat Parameter	Components					
	1	2	3	4	5	6
Temperature	-	-	0.583	-	-	-
pH	0.687	-	-	-	-	-
DO (%)	-	-	-	-	0.688	-
DO (mg/L)	-0.615	-	-	-	-	-
SC	0.920	-	-	-	-	-
RC	0.925	-	-	-	-	-
Salinity	0.678	-	-	-	-	
Current	-	0.67	-	-	-	-
Depth	-	-	-.524	-	-	-
meansub	-	-	0.674	-.566	-	-
%Sand	-	-.508	-	-	0.518	-
%Bedrock	-	-	0.633	-	-	0.682
%Silt	-	-	-	-	-	-
%Organic	-	-	-	0.577	.504	-
%Clay	-	-	-	0.596	-	-
%Wood	-	0.712	-.510	-	-	-
Eigenvalue	**3.68**	**2.36**	**2.27**	**1.86**	**1.61**	**1.12**
% Variance	**23.0**	**14.8**	**14.2**	**11.6**	**10.1**	**7.0**
Cumulative Variance	**23.0**	**37.8**	**52.0**	**63.6**	**73.7**	**80.7**

Table 3. Factor loading scores, total and cumulative percent variance explained by Principle Components with Eigenvalues >1.0, and contribution of habitat parameters to PCs. Habitat parameter abbreviations are as follows: DO (%) = % DO saturation, SC = specific conductivity, (corrected to 25°C), RC = relative conductivity, meansub = mean size of measured substrate particles.

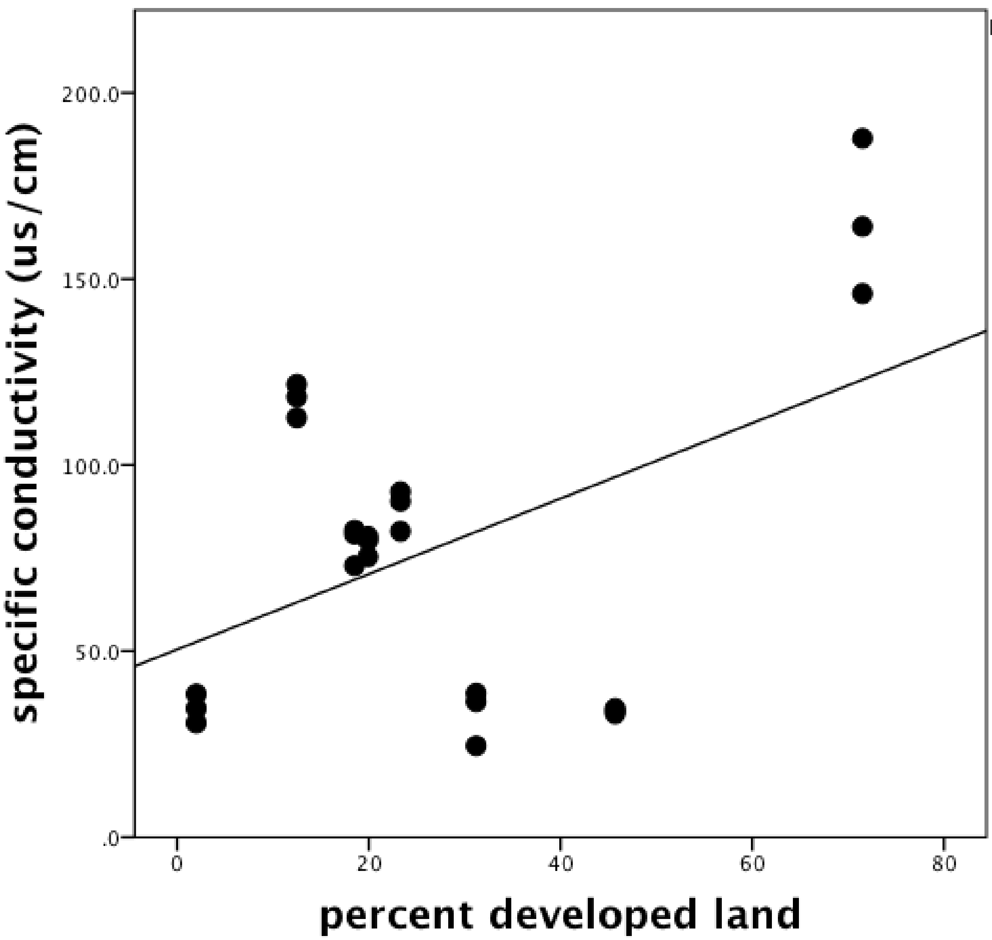

Figure 6. Association between percent riparian developed landuse and specific conductivity at invertebrate and water chemistry sampling sites ($r_s = 0.55$, $p = 0.015$).

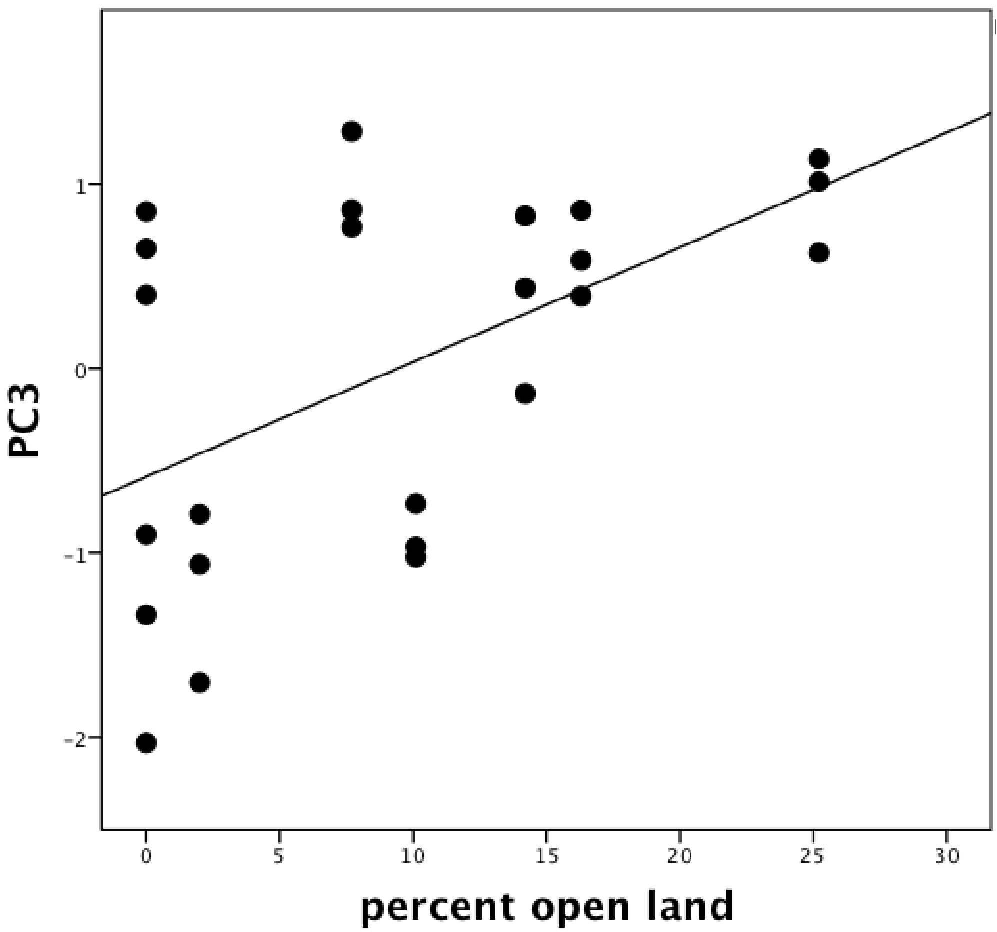

Figure 7. PC_3 (temperature, percent bedrock substrate and mean particle size) positively correlated with open land use in a 100 m riparian zone around sampling sites ($r_s = 0.44$, $p = 0.30$).

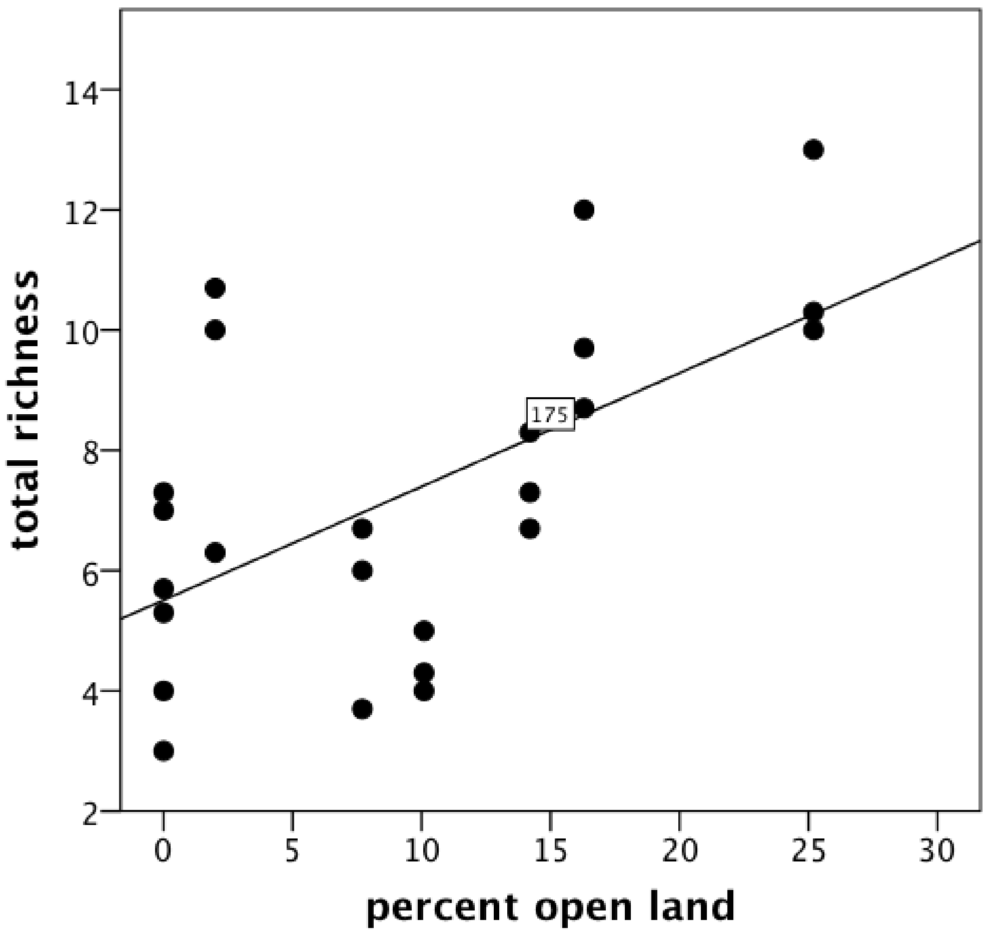

Figure 8. Total richness positively correlated with open land use in a 100 m riparian zone around sampling sites ($r_s = 0.56$, $p = 0.004$).

Vita

Kathryn Ann Rifenburg was born in Atlanta, Georgia, in 1986. She attended elementary through high school in Lilburn and graduated from Parkview High in 2003. Ms. Rifenburg enrolled at the Appalachian State University, North Carolina, to study Environmental Biology and Ecology with a concentration in business and was awarded a Bachelor's of Science degree in May 2009. Through 2009 to the summer of 2010, Ms. Rifenburg volunteered as an AmeriCorp VISTA in Douglas, WY. In the fall of 2010, Ms. Rifenburg accepted a graduate assistantship with Dr. Gangloff at Appalachian State University in Boone, North Carolina. The Masters of Science degree in biology was awarded in May 2013. Ms. Rifenburg's parents are James and Monica Rifenburg of Lilburn, Georgia.