Investigating Historical and Contemporary Land Cover Effects on Macroinvertebrate Communities and Water Quality of Virginia Piedmont Streams

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ABSTRACT

I investigated the relationships between historical and contemporary land cover and macroinvertebrate communities, water quality, and nutrient levels in 10 streams in a historically agricultural region of the Virginia Piedmont. Historical (1963) and contemporary (2011) impervious surface, open area, and forested cover were evaluated using aerial photos and GIS data. Macroinvertebrates were collected in the fall of 2012 and spring of 2013. Water quality parameters (temperature, conductivity, alkalinity, hardness, and DO) and nutrient concentrations (NH_3+NH_4, PO_4-P, NO_3-N, Cl, and SO_4) were measured at each site. Overall, forest cover decreased by 6.29%, open area decreased by 1.46%, and impervious surface increased by 4.83% from 1963 to 2011. Macroinvertebrate communities were explored using Principal Coordinates Analysis and were found to be significantly related to 2011 percent impervious surface. Water quality parameters were not significantly related to contemporary or historical land cover. Nitrate was negatively related with 2011 forest cover and positively related with 2011 open area; chloride was positively related with 2011 impervious surface and negatively related with 2011 open area. For the 10 watersheds included in this study, contemporary land cover is a better predictor of macroinvertebrate assemblages and nutrient concentrations than historical land cover.

ACKNOWLEDGEMENTS

This adventure in science would not have been possible without the boundless knowledge and guidance of my advisor, Dr. Fred Benfield. Thank you, Fred, for your honest and far-better-than-tolerable advising, for showing me that balance between work and life is possible in academia, and for never hesitating to deliver a swift kick in the pants the many times I needed it. I am also forever grateful for my committee member Dr. Bryan Brown, for his steadfast reassuring presence in my graduate career and for teaching me that often, the best science isn't born from a classroom or lecture hall, but over a round of drinks and some lively discussion at the pub. Thanks are also due to my final committee member, Dr. Reese Voshell, for imparting a slice of his encyclopedic knowledge of stream macroinvertebrates to me and for providing critical feedback.

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Finally, thank you to my families for your unending, enthusiastic support and love. You all have been my biggest cheerleaders since day zero and I cherish our massive familial network. It takes a village, and I am so, so very grateful for ours.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

INTRODUCTION

Anthropogenic alteration of land cover is a driving force behind global climate change and the reduction of biodiversity at all scales (Vitousek 1994, Vitousek et al. 1997, Foley et al. 2005). Land cover changes can have dramatic effects on streams that drain altered landscapes, including modifications to physical stream structure, chemical attributes of the water, and biotic communities (Frissell et al. 1986, Quinn et al. 1997). Of the many types of anthropogenically-driven land cover change, (e.g., urbanization, deforestation, prescribed burning), agriculture is perhaps the earliest (Anon 1853) and most thoroughly studied.

The immediate effects of agricultural land use are well understood (e.g., McDowell and Omernik 1979, Dance and Hynes 1980, Lenat and Crawford 1994). For example, removal of riparian vegetation, livestock access, and direct human manipulation leads to erosion, bank instability and deepening channels (Dance and Hynes 1980, Williamson et al. 1992, Zaimes et al. 2004, Jackson et al. 2014). Loss of streamside vegetation leads to an increase in light availability, increased periphyton abundance, a decrease in allochthonous inputs, and higher and more variable water temperatures (Dance and Hynes 1980, Quinn et al. 1992). Loss of vegetation at the watershed scale can lead to an increase in surface runoff and more variable (i.e. “flashy”) flows (Muscutt et al. 1993, Osborne and Kovacic 1993).

Changes to stream nutrient levels, particularly nitrogen and phosphorous, are also well studied in agricultural systems. McDowell and Omernik (1979) found that, as the percentage of agricultural land cover in a catchment increased, total inorganic nitrogen, total organic nitrogen, and total orthophosphate also increased. Frequently, agricultural nonpoint source pollution is caused by high livestock density or over-fertilization of crops (Carpenter et al. 1998). Excess nutrients, in turn, cause an overabundance of algae and an increase in litter breakdown rates, which can alter food web structure and macroinvertebrate assemblages (Allan et al. 2004).

Sediment is another major impact of agriculture to streams. As previously mentioned, removal of riparian vegetation and catchment vegetation increases erosion within the watershed and of stream banks. Eroded sediment is moved downstream during high flows, is deposited on the floodplains during floods, or settles in the stream channel during low flows (Sidorchuk and Golosov 2003). The amount of sediment carried by streams varies based on land cover; Costa (1975) found that streams draining active agricultural or construction areas exhibited increased sediment loads, while streams draining reforested watersheds or watersheds where soil conservation practices were applied did not. Streams in areas with long agricultural histories can show high turbidity, uncharacteristically sandy substrate, and unstable banks in the present due to excessive historic sedimentation and low sediment export rates (Jackson et al. 2005). While base levels of suspended solids provide food for many filter-feeding macroinvertebrates, high levels of suspended solids can impair feeding and respiration, reduce density and abundance, and change community structure of macroinvertebrates (Wood and Armitage 1997, Huggins et al. 2007). In addition to directly affecting macroinvertebrates, sediment indirectly affects habitat by homogenizing substrate (Delong and Brusven 1998).

Macroinvertebrates are subject to all of the physical and chemical changes above, so alterations in their community structure, richness, and abundance logically follow agricultural land cover change. Studies have found that as agriculture in a catchment increases, the richness of sensitive Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa decreases, while the richness and abundance of other, less sensitive taxa (e.g., Chironomidae) increase (Dance and Hynes 1980, Lenat and Crawford 1994, Delong and Brusven 1998). As sensitive taxa are replaced by more tolerant taxa, community structure shifts from its native assemblage to a new community, often less diverse but with a greater abundance (Quinn et al. 1997).

Over the last 30 years, there has been an increasing interest in the influence of land use legacies on ecosystems and conservation efforts (Foster et al. 2003). In aquatic systems, ecologists are particularly interested in the legacy impacts of agricultural land use in areas that are no longer predominantly agricultural. Harding et al. (1998) examined macroinvertebrates and fish in streams in the southern Appalachians that had a decrease in agricultural land cover and an increase in forested cover from the 1950s to the 1990s. The authors found that historical land use at the catchment level was the best predictor of fish and macroinvertebrate diversity, better than contemporary land cover at the riparian and catchment levels (Harding et al. 1998). Maloney et al. (2008) came to similar conclusions about several watersheds in the southeastern plains ecoregion that were formerly used for agriculture, silviculture, or military training exercises. The authors found that variables associated with macroinvertebrates, fish and primary productivity were strongly correlated with historical land use, and concluded that former land use continues to influence physical and chemical properties of the stream, which then influence the biota (Maloney et al. 2008). Sponseller et al. (2001) attributed an inability to predict macroinvertebrate communities in Appalachian streams with contemporary land cover at any scale to a lack of consideration for historical land use data, and suggested that macroinvertebrate assessments in streams in developing catchments are incomplete without the incorporation of land use at appropriate temporal scales.

I sought to examine relationships between historical agriculture and macroinvertebrates, nutrients, and water quality in the Piedmont ecoregion of Virginia. Using the above studies as starting points, I asked the following question: During which time period, historical or contemporary, is land cover most strongly related to 1) macroinvertebrate assemblages, 2) nutrient concentrations, and 3) water quality parameters?

METHODS

Study Location

This study was performed in the Roanoke River basin located in the outer Piedmont sub-province of Virginia, USA. This region is characterized as a broad, rolling plateau that has been carved into gentle hills and valleys by water erosion (Legrand 1960). The underlying geology of the study streams is mixed Precambrian and early Paleozoic rock with portions of Ordovician, Triassic, and Quaternary rock interspersed. The streams drain catchments composed of mainly biotite gneiss, amphibolite, and schists, but some of the streambeds are composed exclusively of Quaternary alluvium and terrace deposits (Henika and Thayer 1983, Marr Jr. 1984).

All sample sites were located in Pittsylvania County, Virginia. This agrarian area has been moderately to intensively farmed since before European exploration in the 1600s, beginning with Sioux agriculture and progressing to modern clear-cutting and row-cropping (Clement 1929). By 1860, large tobacco plantations were spread throughout the area. Even in 1951, farming tobacco was a large source of income for the county's 102,000 residents (Legrand 1960). Personal corn and hay fields, vegetable gardens and cattle pasture added to the agricultural stress on surrounding streams. Presently, few intensely farmed areas remain in Pittsylvania County, but pine plantations, cattle pastures, fallow fields, personal gardens and mown lawns are abundant.

Site Selection and Land Cover Analysis

The most recent United States Geological Survey elevation model (USGS 1/3 arc-second National Elevation Dataset) was used to delineate the drainage basins of all 2nd- or 3rd-order tributaries of the Banister and Pigg Rivers in Pittsylvania County (Gesch 2007). Ten watersheds of approximately the same area were selected for further analysis, and sampling locations were selected on each stream based on ease of access (Fig. 1).

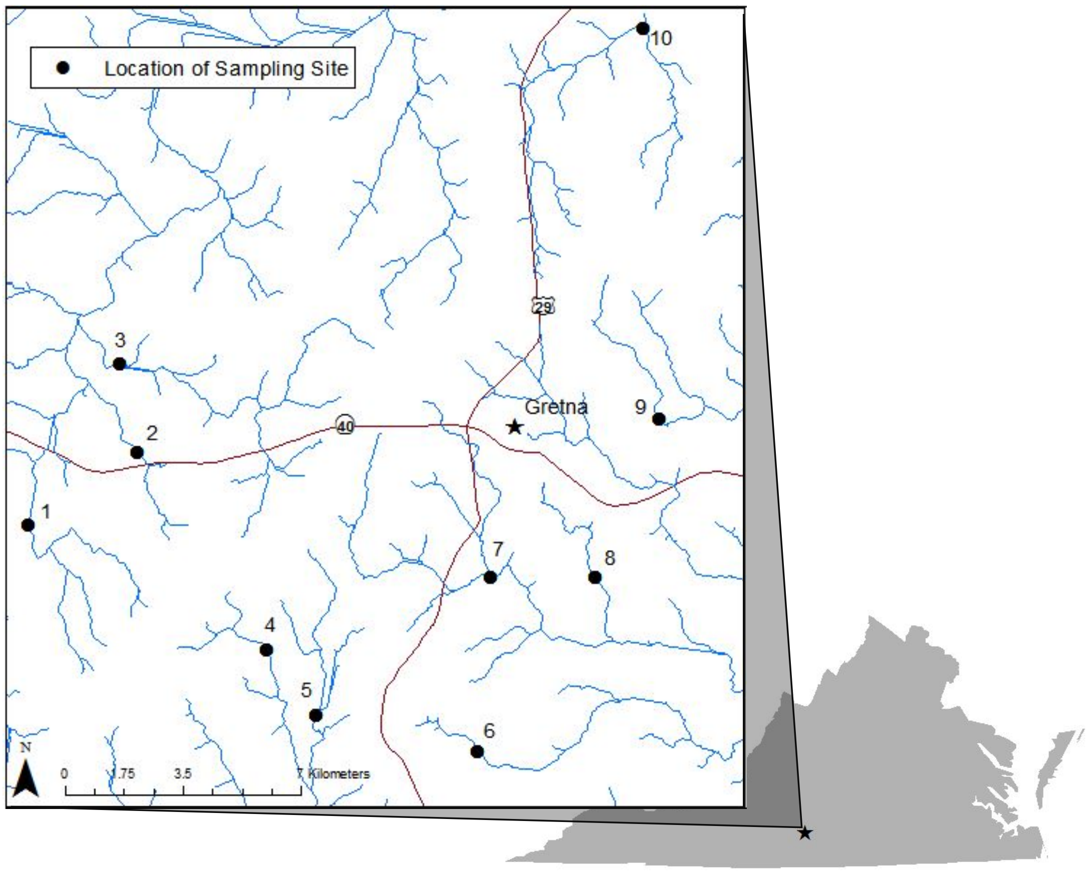

Fig. 1: Study area and sampling sites located on streams near Gretna, Virginia. Site names: 1. Harpen Creek, 2. Potter Creek, 3. Fryingpan Creek, 4. Pole Bridge Branch, 5. Cherrystone Creek, 6. Mill Creek, 7. Whitethorn Creek, 8. Long Branch, 9. West Fork Stinking River, 10. Little Sycamore Creek

Digital aerial photos of the study area from 1963 were obtained using the US Geological Survey's Earth Explorer online tool. These photos were georeferenced in ArcGIS 10 using coordinates provided in the metadata of the photos and ArcGIS 9.2 georeferencing protocol (2008 Environmental Systems Research Institute, Redlands).

Using the USGS National Elevation Dataset as a reference, land use in the 10 watersheds was delineated and land cover areas calculated. Areas of a distinct land cover type were traced by hand in the program using a Monoprice Graphic Drawing Tablet and at a consistent 1:1000 scale. Land cover types included in the historical data set were impervious surface (roads, parking lots, houses, and other impermiable anthropogenic surfaces), open (mown or maintained land, pasture, and active agriculture), and forested (deciduous forest, coniferous forest, and woody wetland areas). The percentage of each land cover type in the catchment upstream of each sampling site was calculated.

The National Land Cover Database 2011 dataset (Jin et al. 2013) was used to calculate the percentage of each of 15 land cover categories, which were reclassified into open (grassland/herbaceous, pasture/hay, and cultivated crops), forested (deciduous forest, evergreen forest, mixed forest, shrub/scrub, woody wetlands, and emergent herb wetlands), and impervious (open developed, low intensity developed, medium intensity developed, high intensity developed, and barren [rock, sand, or clay]) surfaces in each watershed. The NLCD categories were reclassified to facilitate comparisons between the more detailed contemporary land cover data set and the less detailed historical land cover data set. Percent change in open, forested, and impervious area from 1963 to 2011 was calculated for each watershed independently and for all 10 watersheds overall.

Water Quality and Nutrient Analysis

To investigate the effects of land use on water quality, the following parameters were measured at each site: temperature, total alkalinity, hardness, chloride, nitrate, sulfate, ammonium, phosphate, conductivity, and dissolved oxygen. In the summer and fall of 2012 and the spring of 2013, HACH kits were used to measure total alkalinity and hardness in the field, and a YSI Professional Pro meter was used to measure temperature, conductivity, and dissolved oxygen. In the spring of 2013, 3 filtered (Whatman 0.7 μm GF/F w/GMF) and unfiltered water samples were collected from each site, frozen for transportation and storage, and analyzed for chloride, sulfate (SO_4), nitrate (NO_3-N), total ammonium (NH_3+NH_4), and phosphate (PO_4-P) concentrations using a Lachat XYZ Autosampler (ASX 520 Series) and a Dionex Ion Chromatography system. Unfiltered water samples were analyzed for total suspended solids (TSS) using procedures standard to our laboratory (Webster et al. 2012). TSS, phosphate, nitrate, ammonium, and sulfate at each site were compared with one-way analysis of variance (ANOVA) tests and Tukey's Honest Significant Difference (HSD) tests when appropriate. Water quality metrics and nutrient concentrations were individually regressed against land cover percentages using the statistical program R to look for correlations with 1963 and 2011 land cover data (2011, R Foundation for Statistical Computing, Vienna).

Macroinvertebrate Sampling

Macroinvertebrates were quantitatively sampled following the standard practice of our laboratory (e.g. Harding et al. 1998, Burcher and Benfield 2006, Gardiner et al. 2009) as follows: a 0.41 m^2 frame was placed on the streambed and the area encompassed by the frame was thoroughly disturbed for 2 minutes. Macroinvertebrates dislodged were captured in a rectangular net (250 μm mesh) placed at downstream the edge of the frame. This process was repeated 3 times at each sampling site in the spring of 2013. Macroinvertebrates were qualitatively sampled once in late fall of 2012 by thoroughly disturbing all observable habitat types 10 m upstream and downstream of the quantitative sampling site. Disturbed habitats were swept with a dip net to collect dislodged invertebrates. Fifteen minutes of search and collection was performed at each stream.

Macroinvertebrates were preserved in 80% ethanol in the field. Samples were washed to remove fine sediments and the macroinvertebrates were sorted from debris, identified to the lowest practical taxonomic level using appropriate keys (i.e., to genus where practical), and enumerated (Merritt et al. 2008). Taxa richness, Simpson's Index of Diversity, %EPT taxa, and dominant taxon (taxon with the highest density) were calculated for each site using the quantitatively sampled macroinvertebrate data, and adjusted %EPT taxa was calculated by removing tolerant EPT taxa (those with a tolerance value of 3 or greater) from the calculation. Macroinvertebrate metrics were regressed against land cover percentages to look for correlations with 1963 and 2011 land cover.

A Principal Coordinates Analysis (PCO) was performed in R on the quantitatively sampled macroinvertebrate data to represent communities in multivariate space based on similarity (2011, R Foundation for Statistical Computing, Vienna). No data transformations were used and distance matrices were calculated using a Jaccard distance metric and the *vegdist* function in the R package *vegan* (Oksanen et al. 2011). The PCO was performed using the *pco* function in the R package *labdsv* (Roberts 2010). PCO axes 1 and 2 were then regressed against land cover percentages, nutrients, and water quality measurements to look for correlations between the measured variables and macroinvertebrate community structure. The PCO axes were also regressed against watershed area to elucidate any relationships, and were plotted against the macroinvertebrate matrix to find the taxa most strongly correlated with each axis.

RESULTS

Land Cover

Catchment area above the sample sites ranged from 693 to 3882 ha. The study watersheds in 1963 were primarily forested with substantial portions of open area and small patches of impervious surface (Fig. 2). Forested area ranged from 45.2 to 74.6%, open area from 22.4 to 50.4%, and impervious surface from 2.3 to 4.4%. In 2011, the study watersheds were more equally forested and open, with a considerable amount of impervious surface present (Fig. 3). In 2011, forested area ranged from 26.9 to 54.4%, open area from 35.6 to 63.7%, and impervious surface from 5.4 to 12.9%.

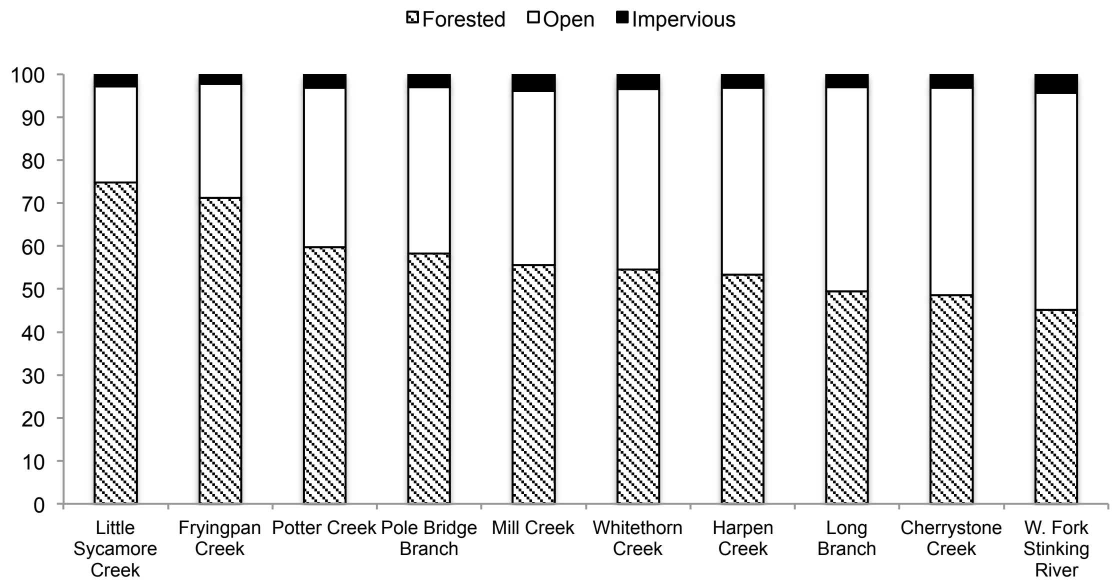

Fig. 2: Percentage of 3 land cover types in each watershed in 1963. Sites are in order from most to least forest cover.

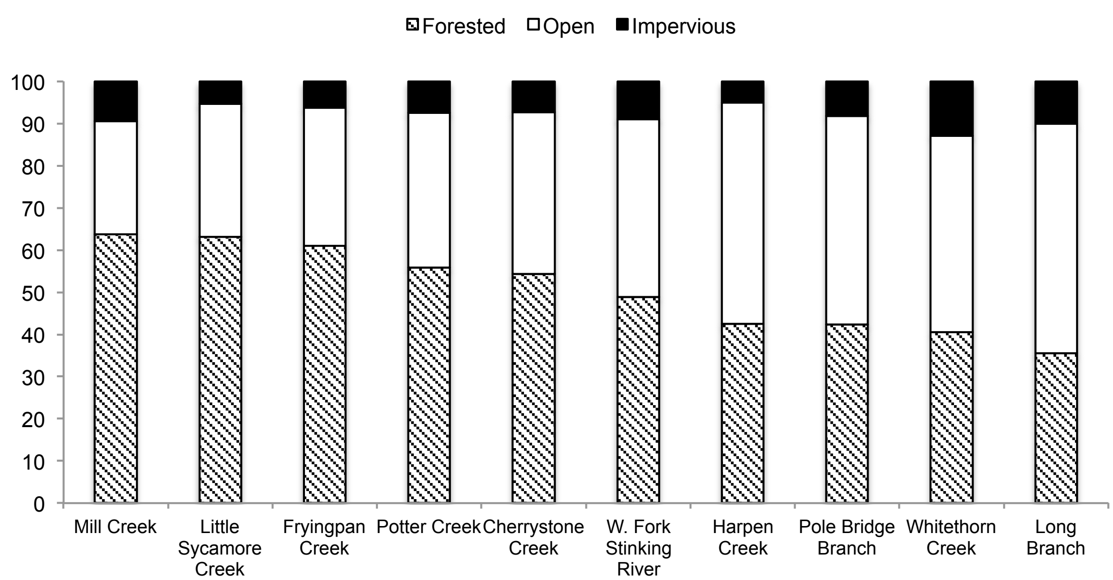

Fig. 3: Percentage of 3 land cover types in each watershed in 2011. Sites are in order from most to least forest cover.

From 1963 to 2011, forested area decreased in 7 of the 10 study watersheds with a maximum decrease of 16.0% and increased in 3 of the 10 watersheds, with a maximum increase of 8.1%. Open area increased in 6 of the 10 watersheds with a maximum increase of 10.8% and decreased in 4 of the 10 with a maximum decrease of 13.6%. All 10 watersheds exhibited an increase in impervious surface cover, with a maximum increase of 9.4% (Fig. 4). Overall, the entire study area showed a decrease in open area of 1.46%, a decrease in forested area of 6.29%, and an increase in impervious surface area of 4.83%.

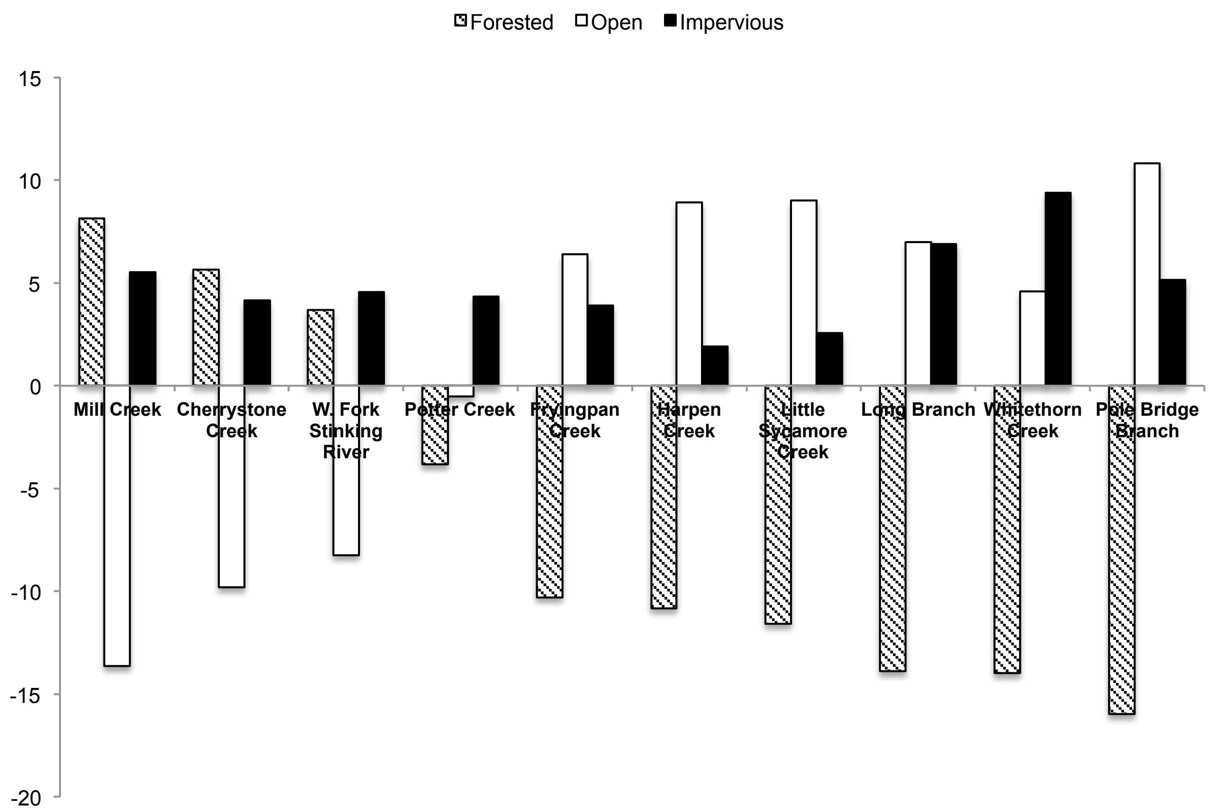

Fig. 4: Percent change in land cover from 1963-2011. Sites are in order from greatest positive to greatest negative change in forest cover.

Water Quality and Nutrients

Water temperature ranged from 5.1 to 21.5 °C with an overall, among-site mean of 13.6 °C (Fig. 5). The minimum dissolved oxygen concentration at any site was 7.9 mg/L and the maximum was 12.3 mg/L, with a mean of 9.9 mg/L (Fig. 6). Conductivity ranged from 33.0 to

264.0 μS/cm, with an overall among-site mean of 134.8 μS/cm (Fig. 7). Total alkalinity ranged from 20.0 to 60.0 mg/L $CaCO_3$ with an average of 32.6 mg/L $CaCO_3$ among the sites, and total hardness ranged from 8.0 to 60.0 mg/L $CaCO_3$ with an average of 29.1 mg/L $CaCO_3$ among the sites (Fig. 8). TSS ranged from 3.1 to 12.2 mg/L and was not significantly different among the sites ($F_{9,16}$=2.75, p =0.06, Fig. 9). No water quality metrics were related to land cover percentages at the α=0.05 level of significance.

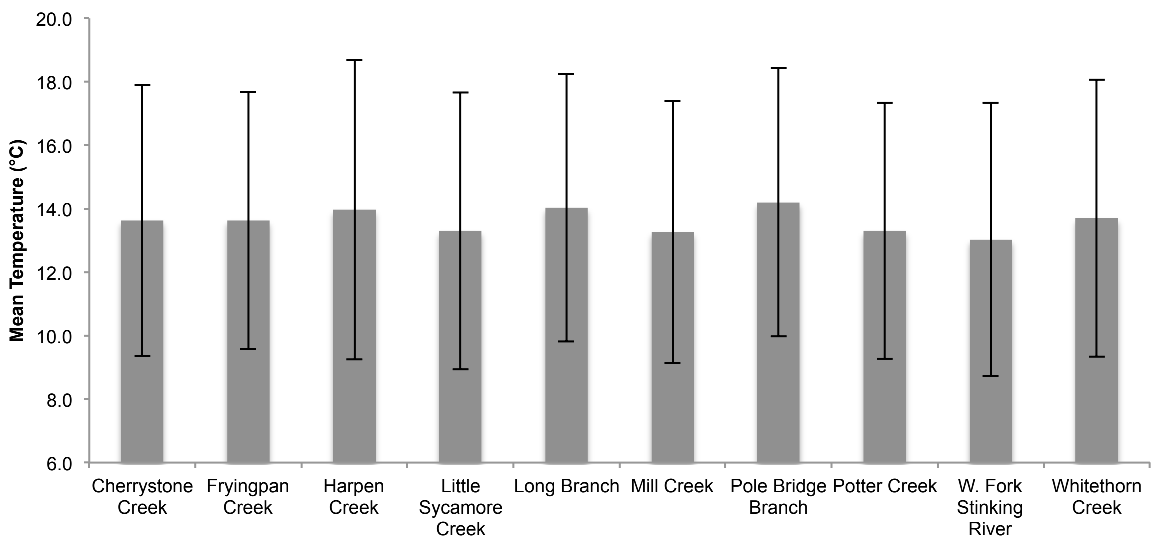

Fig. 5: Mean water temperatures (°C) in 2012-2013. Error bars represent ±1 SE.

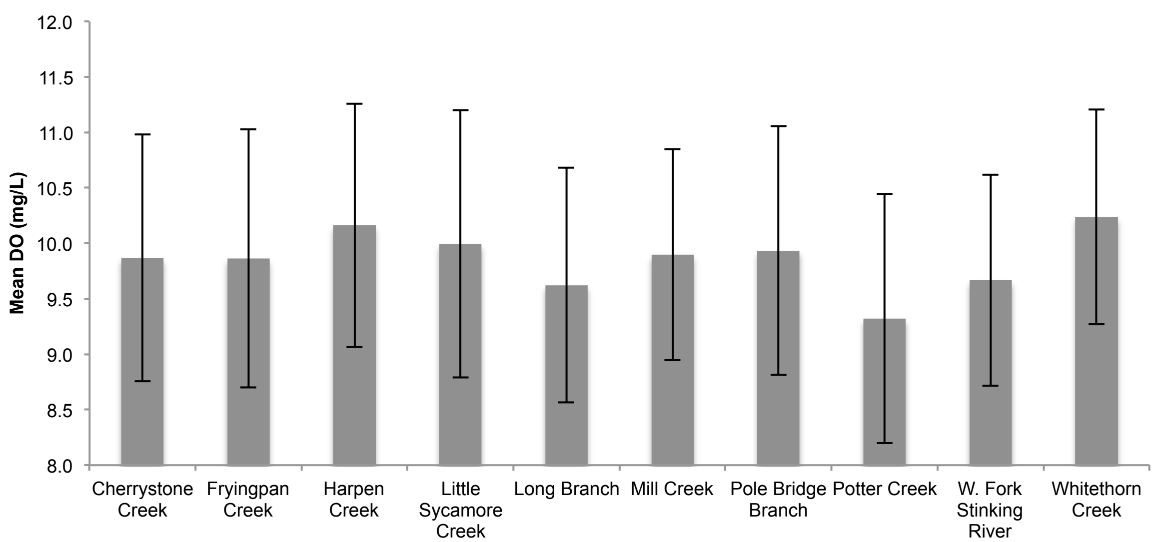

Fig. 6: Mean dissolved oxygen (mg/L) in 2012-2013. Error bars represent ±1 SE.

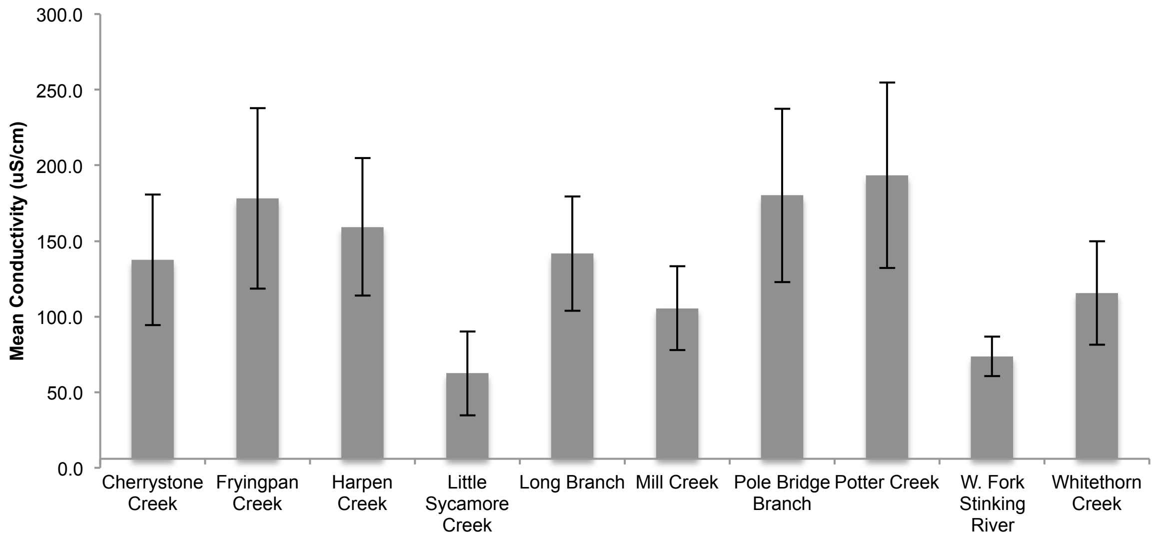

Fig. 7: Mean conductivity (µS/cm) for 2012-2013. Error bars represent ±1 SE.

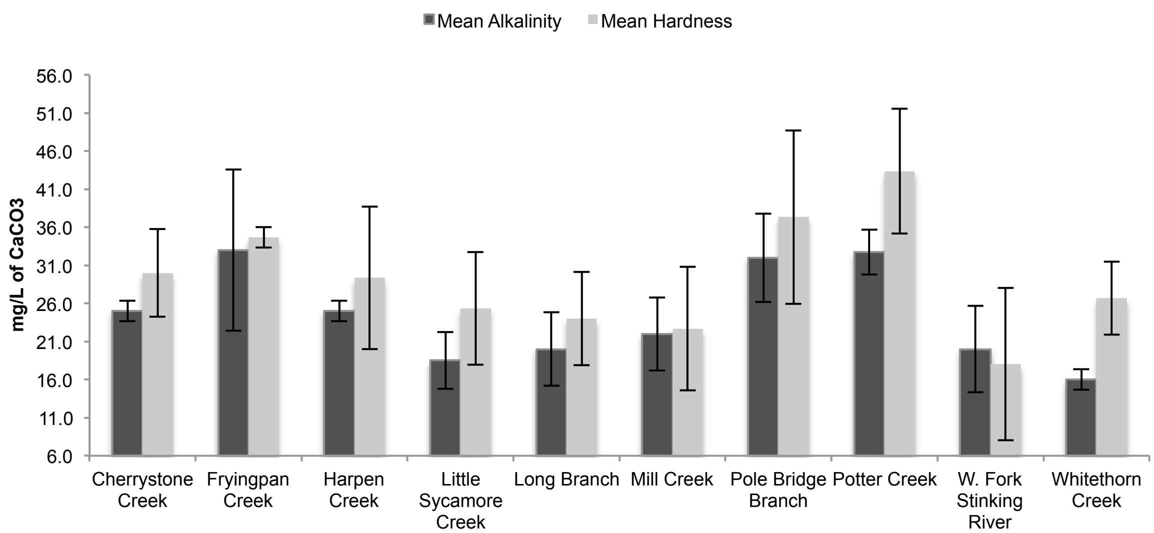

Fig. 8: Mean alkalinity and hardness (mg/L $CaCO_3$) in 2012-2013. Error bars represent ±1 SE.

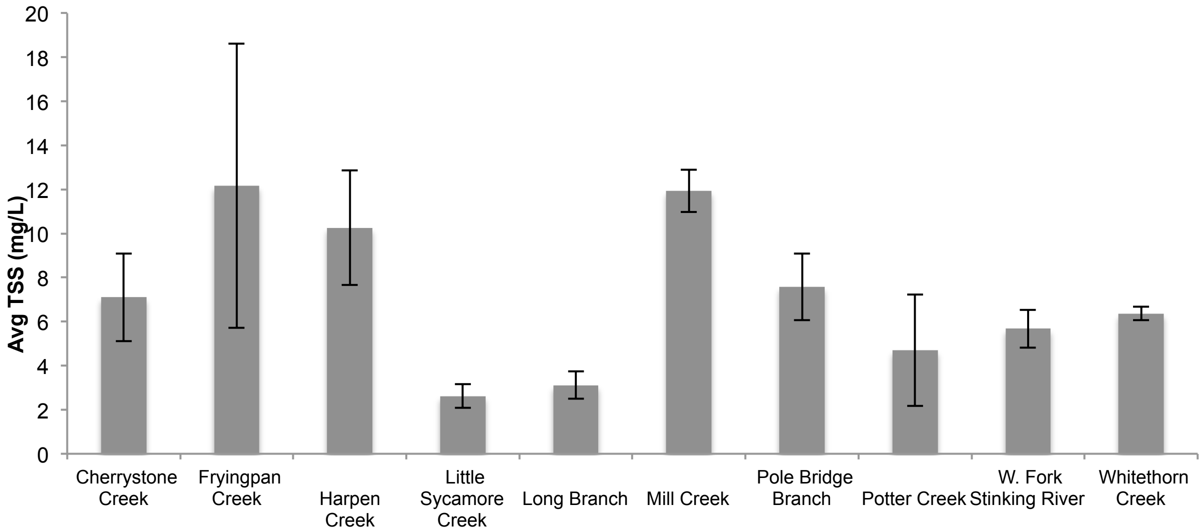

Fig. 9: Total suspended solids (mg/L) in spring 2013. Error bars represent ±1 SE.

Phosphate (as orthophosphate) was not present in measurable concentrations at any site. Chloride concentration was significantly different among the sites ($F_{9,20}$=19.12, $p < 0.001$, Fig. 10), as was nitrate concentration, ($F_{9,20}$=2785.6, $p < 0.001$, Fig. 11), sulfate concentration ($F_{9,20}$=87.2, $p < 0.001$, Fig. 12), and ammonium concentration ($F_{9,20}$=15.9, $p < 0.001$, Fig. 13). Chloride was positively correlated with impervious surface in 2011 (r^2= 0.53, p =0.02, Fig. 14) and negatively correlated with forested area (r^2= 0.43, p =0.04, Fig. 15). Nitrate was positively correlated with open area in 2011 (r^2= 0.44, p =0.04, Fig. 16) and negatively correlated with forested area in 2011 (r^2= 0.46, p =0.03, Fig. 17). No further significant relationships between nutrient concentrations and land cover percentages were found at the α=0.05 level of significance (Table 1).

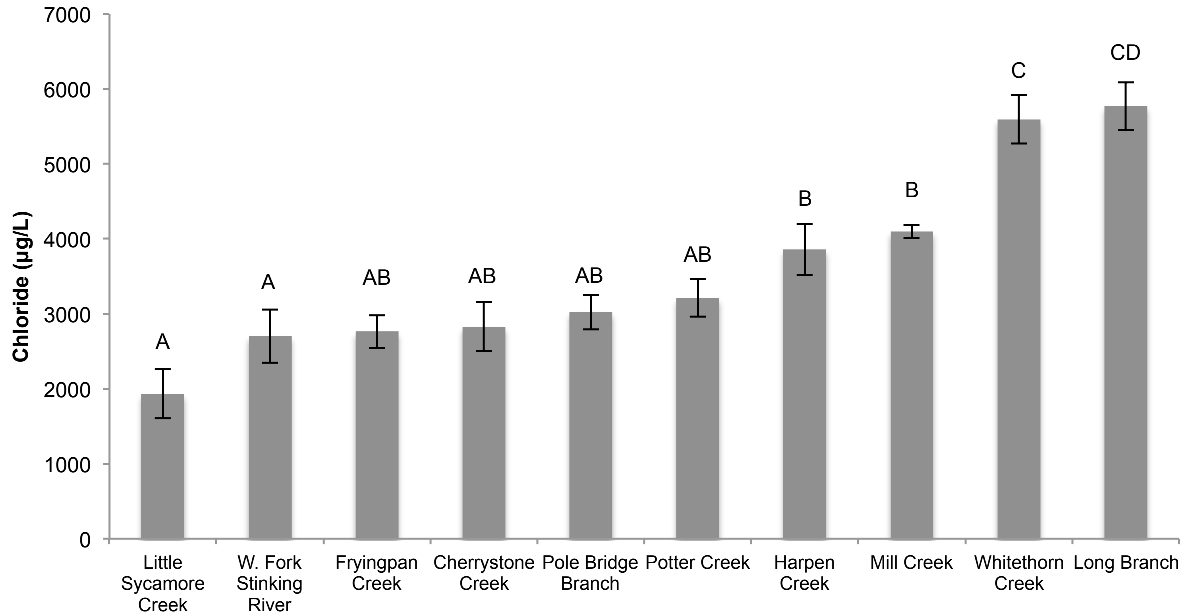

Fig. 10: Chloride concentrations (µg/L) in spring 2013. Error bars represent ±1 SE and letters represent significant differences determined with Tukey's HSD

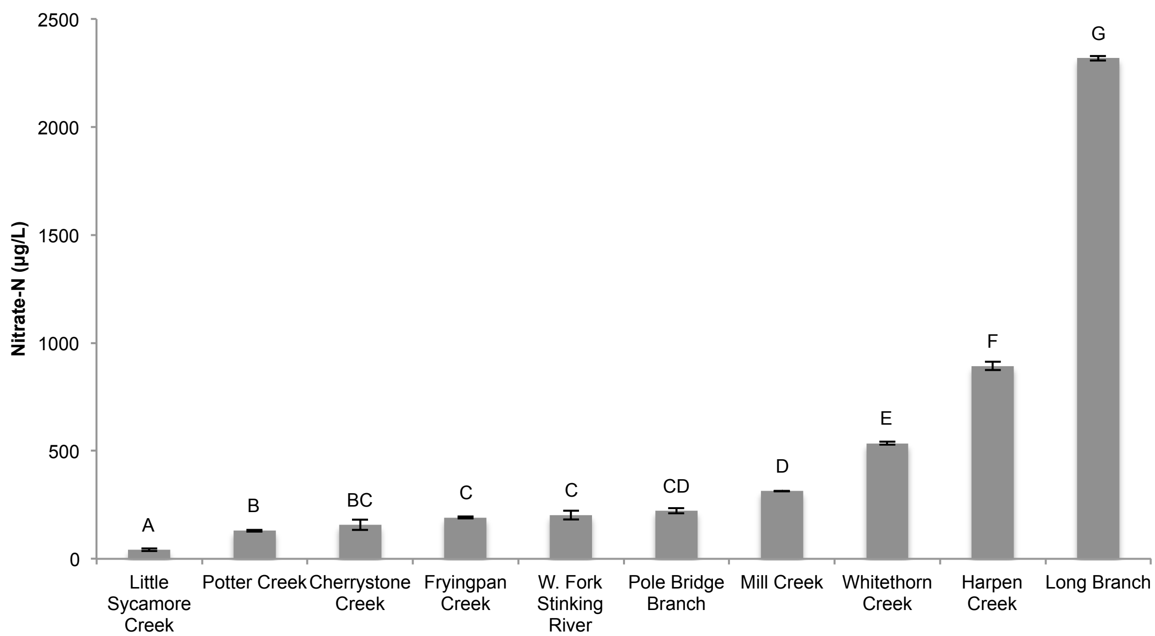

Fig. 11: Nitrate-N concentrations (µg/L) in spring 2013. Error bars represent ±1 SE and letters represent significant differences determined with Tukey's HSD.

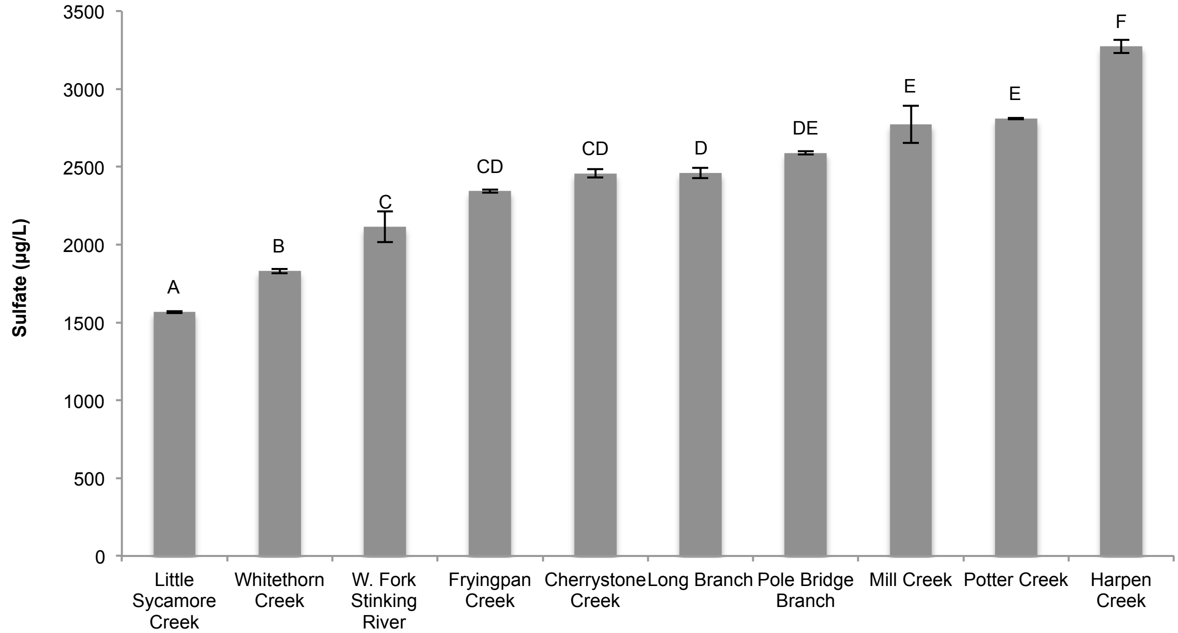

Fig. 12: Sulfate concentrations (µg/L) in spring 2013. Error bars represent ±1 SE and letters represent significant differences determined with Tukey's HSD.

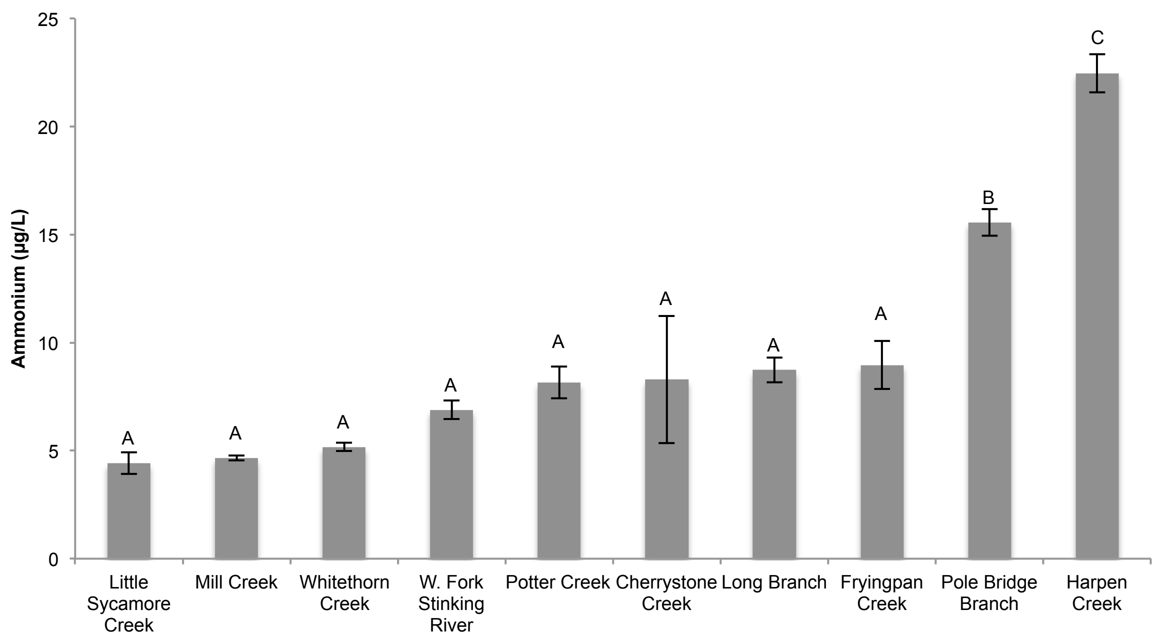

Fig. 13: Ammonium concentrations (µg/L) in spring 2013. Error bars represent ±1 SE and letters represent significant differences determined with Tukey's HSD.

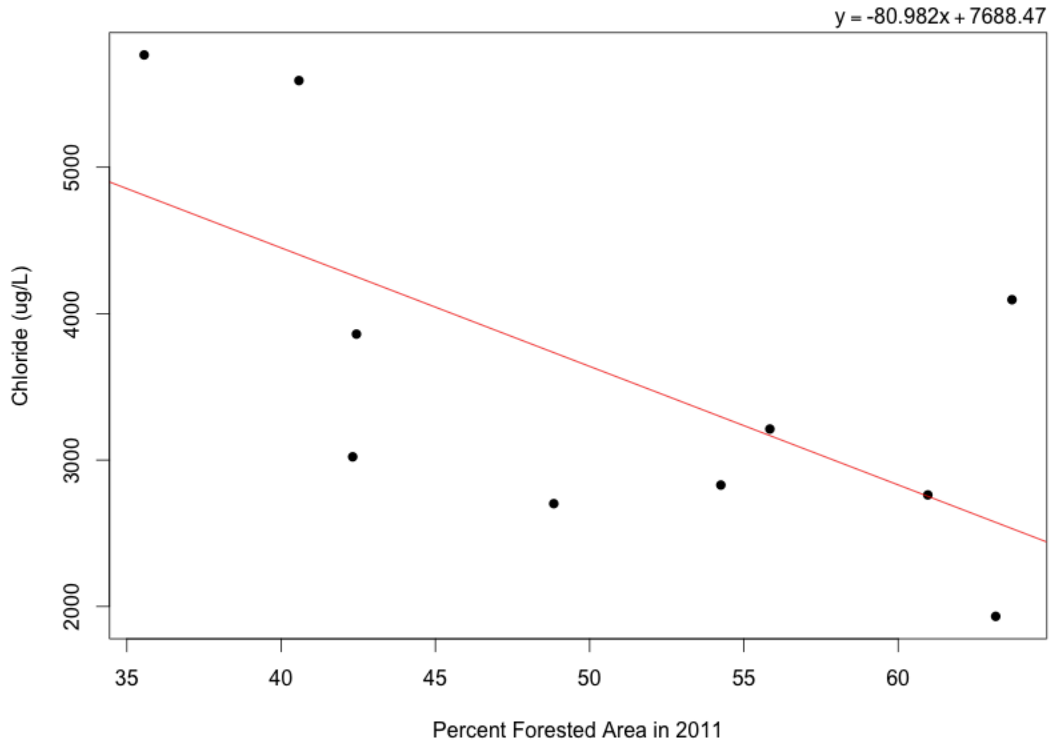

Fig. 14: Regression of the percent forested area in 2011 and chloride concentrations (µg/L) in spring 2013. The solid red line represents the equation of the linear model (given above plot) used to predict chloride concentrations from percent forested area in 2011, r^2=0.43 and p=0.04.

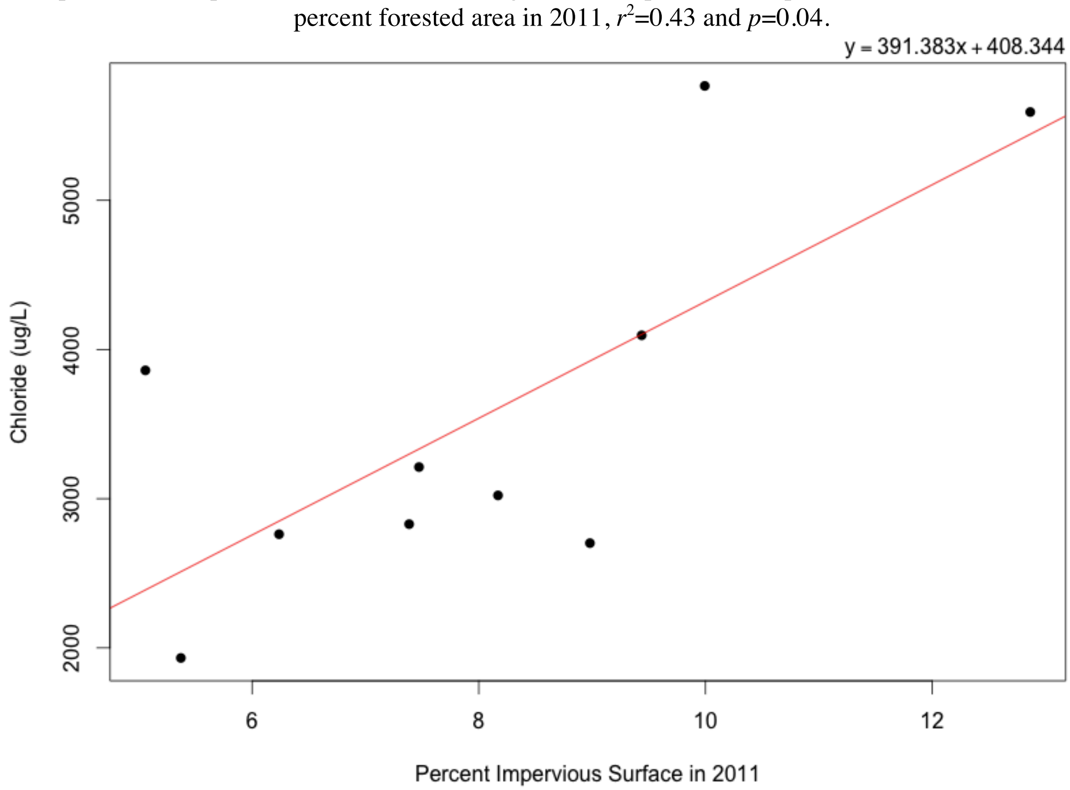

Fig. 15: Regression of the percent impervious surface in 2011 and chloride concentrations (µg/L) in spring 2013. The solid red line represents the equation of the linear model (given above plot) used to predict chloride concentrations from percent impervious surface in 2011, r^2=0.53 and p=0.02.

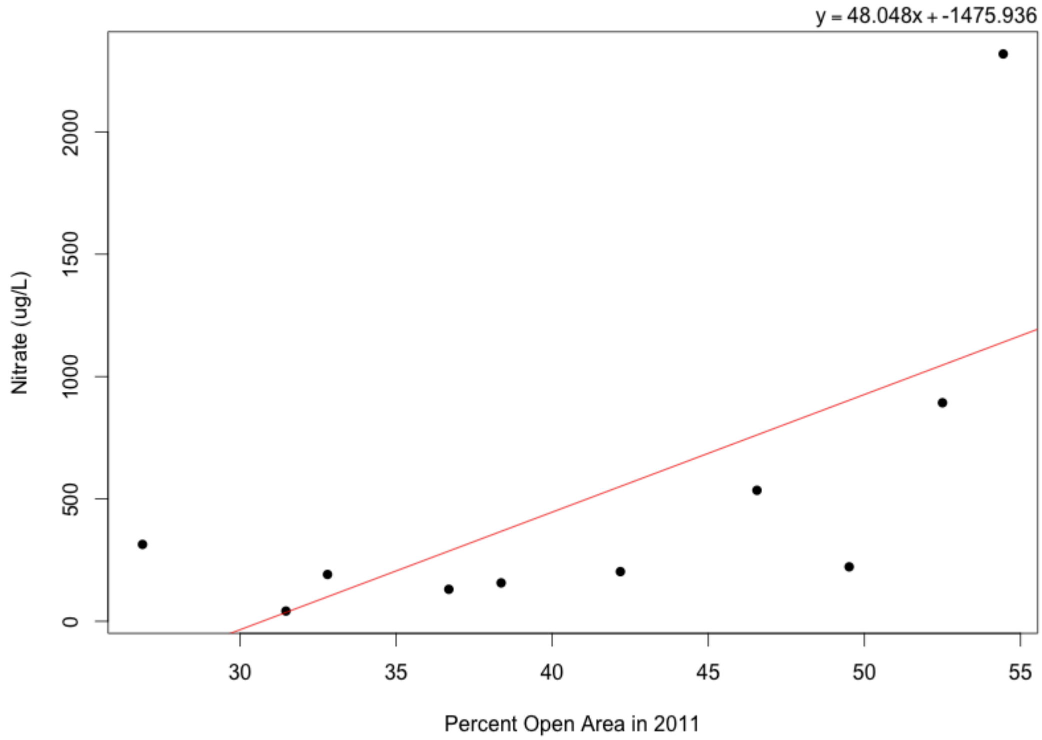

Fig. 16: Regression of the percent open area in 2011 and nitrate-N concentrations (μg/L) in spring 2013. The solid red line represents the equation of the linear model (given above plot) used to predict nitrate concentrations from percent open area in 2011, r^2=0.44 and p=0.04.

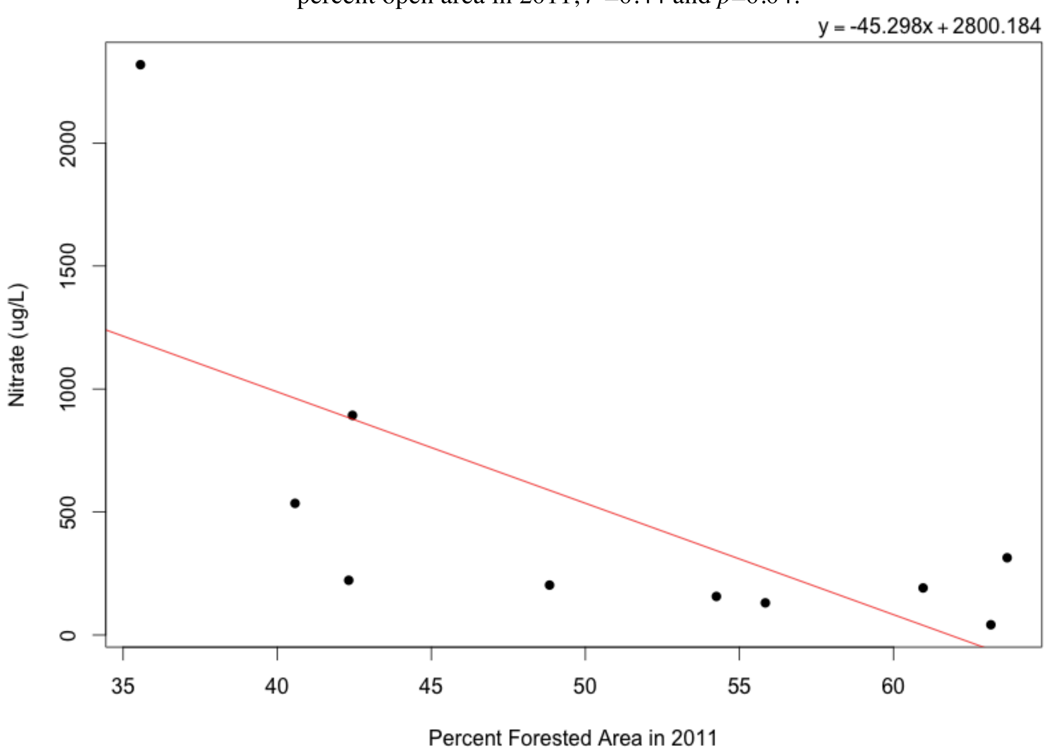

Fig. 17: Regression of the percent forested area in 2011 and nitrate-N concentrations (μg/L) in spring 2013. The solid red line represents the equation of the linear model (given above plot) used to predict nitrate concentrations from percent forested area in 2011, r^2=0.46 and p=0.03.

Table 1: Results of simple linear regressions between land cover percentage and nutrient concentration. Only land cover categories and nutrient concentrations with at least one significant relationship are shown. Numbers reported are the r^2 value followed by the p-value and a (-) to indicate a negative relationship or a (+) to indicate a positive relationship. Values shown represent significance at α=0.05. Dashes indicate no significant relationship.

Land Cover Type	Chloride	Nitrate
Open, 2011	-	0.44, 0.04 (+)
Forested, 2011	0.43, 0.04 (-)	0.46, 0.03 (-)
Impervious, 2011	0.53, 0.02 (+)	-

Macroinvertebrates

A total of 12,791 macroinvertebrates, comprising 97 taxa, were identified. Taxa richness in the quantitative samples ranged from 22 to 40 taxa, and density (no./m^2) ranged from 328 to 1780 (Table 2). The dominant taxon at each site was either Chironomidae or Simuliidae. Simpson's Index of Diversity ranged from 0.62 to 0.84, with larger values indicating greater diversity. Taxa richness in the qualitative samples ranged from 23 to 37 taxa. Percent EPT taxa ranged from 12.8 to 46.0%, and adjusted %EPT taxa ranged from 5.0 to 32.1%. Taxa richness in the quantitative samples was positively correlated with percent impervious cover in 2011 (r^2= 0.45, p=0.03, Fig. 18), but no other macroinvertebrate metric was significantly related to land cover in either time period.

Table 2: Macroinvertebrate metrics for quantitative samples collected spring 2013. Highest values are in bold, lowest values are underlined.

	Density (no./m^2)	Taxa Richness	Simpson's Index of Diversity	%EPT Taxa	Adjusted %EPT Taxa	Dominant Taxon
Cherrystone Creek	1199	33	0.79	34.6	14.7	Simuliidae
Fryingpan Creek	1006	27	0.66	16.7	11.1	Simuliidae
Harpen Creek	**1780**	27	<u>0.62</u>	<u>12.8</u>	<u>5.0</u>	Simuliidae
Little Sycamore Creek	590	26	0.83	36.2	25.6	Chironomidae
Long Branch	<u>328</u>	26	0.68	32.5	15.9	Chironomidae
Mill Creek	844	31	**0.84**	40.1	21.2	Chironomidae
Pole Bridge Branch	933	29	0.78	29.9	13.8	Chironomidae
Potter Creek	864	<u>22</u>	0.67	36.6	**32.1**	Chironomidae
W. Fork Stinking River	736	30	0.80	**46.0**	27.1	Chironomidae
Whitethorn Creek	1681	**40**	0.73	31.5	11.1	Chironomidae

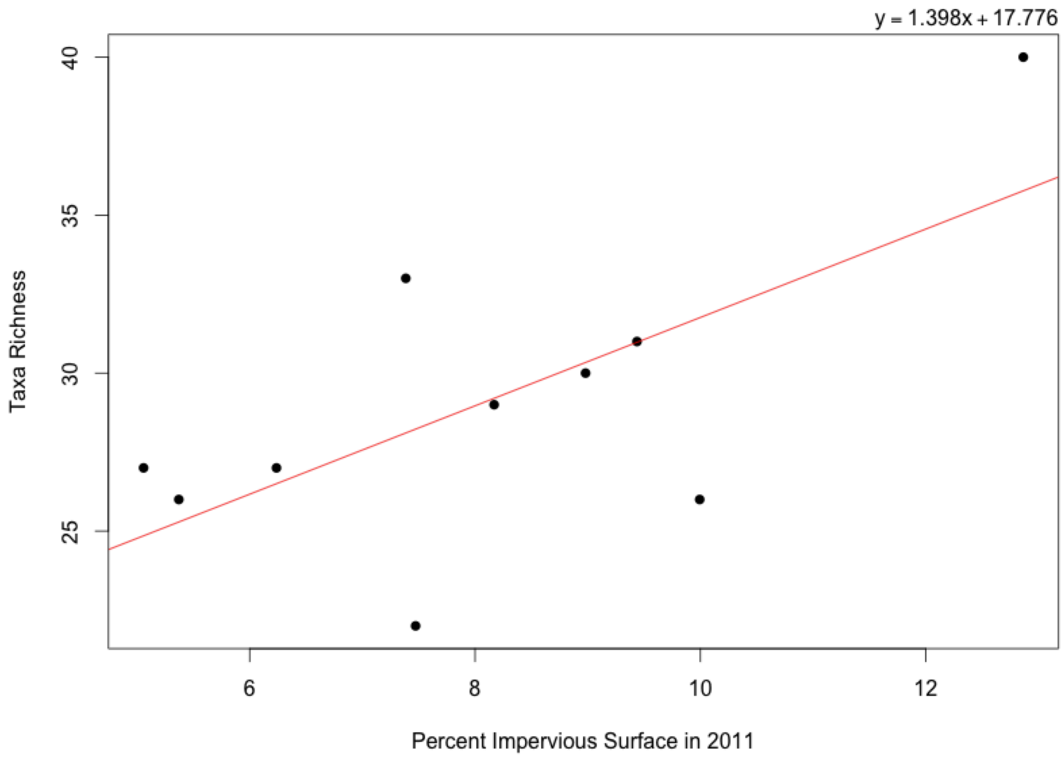

Fig. 18: Regression of the percent impervious surface in 2011 and macroinvertebrate taxa richness. The solid red line represents the equation of the linear model (given above plot) used to predict taxa richness from percent impervious surface in 2011, r^2=0.45 and p=0.03.

Fig. 19A displays community similarity of the quantitative macroinvertebrate samples in multivariate space. PCO axes 1 and 2 explained a combined 51.1% of the variance. PCO axis 1 and percent impervious cover in 2011 were negatively correlated (r^2=0.53, p-value=0.02, Fig. 19B), but no other combinations of PCO axes and land cover percentages were very strongly correlated. No significant correlations were found between PCO axis 1 or PCO axis 2 and dissolved oxygen, conductivity, alkalinity, hardness, chloride, sulfate, ammonium, nitrate or watershed area. PCO axis 1 was most strongly correlated with *Goera* spp. (0.531), *Lanthus* spp. (0.526), Ancylini (-0.970), and unidentified (very small instar) Plecoptera (-0.958). PCO axis 2 was most strongly correlated with *Anchytarsus* spp. (0.652), Tipulidae (0.632), *Hemerodromia* spp. (-0.799), and Leuctridae/Capniidae (-0.639).

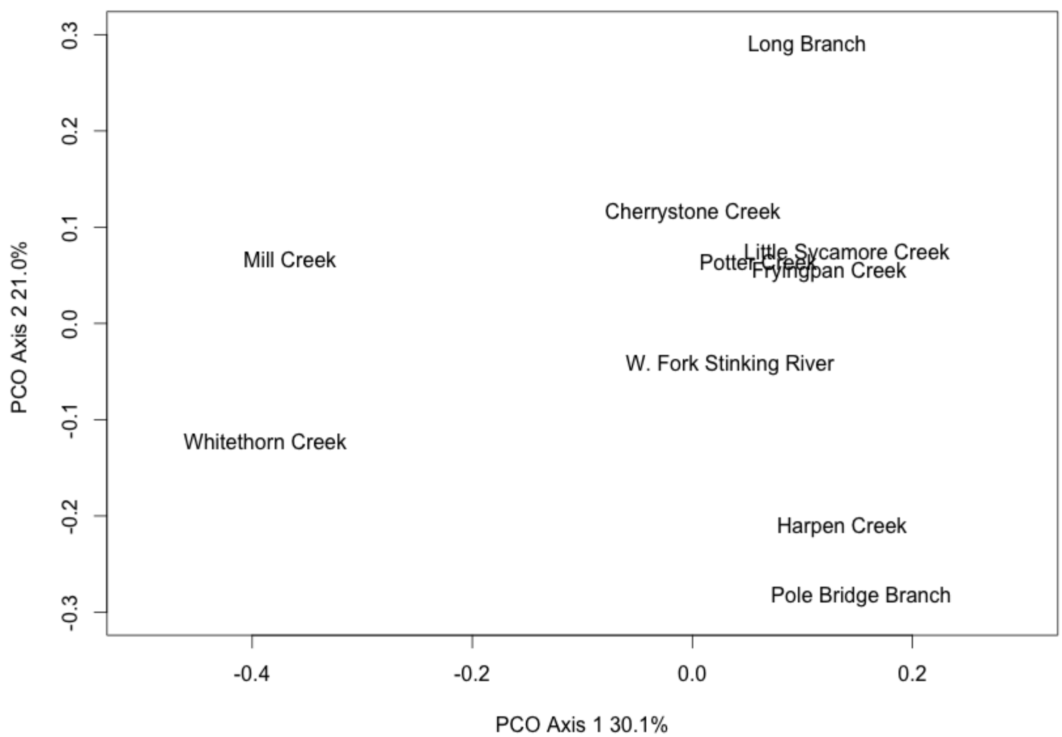

Fig. 19A: PCO of untransformed quantitative macroinvertebrate data using a Jaccard distance metric. PCO Axis 1 explains 30.1% of the variance and PCO axis 2 explains 21.0%.

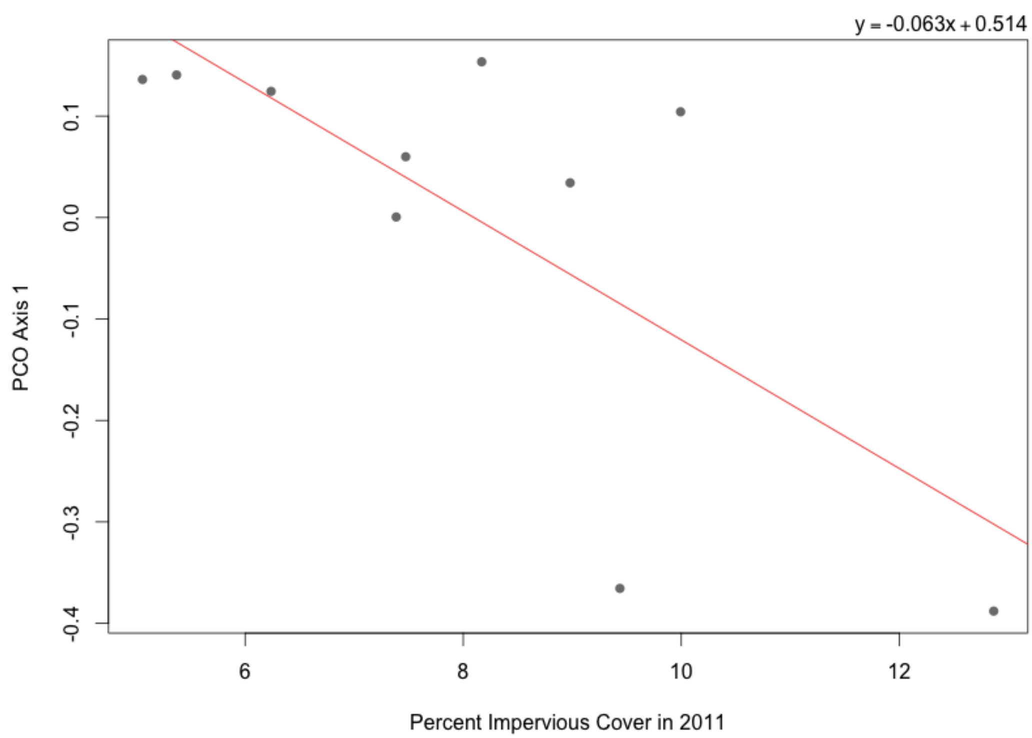

Fig. 19B: Regression of the percent impervious cover in 2011 and PCO axis 1 from Fig. 19A. The solid red line represents the equation of the linear model (given above plot) used to predict PCO axis 1 from percent impervious cover in 2011, r^2=0.53 and p=0.016.

DISCUSSION

Land Cover

The overall increase in impervious surface and decrease in forest cover from 1963-2011 was far from what I expected based on personal observations of current forest cover in the watersheds and what other authors have reported in similar land cover change. For example, there was so little impervious surface in the watersheds studied by Maloney et al. (2008) that they excluded it from their analyses, and the overall percent recovering land cover (including forested/scrub/shrub area) increased by 21% during the 55-year time period over which they evaluated land cover change. All of the watersheds included in a study by Harding et al. (1998) exhibited increases in forest cover in the 30 m riparian zone adjacent to the streams from 1950 to 1990, even those in currently agricultural watersheds. However, these two studies took place in different ecoregions than my study, the Southern plains ecoregion and the Appalachian ecoregion, respectively, which experienced different cultural demands for land use from the 1950s to the late 1990s. The Southern plains experienced a high rate of land cover change from 1950 to 2000, but much of that change was due to repurposing rural land for different agricultural or silvicultural use, and the Southern plains still experienced an overall increase in forest cover during the study period (Brown et al. 2005). The Appalachians (including the Blue Ridge ecoregion) experienced the lowest land cover change rate out of the seven eastern ecoregions studied by Brown et al. (2005), and Loveland et al. (2002) found that the Appalachian region experienced a less than 2% decrease in forest cover and a less than 2% increase in urban cover. The Piedmont ecoregion, however, experienced one of the highest rates of land cover change from 1950-2000, mostly due to urbanization spreading to previously forested areas (Loveland et al. 2002, Brown et al. 2005). Both Loveland et al. (2002) and Brown et al. (2005)

found an increase in urban land cover of about 4.5% from 1973 to 2000, which is very close to the overall increase in impervious surface of 4.8% for the watersheds included in this study.

Water Quality

Temperature and dissolved oxygen measurements fell within ranges found by other authors examining mixed land-use watersheds similar to those in this study (e.g., McDowell and Omernik 1979, Lenat 1984, Goonetilleke et al. 2005). Hardness remained very close to the mean value for Piedmont streams of 23 mg/L $CaCO_3$ established by the Environmental Protection Agency (Harned 1988). Alkalinity measurements were similar to and followed the same seasonal trends by site as the hardness measurements, with no unexpectedly low or high values at any site. Total suspended solids were lower than expected at all sites, and there was a smaller difference than expected among the sites. This deviation from the expected could be because water samples were taken during low discharge, so there was less sediment suspended in the water column.

Conductivity was generally higher than values reported for agricultural streams by Lenat (1984) and Lenat and Crawford (1994), which ranged from 18 to 68 μS/cm, but fell into the range of those reported for urban streams by Lenat (1994) and Goonetilleke et al. (2005), which ranged from 85 to 263 μS/cm. Sulfate concentration was well within the limits established for Virginia drinking water (State Water Control Board 2011), but the amounts measured may be due to the use of sulfur-containing agricultural fertilizers (Binford 2006).

Chloride concentration at all but one site was higher than the nationwide range of 0-2500 μg/L for unpolluted waters (Richards et al. 2010). The positive correlation between chloride and impervious surface in 2011 and the negative correlation between chloride and forested area in 2011 are both expected and logical: as impervious surface increases, so does the amount of road pollution draining into a stream. The absence of orthophosphate in the streams seems unusual in a semi-agricultural context, but phosphorus may still have been present in the streams in the form of total phosphorus. Nitrate concentration was generally below the national background level except for the two sites with the highest concentration, which seem to be driving both the positive correlation between nitrate concentration and open area in 2011, and the negative correlation between nitrate concentration and forested area in 2011 (Dubrovsky and Hamilton 2010). Though driven by a few sites, these two trends have been reported and validated by authors in this field for decades (e.g., McDowell and Omernik 1979, Carpenter et al. 1998, Allan et al. 2004, Mueller et al. 2005).

Macroinvertebrates

The only significant relationship between the univariate macroinvertebrate metrics and land cover percentages was a positive correlation between taxa richness and percent impervious cover in 2011, which directly contrasts the majority of published work on taxa richness in impaired watersheds (e.g., Dance and Hynes 1980, Lenat and Crawford 1994, Delong and Brusven 1998). Similarly to the correlations between contemporary land cover and nitrate discussed above, this positive correlation seems to be driven by a single point, Whitethorn Creek, which had the greatest amount of impervious surface in 2011 but also the greatest taxa richness. However, there are a few studies that have found similarly puzzling relationships between macroinvertebrates and environmental conditions generally regarded as negative. In a study comparing an undisturbed, forested stream to one recovering from clear-cutting and heavy fertilization, Haefner and Wallace (1981) noted that many taxa that were usually grouped with the most sensitive organisms (Ephemeroptera, Plecoptera, Trichoptera, etc.) were found more frequently and in higher numbers in the impaired stream. Examples of such taxa included ephemerellids, baetids, peltoperlids, nemourids, *Lanthus* spp., *Courdulegaster* spp., hydropsychids, and several other taxa that are more commonly grouped with more tolerant organisms, such as elmids, simuliids, tipulids, and chironomids (Haefner and Wallace 1981). Many of these more tolerant EPT taxa were found in the streams included in this study, as can be seen in the large differences between %EPT and adjusted %EPT values calculated for most of the streams.

The PCO seemed to split the watersheds into two groups: Mill Creek and Whitethorn Creek in the first, and the remaining 8 sites in the second. Thirty percent of the variance in the ordination is explained by the axis that divides the two groups, but the driving force behind that division is not obvious. Regressing the first PCO axis against land cover was informative, as I learned that PCO axis 1 is negatively correlated with percent impervious cover in 2011. Mill Creek and Whitethorn Creek have two of the highest percentages of contemporary impervious cover, so the trend may be that simple.

Conclusions

Based on the land cover changes calculated from 1963-2011, it seems that the streams in this region of the Piedmont are transitioning from agricultural to urban impairment, and the parameters I measured reflect that. Impervious surface increased by 4.8% overall, congruent with the overall increase of 4.5% seen in the rest of the Piedmont. The three contemporary land cover percentages were the best predictors for chloride, nitrate, macroinvertebrate communities, and macroinvertebrate taxa richness, but I found no significant relationships between historical land cover percentages and any variables measured. For the watersheds included in this study, the answer to my initial question is this: 1) macroinvertebrate assemblages are most strongly related to contemporary impervious cover; 2) nutrient concentrations and most strongly related to contemporary land cover in all three categories; and 3) water quality parameters are not strongly related to contemporary or historical land cover.

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APPENDIX A - Annotated List of Figures

Fig. 16: Regression of the percent open area in 2011 and nitrate-N concentrations (μg/L) in spring 2013. The solid red line represents the equation of the linear model (given above plot) used to predict nitrate concentrations from percent open area in 2011, r^2=0.44 and p=0.04.

Fig. 17: Regression of the percent forested area in 2011 and nitrate-N concentrations (μg/L) in spring 2013. The solid red line represents the equation of the linear model (given above plot) used to predict nitrate concentrations from percent forested area in 2011, r^2=0.46 and p=0.03.

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Fig. 19A: PCO of untransformed quantitative macroinvertebrate data using a Jaccard distance metric. PCO Axis 1 explains 30.1% of the variance and PCO axis 2 explains 21.0%.

Fig. 19B: Regression of the percent impervious cover in 2011 and PCO axis 1 from Fig. 19A. The solid red line represents the equation of the linear model (given above plot) used to predict PCO axis 1 from percent impervious cover in 2011, r^2=0.53 and p=0.016.