The effects of an altered forest composition due to white-tailed deer herbivory on the condition of a Pennsylvania headwater stream ecosystem within the Chesapeake Bay Watershed

Perry County, Pennsylvania

By
Kerstin Martin

Submitted in partial fulfillment of honors requirements
for the Department of Environmental Studies

Harold Wingert, Supervisor
Candie Wilderman, Reader
Gregory Howard, Reader

May 6, 2011

ABSTRACT

Two headwater streams in Pennsylvania were evaluated to determine whether an altered forest composition due to selective browsing by white-tailed deer (*Odocoileus virginianus)* impacts first-order streams. The streams are located in two adjacent and similar forests. The forests were evaluated based on species and age composition, stand basal area (SBA), canopy cover, and understory visibility. The forest at Reineman Wildlife Sanctuary (RWS), where hunting has been prohibited since 1971, was found to lack a diverse understory due to deer browsing and is moving towards a forest dominated by low herbaceous species that are not preferred by deer; the forest at State Game Lands 170 (SGL 170) remains diverse in species and age composition. Water chemistry data including temperature, conductivity, dissolved oxygen, percent saturation of dissolved oxygen, pH, nitrates, phosphates, alkalinity, and hardness, were collected for each stream and nitrate concentrations at the RWS stream were found to be twice as high as at the stream at SGL 170 ($p = 0.0001$). Nitrate levels in soil and precipitation samples also support higher concentrations at RWS than at SGL 170. This could be due to the lack of vegetation to take up nutrients. Macroinvertebrate communities were also evaluated: several metrics suggest that RWS may support a slightly more pollution-sensitive and richer community. However, a cluster analysis shows no significant differences between the forests and suggests that time of year has a greater influence on the macroinvertebrates than location does. Similar canopy trees in both forests may still contribute similar organic material to the streams.

TABLE OF CONTENTS

LIST OF TABLES AND FIGURES

INTRODUCTION

The decline of the Chesapeake Bay was first widely acknowledged after Hurricane Agnes hit the United States eastern shore in 1972. The storm deposited more sediment in one week than what the upper bay normally receives over 50 years, subsequently destroying significant amounts of submerged aquatic vegetation (SAV) (Gottlieb and Schweighofer 1995). However, the most important effect of Agnes was to expose the already vulnerable condition of the Bay, which can almost entirely be attributed to human activity. Although there are many regional problems in the bay, the universal damaging factor is hypoxia through eutrophication (Boesch et al. 2001). Eutrophication occurs when an excess of nutrients enters the water and causes more growth than the ecosystem can tolerate; when the higher level of organic materials begin to decompose they require more oxygen than is naturally available, leading to hypoxia and limiting aquatic life (Nixon 1995). As the human population in the Chesapeake Bay watershed area steadily rises, the amount of nutrients entering the water is also increasing (Phillips 2005). The three primary anthropogenic sources of nutrients today are discharge from agricultural runoff, publicly owned treatment works, and air pollution (Boesch et al. 2001).

The land use in the immediate Bay area has a heavy impact on the amount of nutrients entering the Chesapeake Bay, but land management farther north in the watershed also has a significant influence on the health of the bay. Pennsylvania contains about 40% of the watershed's 64,000 square mile area and is largely used for agriculture and forest (Figure 1) (McAfee 2011). Forests are the best land-use known to minimize nutrient and sediment runoff and are therefore critical in protecting the natural relationship between the terrestrial, aquatic, and estuarine ecosystems (Lowrance et al. 1984, Mulholland 1992). Stream and forest ecosystems are so interdependent that they are increasingly accepted as one ecosystem; because of this relationship, changes to the forest riparian zone can directly impact the physical and chemical composition of the stream (Sweeney 1993, Cushing and Allan 2001).

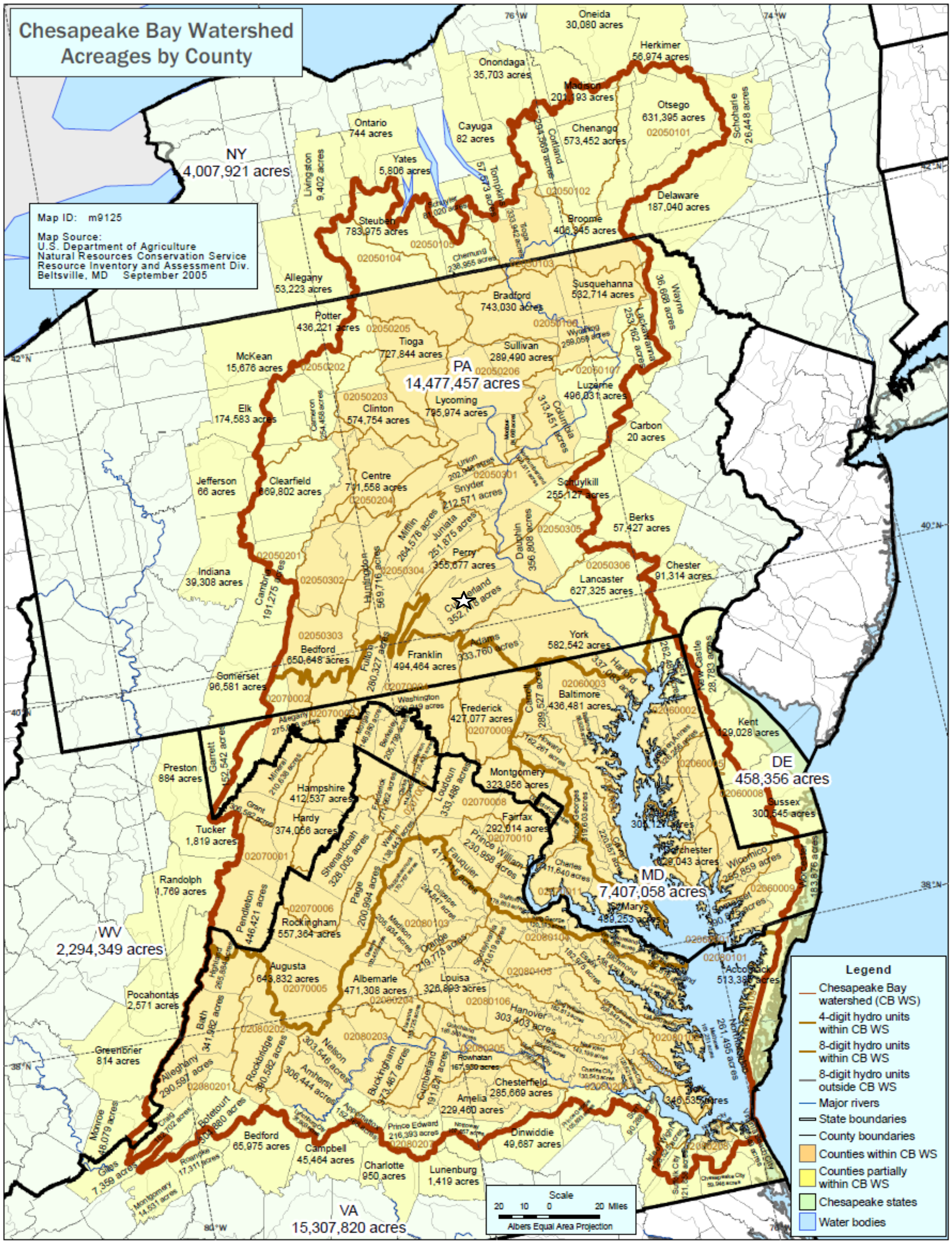

Figure 1. Map of Chesapeake Bay watershed with noted land area.

The River Continuum Concept (RCC) was developed in the 1980s to describe the typical functions of a stream from headwater to mouth and includes the impacts of the surrounding land (Vannote et al. 1980). Starting in a mountainous, forested area, the stream is narrow, fast moving, and well-shaded. Headwater streams are allochthonous, requiring a healthy riparian zone to maintain the stream ecosystem (Cushing and Allan 2001). The riparian zone includes the banks and floodplain vegetation, as well as any vegetation from tall trees or steep stream banks that may enter the stream, even that from beyond the floodplain (Minshall 1994).

One crucial role of riparian zones is to filter nutrients. Forests do this in several ways: tree roots take up nutrients moving through the groundwater before they can reach the stream; roots modify the composition and structure of the soil, which filters the nutrients; above ground plant structures block sediment erosion; canopy trees change the way nutrients are processed in the stream because of shading. Damaged riparian zones have a compromised filtering capacity and can result in an increase in the concentration of nutrients reaching the stream (Sweeney 1993, Lowrance et al. 1997). This changed concentration of nutrients in a stream can affect what kind of macroinvertebrates can live there (Sweeney 1993).

Streamside forests also provide the organic material on which macroinvertebrates are heavily dependent for energy. The macroinvertebrate populations found in this ecosystem are generally composed of insects in the shredder and collector functional feeding groups (Cummins 1974). Shredders are crucial in the breakdown of coarse particulate organic matter (CPOM), including leaves, twigs, bark and other debris, which collect on obstructions in the stream (Cummins et al. 1989). Shredders include most plecoptera (stoneflies), some tricoptera (caddisflies), and others such as tipulidae (crane fly, order dipteria), and usually compose about 35% of the macroinvertebrate community. Collectors feed on the resulting fine particulate organic matter (FPOM) and include most mayflies and some caddisflies. These compose about 45% of the community. The remaining feeding groups include predators (about 15%) and scrapers (about 5%). Because forests are so crucial to the existence of these macroinvertebrates, a significant change in forest composition could alter the population (Cushing and Allan 2001).

There are other functions of the forest that are necessary to the aquatic insect community. This includes temperature regulation: meadow streams, for example, have been found to be 3-5°C warmer than forest streams for large portions of the year (Welsch 1991). Changes in the forest, such as an inability to fill canopy gaps, might allow more sunlight to reach the water and in turn affect biotic community (Sweeney 1993). In addition, it has been found that the tree roots provide habitat for some species of macroinvertebrates (Rhodes and Hubert 1991).

Because riparian zones are such critical ecosystems, the Pennsylvania Department of Conservation and Natural Resources (DCNR) updated its State Forest Resource Management Plan in 2007 to include an extensive guideline for managing them. These guidelines are useful for public lands, but private landowners are not required to adhere to them (DCNR 2007).

Furthermore, while factors such as increased private development and the timber industry impact the health of Pennsylvania's riparian zones, high populations of white-tailed deer (*Odocoileus virginianus)* also affect both public and private forested headwater streams. It is well documented that high deer populations change the composition of forests (Hough 1965, Tilghman 1989; Fergus 2000; Horsley at al. 2003; Petersen and Wallis 2004). The understory, including seedlings, saplings, and pole trees, is critical to the regeneration of a forest. In a healthy forest, the underlying trees compete to fill canopy gaps quickly (Perry 1994, Pedersen and Wallis 2004). However, deer selectively browse on underbrush, seedlings, and saplings, which can prevent those species from regenerating (Horsley et al. 2003). Table I shows some browsing preferences of white-tailed deer.

Common Name	Browsing preference (spring/summer)	Browsing preference (fall/winter)
Blackberry (herbaceous)	high	high
Ferns	not preferred	not preferred
Ash, White	low/moderate	high
Beech, American	low	high
Birch, Black	low/moderate	high (late fall)
Birch, Yellow	low/moderate	high (late fall)
Dogwood	moderate	high
Gum, Black	high	high
Hawthorn	low	high
Hickory	low	low
Maple, Red	low/moderate	high
Maple, Striped	low	low
Maple, Sugar	low/moderate	moderate
Oak, Chestnut	moderate	high
Oak, Black	moderate	high
Oak, Red	moderate	high
Oak, White	moderate	high
Pine, White	low	moderate
Sassafras	moderate	high
Tulip Magnolia	high	high

Table I. List of deer preferences of some species (modified from Table 3 in Horsley et al. 2003 and from Lantham et al. 2005).

In addition to the impacts of browsing, herbaceous species not preferred by deer, such as Asian stilt grass, compete for sunlight with the native seedlings that remain (Pedersen and Wallis 2004). These two factors lower tree species diversity, which causes the forest to be less resilient to stress and decreases habitat diversity (Horsley et al. 2003). One study also found that deer herbivory actually increased on plants growing in soils with higher nitrogen levels (Tripler 2002). This could create a positive feedback loop where the lowered filtering capacity of a damaged forest may attract more browsing. Overall, the relatively new development of increased white-tailed deer populations negatively impacts the health of forests.

White-tailed deer populations in Pennsylvania have fluctuated ever since colonists eradicated their natural predators, such as wolves and mountain lions, from within the state. In the early 1900s, the Pennsylvania Game Commission (PGC) began stocking deer and regulating the hunting seasons (PCG 2003). The deer populations quickly recovered without the natural predators and because land development such as agriculture and suburbs provided ideal forest-edge habitat (Horsley et al. 2003). As of 2001, the PGC enforces antlered/antlerless seasons in September through January, depending on the firearm (PCG 2003). In an interview, District Forester for Michaux State Forest Mike Kusko noted that deer populations are never evenly spread everywhere and that herds are very localized, which can make monitoring difficult (Kusko 2009). For this reason additional permits are distributed based on Deer Management Assistance Programs (DMAPs), which the DCNR began using in 2003 (DCNR 2007, Kusko 2009). This voluntary program helps both public and private landowners track deer populations and determine whether more hunting is necessary.

Hunting is the state's primary method of deer management because it is thought to be most effective, efficient, and least expensive (PCG 2003); however, other strategies such as deer exclosures are sometimes also employed (Latham et al. 2005). State forests must balance the sustainability of the forest and wildlife resources with the interests of the public and the state: in 2001, hunting generated over $500 million in Pennsylvania (PCG 2003). In addition to the complexity of managing public lands, private landowners independently choose whether or not to permit hunting on their properties. The variables in different land ownership often present difficulties in managing deer populations (Witmer and DeCalesta 1991).

It is estimated that only about 23% of the forests are state-owned by divisions such as DCNR and the PGC and 4% are federally owned, mainly by the National Forest Service in Allegheny National Forest. Seventy-one percent of forests are owned privately or non-corporately (McWilliams et al. 2004). While state-owned forests have written guidelines for management outlined by the DCNR, private owners generally do not. In fact, only 8 % of private landowners have sought professional management advice and only 2% have written management plans (McWilliams et al. 2004).

As a result of current deer management strategies, deer populations now average 7.7 to 14.8 deer per km^2 across the state, but it is not uncommon for densities to be well over 60 deer per km^2 (Porter et al. 1994, deCalesta and Stout 1997, Diefenbach et al. 1997). These densities are generally higher than the calculated and observed ecological carrying capacity of white-tailed deer: Tilghman (1989) found that forest vegetation began to decline at densities ranging from 4 to 6 deer per km^2. The 2005 *Report by the Deer Management Forum* for Audubon Pennsylvania and Pennsylvania Habitat Alliance suggested 7.7 deer per

km^2 as an initial goal for reducing deer populations (Latham et al. 2005); based on Native American deer harvests, 7.7 deer per km^2 is the estimated pre-colonist deer density (USDA). The need for more stringent deer management has been known for decades, but little has changed (Latham et al. 2005).

It is clear that productive forests in Pennsylvania are a necessary component of the Chesapeake Bay's health. However, there have been few studies on whether forests with altered compositions due to deer browsing have an impact on the streams running through them. This study seeks to link the relationship between deer management and forest health, and the impact on headwater streams. This study is important to groups such as the Chesapeake Bay Foundation, the Chesapeake Bay Program, and the Alliance for the Chesapeake Bay, who seek better watershed management upstream of the bay (Blankenship 2006). If the Chesapeake Bay watershed is impacted by Pennsylvania's deer management, the argument to change management practices becomes more compelling. This study would also aid groups such as Audubon Pennsylvania, the Pennsylvania Game Commission (PGC), the Department of Natural Resource Conservation (DCNR), all of which have an interest in managing Pennsylvania's resources to best benefit the ecosystem and people (Latham et al. 2005, PGC 2003, Benner 2006). Conversely, there are also groups that wish to allow deer populations to remain at current populations or even increase. These include some hunters, who believe deer populations are not high enough to sustain an enjoyable hunting season (Diefenbach et al. 1997). The economic benefit of hunting in Pennsylvania may also influence some departments such as the PGC in advocating for high deer populations.

METHODS AND MATERIALS

Study area

Two sites in Perry County, Pennsylvania were studied (Figure 2).

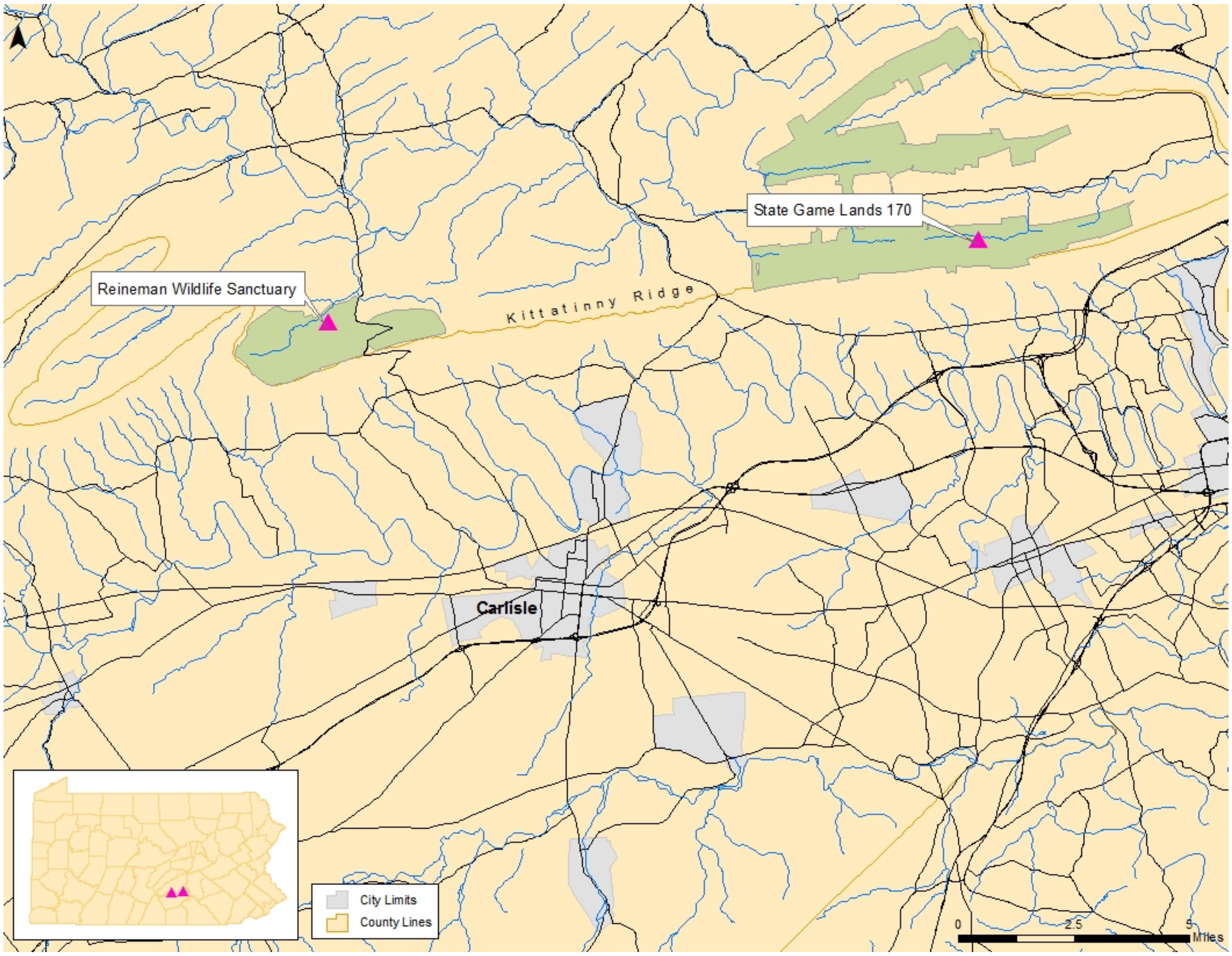

Figure 2. Location of RWS and SGL 170.

The sites were chosen because the geology, climate, and forest type are nearly identical. Both are oak-mixed hardwood forests and have headwater streams running through them; Trout Run flows through SGL 170 and an unnamed stream runs through RWS. The watershed above the study site at RWS is 1.4 km^2 (282.7 acres), while the watershed above the SGL 170 study site is 2.3 km^2 (532.9 acres). The streams have shale/sandstone streambeds and the study sites are at an elevation of approximately 200-220m. The stream at

RWS has a slope ~3% and SGL 170 has a slope ~0.5% (Figure 3).

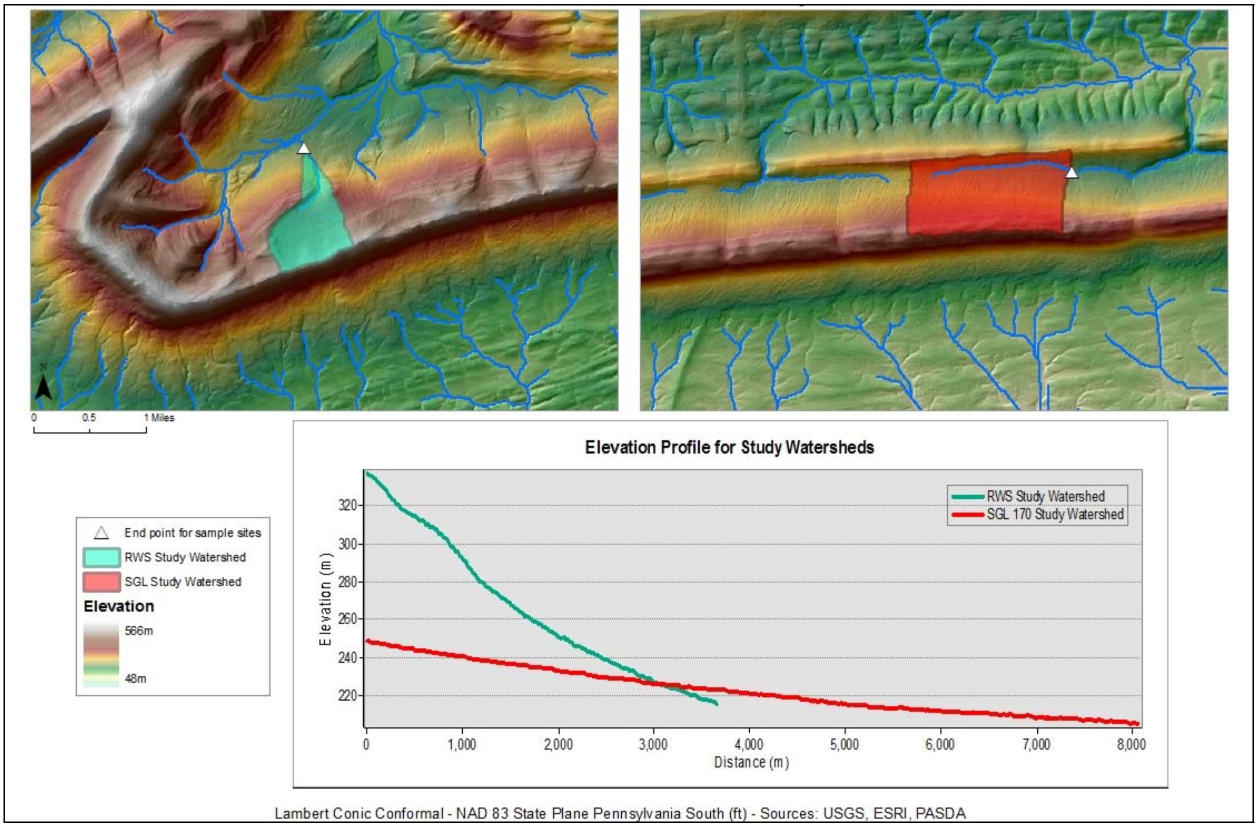

Figure 3. Elevation profile and model for RWS and SGL 170 study watersheds.

Deer hunting is prohibited at RWS while SGL 170 is actively hunted in accordance with the Pennsylvania Game Commission designated seasons. In 2004, the deer population density at RWS was around 1500 deer/km^2 and the population density at SGL 170 is approximately 200 deer/ km^2; these numbers may be higher because the samples were taken around a field (Pedersen and Wallis 2004). The single variable of deer management practices allows for a comparison of the effects of white-tailed deer on the forests and the possible subsequent effects on the streams.

Field and laboratory methods

At each location, forest data were collected to confirm the visible difference in species and age composition. In 2009, five 200m^2 transects of forested land running perpendicular to the respective streams were measured. Three random transects were measured on one side of the stream and two on the other; each individual transect measured 100m x 2m. In 2010, two 200m^2 transects were measured at each location. Within these transects we counted the trees, categorized them by age (seedling, sapling, pole, and canopy trees), and identified the species. For the 2009 transects we also measured diameter at breast height (DBH), canopy cover, and understory visibility using the instruments listed in Table II. The results of the five transects from 2009 and the two transects from 2010 were composited in the results.

Measurement	Instrument or Method	Brand
tree species identification	dichotomous guide	prepared by Gene Wingert
DBH	DBH tape	Forestry Suppliers, Inc. 8m tape
canopy cover	densitometer	Geographic Resources Solutions
understory visibility	spot board	developed by Gene Wingert
forest transect length	100m measuring tape	Keson Ultra Glass 165 ft. tape
forest transect width	two 1m sticks	standard wooden meter stick

Table II. Methods and instruments used for forest transect measurements.

In fall 2009, understory visibility was measured to quantify the effect of deer browsing on understory vegetation. We measured visibility with a spot board. A spot board is a $1m^2$ piece of plywood painted red with nine evenly spaced yellow circles, 15cm in diameter (Figure 4).

Figure 4. Using the spot board method at SGL 170. Source: Gabrielle Ostermayer.

The spot board was held at 10m intervals along the 100m transects and was moved laterally 1m for each interval (Figure 5).

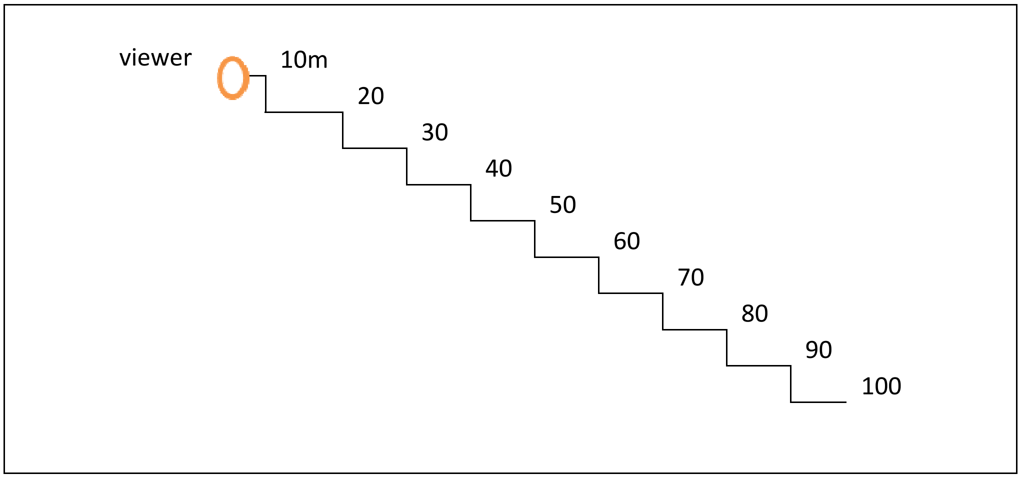

Figure 5. Diagram of spot board method (plan view).

Two points are given for every fully visible spot, one for each partially visible spot, and none for invisible spots. Percent visibility is then calculated as:

$$\frac{\text{total \# points}}{18} \times 100$$

Canopy cover was measured using a densitometer. The densitometer allows light to filter through the eye piece: along five 100m transects, approximately 50 observations about whether the sky was visible or not were made. The observations were composited and the proportion of observations of sky was compared to those where sky was not visible.

Types of underbrush, which may be affected by deer browsing, were also noted within two 100mx 2m transects in 2009 and in 2010. Quantities were not measured because of limited time in 2009. In 2010 a quantitative study on the underbrush based on presence was completed.

In fall 2010, leaf litter input was measured using 3 baskets for vertical leaf fall collection and 3 baskets for lateral leaf movement collection for each location. Leaves were collected, rinsed, and dried at 50℃ for 48hours. This study compared the masses of leaf litter input into each stream to measure whether there is a difference in how much organic matter the streams are receiving. The study was interrupted because the baskets were stolen before significant data could be collected, so the results and discussion have been omitted for this paper.

Fecal coliform bacteria tests were done on stream water on 3 occasions within a 3-week span in November 2010. Membrane filtration analysis was used and bacteria were placed in the incubator for 24 hours at 44.5℃ (Rose et al 1974).

Water chemistry data were collected at both creeks on seven occasions at three varying sites. In the field we measured temperature, dissolved oxygen (DO), conductivity, and pH and collected 1L water samples (Table III). In the lab we tested nitrates, phosphates, hardness, and alkalinity levels using the methods and instruments listed in Table III. Excluding nitrate and phosphate concentrations, not all parameters were measured on all dates due to instrument malfunction and time. On one occasion, precipitation was collected before entering the soil and was tested for phosphate and nitrate concentration.

Measurement	Instrument or Method
Temperature	YSI 550A
Conductivity	YSI 30
Dissolved oxygen	YSI 550A
pH	LaMotte Chemical pH Meter
Alkalinity	Lamotte 4491-DR
Hardness	Hach Method 8213 using digital titrators
Nitrogen	Hach DR 2800 Portable Spectrometer
Phosphorus	Hach DR 2800 Portable Spectrometer

Table III. Instruments and methods used for water chemistry measurements.

On two occasions (December 2010 and March 2011) soil samples were taken from within 5m of the stream bank. These were sent to the Penn State Cooperative Extension for Cumberland County for analysis of pH, acidity, nitrogen, and phosphorus.

Macroinvertebrates are biotic indicators of the health of a stream (Hynes 1970). Macroinvertebrates were collected at three varying locations along each stream. They were collected using a D-net or Surber Sampler net and preserved in isopropanol. Macroinvertebrates were classified to the family level using Voshell, 2002 and McCafferty, 1981. Ten metrics were chosen to analyze the data. Table IV shows the metrics used and their relevance in evaluating macroinvertebrate communities.

Metric	Notes	Importance
Taxa richness	Variety of taxa found.	A higher variety indicates a more stable habitat.
Hilsenhoff Biotic Index	Tolerance to organic pollution (scale 0-10, with 0 least tolerant) (Hilsenhoff 1983)	Indicates presence of organic pollution
Modified EPT index	Number of Ephemeroptera, Plecoptera, and Tricoptera present, with Hilsenhoff values ≤3 (Voshell 2002)	Indicates pollution sensitivity, scores range 0-10 with 0 being most sensitive.
% Modified Mayflies	Percent of mayflies compared with the community, with Hilsenhoff values ≤3.	Mayflies are especially sensitive to organic pollution and acidification- indicates these disturbances.
% Dominant Family	Percentage of individuals in the dominant family compared with the community.	Indicates whether any one species is dominant; could indicate disturbance to ecosystem.
% Shredders	Percentage of individuals in shredder feeding group to the community.	Can indicate whether an expected amount of CPOM is processed in the stream.
Simpson Index	$D = \sum \frac{n(n-1)}{N(N-1)}$, where n is total number of organisms of a particular family, and N is the total number of organisms of all families. (Hauer and Lamberti 2007).	Indicates whether the population is evenly distributed among the present taxa.
Shannon-Wiener Index	$H = -\sum P_i(\ln P_i)$, where P_i is the proportion of each species in the sample (Hauer and Lamberti 2007).	Indicates evenness and richness.
Sorensen's coefficient of community	CC = 2c / (a+b), where c is the number of species shared by both locations and a and b are the number of species at each location (Greig-Smith 1983).	Compares two different locations in terms of richness.

Table IV. Metrics used to evaluate macroinvertebrates and their purposes.

A cluster analysis using Ward's method was also run using PASW Statistic 18 from IBM. Ward's method bases the distance between two clusters on how much the sum of squares increases when the two clusters are combined (Punj and Stewart 1983). The macroinvertebrates were sorted by date, collection site, and stream.

RESULTS AND DISCUSSION

Forest Data

Because the environmental characteristics other than deer population are similar between the forests at RWS and SGL 170, a positive relationship between high deer populations and altered forest composition is supported by the data (Table V, Figure 6, Table VI, Figure 7); this is also supported by the literature. Red maple, which is not preferred by white-tailed deer, had the highest seedling count (423 individuals) at RWS. In addition to being a less preferred species, deer browsing of other species reduces the competition for resources for the red maples. However, the lower counts of sapling and pole red maples indicate that they are eventually browsed or outcompeted by herbaceous undergrowth. Most of the preferred species (American beech, black birch, and striped maple) at RWS had lower total counts for seedlings, saplings, and poles compared with SGL 170.

Species	Seedling	Sapling	Pole	Canopy	Total
Ash, White	0	1	4	2	7
Beech, American	10	5	4	3	22
Birch, Black	2	2	0	4	8
Birch, Yellow	0	5	2	1	8
Cherry, Black	1	0	0	0	1
Dogwood	0	0	1	0	1
Gum, Black	0	1	0	0	1
Hackberry	0	0	0	0	0
Hawthorn	29	1	0	0	30
Hickory	5	0	2	1	8
Hickory, Shagbark	0	0	1	0	1
Maple, Red	423	1	0	4	428
Maple, Striped	0	0	0	0	0
Maple, Sugar	1	4	0	0	5
Oak, Chestnut	2	7	1	6	16
Oak, Black	0	1	0	2	3
Oak, Red	0	0	0	0	0
Oak, White	0	0	2	4	6
Pine, White	0	0	0	2	2
Sassafras	0	0	0	0	0
Tulip Magnolia	11	3	1	1	16
Walnut, Black	1	0	1	2	4

Table V. Composite species and age counts for transects at RWS 2009, 2010.

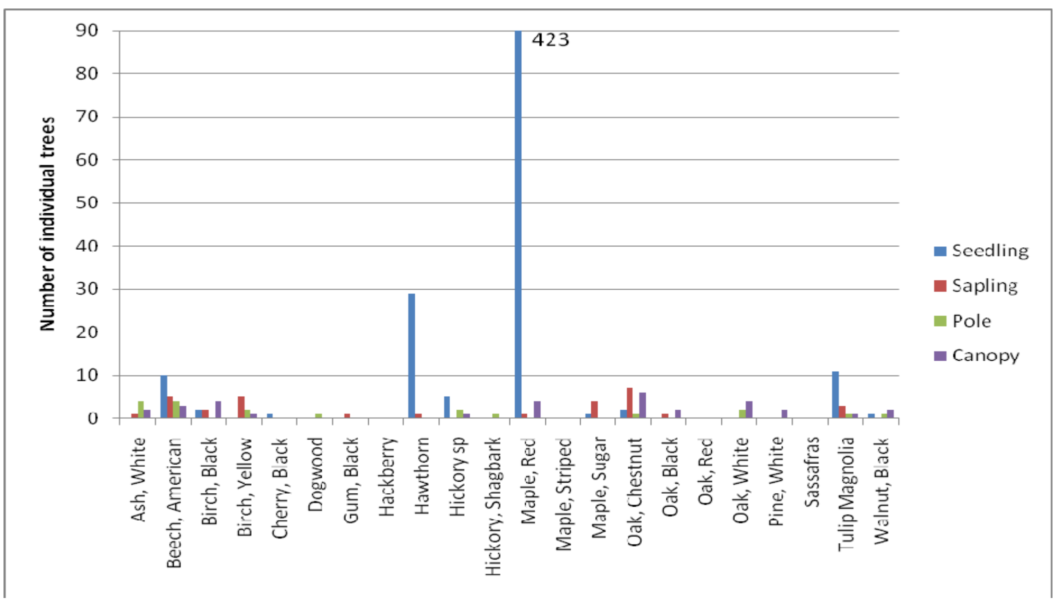

Figure 6. Composite species and age counts for transects at RWS 2009, 2010.

Species	Seedling	Sapling	Pole	Canopy	Total
Ash, White	1	14	0	0	15
Beech, American	8	82	3	1	94
Birch, Black	4	61	12	11	88
Birch, Yellow	0	4	0	0	4
Cherry, Black	0	0	0	0	0
Dogwood	0	2	0	0	2
Gum, Black	4	49	1	0	54
Hackberry	0	1	0	0	1
Hawthorn	5	1	1	0	7
Hickory sp	14	53	3	1	71
Hickory, Shagbark	0	0	0	0	0
Maple, Red	50	65	6	3	124
Maple, Striped	3	11	0	0	14
Maple, Sugar	0	0	0	0	0
Oak, Chestnut	0	21	0	0	21
Oak, Black	1	78	0	0	79
Oak, Red	13	46	2	2	63
Oak, White	7	72	1	3	83
Pine, White	33	8	0	0	41
Sassafras	0	5	0	0	5
Tulip Magnolia	1	14	6	6	27
Walnut, Black	6	0	2	0	8

Table VI. Composite species and age counts for transects at SGL 170 2009, 2010.

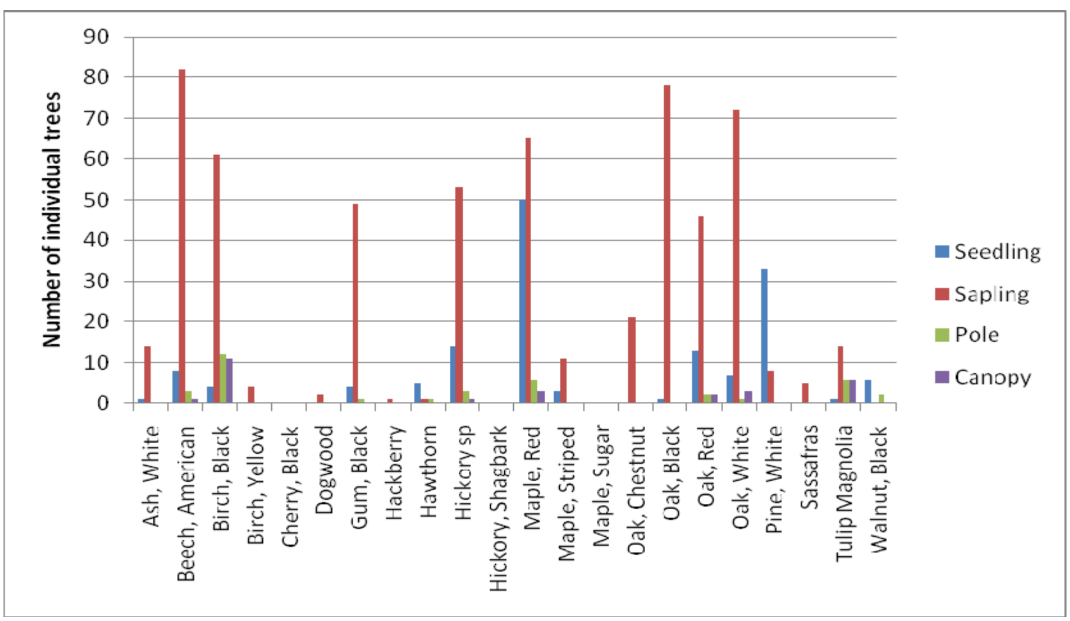

Figure 7. Composite species and age counts for transects at SGL 170 2009, 2010.

The overall numbers of trees within each age category show a changing forest dynamic. The sites at RWS and SGL 170 show similar numbers of canopy trees, 32 and 27 respectively (Table VII, Figure 8).

Site	Seedling	Sapling	Pole	Canopy
RWS	485	31	19	32
SGL 170	150	587	37	27

Table VII. Total number of trees within each age category at each site.

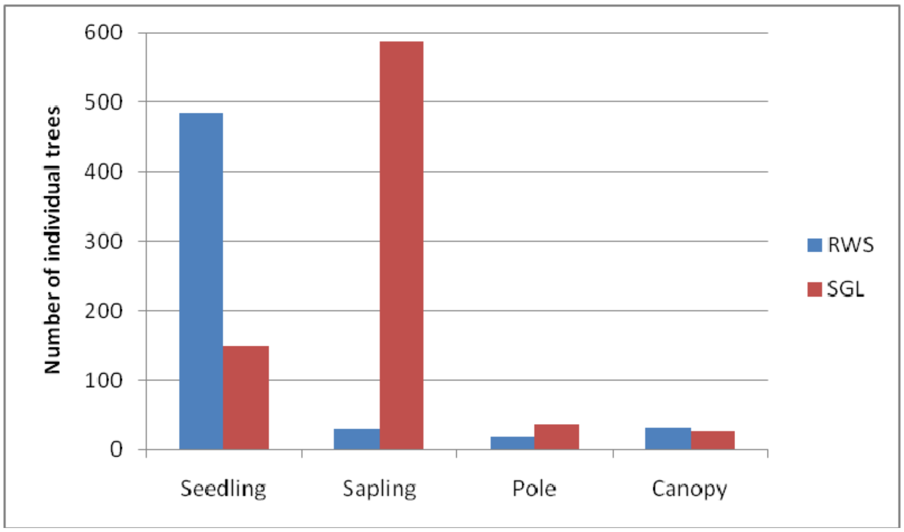

Figure 8. Total number of trees within each age category at each site for 2009 and 2010.

Stand basal area (SBA) was calculated for all canopy trees for both sites for the 2009 data (Table VIII). It is possible that logging took place at SGL 170, which may explain there are fewer canopy trees and why they are less developed.

Site	# Canopy Trees	Stand basal area (m^2/ha)
RWS	21	3.08
SGL 170	18	1.74

Table VIII. Number of canopy trees and the SBA at each site (2009).

Using chi-square statistics, p-values were calculated for age categories and species richness. Both tests show significant evidence that the differences between the forests at SGL 170 and RWS were greater than the differences within each forest ($\alpha = 0.05$) (Table IX).

Test	X^2	df	p-value
Species richness	645.1	43	<0.00
Age diversity	662.5	7	<0.00

Table IX. Species richness and age category diversity between RWS and SGL 170 (2009 and 2010 composite).

The spot board test, done in October 2009, confirms the lack of understory at RWS compared with SGL 170. Table X and Figure 9 show the percent visibility at intervals of 10m. Both the RWS spot board transects had some visibility throughout the entire 100m (44.44% and 66.67% visibility), while visibility at both transects at SGL 170 ceased at 40m. The lacking understory will hinder replacement of the canopy.

Distance (m)	RWS Site 1	RWS Site 2	SGL 170 Site 3	SGL 170 Site 4
10	100.0	100.00	61.11	88.89
20	94.44	100.00	61.11	38.89
30	77.78	100.00	11.11	11.11
40	100.00	100.00	0.00	0.00
50	94.44	50.00	0.00	0.00
60	77.78	50.00	0.00	0.00
70	55.56	66.67	0.00	0.00
80	66.67	33.33	0.00	0.00
90	61.11	33.33	0.00	0.00
100	44.44	66.67	0.00	0.00

Table X. Percent visibility using spot board at 10m intervals for two transects at each location.

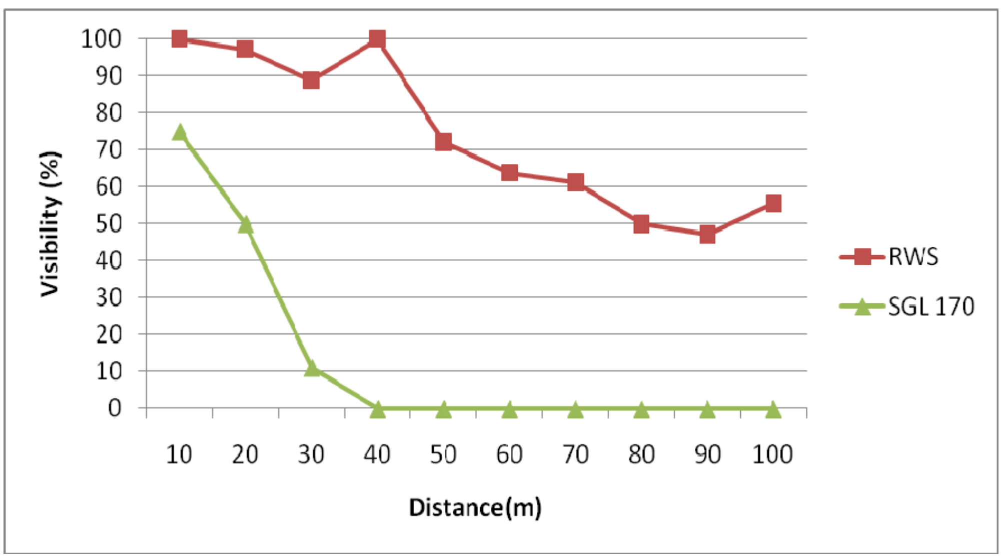

Figure 9. Percent visibility at 10m intervals at each study site.

Table XI shows the percent canopy cover for both sites in 2009. SGL 170 had an average of 95.70% canopy cover, while RWS had an average of 87.91%. These results demonstrate that canopy gaps are filled more quickly at SGL 170 than at RWS.

Sample Site	RWS (%)	SGL 170 (%)
1	95.16	n/a
2	87.50	100.00
3	62.16	100.00
4	94.74	90.70
5	100.00	92.11
Average	87.91	95.70
Difference	7.79	

Table XI. Percent canopy cover for both sites.

The herbaceous underbrush found at RWS and SGL 170 in 2009 is listed in

Species	Presence	
	SGL	RWS
Arrowwood (*Viburnum dentatum*)	x	
Asian barberry (*Berberis vulgaris*)		x
Asian stilt grass (*Microstegium vimineum*)*		x
Asian wine berry (*Rubus phoenicolasius*)		x
Blackberry (*Rubus fruticosus*)	x	
Blueberry (*Vaccinium* genus)	x	
Cinnamon fern (*Osmunda cinnamomea*)	x	
Dewberry (*Rubus caesius*)	x	
Green briar (*Smilax rotundifoli*)	x	
Hay-scented fern (*Dennstaedtia punctilobula*)*	x	x
Huckleberry (*Gaylussacia baccata*)	x	
Indian pipe *(Monotropa uniflora)**	x	
Jack-in-the-pulpit (*Arisaema triphyllum*)		
Partridge berry (*Mitchella Repens)*	x	
Pink Lady's Slippers (*Cypripedium acaule*)	x	
Royal fern (*Osmunda regalis)*	x	
Sarsaparilla (*Aralia nudicaulis*)	x	
Solomon's Seal (*Polygonatum*)	x	
Spice bush (*Lindera benzoin*)	x	
Stinging nettle (*Urtica dioica*)		x
Swamp azalea (*Rhododendron viscosum*)	x	
Teaberry (*Gaultheria procumbens*)	x	
Viburnum (*Acer folium*)	x	
Virginia Creeper (*Parthenocissus quinquefolia*)*	x	
Virginia Cucumber Root (*Medeola virginiana*)	x	
White Snake Root (*Eupatorium rugosum*)*		x
Winterberry (*Ilex verticillata*)		x
Witchhazel (*Hamamelis virginiana*)*		x

Table XII; quantities were not measured. SGL 170 had 20 different kinds of plants while RWS had 8.

Species	Presence	
	SGL	RWS
Arrowwood (*Viburnum dentatum*)	x	
Asian barberry (*Berberis vulgaris*)		x
Asian stilt grass (*Microstegium vimineum*)*		x
Asian wine berry (*Rubus phoenicolasius*)		x
Blackberry (*Rubus fruticosus*)	x	
Blueberry (*Vaccinium* genus)	x	

Cinnamon fern (*Osmunda cinnamomea*)	x	
Dewberry (*Rubus caesius*)	x	
Green briar (*Smilax rotundifoli*)	x	
Hay-scented fern (*Dennstaedtia punctilobula*)*	x	x
Huckleberry (*Gaylussacia baccata*)	x	
Indian pipe (*Monotropa uniflora*)*	x	
Jack-in-the-pulpit (*Arisaema triphyllum*)		
Partridge berry (*Mitchella Repens*)	x	
Pink Lady's Slippers (*Cypripedium acaule*)	x	
Royal fern (*Osmunda regalis*)	x	
Sarsaparilla (*Aralia nudicaulis*)	x	
Solomon's Seal (*Polygonatum*)	x	
Spice bush (*Lindera benzoin*)	x	
Stinging nettle (*Urtica dioica*)		x
Swamp azalea (*Rhododendron viscosum*)	x	
Teaberry (*Gaultheria procumbens*)	x	
Viburnum (*Acer folium*)	x	
Virginia Creeper (*Parthenocissus quinquefolia*)*	x	
Virginia Cucumber Root (*Medeola virginiana*)	x	
White Snake Root (*Eupatorium rugosum*)*		x
Winterberry (*Ilex verticillata*)		x
Witchhazel (*Hamamelis virginiana*)*		x

Table XII. List of underbrush found at RWS and SGL 2009. *Indicates not preferred by deer.

Type	Count	
	SGL	RWS
Arrowwood (*Viburnum dentatum*)	2	
Asian barberry (*Berberis vulgaris*)		9
Asian stilt grass (*Microstegium vimineum*)*	1	25
Azalea (genus *Rhododendron*)	8	
Blackberry (*Rubus fruticosus*)	8	
Clearweed (*Pilea pumila*)		1
Creeping Cintfoil (*Potentilla reptans*)		1
Fernmoss (*Thuidium delicatulum*)*	1	1
Greenbriar (*Smilax rotundifoli*)		1
Hairycap Moss (*Polytrichum commune*)	1	1

Hayscented Fern (*Dennstaedtia punctilobula*)*	18	15
Huckleberry (*Gaylussacia baccata*)		22
Indian Pipe *(Monotropa uniflora)**		1
Mountain Laurel (*Kalmia latifolia*)	2	
Mulitflora Rose (*Rosa multiflora*)		5
Partridge Berry (*Mitchella Repens)*	4	7
Rattlesnake Plant (*Calathea insignis*)	3	1
Rock Polypody Fern (*Polypodium virginianum*)*	1	20
Sarsaparilla (*Aralia nudicaulis*)	12	4
Smart weed (*polgonum* sp.)	18	3
Solomon's Seal (*Polygonatum*)	12	
Spicebush (*Lindera benzoin*)	37	
Viburnum (*Acer folium*)	23	
Violet (*Viola* sp.)	2	7
Virginia Creeper (*Parthenocissus quinquefolia*)*	1	
Virginia Cucumber Root (*Medeola virginiana*)		3
White Snake Root (*Eupatorium rugosum*)*	1	1
Wild pea (*Lathyrus ochroleucus*)		4
Wineberry (*Rubus phoenicolasius*)*		5
Witchhazel (*Hamamelis virginiana*)*	6	4
Wood Fern (genus *Dryopteris)**		4

Table XIII shows the composited underbrush found in 2010 in both locations. SGL 170 had 20 different species while RWS had 23 species.

Type	Count	
	SGL	RWS
Arrowwood (*Viburnum dentatum*)	2	
Asian barberry (*Berberis vulgaris*)		9
Asian stilt grass (*Microstegium vimineum*)*	1	25
Azalea (genus *Rhododendron*)	8	
Blackberry (*Rubus fruticosus*)	8	
Clearweed (*Pilea pumila*)		1
Creeping Cintfoil (*Potentilla reptans*)		1
Fernmoss (*Thuidium delicatulum*)*	1	1
Greenbriar (*Smilax rotundifoli*)		1
Hairycap Moss (*Polytrichum commune*)	1	1
Hayscented Fern (*Dennstaedtia punctilobula*)*	18	15
Huckleberry (*Gaylussacia baccata*)		22
Indian Pipe *(Monotropa uniflora)**		1
Mountain Laurel (*Kalmia latifolia*)	2	
Mulitflora Rose (*Rosa multiflora*)		5
Partridge Berry (*Mitchella Repens)*	4	7

Rattlesnake Plant (*Calathea insignis*)	3	1
Rock Polypody Fern (*Polypodium virginianum*)*	1	20
Sarsaparilla (*Aralia nudicaulis*)	12	4
Smart weed (*polgonum* sp.)	18	3
Solomon's Seal (*Polygonatum*)	12	
Spicebush (*Lindera benzoin*)	37	
Viburnum (*Acer folium*)	23	
Violet (*Viola* sp.)	2	7
Virginia Creeper (*Parthenocissus quinquefolia*)*	1	
Virginia Cucumber Root (*Medeola virginiana*)		3
White Snake Root (*Eupatorium rugosum*)*	1	1
Wild pea (*Lathyrus ochroleucus*)		4
Wineberry (*Rubus phoenicolasius*)*		5
Witchhazel (*Hamamelis virginiana*)*	6	4
Wood Fern (genus *Dryopteris)**		4

Table XIII. List of underbrush found in RWS and SGL 170 in 2010. *Indicates not preferred by deer.

Using the Sorensen Community Similarity Index for the 2010 data, which measures the presence/ absence of species, there is a 55.8% similarity between the two sites. Using Simpson's Index, SGL 170 had a probability of 0.11 and RWS of 0.09 that two individuals picked at random would be the same species. While the two sites were not very similar, based on the data, the species at each site were evenly distributed. Based these observations, there is evidence that deer browsing impacts the types of herbaceous understory species that exist at RWS.

Fecal Coliform Data

Table XIV and Figures 10-12 show the fecal coliform bacteria concentrations from three test dates. SGL 170 consistently had higher concentrations. This could be due to a possible higher diversity of fauna that the habitat at SGL 170 can support.

Replicate	SGL Nov. 4	RWS Nov. 4	SGL 170 Nov. 11	RWS Nov. 11	SGL Nov. 18	RWS Nov. 18
1A	124	38	11	0	19	6
1B	87	23	15	0	25	1
1C	81	29	21	0	29	2
2A	51	24	13	1	20	3
2B	55	50	15	0	18	4
2C	41.6	29	13	0	16	1
3A	63	30	5	0	24	1
3B	74	34	6	0	25	2

3C	70	36	7	0	18	2

Table XIV. Fecal coliform bacteria counts for RWS and SGL 170 2010.

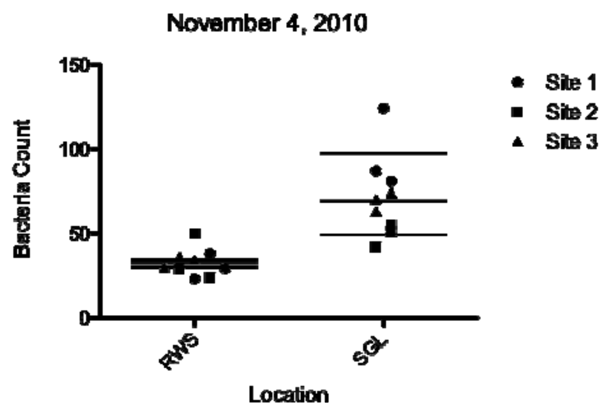

Figure 10. Fecal coliform bacteria count for RWS and SGL 170 for November 4, 2010. The bars indicate the average of each site.

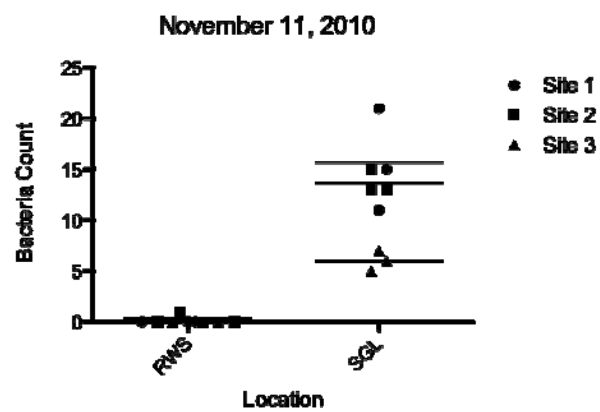

Figure 11. Fecal coliform bacteria counts for RWS and SGL 170 for November 11, 2010. The bars indicate the average of each site.

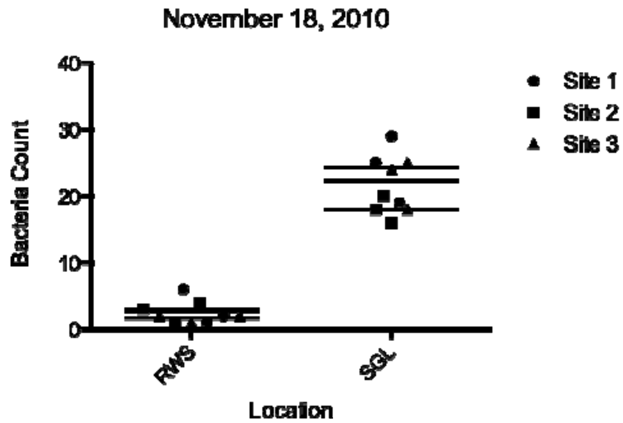

Figure 12. Fecal coliform bacteria counts for RWS and SGL 170 for November 18, 2010. The bars indicate the average of each site.

Water Chemistry Data

On seven occasions between 2009 and 2011, water samples were taken and analyzed (Table XV, Table XVI).

Measure	Sept. 28 2009	Oct. 18 2009	Oct. 26 2009	Oct. 7 2010	Feb. 10 2011	Feb. 17 2011	Mar. 3 2011	Average
Temperature (°C)	14.8	8.5	10.6	12.6	-1.97	15.37	n/d	9.98
Conductivity (umohs)	30.1	22.4	30.0	22.53	n/d	n/d	n/d	26.27
DO (mg/L)	9.5	0.0	3.8	n/d	n/d	n/d	n/d	4.43
% Saturation DO (%)	93.7	108.0	32.8	n/d	n/d	n/d	n/d	78.16
pH	5.9	6.9	6.0	5.88	n/d	6.82	n/d	6.30
Nitrates (mg/L)	0.6	0.8	0.6	0.57	0.74	0.98	0.97	0.75
Phosphates (mg/L)	0.15	0.05	0.05	0.10	0.17	0.23	0.12	0.13
Alkalinity (mg/L)	16.2	19.6	14.2	n/d	n/d	n/d	n/d	16.67

Hardness (mg/L)	7.8	8.2	9.5	n/d	n/d	n/d	n/d	8.50

Table XV. Water chemistry data for RWS for 2009 and 2010.

Measure	Sept. 28 2009	Oct. 18 2009	Oct. 26 2009	Oct. 7 2010	Feb. 10 2011	Feb. 17 2011	Mar. 3 2011	Average
Temperature (°C)	14.5	7.4	9.2	11.50	-0.27	14.67	n/d	9.5
Conductivity (umohs)	29.7	20.6	38.7	24.73	n/d	n/d	n/d	28.4
DO (mg/L)	9.5	0.0	6.5	n/d	n/d	n/d	n/d	5.3
% Saturation DO (%)	92.4	102.1	56.5	n/d	n/d	n/d	n/d	83.7
pH	6.2	6.5	5.7	4.78	7.17	6.39	n/d	6.1
Nitrates (mg/L)	0.2	0.2	0.3	0.18	0.41	0.43	0.32	0.29
Phosphates (mg/L)	0.3	0.1	0.0	0.19	0.12	0.24	0.12	0.1
Alkalinity (mg/L)	16.2	19.8	10.2	n/d	n/d	n/d	n/d	15.4
Hardness (mg/L)	6.0	7.6	9.7	n/d	n/d	n/d	n/d	7.7

Table XVI. Water chemistry data for SGL 170 for 2009 and 2010.

The two-tailed t-test shows insignificant evidence for differences between the streams except for the nitrate concentration ($p = 0.0001$) (Table XVII).

Measure	p-value
Temperature (°C)	0.89
Conductivity (μmohs)	0.65
DO (mg/L)	0.83
% Saturation DO (%)	0.85
pH	0.69
Nitrates (mg/L)	0.00
Phosphates (mg/L)	0.58
Alkalinity (mg/L)	0.72
Hardness (mg/L)	0.57

Table XVII. P-values for the water chemistry data 2009.

Precipitation samples were also collected from a rain event (October 4-6, 2010). Table XVIII shows the nitrate and phosphate levels.

Site	Nitrate (mg/L)	Phosphate (mg/L)
RWS	0.35	0.56
SGL 170	0.28	0.13

Table XVIII. Precipitation water chemistry for RWS and SGL 170 2010.

Soil samples were taken on the banks of the streams. The nitrate concentrations are shown in Table XIX.

Site	Dec. 2010 Nitrate (mg/L)	Mar. 2011 Nitrate (mg/L)	Average	p-value
RWS	7.0	2.2	4.6	0.35
SGL 170	0.6	0.8	0.7	

Table XIX. Nitrate concentration in soil samples.

The nitrate levels were significantly higher at RWS (average 0.75mg/L) compared to SGL 170 (average 0.29mg/L). Similar observations hold for the soil nitrate concentration. Although not statistically significant ($p = 0.35$) for this sample size, further data might suggest a more significant difference. The higher concentrations of nitrates in the stream and soil samples suggest that the changing forest composition at RWS may contribute to increased nitrogen runoff into the forest stream. In addition, a preliminary test on the nitrate and phosphate concentrations of the incoming precipitation show that nitrate concentrations are similar in both areas before making contact with the soil. This further supports the hypothesis that it is a variable in the terrestrial ecosystem that changes the concentration of nitrates.

The higher nitrate concentration at the RWS stream in particular could be because the filtering capacity of the RWS forest has already been compromised by the changing understory composition. If the nitrate results are consistent with other headwater streams in similar forests within the Chesapeake Bay, the amount of nitrogen runoff these streams contribute to the Chesapeake Bay could be significant. While more data are required to confirm this, the deer management practices of some forests could be reexamined based on their impact on the Chesapeake Bay. The differences between the two streams for the other water chemistry parameters were statistically insignificant. This can be attributed to their otherwise similar environments.

Macroinvertebrate Data

Table XX shows the composite data for macroinvertebrates collected on seven occasions between October 2009 and February 2011. A total of 198 individuals were collected for RWS and 106 for SGL 170. The original data can be found in Appendix A.

Order	Family	Common name	RWS	SGL 170	Hilsenhoff Biotic Index (DEP)	Feeding group
DIPTERA	Chironomidae	Bloodworm	2	1	10	CG
	Tipulidae	Crane fly larva	15	27	3	SH
	Simuliidae	Black Fly	3		6	FC
	Ceratopogonidae	Biting midge	2		5	P
	Tabanidae	Horse Fly		1	7	PI
	Athericidae	Aquatic Snipe Fly		1	2	P
	Dixidae	Dixid midge	2	1	2	CG
EPHEMEROPTERA	Heptageniidae	Flat headed mayfly	47	15	4	SC

	Baetidae	Small minnow mayfly	1	1	5	CG, SC
	Ephemeridae	Common burrower mayfly	1	1	4	CG
	Ephemerellidae	Spiny crawler mayfly	8		2	CG
	Leptophlebiidae	Pronggilled mayfly	2		2	CG
	Ameletidae	Ameletid Minnow mayfly	1		0	SC, CG
COLEOPTERA	Psephenidae	Water penny	1		4	SC
TRICHOPTERA	Hydropsychidae	Net spinning caddisfly	30	15	4	FC
	Philopotamidae	Fingernet caddisfly	2		3	FC
	Polycentropodidae	Trumpetnet and tube	2	1	6	FC
	Rhyacophilidae	Freeliving caddisfly		2	0	P
PLECOPTERA	Perlidae	Common stonefly	10	14	1	P
	Capniidae	Slender winter stonefly	38	13	1	SH
	Perlolidae	Perloid stonefly	1		2	P
	Peltoperlidae	Roachlike stonefly	7		1	SH
	Chloroperlidae	Green Stonefly	17	1	1	P, SH
	Taeniopterygidae	Winter Stonefly	1	6	2	SH
ODONATA	Libellulidae	Skimmer dragonfly		2	9	P
ANNELIDA	Oligochaeta	Aquatic worm	1		10	P, CG
AMPHIPODA	Gammaridae	Scud	1		4	CG
DECOPODA		Crayfish	3	4	7	CG
Total			198	106		

Table XX. Macroinvertebrates collected at both sites (composite 2009-2011).

Table XXI shows the results of the metrics used to analyze the macroinvertebrates.

Metric	*RWS*	*SGL 170*
Hilsenhoff Biotic Index	2.84	3.05
PA Modifed EPT Index	10	5
Taxa richness	24	17
% modified mayflies	5.6	0.0
% dominant family	23.7	25.5
% shredders	39.4	44.3
Shannon-Wiener Index	2.4	2.2
Simpson's Index	0.13	0.14
Sorensen's coefficient of community similarity	0.59	

Table XXI. Metrics to evaluate the macroinvertebrate data.

Based on these metrics, both the RWS and SGL 170 streams support rich, even, and pollution-sensitive communities. The Hilsenhoff Biotic Index and PA Modified EPT Index show that both streams have a variety of taxa that are sensitive to organic pollution.

Both streams had a variety of overall taxa, and neither location had one heavily dominant family. The values for the Shannon-Wiener Index further support that stream has an even and rich community. The values for the Simpson Index support an even community as well.

Based on the habitat, a prevalence of mayflies and insects in the shredder feeding group would be expected. The higher percent modified mayflies at RWS could be due to the steeper gradient stream profile there; some mayflies are well-adapted to higher velocity stream habitats especially due to their adaptive gills (Voshell 2002). The high percentage of shredders for both sites is indicative of a large amount of organic material entering the stream to be processed. Because the tree canopy structure has not yet been significantly altered due to deer browsing, it can be expected that both streams are receiving similar amounts of CPOM.

Although both streams seem to support rich and even communities, based on the Sorensen's Coefficient of Community Similarity, the macroinvertebrate populations are not as similar as could be expected for two similar forested headwater streams. This could be due to differences in the stream gradient and velocity, or in sampling errors. However, these metrics do not account for temporal changes.

Figure 13. shows a cluster analysis dendrogram for the individual sampling sites on each stream for each collection date.

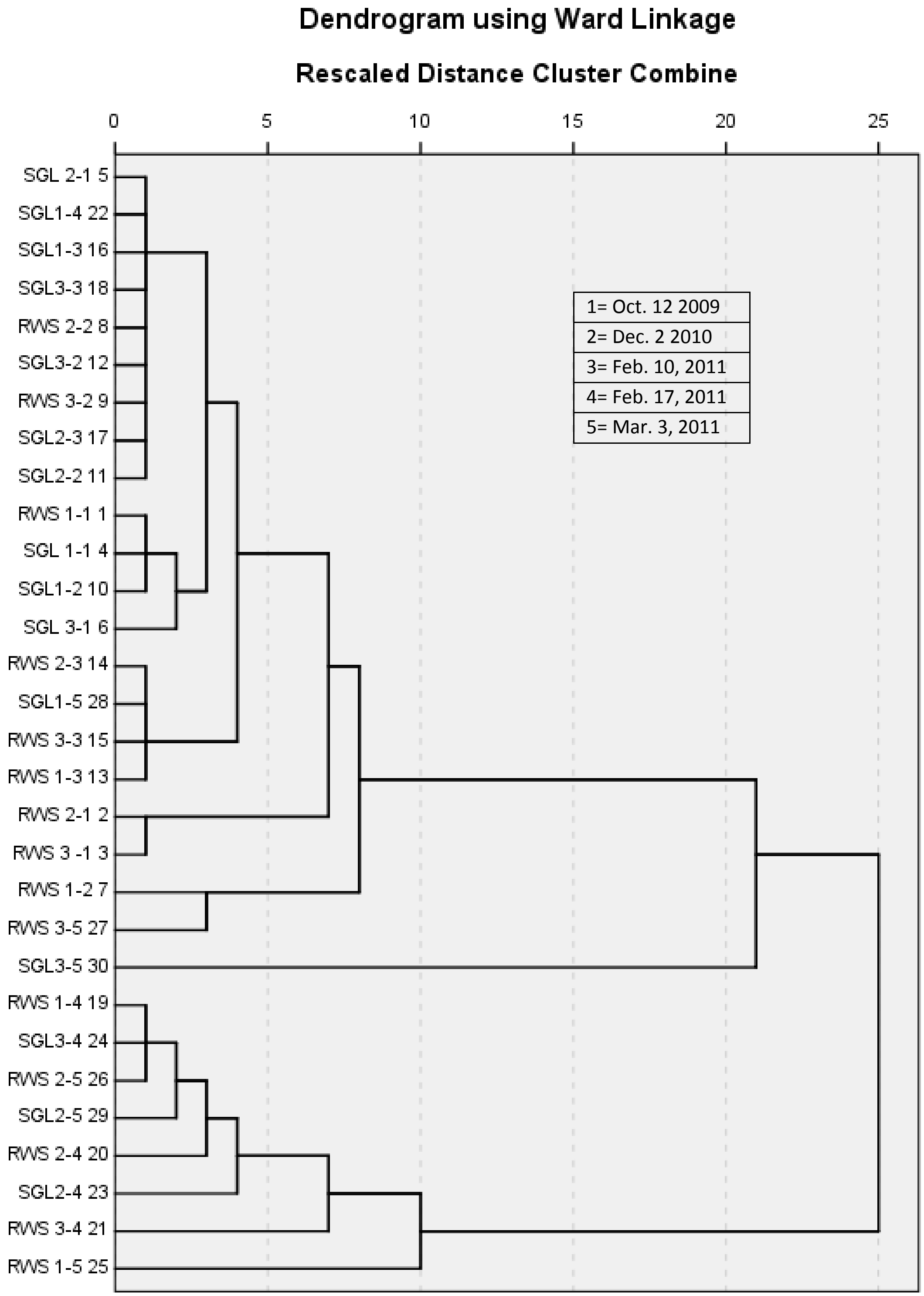

Figure 13. Cluster analysis dendrogram for the individual sampling sites on each stream for each collection date.

The cluster analysis suggests that there could be varying differences by stream in the macroinvertebrate communities for collection dates October 12, 2009, December 2, 2010, and February 10, 2011. For example, the first cluster has 7 of 9 sites at the SGL 170 stream, mostly for the December 2 and February 10 dates. The second, third, and fourth clusterings also show more tendency to clustering by stream location and then by date. The last larger clustering shows a strong tendency towards similarity by the February 17 and March 3 dates rather than by location. In looking at the raw data (Appendix A), however, it becomes apparent that the quantity of data impacts the cluster analysis: the most individuals were collected in the February 17 and March 3 collections, while far fewer were collected on the other dates. Because of the strong correlation between dates rather than locations for February 17 and March 3, the data suggests that the seasonal patterns have a greater influence on macroinvertebrate communities than the location does. Because of the low numbers of individuals collected on the first three dates, it is difficult to draw a conclusion as to the importance of location on the macroinvertebrates based on the cluster analysis.

From the metrics and the cluster analysis, it seems that the stream location does not significantly impact the sensitivity, richness, and distribution of the macroinvertebrates. Changes in seasonal patterns may have the greatest influence on macroinvertebrates; this can be expected because of changes in the availability of organic material and because of hatching cycles.

Because the forest at RWS will likely continue to change composition due to deer herbivory; the amount and type of organic matter entering the stream will also change. This may impact the community of macroinvertebrates in the future.

CONCLUSIONS

The headwater streams in Pennsylvania forests have an impact on the condition of the Chesapeake Bay. While forests have historically had the least negative impact on water systems, some forests are changing due to increased white-tailed deer populations. The ability of the RWS forest to regenerate has diminished because the understory is selectively browsed by a high deer density. The filtering capacity of the altered forest at RWS may diminish as a result. Water chemistry tests from this study show that the nitrate concentrations are already significantly higher in the forest at RWS than at SGL 170. Soil and precipitation samples confirm this trend. Macroinvertebrate samples do not indicate significant or consistent differences in richness, evenness, and pollution tolerance between the two sites; this is likely because the canopy trees that contribute organic matter to the streams remain similar from previous deer management practices.

While the changed forest does not seem to have impacted the biotic community, the data suggest that it has impacted the concentration of nitrates. If this pattern holds true for other forested headwater streams altered by high deer populations, there could be a significant influx in nutrients to the Chesapeake Bay. With the already damaging levels of nutrients entering the bay, this could necessitate a change in forest management practices and legislation.

This was a preliminary study in connecting the forest and stream ecosystems. Studies should be continued to examine the connection between the alteration of forests by white-tailed deer and the health of forest streams and the greater watershed. These future studies should include collecting data from a wider variety of locations, including sites farther upstream on the streams studied here as well as on other streams. In addition, soil samples could be extended more systematically to determine the pattern of soil nitrate concentrations relative to the distance from the stream and within the forests.

RESOURCES

Benner, J.M. 2006. *Browsing impact report for the Pennsylvania State Forests*. Bureau of Forestry DCNR.

Blankenship, K. 2006. State of the Chesapeake forests: Public perception often fails to look at importance of forests beyond trees. *Bay Journal* October 2006.

Boesch, D.F., R.B. Brinsfield, R.E.Magnien. 2001. Chesapeake Bay eutrophication: Scientific understanding, ecosystem restoration, and challenges for agriculture. *Journal of Environmental Quality* 30: 303-320.

Chesapeake Bay Program website “Nutrients”. Available: http://www.chesapeakebay.net/nutrients.aspx?menuitem=15216

Cummins, K.W. 1974 Structure and function of stream ecosystems. Bioscience 24: 631- 641.

Cummins, K.W., M.A., Wilzbach, D.M. Gates, J.B. Perry, and W.B. Taliaferro.1989. Shredders and riparian vegetation. *Bioscience* 39: 24-30.

Cushing, C.E. and J.D. Allan. 2001. “The Ecology of Rivers and Streams” in *Streams: Their Ecology and Life*. San Diego, CA: Academic Press.

deCalesta, D. S., and S. L. Stout. 1997. Relative deer density and sustainability: a conceptual framework for integrating deer management with ecosystem management. *Wildlife Society Bulletin* 25(4): 252–258.

Department and Conservation of Natural Resources (DCNR) and U.S. Forest Service. 2004. *A snapshot of Pennsylvania’s updates forest inventory 2004*.

Department and Conservation of Natural Resources (DCNR), Bureau of Forestry. 2007. *State Forest Resource Management Plan*. Harrisburg, PA.

Diefenbach, D.R., Palmer, W.L., Shope, W.K. 1997. Attitudes of Pennsylvania sportsmen towards managing white-tailed deer to protect the ecological integrity of forests. *Wildlife Society Bulletin* 25(2): 244-251.

Fergus, C. 2000. *Wildlife of Pennsylvania and the Northeast*. Mechanicsburg, PA: Stackpole Books.

Gottlieb, S.J. and M.E. Schweighofer. 1996. Oysters and the Chesapeake Bay ecosystem: A case study for exotic species introduction to improve environmental quality? *Estuaries* 19(3): 639-650.

Horsley, S.B., S.L. Stout, D.S. deCalesta. 2003. White-tailed deer impact on the vegetation dynamics of a northern hardwood forest. *Ecology Applications* 13(1): 98-118.

Hough, A.F. 1965. A twenty-year record of understory vegetational change in a virgin Pennsylvania forest. *Ecology* 46(3): 370-373.

Hynes, H.B.N. 1970. *The Ecology of Running Waters*. Toronto: University of Toronto Press.

Kusko, M. 2009. Interview by Kerstin Martin. Tape recording. October 30. Bureau of Forestry
DCNR, Fayetteville, PA.

Latham, R.E., J. Beyea, M. Benner, C.A. Dunn, M.A. Fajvan, R.R. Freed, M. Grund, S.B. Horsley, A.F. Rhoads, B.P. Shissler. 2005. Managing white-tailed deer in forest habitat from an ecosystem perspective: Pennsylvania case study. *Report of the Deer Management Forum* for Audubon Pennsylvania and Pennsylvania Habitat Alliance.

Lowrance, R., R. Todd, J. Fail Jr., O. Hendrickson Jr., R. Leonard, L. Asmussen. 1984. Riparian Forests as Nutrient Filters in Agricultural Watersheds. *BioScience* 34(6): 374-377.

Lowrance, R., L.S. Altier, J.D. Newbold, R.R. Schnabel, P.M. Groffman, J.M. Denver, D.L. Correll, J.W. Gilliam, J.L. Robinson, R.B. Brinsfield, K.W. Staver, W. Lucas, A.H. Todd. 1997. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. *Environmental Management* 21 (5): 687-712.

McAfee, R. 2011. “Chesapeake Bay Conservation”. Natural Resource and Conservation Service (NRCS) website Available:
http://www.pa.nrcs.usda.gov/chesapeake_bay.html

McCafferty, W. P. 1981. *Aquatic Entomology*. Boston: Jones and Bartlett Publishers, Inc.

McWilliams, W.H., S.P Cassell, B.J. Butler, M.L. Hoppus, S.B. Horsley, A.J. Lister, T.W. Lister, R.S. Morin, C.H. Perry, J.A.Westfall, E.H. Wharton, C.W.Woodall. 2004. Pennsylvania’s Forest 2004. Resource Bulletin Northern Research Station.

Minshall, G.W. 1994. Stream riparian ecosystems: rationale and methods for basin level assessments of management effects. Pages 153-177 *in* M.E. Jensen and P.S. Bourgeron (Eds.) *Eastside Forest Ecosystem Health Assessment, Volume II: Ecosystem Management: principles and applications.* General Technical Report

PNW-GTR-318. U.S. Forest Service, Pacific Northwest Research Station, Portland, OR.

Mulholland, P.J. 1992. Regulation of Nutrient Concentrations in a Temperate Forest Stream: Roles of Upland, Riparian, and Instream Processes. *Limnology and Oceanography* 37(7): 1512-1526.

Nixon, S.W. 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. *Ophelia* 41:1994-219.

Pedersen, B.S., and A.M.Wallis. 2004. Effects of white-tailed deer herbivory on forest gap dynamics in a wildlife preserve, Pennsylvania, USA. *Natural Areas Journal*. 24: 82-94.

Pennsylvania Game Commission (PGC), Bureau of Wildlife Management. 2003. Population management plan for white-tailed deer in Pennsylvania (2003-2007).

Perry, D.A. 1994. *Forest Ecosystems*. Baltimore: John Hopkins University Press.

Phillips, S.W. 2005. U.S. Geological Survey Chesapeake Bay Science Plan 2006-2011. Open-File Report: 2005-1440.

Porter, W. F., M. C. Coffey, and J. Hadidian. 1994. In search of a litmus test: wildlife management on the U.S. national parks. *Wildlife Society Bulletin* 22(2): 301–306.

Punj, G. and D.W. Stewart. 1983. Cluster analysis in marketing research: review and suggestions for application. *Journal of Marketing Research* 20 (2): 134-148.

Rhodes, H.A., and W.A. Hubert. 1991. Submerged undercut banks as macroinvertebrate habitat in a subalpine stream. Hydrobiologia 213: 149-153.

Rose, R.E., Geldreich, E.E., Litsky, W. 1974. Improved Membrane Filter Method for Fecal Coliform Analysis. *Applied Microbiology* 29 (4): 532-536.

Sweeney, B.W. 1993. Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in eastern North America. *Proceedings of the Academy of Natural Sciences of Philadelphia* 144: 291-340.

Tilghman, N.G. 1989. Impacts of White-Tailed Deer on Forest Regeneration in Northwestern Pennsylvania. *The Journal of Wildlife Management* 53 (3): 524-532.

Tripler, C.E., C.D. Canham, R.S.Inouye, J.L.Schnurr. 2002. Soil nitrogen availability, plant luxury consumption, and herbivory by white-tailed deer. *Oecologia* 133: 517-524.

United States Department of Agriculture (USDA), Forest Service. 1995. Proceedings of the 10th Central Hardwood Forest Conference. Morgantown, WV. General Technical Report NE-197.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37: 130-137.

Voshell, J. R. 2002. *A Guide to Common Freshwater Invertebrates of North America.* Blacksburg, Virginia: The McDonald and Woodward Publishing Company.

Welsch, D. 1991. Riparian forest buffers: Function and design for protection and enhancement of water resources. USDA Forest Service Report No. NA-PR-07-91.

Witmer, G.W. and D.S. DeCalesta. 1991. The need and difficulty of bringing the Pennsylvania deer herd under control. *Proceedings of the Eastern Wildlife Damage Control Conference* 5:130-137.1.

APPENDIX A

Macroinvertebrates collected on seven different dates.

Order	FAMILY	RWS 10-12-09			SGL 10-12-09			RWS 12-2-2010			SGL 12-2-10		
		Site 1	Site 2	Site 3	Site 1	Site 2	Site 3	Site 1	Site 2	Site 3	Site 1	Site 2	Site 3
Diptera	Chironomidae		1										
	Tipulidae	2	2	1	2		4				2	1	
	Simuliidae	1											
	Ceratopogonidae												
	Tabanidae												
	Athericidae												
	Dixidae												
Ephemeroptera	Heptageniidae						1						1
	Baetidae							1				1	
	Ephemeridae		1										1
	Ephemerellidae												
	Leptophlebiidae												
	Ameletidae												
Coleoptera	Psephenidae	1											
Tricoptera	Hydropsychidae	2	8	5	1	2	2	3		1	1		
	Philopotamidae									1			
	Polycentropodidae				1			2					
	Rhyacophilidae										2		
Plecoptera	Perlidae		1			1	2		1	2	3	2	
	Capniidae	1											
	Perlolidae		1				3						
	Peltoperlidae			2									
	Chloroperlidae							7	1			1	
	Taeniopterygidae												
Odonata	Libellulidae												
Annelida	Oligochaeta (Class)		1										
Amphipoda	Gammaridae			1									
Decopoda	Astacidae/Cambaridae	2	1								2	1	1

Order	FAMILY	RWS 2-10-11			SGL 2-10-11			RWS 2-17-11			SGL 2-17-11		
		Site 1	Site 2	Site 3	Site 1	Site 2	Site 3	Site 1	Site 2	Site 3	Site 1	Site 2	Site 3
Diptera	Chironomidae	1											
	Tipulidae												1
	Simuliidae							1	1				
	Ceratopogonidae									2			
	Tabanidae											1	
	Athericidae												
	Dixidae												
Ephemeroptera	Heptageniidae	2	2	3				7	7	2		6	5
	Baetidae												
	Ephemeridae												
	Ephemerellidae								1	7			
	Leptophlebiidae												
	Ameletidae												
Coleoptera	Psephenidae												
Tricoptera	Hydropsychidae							1	4	2	2		2
	Philopotamidae	1											
	Polycentropodidae												
	Rhyacophilidae												
Plecoptera	Perlidae	1	2		1	2			1		1	1	
	Capniidae	2	4	2				4	8	8		1	5
	Perlolidae												
	Peltoperlidae	2											
	Chloroperlidae			1									
	Taeniopterygidae										1	5	
Odonata	Libellulidae	1											
Annelida	Oligochaeta (Class)												
Amphipoda	Gammaridae												
Decopoda	Astacidae/Cambaridae												

Order	FAMILY	RWS 3-3-11			SGL 3-3-11		
		Site 1	Site 2	Site 3	Site 1	Site 2	Site 3
Diptera	Chironomidae						
	Tipulidae	3	3	3		1	16
	Simuliidae						
	Ceratopogonidae						
	Tabanidae						
	Athericidae					1	
	Dixidae		1	1		1	
Ephemeroptera	Heptageniidae	15	5	4	1	2	
	Baetidae						
	Ephemeridae						
	Ephemerellidae						
	Leptophlebiidae	1	1				
	Ameletidae		1				
Coleoptera	Psephenidae						
Tricoptera	Hydropsychidae	2	2			4	1
	Philopotamidae						
	Polycentropodidae						
	Rhyacophilidae						
Plecoptera	Perlidae			2	1	0	1
	Capniidae	2	3	4	2	4	3
	Perlolidae						
	Peltoperlidae	1					
	Chloroperlidae		2	6			
	Taeniopterygidae	1					
Odonata	Libellulidae						
Annelida	Oligochaeta (Class)						
Amphipoda	Gammaridae						
Decopoda	Astacidae/Cambaridae						