IMPACTS OF TRANSPORTATION INFRASTRUCTURE ON STORMWATER AND SURFACE WATERS IN CHITTENDEN COUNTY, VERMONT, USA

A Thesis Presented

by

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to

The Faculty of the Graduate College

of

The University of Vermont

In Partial Fulfilment of the Requirements
For the Degree of Master of Science
Specializing in Natural Resources

January, 2016

Defense Date: November 11, 2015
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ABSTRACT

Transportation infrastructure is a major source of stormwater runoff that can alter hydrology and contribute significant loading of nutrients, sediment, and other pollutants to surface waters. These increased loads can contribute to impairment of streams in developed areas and ultimately to Lake Champlain. In this study we selected six watersheds that represent a range of road types (gravel and paved) and road densities (rural, suburban, and urban) present in Chittenden County, one of the most developed areas in Vermont. The location and density of road networks were characterized and quantified for each watershed using GIS analysis. Monitoring stations in each watershed were constructed and instrumented to measure discharge and water quality parameters continuously from spring through early winter. Storm event composite samples and monthly water chemistry grab samples were collected and analyzed for total nitrogen, total phosphorus, chloride, and total suspended sediments. Results from this study show that road type and road density are closely linked with the level of impairment in each watershed. Total phosphorus and total nitrogen from storm event composite samples and monthly grab samples significantly increased along a gradient of increasing road network density. Chloride concentrations increased several orders of magnitude along this same gradient. With the exception of Alder Brook where total suspended sediment (TSS) concentrations tended to be high, there were no significant differences in TSS concentrations between rural and developed watersheds. The elevated storm event TSS concentrations in the rural streams suggest that the unpaved roads in the rural watersheds contribute to stormwater runoff loads and that sediment control, at least in the developed watersheds, might be fairly effective. The overall results from this study show that local roads are a significant source of impairment for streams in the Chittenden County area. Most of these roads are municipal roads that are not under management of the Vermont Agency of Transportation. Thus, local actions will be necessary to reduce runoff and pollutant loading from these roads.

DEDICATION

This thesis is dedicated to my mother Holly and my grandmother Josephine. They have both cultivated my love and appreciation of the environment and pushed me to be a better scientist. This work is also dedicated to my wife Sarah who shares my love of Vermont and passion for protecting our environment and natural resources. I very much look forward to the adventures we have in store for us.

ACKNOWLEDGEMENTS

I would like to acknowledge and thank my advisor and mentor Dr. Breck Bowden. Breck has shaped my academic and professional path since his arrival at the University of Vermont. My thesis advisors Drs. Mary Watzin and Donald Ross also provided important advice and support for this project. The extensive field and lab work required for the project required an army of researchers, friends, and family. Colin Penn, Patrick Tobin, Dan Freedman, Bianca Rodriguez, and Kelsey McAuliffe helped me through floods, mosquitos, snow, and ice. Susan Fuller and Leman Bronson helped with laboratory assistance and sample analysis. I would like to thank Steve Fiske and the Vermont Department of Environmental Conservation BASS lab for collaboration on the benthic macroinvertebrate sampling and analysis. The Bowden Watershed Research Team provided invaluable feedback and support over the years and the diverse background of its members was an invaluable resource. This project was supported by funding from the US DOT through the Transportation Research Center at the University of Vermont.

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CHAPTER 1: COMPREHENSIVE LITERATURE REVIEW

Introduction

This literature review will briefly discuss the suite of biological, chemical, and physical effects of urbanization on surface waters. These impacts are commonly known as the "urban stream syndrome" and are well described in scientific literature; however these studies were typically conducted in watersheds with much higher development densities than are present in Vermont. Next, the review will focus on the types of impacts specifically related to road networks. While dirt and gravel roads have a unique set of associated impacts, much of the discussion of these roads is based on studies of temporary logging roads, not the types of well-maintained gravel roads found throughout rural Vermont.

Urbanized areas have disproportionately large ecological footprints even though they only cover 2% of the earth's land surface (Paul and Meyer 2001). As a result of these large ecological footprints, very few watersheds remain unimpacted by anthropogenic activities (Eyles and Meriano 2010). Streams and rivers are the low points of the landscape and are therefore potentially affected by any anthropogenic changes to the natural environment. Over 130,000 km of streams and rivers in the U.S. are impacted by urbanization (Paul and Meyer 2001). Chittenden County in northwestern Vermont is the most developed county in the State with a population of 156,545 in 2010 (U.S Census Bureau 2010). Several streams in Chittenden County and Lake Champlain are listed as impaired waters as a result of stormwater runoff (VTDEC 2010).

Transportation infrastructure is an important component and driver of urbanization. Road networks are constructed and expanded concurrently with urbanization, and existing roads are shown to drive future development patterns (Wheeler et al. 2005). Numerous recent studies have identified transportation infrastructure as a major source of stormwater runoff and pollutant loading to receiving waterbodies (Wheeler et al. 2005; Kang and Marston 2006; Eyles and Meriano 2010). Transportation infrastructure is a ubiquitous feature of the developed landscape and can represent as much as half of the impervious surfaces in developed watersheds (Eyles and Mariano 2010). The 6,300,000 km of public roads in the U.S. directly affect approximately 20% of the landscape and 50% of the U.S. land area is within 382 m of a road (Wheeler et al. 2005).

Characterizing Watershed Urbanization

The total impervious area (TIA) in a watershed is the most prevalent metric for characterizing the level of urbanization and therefore the degree of impact on receiving waterbodies (CWP 2003; Kang and Marston 2006). The most common method for quantifying TIA, especially in larger watersheds, is through the use of spatial land use/ land cover classification data (CWP 2003). Each land use class (forest, commercial, light residential, etc.) has an associated TIA value and the relative percent land cover for each class is summed and weighted to estimate watershed TIA. The TIA values commonly used for these studies were calculated from direct measurements of orthophotos from numerous datasets, mainly in the 1970’s. The accuracy of %TIA estimates are closely linked to data quality and to the level of data processing effort; however these original studies noted that considerable variation was present within each land use class (Brabec et al. 2002; CWP 2003).

Estimates of TIA represent the paved impervious surfaces in the watershed (Booth and Jackson 1997; Kang and Marston 2006). These calculations ignore compacted impervious surfaces and the complex network of pipes and connectivity present in a developed watershed (Booth and Jackson 1997; CWP 2003; Kang and Marston 2006). Effective Impervious Area (EIA) is a more detailed and comprehensive method for quantifying watershed imperviousness, taking into account the complex drainage networks that define an urban watershed (Hatt et al. 2004; Jacobson 2011). EIA calculations require ground truthing of drainage infrastructure and connectivity and are much more time intensive (Booth and Jackson 1997; Brabec et al. 2002; CWP 2003). As a result of the different methods and range of effort level required for calculating EIA or TIA, it can be challenging to compare findings from different studies of watershed impacts from development (CWP 2003; Jacobson 2011).

Urban Stream Syndrome

A broad range of scientific studies have found strong evidence that increasing watershed development leads to physical, biological, and chemical impairment in receiving streams and rivers, known as the “urban stream syndrome” (Meyer et al. 2005; Walsh et al. 2005). The location of streams in the landscape makes them particularly vulnerable to the myriad impacts of the urban stream syndrome (Walsh et al. 2005). Most studies consider mid to high levels of urbanization to be as high as 30-40% TIA. Much less research has been conducted in watersheds with lower levels of development as is typically found in Vermont (Wang et al. 2003; Wheeler et al. 2005; Chadwick et al. 2006).

Significant relationships between TIA and a range of biotic and abiotic factors have been documented in numerous studies; many of these relationships suggest a threshold response at a specific level of watershed development (Booth and Jackson 1997; Paul and Meyer 2001; Brabec et al. 2002; CWP 2003). A threshold of 10% watershed development has been associated with a wide range of physical (i.e. altered hydrology), chemical (i.e. increased nutrient concentrations), and biological impacts (i.e. loss of pollution sensitive organisms) (Booth and Jackson 1997; Paul and Meyer 2001; Wang et al. 2003; Wheeler et al. 2005). However, many studies have observed watershed impacts at very low levels of development and suggest response thresholds as low as 5% particularly for some biologic and physical stream indicators (Paul and Meyer 2001; Wang et al. 2003; Schiff and Benoit 2007). Recent updates of the impervious cover model and threshold responses defined by CWP (2003) show the range of observed impacts at a given level of watershed imperviousness (white cone in Figure 1) and indicate a range of imperviousness where overall stream quality is expected to decrease in contrast to the specific degradation thresholds previously included in the model (Schueler et al. 2009).

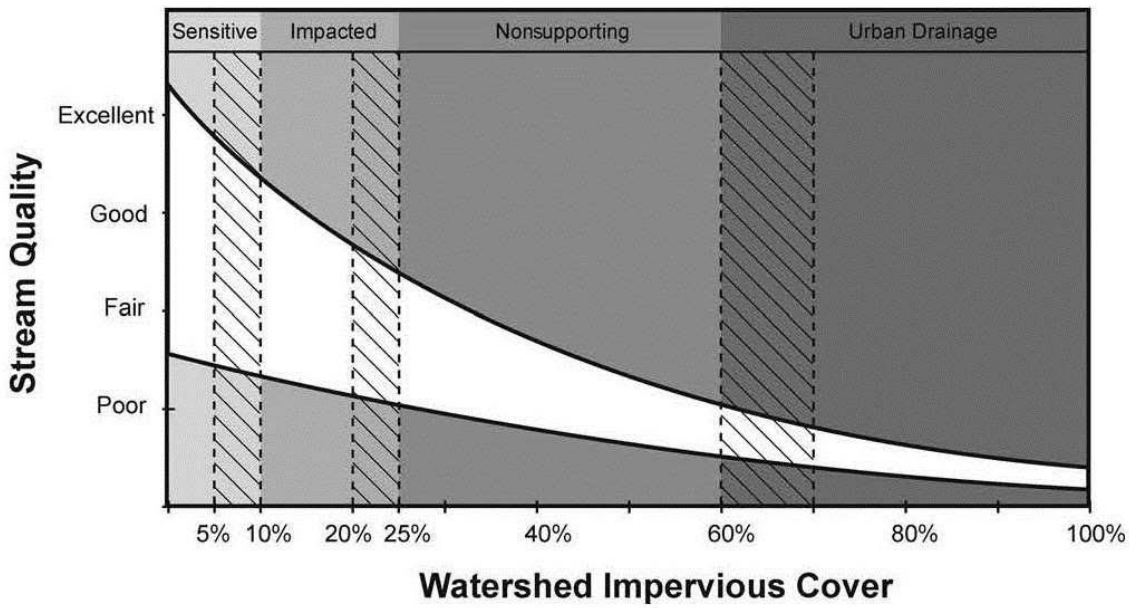

FIGURE 1. Summary of impervious cover impacts on stream quality from Schueler et al. (2009).

Water Chemistry Impacts.

Stormwater drainage infrastructure collects and concentrates the diffuse sources of nutrients and other chemicals across an urbanized watershed and routes these directly to receiving waters (Hatt et al. 2004; Berhnardt et al. 2008). Effective impervious area calculations which account for drainage infrastructure and increased hydrologic connectivity are shown to best explain chemical loading and impacts in developed watersheds (Hatt et al. 2004). Most studies have found that stream chemistry impacts are typically observed at higher levels of urbanization (30-40% TIA) and that highly urbanized watersheds may have extensive chemical and nutrients impairments, including levels of total phosphorus (TP) and total nitrogen (TN) higher than those observed in agricultural watersheds (Paul and Meyer 2001; Wheeler et al. 2005). Chloride and a suite of chemicals that can be toxic to stream biota are well documented components of the urban stream syndrome and both are predominantly generated on roads and parking lots and will be discussed in a later section describing road network specific water chemistry impacts.

Phosphorus (P) is a critical nutrient in freshwater aquatic systems and increased loading of this nutrient can dramatically change receiving waters. Wastewater treatment plants are a major point-source of P in some watersheds; however most of the P loading originates from diffuse sources such as lawn fertilizer, animal waste, and septic systems (Paul and Meyer 2001). Phosphorus loading from septic systems can be particularly important in rural watersheds and may lead to significant nutrient enrichment of smaller streams (Withers et al. 2011). Phosphorus concentrations in urban streams are typically closely linked to sediment, sources of which will be discussed in a following section.

Nitrogen (N) has a complex life cycle in urban watersheds and many recent studies have been devoted towards understanding N dynamics in these streams (Mulholland and Webster 2010). Nitrogen loading in developed watersheds shares many of the same sources as P, and biologically available forms of N (fixed N) are also important nutrients in freshwater systems. Developed watersheds also receive significant loading of NH_3, NO_x, and N_2O that is fixed from inert N_2 in automobile engines. The total inputs of anthropogenic fixed N in urban watersheds can equal or exceed natural sources (Bernhardt et al. 2008; Collins et al. 2010). Drainage infrastructure and stream habitat simplification in urban watersheds increase the hydrologic connectivity between N sources and surface waters. This creates a direct pathway for N export and removes or reduces the opportunity for natural denitrification processes in riparian soils or instream on features such as debris jams (Walsh et al. 2005; Bernhardt et al. 2008; Collins et al. 2010).

Total Suspended Sediment (TSS) loading has a variable response across studies of urban watersheds (Walsh et al. 2005). This is in part due to shifts in TSS sources based on the development trends within a watershed. During the clearing and building phase of watershed development, the primary loading is fine sediments from terrestrial sources. Following urbanization, increased flashiness and increased peak flows lead to stream morphological changes and stimulate channel, bed, and bank erosion (Booth and Jackson 1997; Schoonover et al 2007).

Biological Impacts.

Watershed urbanization is shown to dramatically reduce riparian and instream habitat complexity and function, and to impact the biodiversity and community composition of fish and benthic macroinvertebrate communities (Paul and Meyer 2001; Wang et al. 2003; Walsh et al. 2005; Wheeler et al. 2005; Chadwick et al. 2006). Degradation of the riparian zone is observed at the lowest levels of watershed development (<5% TIA) and the riparian zone may be completely removed along surface waters in watersheds with higher development (Wheeler et al. 2005). Any impacts to the riparian zone are likely to decrease runoff and nutrient attenuation, de-stabilize streambanks, reduce stream shading, and reduce important inputs of woody debris to the channel (Booth and Jackson 1997; Paul and Meyer 2001; Walsh et al. 2005; Wheeler et al. 2005; Schiff and Benoit 2007).

Benthic macroinvertebrates are the most studied component of urban streams. Because of their widespread research base, macroinvertebrates are one of the most useful tools for comparing interregional variation in responses to urbanization or other land use change (Walsh et al. 2005). Increased TIA is linked to shifts in macroinvertebrate community from high diversity of sensitive species to populations dominated by pollution tolerant taxa (Chadwick et al. 2006). Organisms from the families Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) represent a large portion of the pollution sensitive macroinvertebrate community and therefore are most responsive to urbanization (Wheeler et al. 2005). Macroinvertebrate communities in impaired streams have altered functional feeding group composition that can strongly impact nutrient cycling, processing of organic matter, and secondary production (Chadwick et al. 2006). Leaf shredding taxa are typically the least pollution tolerant and the loss of these organisms can impact the processing of terrestrial organic matter and the overall food web within the stream (Wheeler et al. 2005). However, some studies have found that reduced biological processing rates for terrestrial organic matter may be offset by increased physical abrasion during high flow periods associated with streams in developed watersheds (Meyer et al. 2005; Chadwick et al. 2006).

Fish communities show a very similar response to urban impacts with decreased biodiversity, loss of pollution sensitive species, and a loss of functional feeding group diversity (Wheeler et al. 2005). Stream temperature increases, from the loss of riparian shading and from warm water inputs through connected impervious surfaces, are a key factor for fish communities (CWP 2003; Wang et al. 2003; Herb et al. 2008). A study in

Vermont found that fish communities were impacted by geomorphic instability and habitat fragmentation associated with development (Sullivan et al. 2006).

Physical Impacts.

Stream hydrology alterations are the most widespread and consistently visible changes associated with urbanization (Walsh et al. 2005). The changes begin with the conversion of natural lands to a developed setting (i.e. paving) which causes a drastic reduction in the permeability of the soil and increases the amount of rainfall that flows over land surfaces as runoff (Paul and Meyer 2001). The drainage infrastructure that accompanies watershed development further impacts stream hydrology by creating pathways that increase the rate of delivery of runoff to receiving waters (Hatt et al. 2004; Galster et al. 2006). Until very recently, the primary purpose of drainage infrastructure was to convey water away from buildings and roads as quickly as possible; current stormwater management efforts now focus on slowing and storing stormwater runoff where possible (Booth and Jackson 1997; Brabec et al. 2002). Stream channel modification (i.e. straightening) and floodplain disconnection or degradation are associated with all levels of development and increase the speed and efficiency of drainage through a watershed (Paul and Meyer 2001; Schiff and Benoit 2007; Jacobson 2011).

The hydrologic impacts associated with urbanization result in stream hydrographs that are altered during high flow events and under baseflow conditions. Urban streams are described as “flashy” and have taller and narrower flow peaks during runoff events as compared to undeveloped watersheds as shown in Figure 2 (Paul and Meyer 2001; Walsh et al. 2005). Reduced infiltration in these watersheds decreases groundwater recharge from precipitation leading to reduced baseflow in receiving streams (Paul and Meyer 2001; Konrad and Booth 2005; Wheeler et al. 2005; Jacobson 2011). Reduced summer baseflow amplifies the stream temperature, dissolved oxygen, and chemical impacts also associated with the urban stream syndrome and represent a major stressor for aquatic biota (Richter et al. 1996; Herb et al. 2008).

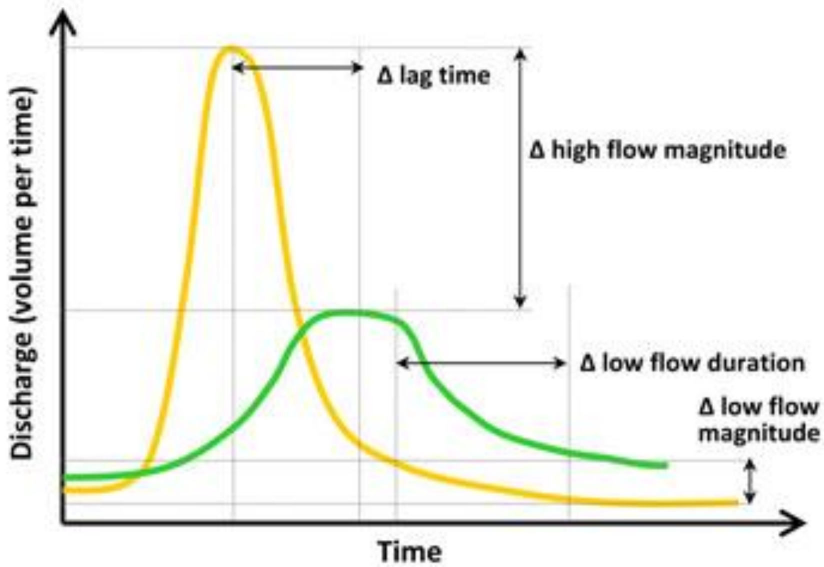

FIGURE 2. Hypothetical hydrograph for an urban watershed (yellow) as compared to an undeveloped watershed (green) from USEPA (2012).

Stream geomorphic processes are impacted by the altered sediment supply and hydrology in urban watersheds. The construction phase of watershed development introduces large volumes of terrestrial sediment to stream channels leading to aggradation. Floodplain access may actually increase during this phase as bar building processes decrease the channel area. Sediment supply decreases following the completion of construction and the stream channel enters the erosional phase (Paul and Meyer 2001).

The increased frequency of high flow events associated with flashy streams cause channel forming flows to occur more frequently than in undeveloped watersheds (Paul and Meyer 2001; Walsh et al. 2005). These events lead to channel incision as the channel bed and banks erode, resulting in streams that are typically deeper and wider than undeveloped streams (Booth and Jackson 1997; Wheeler et al. 2005; Kang and Marston 2006; Jacobson 2011). As erosion becomes the dominant geomorphic process, channel derived fine sediment loading can increase by several orders of magnitude and have widespread impacts on biota (CWP 2003; Wheeler et al. 2005; Schoonover et al. 2007).

Riparian forests are typically impacted at even the lowest levels of development. Removal of native woody vegetation along stream banks and adjacent floodplain reduces bank protection and further increases erosion (Kang and Marston 2006). If banks are re-vegetated after development is it typically by shallow-rooting plants and grasses that provide much less stability than native vegetation (Booth and Jackson 1997). The linear nature of road networks and the prevalence of roads constructed near or immediately adjacent to surface waters increases impacts to the riparian buffer and many other road-specific geomorphic impacts that are discussed later in this review.

Spatial Scale of Watershed Impacts

Many studies have found spatial scale to be an important consideration for predicting and understanding different impacts from development (Urban et al. 2006; Schiff and Benoit 2007). Impervious surfaces that are located close to surface waters are very likely to be directly connected via overland runoff or drainage infrastructure (Schiff and Benoit 2007; Wemple 2013). Studies have defined a buffer zone ranging from 30m to 100m to describe connected impervious surfaces with the greatest impact on receiving waters (Wang et al 2003; Schiff and Benoit 2007; Wemple 2013). Schiff and Benoit suggest that a 100m buffer area is the best predictor of biota and stream habitat impacts, and that stream chemistry impacts are best correlated with watershed scale predictors (2007). Macroinvertebrate community responses at very local scales have also been observed from samples collected upstream and downstream of a single road crossing (Wheeler et al. 2005).

Specific Impacts from Transportation Infrastructure

Road networks represent a major feature in the developed landscape. Many of the impacts observed with watershed urbanization are closely linked with transportation infrastructure (Eyles and Meriano 2010). A study of roads networks in rural Vermont watersheds found that 33-75% of roads were within 50m of streams and were likely directly hydrologically connected (Wemple 2013). The linear nature of roads and the associated drainage infrastructure can significantly increase the portion of connected impervious surfaces within a watershed and can extend the drainage area outside of the topographic watershed boundary (Noll and Magee 2009). Roadway drainage infrastructure may also intercept natural topographic flow paths and bypass floodplains and other storage areas (Wemple 2013).

Three phases of road impacts are described in Angermeier et al. (2004) and Wheeler et al (2005): road construction, road presence, and watershed urbanization. The road construction phase begins at the onset of development within a watershed and is characterized by short-term and typically local impacts from severe fine sediment loading and mechanical disturbance. The road presence phase will be discussed in greater detail; this is characterized by long-term impacts to hydrology, geomorphology, and stream chemistry, resulting in significant and permanent responses from biotic communities. Intermittent maintenance activities and seasonal road treatments (winter deicing practices) can further stress biota. The final phase of road impacts is the general increase in watershed development that is facilitated and directed by transportation infrastructure, these impacts were described earlier as the "urban stream syndrome".

Stream geomorphology is impacted during road construction as channelization and bank armoring are frequently used to lock the channel in place and protect infrastructure. Streambank and riparian vegetation may also be removed or degraded (Wheeler et al. 2005; Sullivan et al. 2006). Bridges and culverts at road crossings can have major geomorphic impacts on streams that can extend upstream and downstream far beyond the actual structure. Structures with fixed bottoms (e.g. culverts) can cause sudden changes in channel slope (Paul and Meyer 2001; Wheeler et al. 2005). Bridges and more commonly culverts may constrict channel flow and floodplain width, altering the downstream transport of organic matter, woody debris, sediment, and floodwaters (Angermeier et al 2004; Wheeler et al. 2005). Road crossings are particularly vulnerable to debris jamming during large flood events potentially causing catastrophic damage to adjacent infrastructure and the downstream channel (Wheeler et al. 2005).

Many of the water chemistry impacts associated with urbanization are closely associated with roads and automobiles. In non-industrial watersheds, traffic residues on roads are the most common source of heavy metals (cadmium, chromium, copper, iron, lead, nickel, and zinc) and polycyclic aromatic hydrocarbons (PAHs), which are important toxicity concerns for stream biota and human health (Wheeler et al. 2005; Yang 2010). Chemicals stored in road dust and oil and grease from crankcase drippings are typically stored on the road surface or in roadside ditches. These chemicals rapidly move into receiving surface waters during runoff events and can cause acute or chronic toxicity, particularly in smaller streams (Wheeler et al. 2005; Eyles and Meriano 2010). Hazardous material spills are a concern for waterways near major roads. On average approximately 10,000 accidents involving hazardous materials occur annually on U.S. roads. Bridges are inherently higher-risk for hazardous waste spills especially during winter months (Wheeler et al. 2005).

Sediment Impacts from Non-Paved Roads.

Non-paved roads are an important component of the transportation network in rural watersheds. These roads are a large and continuous source of sediment loading to surface waters through erosion of the road surface (Lane and Sheridan 2002). Sediment generation from temporary logging roads has been well studied; however less research has been conducted on maintained non-paved roads in rural settings (Lane and Sheridan 2002; Luce 2002; Sheridan and Noske 2007; Jordan and Martinez-Zavala 2008; Wemple 2013). A recent study of non-paved roads in rural Vermont watersheds found very high loading rates of sediment and phosphorus from roads, especially in steeper watersheds (Wemple 2013). Intensive sampling of storm event runoff from a series of road segments estimated annual sediment loading rates ranging from 1 to over 100 MT/km/year. Extrapolating these results and the results from an inventory of discrete hydrogeomorphic impairments, non-paved roads are estimated to produce 31% of the sediment load and 11% of the phosphorus load to the Winooski River, a large watershed with a wide range of forest, agriculture, and developed land (Wemple 2013). Non-point pollution from non-paved roads has been identified as a major concern in basin action plans in Vermont and will likely become an important research and management priority (VVCAP 2009).

Winter Deicing Chemical Impacts.

Winter deicing chemical application is a critical maintenance activity in colder regions such as Vermont. Nationally, over $2 billion is spent each year on winter maintenance chemicals, materials, and labor. Sodium chloride (NaCl) is the most widespread and affordable deicing chemical; however other chemicals (i.e. $CaCl_2$ and $MgCl_2$) may be used for specific areas or during very cold periods. Deicing chemicals applied to roads are transported as snow and ice melts. Salt residue may remain on the road and on nearby soils and vegetation, or dissolved salts enter shallow groundwater or flow directly to streams in surface runoff. Drainage infrastructure associated with roads is very important for determining the speed of delivery of deicing chemicals to receiving surface waters (Denner et al. 2009).

All surface waters contain some natural concentration of Cl, primarily from rock weathering, windblown dust, and precipitation. Anthropogenic sources not related to deicing also include, sewage, septic, and agriculture (Denner et al. 2009). A study of chloride sources in upstate New York found that over 90% of the Cl contributions in a rural watershed were derived from road salt, and that less than 2% come from natural sources (Kelly et al. 2008).

Chloride concentrations in Vermont surface waters have been observed near and above the EPA criterion for chronic exposure of 230 mg/L (USEPA 1998; Denner et al. 2009). These concentrations were observed in a small tributary to Alder Brook that drains a large area of impervious surfaces. The Denner study found that peak chloride loading occurred during the spring snowmelt. Chloride concentrations in streams tend to be inversely proportional to discharge and many studies have found that concentrations peak during the late summer period when stream flow is lowest and groundwater contribution is proportionately greatest (Wheeler et al. 2005; Kelly et al. 2008; Denner et al. 2009; Eyles and Meriano 2010). A study in upstate New York found stream chloride concentrations that were increasing at a faster rate than watershed application, suggesting a buildup of subsurface chloride and a multiple year lag time to reach surface waters (Kelly et al. 2008). State and municipal road maintenance crews in Vermont are well aware of chloride impacts and employ many practices and technologies to minimize salt application while maintaining safe driving conditions. Vermont is the only state to require an environmental permit for salt application and there is no “bare road” designation for snow clearing. Despite these efforts, chloride is a major management concern for surface waters in the Lake Champlain basin and across Vermont (Shambaugh 2008; Denner et al. 2009).

CHAPTER 2: JOURNAL ARTICLE

Project Background

Numerous studies have been conducted that clearly describe the impact of urbanization at broad (catchment) to fine (parking lot) scales (as reviewed in Paul and Meyer 2001). These studies have quantified and described myriad physical, biological, and chemical impacts to streams collectively known as the urban stream syndrome (Walsh et al. 2005). Typically characterizations of impacts from development are based on the percentage of the watershed covered in impervious surfaces (%TIA) the most common variable used to estimate the level of development and associated impacts. Methods for estimating watershed imperviousness vary widely in required effort and in accuracy. As such, it can be challenging to compare results across studies (Brabec et al. 2002).

The effects of dense urbanization on watersheds are well studied. Less research has been done for watersheds that are characterized by lower levels of urbanization (Wang et al. 2003). In studies of densely urbanized areas it is not uncommon to consider 30-40% TIA to be a moderate level of development (Chadwick et al. 2006). By contrast, the level of development in smaller, less populous states like Vermont is considerably lower; typically 10-20% TIA in the most developed watersheds in Vermont. However, despite the lower level of overall development in Vermont's urbanized areas (measured as TIA), roads still constitute an important fraction of the developed area and may be an important source of impairment to local streams. The specific impacts of roads are not often isolated in studies of moderately developed urban and suburban watersheds. Road network information is easily and publicly available and the close association with overall watershed development suggests that road networks may be a useful and simple indicator of potential surface water impacts within a watershed. Specifically, we reason that:

- Road surfaces are major sources of runoff and stormwater pollutants.
- Roads are frequently one of the largest sources of watershed imperviousness, especially in the low to moderate levels of development most commonly found in Vermont.
- Roads are the primary source of chloride loading to Vermont streams due to winter deicing activities.
- Roads are frequently associated with drainage infrastructure that provides direct or expedited pathways for runoff to enter streams.
- Road networks are simple to map and can be consistently applied to a wide range of watershed development levels.

Local Context

In the Lake Champlain Basin and Chittenden County, sediment, nutrients, and chloride from road salting are the pollutants that most often are cause for management concern. Because the stormwater generated from impervious surfaces itself alters the hydrology and geomorphology of streams and rivers, the State of Vermont's stormwater control approach focuses first on controlling the discharge of water in developed watersheds (VTDEC 2013). The water as well as the sediments, nutrients, and other pollutants in stormwater alter habitat quality and the species composition of streams and rivers (e.g., Jackson et al. 2001; Par and Mason 2003; Sullivan et al. 2006) but equally, the increased pollutant load moving downstream frequently leads to impaired receiving waters, as it the case for Lake Champlain (VTDEC and NYSDEC, 2002). To make good decisions about transportation futures, managers need information about the magnitude of these problems and how different road types and densities affect stream networks and receiving waters.

Research Goal, Objective, and Hypotheses

The primary goal of this research was to evaluate the effects of the transportation network on water quality and freshwater ecosystem integrity. We characterized the effects of road type and road density on water quality, stream stability, and the pollutant load exported to Lake Champlain. We hypothesized that:

- Metrics generated from relatively simple spatial analysis of road networks and streams can be used to predict water quality impacts and stream conditions at a watershed scale.
- Water quality indicators will decrease along a gradient of increasing road density.
- Unpaved roads will generate proportionately more sediment loading than paved roads.

Study Area

Chittenden County is located in northwestern Vermont and is closely connected to two of the most important aquatic resources within the state: Lake Champlain and the Winooski River. Stormwater runoff and water quality impacts on receiving waterbodies are a primary concern for residents and municipalities within the county. Chittenden County contains the highest density of development in Vermont, including segments of Allen Brook and Potash Brook which have been listed by the state of Vermont as "impaired" by stormwater runoff. The Vermont Department of Environmental Conservation has developed total maximum daily load (TMDL) plans for some of these streams and is actively working to improve water quality and reduce hydrologic impairment in these watersheds. More information is available at: http://www.watershedmanagement.vt.gov/stormwater/htm/sw_impairedwaters.htm.

The watersheds selected for this study spanned the range of the road network density and road type found within Chittenden County. Watershed areas range from 13 to 53 km^2 and stream order is 3rd or 4th. Table 1 includes general watershed characteristics and land use summaries in order of increasing development and road network density (VCGI 2011). A map of the study watersheds and monitoring station locations is shown in Figure 3. In this table and subsequent tables the study watersheds are organized from least to most developed.

TABLE 1. Watershed and landuse characteristics for the study area.

Watershed	Size (km^2)	Order	M/H Dev**	L/O Dev*	Forest	Shrub/ Scrub	Pasture	Crop	Water/ Wetland
Snipe	13.1	4	0%	2%	93%	1%	1%	0%	3%
Mill	29.8	3	0%	4%	85%	1%	4%	2%	4%
Alder	25.5	4	3%	15%	41%	1%	28%	7%	5%
Allen	16.8	3	1%	15%	41%	2%	30%	4%	7%
Muddy	53.3	4	7%	12%	36%	2%	24%	8%	11%
Potash	18.2	3	27%	42%	11%	1%	12%	5%	2%

*Denotes high intensity and moderate intensity development.
** Denotes low intensity development and urban open space

The two rural watersheds (Mill Brook and Snipe Island Brook) are primarily forested and contain low densities of gravel and dirt roads. Both watersheds contain low densities of single-family homes and minimal agriculture. The watersheds are the steepest in this study and the main stream channels are located in narrower valleys typically shared with roads.

Alder and Allen Brook drain primarily suburban watersheds with moderate forest and agricultural cover. The lower portion of Alder Brook closely follows a 4-lane highway (I-289). Both watersheds drain areas of suburban residential development typical to Chittenden County.

The Muddy Brook watershed has the most variable land use. The upper watershed (southern) drains rural and suburban areas; the lower watershed (northern) drains dense commercial areas and has a high concentration of major roads (Rt. 2 and I-89).

Potash Brook is one of the most developed watersheds in Vermont and represents the highest degree of residential and commercial development in this study. The entirety of I-189 and significant stretches of I-89, Rt. 2, Rt. 7, and Rt. 116 area located within the watershed.

Allen Brook and Potash Brook are listed by VTDEC as biologically impaired due to stormwater and have approved Total Maximum Daily Loads (TMDLs) prepared by VTDEC for the U.S. Environmental Protection Agency (2006; 2008).

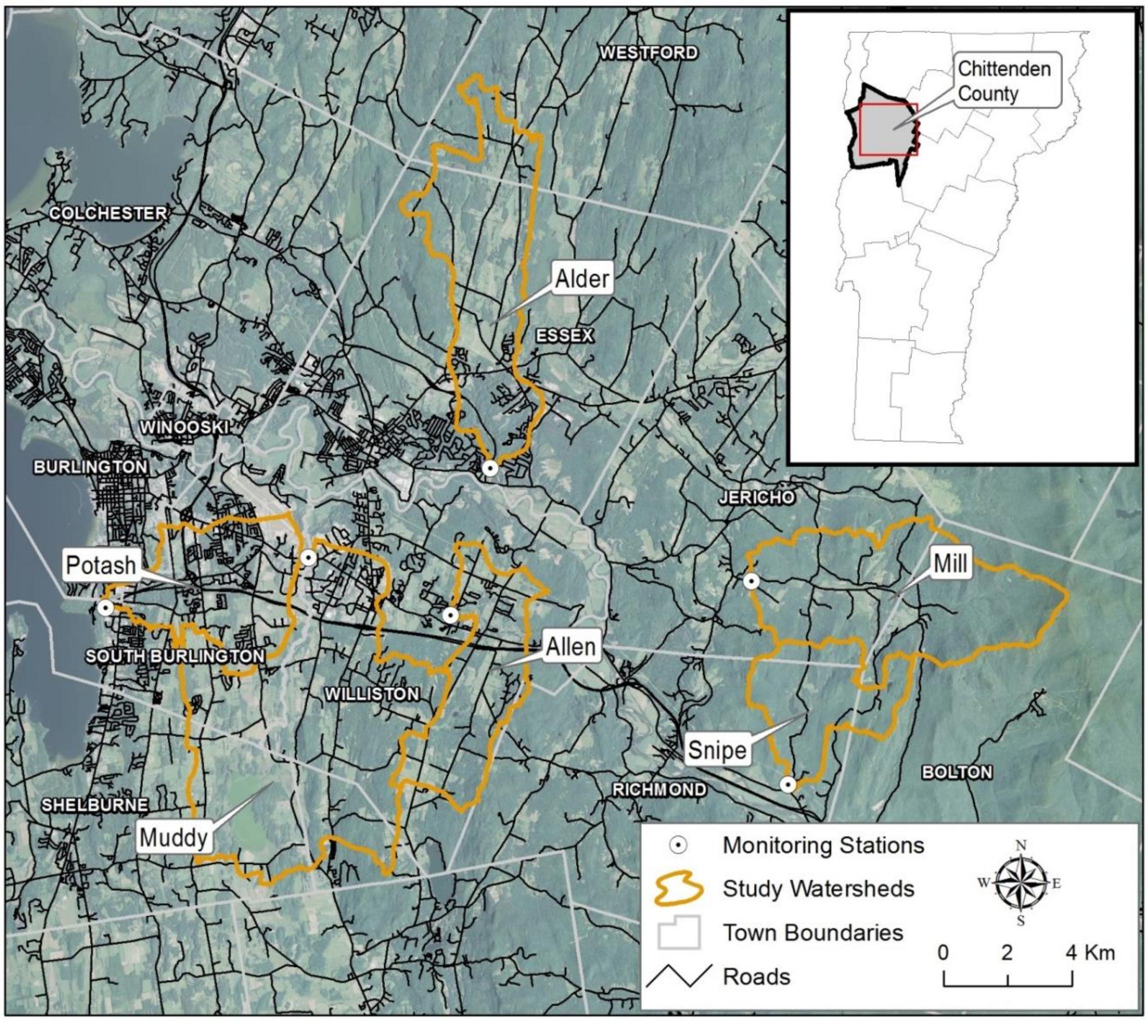

FIGURE 3. Location of the study area in northwestern Vermont including the six monitoring stations with upstream watershed boundaries.

METHODS

Monitoring Station Location, Construction, and Instrumentation

Locations for continuous monitoring stations were identified based on channel stability, substrate, protection from flooding, access, and security. Each monitoring station included an ISCO 6712 auto-sampler with an ISCO 720 pressure transducer and a YSI 6600 OMSv2 sonde outfitted with temperature, specific conductance, and optical dissolved oxygen (DO) sensors. Weather stations were installed near the monitoring stations and were instrumented with HOBO micro-stations, photosynthetically active radiation sensors (PAR), and tipping rain buckets (0.2mm increment). ISCO auto-samplers were installed above the flood-prone elevation and were tethered and locked to trees. Pressure transducer and suction lines were housed in flexible plastic conduit and were staked to the bank with rebar and mounted to a PVC carrier staked to the stream bottom with 4' rebar. YSI sondes were bolted in to heavy PVC holders that were staked to the stream bottom in the thalweg with 4' rebar. Equipment was installed as early as spring flow levels allowed and remained in place until the onset of anchor ice in late December. Monitoring stations in all watersheds were operated from June-December 2008, April-December 2009, and April-December 2010.

Continuous Monitoring and Maintenance

Stream temperature, specific conductance, dissolved oxygen saturation, and stage height were measured continuously at 5 minute intervals. YSI sondes required weekly maintenance for cleaning, DO calibration, and battery changes. Field calibrations of DO followed the manufacturer listed calibration methods: wrap sonde in a wet towel to stabilize temperature and DO saturation, and then calibrate using a handheld digital barometer. Conductivity sensors were calibrated in the lab using 3 standards (1, 100, 1000 us/cm) at the beginning of each field season and checked for drift at the end of each season. Dissolved oxygen calibrations were rarely greater than 1% and no sensors indicated annual conductivity drift greater than 1%.

Discharge Rating Curves

Stage/Discharge rating curves were established for each stream using approximately 10-15 manual area-velocity discharge measurements taken at a range from baseflow to highest wadeable flow levels. Discrete discharge measurements were collected with a Sontek Flowtracker 2D ADV. Rating data were plotted in Microsoft Excel and fitted with single or two-part power curves following standard USGS stream rating methods (Turnipseed and Sauer 2010).

Storm Event Sampling

ISCO auto-samplers were programmed to collect stage triggered, time-paced, single composite samples into a 9L plastic jug. Stage triggers and sampling intervals were programmed before each storm event and an enable triggering stage threshold was selected based on current flow conditions and predicted storm forecast. Samplers collected up to 36 samples (200ml per sample) into the composite jug. We acknowledge that flow-weighted sampling would be preferable to more accurately estimate solute loads. However, in most cases we did not have good information about the discharge characteristics of these streams prior to study and could not develop the necessary rating curves to program samplers for volume weighted samples a priori. In addition, it is logistically difficult to simultaneously collect good flow-weighted sample at multiple flashy and poorly characterized developed watersheds. Thus, there is a high risk that samples and data will be lost at one or more sites during any given storm. Given that our primary objective was a comparative study of differently developed watersheds and not a quantitative study of area-specific loading from the study watersheds, we concluded that this tradeoff was acceptable. It is likely that by compositing samples taken over regular time intervals during storms, we have underestimated the true loads of sediment and nutrients. Thus, the actual differences among our study watersheds may be larger than we have reported.

We found that a 30 minute sampling interval was ideal for most storms and successfully captured the rising limb, peak, and most of the falling limb without over sampling any particular period of the storm. Stage enable triggers were typically set to 2cm above the stream level prior to the storm. A total of 28 to 35 storm events were successfully sampled at each site.

Monthly Grab Sampling

Monthly water quality grab samples were collected for December 2008 through January 2011 to provide a year-round water chemistry record. This sampling included winter months when continuous monitoring stations were decommissioned due to freezing conditions. Grab samples were collected by directly filling sampling containers or using a clean 9 liter jug during icing conditions. All samples were collected under stable low flow conditions to best characterize baseflow and seasonal water quality conditions.

Water Sample Processing and Analysis

Composite storm event and baseflow grab samples were split into individual sample bottles for each analyte and were stored at the University of Vermont Rubenstein Lab until analysis. Total phosphorus (TP) samples were stored frozen in 150ml plastic bottles. Total nitrogen (TN) samples were stored in 50ml conical tubes, acidified with H_2S0_4, and refrigerated. Chloride samples were collected in 10ml scintillation vials and refrigerated. TSS samples were collected from the 9 liter jugs immediately following vigorous shaking. Water was filtered through pre-combusted 47mm Type 934-AH GF filter paper using a hand vacuum pump and Nalgene 500ml vacuum filter apparatus. Filtered sample volumes ranged from 30ml to 5,000ml depending on sediment load in samples. Filter papers were dried for at least 24 hours in pre-combusted and weighed aluminum tins. Dried samples were allowed to cool in a desiccator and then weighed. Samples were then combusted in a 550^{0}C muffle furnace for 4 hours to remove organic matter and re-weighed.

Frozen TP samples were thawed in a refrigerator and were analyzed on a Lachat auto-analyzer using the Quick Chem Method 10-115-01-4-F, determination of total phosphorus by flow injection analysis colorimetry (acid persulfate digestion method). Total nitrogen samples were analyzed using Lachat Quick Chem method 10-107-04-4-A, determination of nitrate+nitrite in manual persulfate digests. Chloride samples were diluted so that all samples would range from 0-10 mg/l Cl. Dilutions were 1:3 for rural streams and up to 1:50 for urban streams. Dilutions were determined based on specific conductance readings at the time of sample collection. Chloride samples were analyzed by the University of Vermont Agricultural and Environmental Testing Laboratory on a Dionex ion chromatograph.

All TP and TN samples were run in duplicate and standard checks were performed every 10 samples. Any samples with greater than 10% difference were re-run automatically by the Lachat auto-analyzer. Chloride samples were also run in duplicate and included a 10% field replicate.

Benthic Macroinvertebrate Sampling

Benthic macroinvertebrates (BMIs) were sampled at all sites during the late fall index period (September-early October 2010). Samples were collected using a 500µm kick net placed in four locations to best characterize the flow depths, velocities, and substrate size present in a single riffle sequence. The substrate and organic matter at each sampling location was vigorously scrubbed into the net for a total sampling effort time of approximately 2 minutes. Samples were rinsed, sieved, and preserved in 70% ethanol (VTDEC 2004). We partnered with VTDEC Biomonitoring and Aquatic Studies Sections (BASS) to share in sampling effort and analysis costs. VTDEC collected annual samples at Potash Brook, Muddy Brook, and Alder Brook, and BMIs were picked and identified at the VTDEC lab. We collected and processed two replicate samples from Allen Brook, Mill Brook, and Snipe Island Brook and picked samples were analyzed by Rapid Watershed Associates (Schenectady, NY). BMI community composition was characterized using the suite of VTDEC metrics to describe density, diversity (taxa richness, EPT richness), pollution tolerance (index of biotic integrity, %Oligochaeta, Chironomidae metrics), similarity to reference communities, and functional feeding group composition (VTDEC 2004).

Rapid Habitat Assessment

Rapid Habitat Assessments (RHAs) were completed at all sites following the VTDEC guidelines (VTDEC 2009). RHA scores describe woody debris cover, bed substrate cover, scour and depositional features, channel morphology, hydrologic characteristics, connectivity, river bank condition, and riparian condition. Each stream was assessed over an approximate 100m reach centered on the monitoring station. All categories were scored from 0 (worst) to 20 (reference) for each RHA category and the total score was used to assign a habitat condition rating.

GIS Analysis

Watersheds for each study stream were delineated using the ArcHydro tool in ESRI ArcMap 9. All data layers were clipped to watershed boundaries. Road networks were characterized based on road surface type from the VTRANS TRANS_RDS database. Class 4 roads were manually classified based on ground observations and aerial imagery. Road networks were also characterized based on proximity to streams. Road crossings within each watershed were counted and road lengths were measured within a 100m buffer from the stream centerline. We selected a 100m buffer to best capture the portion of roads that directly impact neighboring streams as described by Schiff and Benoit (2007).

We conducted a simple analysis of total impervious area and road impervious area in the study watersheds. Total percent impervious area (%TIA) was estimated for each watershed based on correlations between statewide 30m Landsat imagery and detailed Quickbird satellite imagery that measured impervious area in portions of the study area, as described by Fitzgerald (2007). Road impervious area was estimated for each watershed by manually measuring width for at least 50 randomly selected road segments for each of the four AOT classes within each watershed (highway; paved and ditched; gravel and ditched; gravel/dirt no ditch). A mean width was determined for each surface type by watershed and was multiplied by total length of each road class to estimate total road area within each watershed. This calculation method indicated that road area represented over 90% of the estimated total imperviousness within the Snipe Island Brook watershed. Due to the small size and low level of development we manually measured all non-road imperviousness (less than 100 driveways and rooftops) within the watershed and on the basis of this analysis we adjusted the %TIA from 0.5% to 0.76%.

Data Analysis

Continuous data (5-minute) from ISCO and YSI sensors were converted to Microsoft Excel-readable formats and then compiled into annual master datasets. All data were manually screened for errors. YSI files were trimmed by 1 to 2 readings on the start and end of each weekly deployment to remove data points influenced by sonde temperature changes caused by instrument downloading and calibration. Daily mean values were calculated for all continuous variables. Daily mean streamflow data were used to calculate flow duration curves and additional metrics for baseflow contribution and the flood-peak index (Hauer and Lamberti 2006). Additional metrics were calculated for stream temperature data (maximum daily mean, maximum 7-day mean, and mean of the daily maxima for first 3 weeks of July) as described in Wang et al. (2003).

All water quality data were summarized to generate mean concentrations for storm event and baseflow grab samples. Mean values were tested for significant differences between sites for baseflow and storm event samples, and for differences between baseflow and storm event concentrations within each watershed using ANOVA (α=0.05) with the JMP 10 statistical software package. A multivariate analysis with Spearman's ρ correlation was selected to test the strength of road network variables as predictors for physical, biological, and chemical responses (α=0.05 and 0.1).

RESULTS

GIS Results

Road networks within each watershed were characterized by length within each major AOT category (highway; paved and ditched; gravel and ditched; gravel/dirt no ditch) (Table 2). The Potash Brook watershed contains several major roads and highways (I-89, I-189, US 2, US 7, and VT 116), a high density of residential roads, and a very low density of unpaved roads. Muddy, Allen, and Alder Brooks all drain portions of highway and major roads, large networks of residential roads, and small to moderate densities of unpaved roads. Mill and Snipe Island Brooks drain watersheds with predominantly gravel and dirt roads with only a small stretch of paved roads in the upper Mill watershed.

TABLE 2. Length of road (km) by type in each study watershed.

Watershed	Highway	Paved	Gravel w/ ditch	Gravel/Dirt	Paved Total	Unpaved Total
Snipe	0.0	0.0	4.7	3.0	0.0	7.7
Mill	0.0	1.4	29.2	2.9	1.4	32.1
Alder	8.7	30.9	5.3	0.0	39.6	5.3
Allen	4.3	46.9	16.0	0.4	51.2	16.3
Muddy	10.2	81.5	25.6	1.0	91.7	26.6
Potash	21.8	91.8	0.9	0.0	113.6	0.9

Road network metrics and watershed impervious cover results are shown in Table 3. The road density within 100m of the stream centerline indicates a higher proportion of roads in the rural watersheds closely follow streams. Total impervious area percentages characterize the overall development level for each watershed and show the elevated percentage of watershed imperviousness represented by roads in moderately developed watersheds with higher densities of major roads (Allen and Alder) and in low development rural watersheds (Snipe). Gravel roads in the Allen and Muddy watersheds are located near the headwaters; the portions of the watershed closer to the monitoring stations are dominated by paved roads.

TABLE 3. Characteristics of the road network in each study watershed.

Watershed	Stream-Road Crossings/km^2	Road Density (km/km^2)	Road Density within 100m stream buffer (km/km^2)	%TIA	%TIA from roads
Snipe	0.53	0.63	0.34	0.8	86.2
Mill	0.74	0.99	0.49	1.6	33.1
Allen	1.31	2.56	0.60	4.1	59.6
Alder	0.78	2.20	0.55	4.5	51.8
Muddy	1.33	2.22	0.57	6.0	30.3
Potash	3.41	5.41	1.62	22.0	22.2

Stream Flow

Discharge estimates were collected across a wide range of wadeable flows for all six monitoring stations. We developed one-part or two-part rating curves to best fit discharge (y) to stage height (x) shown in Table 4. Changes in channel dimensions typically located near bankfull at five of the monitoring stations (Allen, Mill, Muddy, Potash, and Snipe) required the use of a lower and upper rating curve to best fit flows above and below bankfull (Figure 4).

TABLE 4. Discharge rating curve equations.

Watershed	**Low Curve**	**R^2**	**High Curve**	**R^2**	**Transition Stage (m)**
Snipe	$y=79.53x^{6.13}$	0.98	$y=10.28x^{4.17}$	0.96	0.35
Mill	$y=7.32x^{2.09}$	0.99	$y=17.22x^{2.93}$	0.96	0.36
Alder	$y=32.46x^{3.00}$	0.99	n/a		n/a
Allen	$y=59.79x^{4.14}$	0.96	$y=10.2x^{2.79}$	0.97	0.28
Muddy	$y=16.08x^{3.50}$	0.98	$y=9.0x^{2.93}$	0.99	0.36
Potash	$y=0.341x^{0.23}$	0.98	$y=0.399x^{0.50}$	0.99	0.30

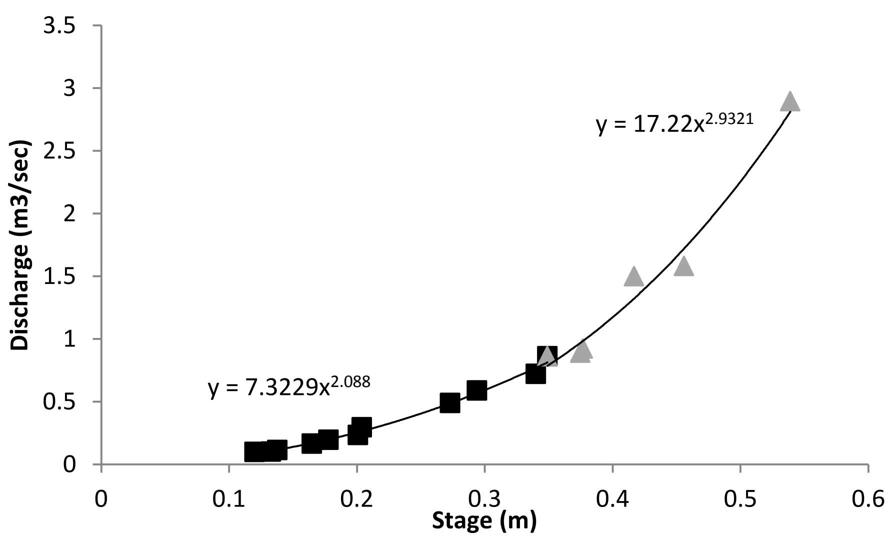

FIGURE 4. Example of a two-part rating curve from Mill Brook.

Discharge above the extent of the rating curve measurements is estimated and therefore we expect a greater potential for error in the highest discharge estimates. We prioritized discharge measurements across the full range of wadeable flows and successfully measured all but the highest observed discharges. The total proportion of time each station was above the extent of our rating curves ranged from 0.14% (Potash) to 2.4% (Snipe) with a mean of 1.4%.

Flow duration curves based on area-normalized hourly mean flows ($m^3/sec/km^2$) were calculated for each site to characterize changes in peak and base flows based on watershed imperviousness (Figure 5). We observed a large decrease in baseflow discharge in the three most developed watersheds as described in Booth and Jackson (1997) and CWP (2003). Although the peak flow volumes in the developed watersheds were higher than the rural watersheds, this relationship was not significant. Allen Brook was observed to have consistently lower mean discharge (Q50) and baseflow (Q90) than the other study watersheds. This is likely due to the well-drained sandy soils prevalent in much of the study watershed. The baseflow contribution metric (Q90/Q50) and the flood peak index both show a shift towards reduced baseflow and increased flood peaks relative to mean flows in the watersheds with higher road density and development (Table 5).

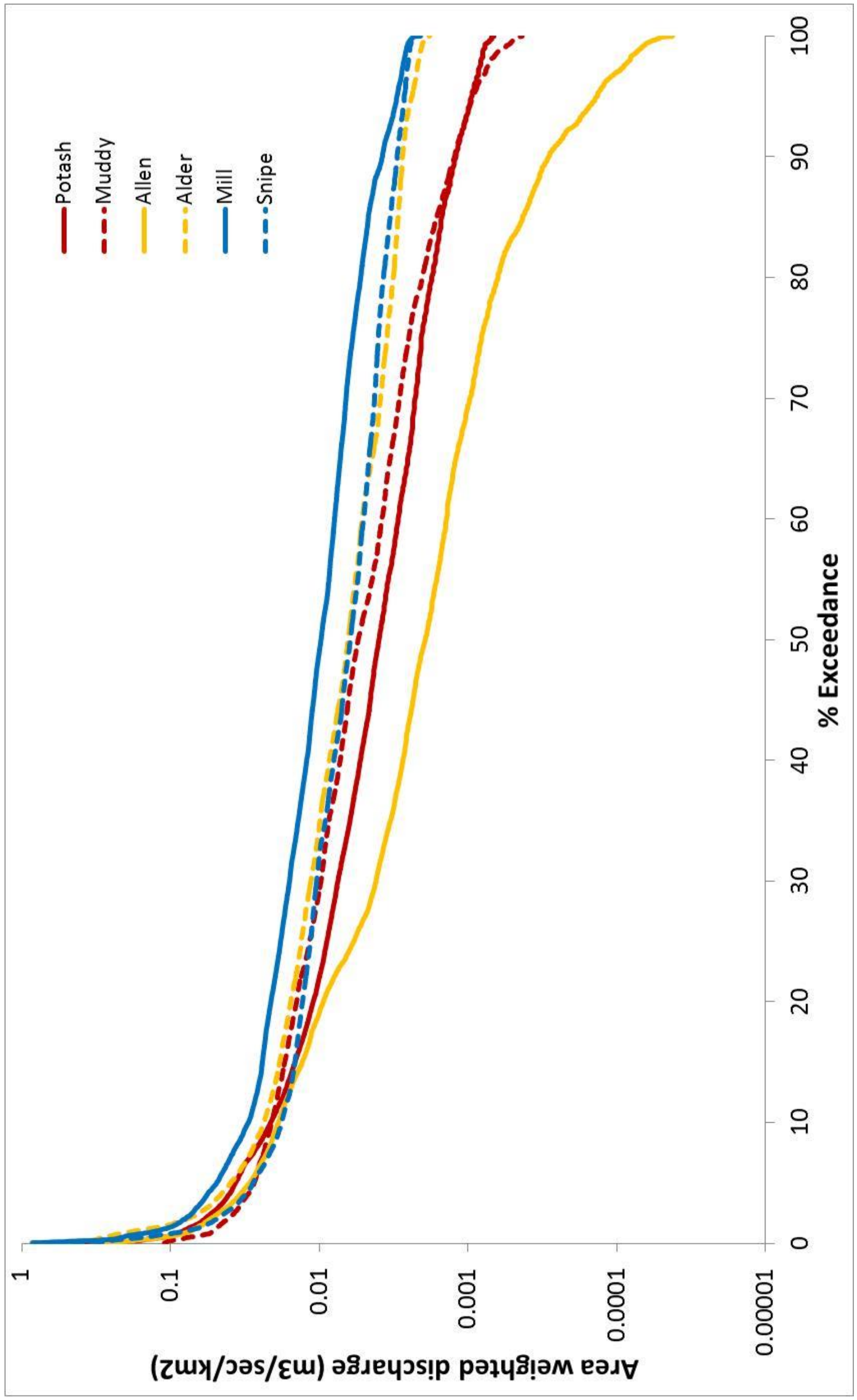

FIGURE 5. Flow duration curves based three-years of hourly flow data.

TABLE 5. Daily mean flow duration summary.

Watershed	Q10	Q50 Median	Q90 Baseflow	Q90/Q50 Baseflow Contribution	Q10/Q50 Flood Peak Index
Snipe	0.018	0.006	0.0030	0.484	5.97
Mill	0.030	0.010	0.0038	0.392	7.90
Alder	0.024	0.006	0.0028	0.438	8.43
Allen	0.019	0.002	0.0003	0.145	65.52
Muddy	0.021	0.006	0.0012	0.225	16.94
Potash	0.021	0.004	0.0012	0.300	17.67

Continuous Water Quality Results

The YSI multi-parameter sondes collected readings of water temperature, dissolved oxygen, and specific conductance at a 5 minute interval. We collected approximately 140,000 sets of these readings at each monitoring station over the duration of the project. These continuous data allow for observation of interactions between numerous parameters over a discrete rainfall event, as well as the characterization of water quality data over longer periods of time. Figure 6 shows 10 days of continuous data from Potash Brook. Regular daily fluctuations in temperature and dissolved oxygen are observed until a moderate storm on September 23, 2009. The storm event causes a small spike in water temperature as runoff is produced on hot surfaces (i.e. pavement and rooftops). Specific conductance is very high in Potash Brook throughout the summer as a result of high salt loading in groundwater, which will be discussed later in this section. Specific conductance levels are highest at low discharges when the groundwater contribution is greatest. The conductivity drops during the storm event as rainwater and surface runoff increase discharge.

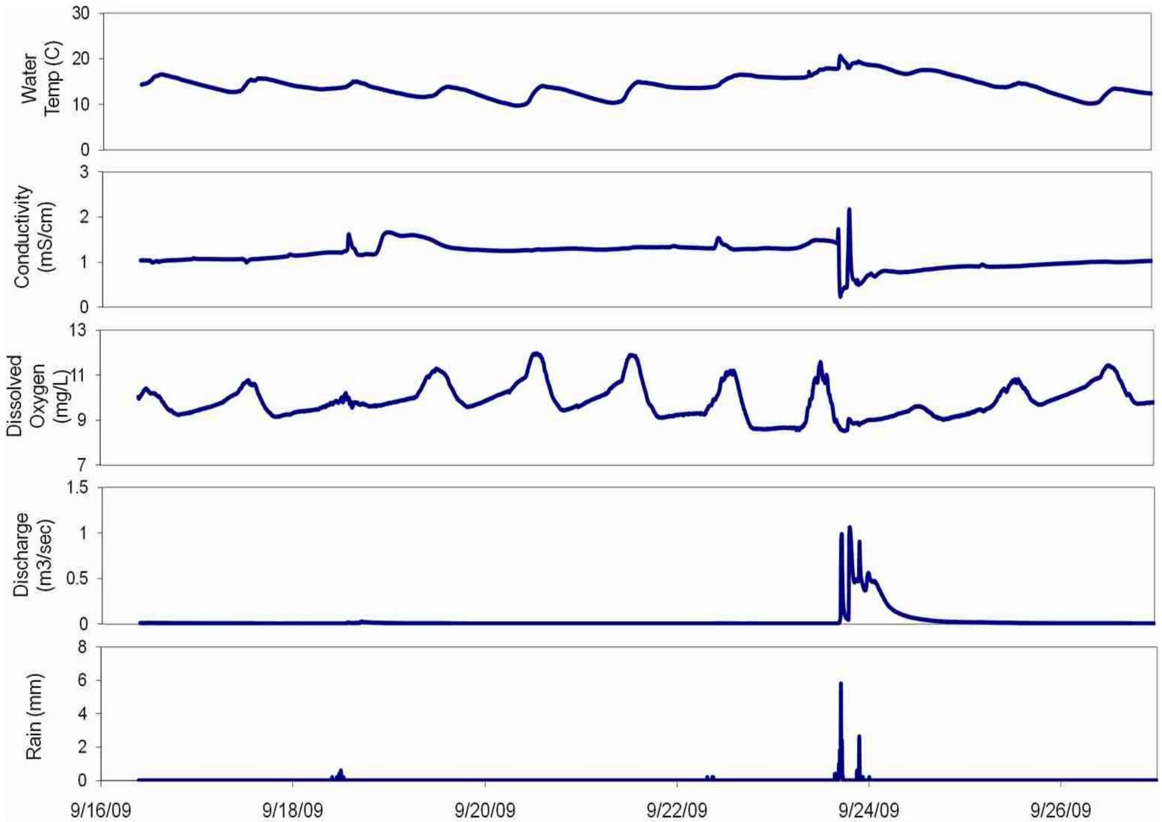

FIGURE 6. Water quality parameter values from continuous monitoring sondes in Potash Brook with a moderate storm on September 29, 2009.

Water temperature and dissolved oxygen concentration data were analyzed for minimum and maximum values for discrete readings, daily means, and seven day means as described in Wang et al. (2003). We observed increased maximum temperatures, decreased dissolved oxygen concentration, and large increases in the daily ranges for both of these parameters in the developed watersheds (Table 6).

TABLE 6. Water temperature and dissolved oxygen concentration summary data.

Watershed	Max Temp °C	7-day Max Temp °C	Max Daily Range	Min DO (mg/L)	7-day Min DO (mg/L)	Max Daily DO Range (mg/L)
Snipe	22.0	20.0	6.7	8.4	8.8	1.8
Mill	25.2	21.9	7.3	7.9	8.5	1.9
Alder	27.5	23.0	7.7	5.9	7.6	5.8
Allen	28.9	25.3	8.5	5.4	7.0	3.8
Muddy	31.1	26.5	7.1	6.7	7.7	5.2
Potash	27.4	23.8	6.2	6.2	8.0	6.0

High temperatures and low dissolved oxygen concentrations are major stressors for aquatic life in developed watersheds (Wang et al. 2003; Herb et al. 2008). We observed increased maximum temperatures in the more developed streams; however this is strongly influenced by shading of the channel within the reach immediately upstream of the monitoring station. Dissolved oxygen concentrations are closely linked with temperature and biological activity within the stream. The more developed watersheds had lower minimum oxygen concentrations and much higher daily variation in concentration. The minimum concentrations we observed in these streams are at or near the requirements (6.0 mg/L) for sensitive fish and macroinvertebrate species (Meador et al. 2008).

Baseflow and Storm Event Water Quality Results.

Baseflow grab samples were collected 2-4 times at each station in 2008 and then once a month at all stations from January 2009 through January 2011, for a total of 28-29 samples per watershed. Time-paced composite storm event samples were collected for 26 – 35 storms at each station. Baseflow concentrations of TP were lower than storm event concentrations at all sites and this difference was significant at Snipe, Alder, Allen, and Potash (Figure 7). Storm event concentrations were significantly higher at Alder and significantly lower at Mill compared to the remaining watersheds. Tukey-Kramer comparison of means for baseflow TP had three significant groups: Muddy was highest; Potash, Allen, Alder, and Mill were moderate; and Snipe had the lowest concentrations.

Concentrations of total nitrogen in baseflow and storm event samples were generally similar for each site; Snipe had significantly higher storm event concentrations (Figure 8). Tukey-Kramer comparison of means for both baseflow and storm event samples had three significant groups: Potash was highest; Alder, Allen, and Muddy were moderate; and Mill and Snipe had the lowest concentrations.

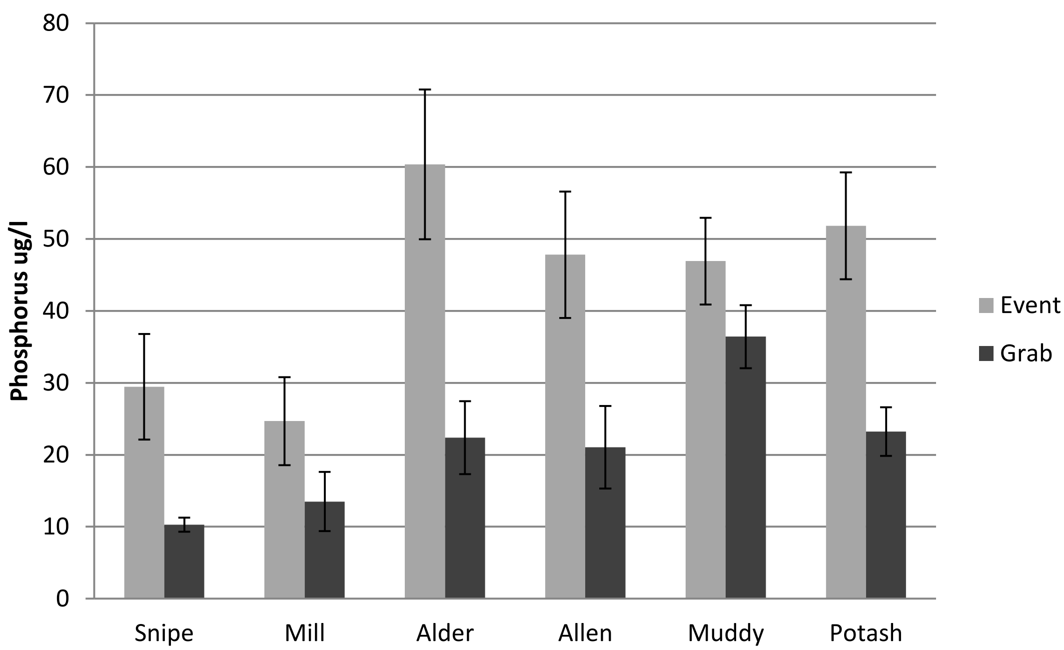

FIGURE 7. Mean total phosphorus concentrations for storm event and baseflow grab samples (error bars represent +/1 1 SE).

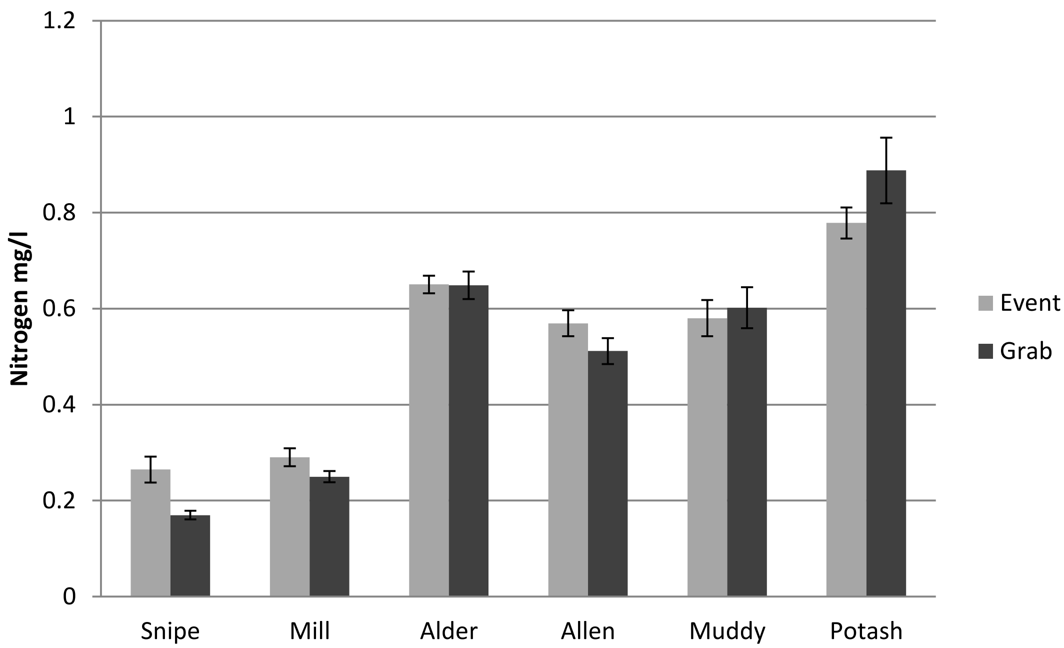

FIGURE 8. Mean total nitrogen concentrations for storm event and baseflow grab samples (error bars represent +/- 1 SE).

Chloride concentrations were significantly higher in the watersheds with paved roads and higher development (Figure 9). Potash had significantly higher concentrations of chloride in baseflow and storm event samples. Muddy had significantly higher storm event concentrations of chloride than Alder and Allen, and Mill and Snipe had the lowest concentrations by over an order of magnitude. Baseflow concentrations produced the following Tukey-Kramer groupings in decreasing order: Potash, Muddy and Alder, Alder and Allen, and Mill and Snipe. Chloride concentrations were significantly higher in baseflow samples at Potash compared to storm event samples. The high concentrations of chloride in the watersheds with large networks of paved roads are directly linked to the extensive use of road salt during icing months. The highest chloride concentration (795 mg/L) in Potash Brook was observed in February; however levels remained very high throughout the summer suggesting that groundwater contributions are a major source of chloride. Similar seasonal patterns were also observed in the remaining watersheds with paved roads. Concentrations in Mill and Snipe were consistent throughout the year indicating minimal contributions from road maintenance. Additional analysis and discussion of chloride data is presented in Appendix A.

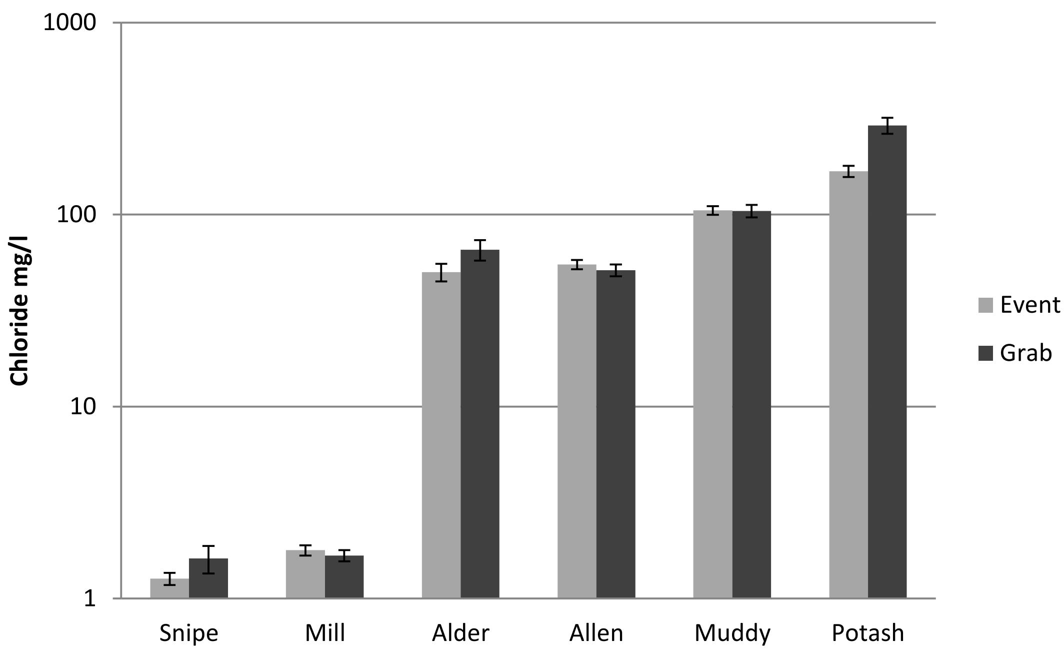

FIGURE 9. Mean Chloride concentrations for storm event and baseflow grab samples (error bars represent +/- 1 SE) *Note logarithmic scale on y-axis.

Total suspended sediment concentrations were significantly higher during storm events for all watersheds (Figure 10). In contrast to the nutrient and chloride results, Alder Brook has significantly higher storm event sediment concentrations and Alder and Muddy have significantly higher baseflow concentrations compared to the other watersheds. The two rural watersheds with primarily dirt/gravel roads have similar sediment concentrations to watersheds with much higher densities of roads and development.

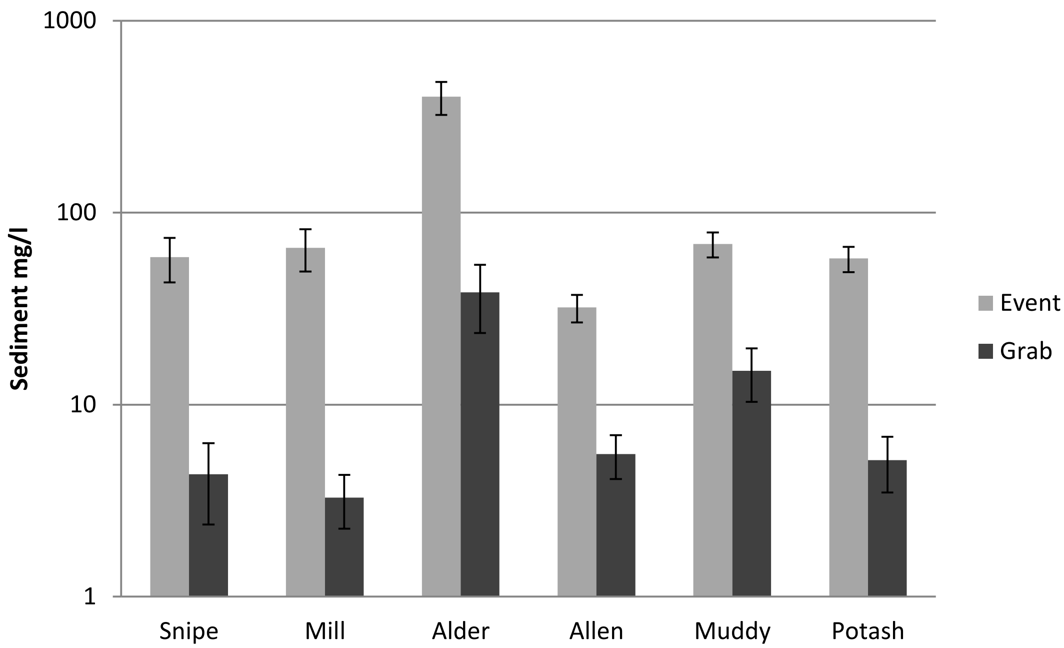

FIGURE 10. Mean total suspended sediment concentrations for storm event and baseflow grab samples (error bars represent +/- 1 SE) *Note logarithmic scale on y-axis

Macroinvertebrate Results.

The total number of unique macroinvertebrate species (richness) and the number of unique species from pollution sensitive Ephemeroptera, Plecoptera, and Trichoptera (EPT) families both decreased in the watersheds with higher road density and development (Table 7). The Index of Biotic Integrity (BI) summarizes the overall pollution tolerance of the BMI community and also followed the same pattern with scores increasing (more pollution tolerant) with increasing road density and development. PMA-O1 is a comparison of the sampled BMI community to the reference community for a given stream type and was least similar in the most developed watersheds and most similar for Mill Brook. Additional BMI community metrics and discussion are provided in Appendix B.

TABLE 7. Benthic macroinvertebrate community results.

Watershed	Species Richness	EPT Richness	BI	PMA-O1
Snipe	46.0	27.5	3.14	73.1
Mill	47.5	29.0	2.80	89.2
Alder	40.0	19.0	3.88	64.5
Allen	42.5	19.0	4.02	73.2
Muddy	41.0	16.0	4.52	63.7
Potash	39.0	13.0	5.45	56.1

RHA Results.

The rapid habitat assessment results show a decrease in RHA rating and condition in the developed watersheds (Table 8). The RHA was conducted on an approximately 100m long reach immediately upstream and downstream of the monitoring station, weighting the importance of the local condition. Sediment deposition was a common impact in the developed streams with decreased riffle and pool variability and increased embeddedness. Bank stability impacts were important at several sites and are frequently a response to increased peak flows.

TABLE 8. Rapid habitat assessment results.

Watershed	RHA Rating	RHA Condition	Impacts
Snipe	72.5	Good	Substrate, Buffer
Mill	89.0	Reference	Deposition, Bank Stability
Alder	68.5	Good	Pools, Deposition, Bank Stability
Allen	72.5	Good	Pools, Channel Alteration, Buffer
Muddy	53.5	Fair	Substrate, Channel Alteration, Buffer
Potash	60.5	Fair	Substrate, Pools, Bank Stability

Road Network/Water Quality Parameter Relationships.

We tested the ability of several road network metrics for predicting water quality, BMI characteristics, habitat quality, and the relationship with watershed imperviousness. Spearman's ρ correlation results that are significant at α=0.10 are shown in gray, significant correlations are shown in black: * denotes p=0.05, ** denotes p=0.01, and *** denotes p<0.001 (Tables 9-11). Total watershed imperviousness (%TIA) is a strong predictor of water quality, BMI, and habitat quality. This supports findings from numerous studies on urbanization and watershed impacts (Booth and Jackson 1997; Paul and Meyer 2001; Wang et al. 2003; Wheeler et al. 2005). Stream crossing density was the most powerful road network predictor of water quality, BMI, and stream habitat quality. Correlations with road/stream crossings were significant for 7-day maximum temperature, maximum dissolved oxygen range, event and baseflow concentrations of TN and Cl, baseflow TP concentration, and BMI biotic integrity, EPT richness (negative).

TABLE 9. Road network and watershed imperviousness correlation.

Predictor	Stream Crossings	Road Density	Road Density 100m
Stream Crossings	--		
Road Density	0.94 **	--	
100m Road Density	0.94 **	1.00 ***	--
%TIA	0.94 **	0.83 *	0.83 *
%Road TIA	0.83 *	0.95 **	0.95 **

TABLE 10. Road network metric correlation with water quality results.

				Event Concentrations			Baseflow Concentrations		
Predictor	**Max Temp 7d**	**Min DO**	**Max DO Range**	**TP**	**TN**	**Cl**	**TP**	**TN**	**Cl**
Stream Crossings	0.83 *		0.83 *		0.83 *	1.00 ***	0.89 *	0.83 *	0.95 **
Road Density	0.77		0.77		0.77	0.94 **		0.77	0.83 *
100m Road Density	0.77		0.77		0.77	0.94 **		0.77	0.83 *
%TIA			0.94 **		0.94 **	0.94 **	0.94 **	0.94 **	1.00 ***
%Road TIA		-0.8 *	0.83 *	0.8	0.83 *	0.83 *		0.83 *	0.77

TABLE 11. Road network metric correlation with BMI and habitat results.

Predictor	**Species Richness**	**EPT Richness**	**PMA-O1**	**BI**	**RHA**
Stream Crossings	-0.77	-0.93 **		0.94 **	-0.75
Road Density		-0.84 *		0.89 *	
100m Road Density		-0.84 *		0.89 *	
%TIA	-0.89 *	-0.93 **	-0.83 *	0.89 *	-0.84 *
%Road TIA	-0.77	-0.75		0.77	

Selected regression plots from the road network metric analysis are shown in Figures 11-13. These regressions show the significant positive correlation of stream/road crossing density with TN and Cl storm event and baseflow concentrations and the significant negative correlation with EPT richness.

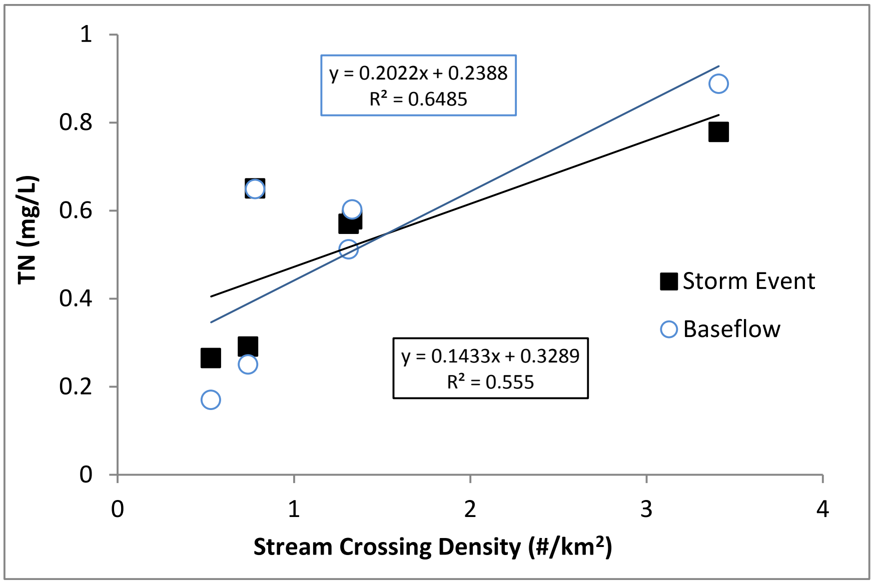

FIGURE 11. Regression of stream crossing density and TN concentrations for baseflow samples (p=0.054) and storm events samples (p=0.089).

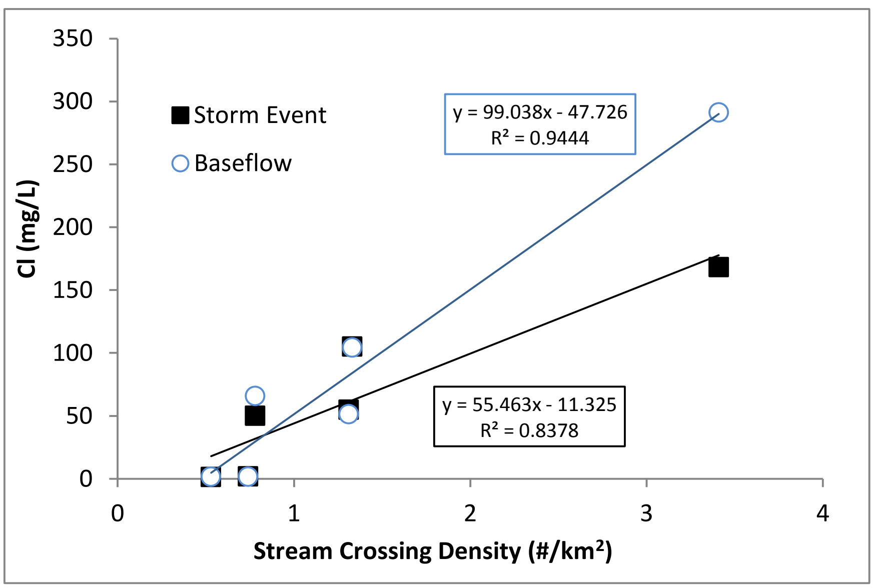

FIGURE 12. Regression of stream crossing density and Cl concentrations for baseflow samples (p=0.001) and storm event samples (p=0.011).

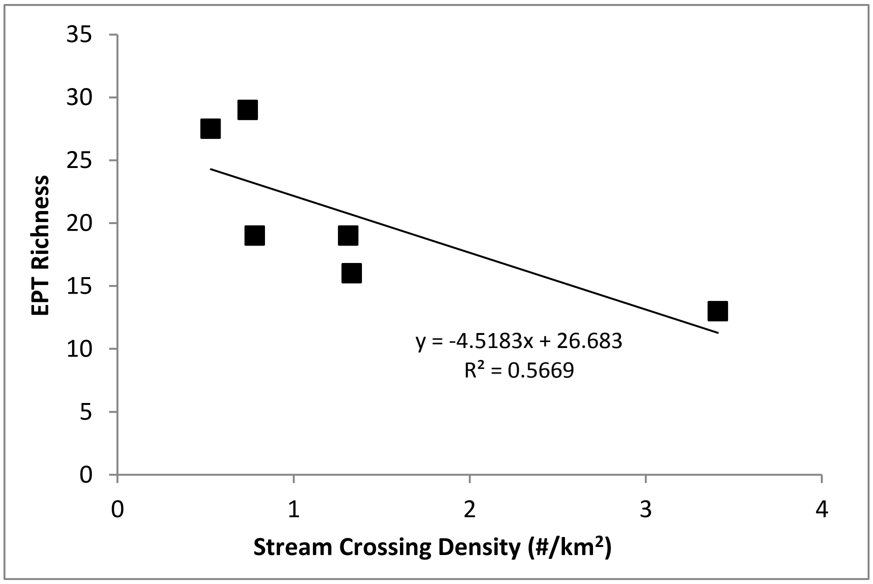

FIGURE 13. Regression of stream crossing density and EPT Richness (p=0.084).

DISCUSSION

Stormwater has become a politically charged issue in Vermont and throughout the country. The transportation network (roads, parking lots, and railways) represents a large percentage of the impervious surfaces across that landscape that can increase the peak flow and contribute significant quantities of pollution to surface waters (Kang and Marston 2006; Eyles and Meriano 2010). While important progress has been made to treat and reduce pollution from point sources, less progress has been made to address nonpoint sources of pollution, which are now responsible for the majority of the pollutant load for many surface waters (VTDEC and NYSDEC 2002). In our study watersheds and typical of watersheds throughout Vermont, the transportation network is managed by a combination of State and Municipal entities, therefore local and statewide efforts and cooperation are required to address runoff and pollution from these sources.

We calculated a series of road density metrics that can be used to easily and consistently quantify potential water quality impacts in a watershed. All of the road density metrics correlated with a range of observed impacts; however the density of stream/road crossings was the most successful predictor for impacts (Table 10). This is in part due to the inherent proximity of road surfaces and drainage infrastructure located at or near every crossing. Drainage infrastructure, such as culverts and ditches, directly channel stormwater runoff to streams and has greater impact than other impervious surfaces in the watershed (Booth and Jackson 1997; Wheeler et al. 2005; Schiff and Benoit 2007). Undersized structures and associated channel alterations near these crossings can cause major physical impacts such as sediment transport interruption, bank erosion, buffer degradation, and deposition (Lane and Sheridan 2002; Wheeler et al. 2005). As expected, the road network metrics are correlated with watershed imperviousness; however the lower level of significance between imperviousness and road density within a 100m stream buffer highlights the potential for underestimating impacts of development in steeper watersheds where a large percentage of the imperviousness may be very close to the receiving waterbody (Table 9).

Concentrations of phosphorus and nitrogen increased along the gradient of development and road network density (Figure 7 and 8). These findings were consistent with numerous studies in urbanized watersheds (Paul and Meyer 2001; Wheeler et al. 2005; Cunningham et al. 2009; Noll and Magee 2009). We also found large increases in chloride concentrations in baseflow and storm event samples (Figure 9). These findings are supported by studies in Vermont and other cold-weather regions where deicing chemicals are applied throughout the winter (Kelly et al. 2008; Cunningham et al. 2009; Denner et al. 2009).

Total suspended sediment concentrations were not correlated with development or road network density. The presence of unpaved roads in the rural watersheds increased suspended sediment concentrations to levels that were statistically indistinguishable from all but one of the developed watersheds (Figure 10). Transport of large amounts of sediment from unpaved roads has been described in several studies; however these watersheds primarily contained temporary logging roads that likely behave differently than maintained rural roads (Lane and Sheridan 2002; Sheridan and Noske 2007; Jordan and Martinez-Zavala 2008). Our results are consistent with findings from a recent study in rural Vermont watersheds that quantified the magnitude of sediment production from maintained gravel roads (Wemple 2013). The sediment generated from these road surfaces is also an important source of TP in rural watersheds. Several studies in Vermont have found that TP in road and ditch sediments (mean of 396mg/kg) and TP in sanding mixes for winter road treatments (mean of 780 mg/kg) is similar to the mean TP content in eroding streambanks (621 mg/kg) (Gaddis and Voinov 2010; Wemple 2013; Ishee et al. 2015).

We did not observe consistent or significant changes in hydrologic metrics within our range of watershed development and road network density; however decreased baseflow in the developed watersheds likely increases the temperature and dissolved oxygen impacts that were significantly correlated with road network metrics (Richter et al. 1996; Wang et al. 2003; Herb et al. 2008). Riparian buffer degradation likely also contributes to the increased water temperature and daily ranges we observed (Angermeier et al, 2004; Walsh et al. 2005; Schiff and Benoit 2007). Peak flow volumes were not significantly higher in the urban watersheds; this is likely due in part to runoff attenuation in stormwater treatment structures and differences in channel and basin slope between developed and rural watersheds.

The levels of imperviousness and road network density in our six study watersheds are representative of the full range of development present in Vermont. We found significant water quality impacts despite levels of development below those classified as “moderate” by previous studies (Chadwick et al. 2006). The relatively low level of total imperviousness likely increases the proportional impact of roads within each watershed. Despite the lower levels of development in our study watersheds, our results showed significant physical, chemical, and biological impacts associated with increasing watershed imperviousness and road network density metrics. Stream crossing density was significantly correlated to many of our stream quality indicators and is very simple to calculate and easily replicated with publicly available spatial data. Most of our findings closely follow the results of numerous studies that link watershed imperviousness to a suite of physical, chemical, and biological impacts known as the “urban stream syndrome” (Paul and Meyer 2001; Wheeler et al. 2005; Cunningham et al. 2009; Denner et al. 2009; Noll and Magee 2009). These results suggest that additional safeguards are necessary to reduce the impacts of roads and associated development on streams in the Chittenden County area of Vermont.

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APPENDIX A

Chloride Concentration and Specific Conductance in Developed Watersheds

The Vermont Department of Environmental Conservation (VTDEC) is in the early stages of developing a listing process for chloride impaired streams which will be integral for future watershed restoration efforts to reduce chloride loading. The concentration of chloride and the specific conductance (SpC) are usually strongly correlated in developed watersheds. Chloride analyses are relatively simple and affordable, but SpC is even easier to measure and can be done at high resolution with readily available, affordable, and robust SpC sensors that can monitor SpC continuously. Deploying a network of SpC sensors throughout a watershed may be an effective tool for isolating chloride loading “hotspots” and increasing the effectiveness of management actions.

USEPA and VTDEC define chronic and acute chloride exposure criteria for fresh water based on chloride concentration over a specified time period. Chronic exposure is defined as exceeding 230mg/L averaged over a 4 day time period, and acute exposure is defined as a 1 hour average of 860mg/L (USEPA, 1988). Both exposures are allowable once in a three year period, but any additional exceedances would trigger a chloride impairment listing. The incorporation of both time and concentration criteria in defining exposure further supports the use of continuous monitoring.

Our study collected a large volume of continuous SpC data and chloride concentration data from water quality sampling, providing a valuable resource to explore the potential for chloride impairment in developed watersheds in Chittenden County. We collected 27-30 baseflow water quality grab samples from each of the developed watersheds. Specific conductance readings at the time of sample collection were recorded for all of the samples that were collected during the ice-free monitoring period (15-17 samples per watershed). While most of the samples from Potash Brook exceeded the chronic exposure concentration, none exceeded the acute threshold. No other streams exceeded the 230 mg/L threshold at the time of sampling. Maximum observed chloride concentrations for each of the developed watersheds are shown in Table 1.

TABLE 1: Maximum observed grab sample chloride concentration.

Watershed	Maximum Observed Cl concentration (mg/L)	Sampling Date
Potash	792	2/25/09*
Muddy	211	8/31/10
Allen	97	8/31/10
Alder	222	12/28/09*

* Denotes that SpC sensors were not deployed during sample collection

Continuous specific conductance data was collected for approximately 20 months in each of the watersheds. These conductance readings were compared to the chloride concentration values from grab samples collected in each watershed using a series of simple regression analyses. Individual regressions were calculated for each watershed as well as the combined dataset for all of the developed watersheds (Table 2 and Figure 1). The regression slopes were tested with an ANCOVA analysis and were not significantly different from each other or from the combined dataset.

TABLE 2: Regression equations and R^2 values for SpC and chloride concentration.

Watershed	Linear Regression Equation	R^2
Potash	$y = 272.07x - 58.89$	0.855
Muddy	$y = 248.64x - 28.99$	0.688
Allen	$y = 174.05x - 12.07$	0.808
Alder	$Y = 177.93x - 8.69$	0.934
Combined Dataset	$y = 175.26x - 10.24$	0.956

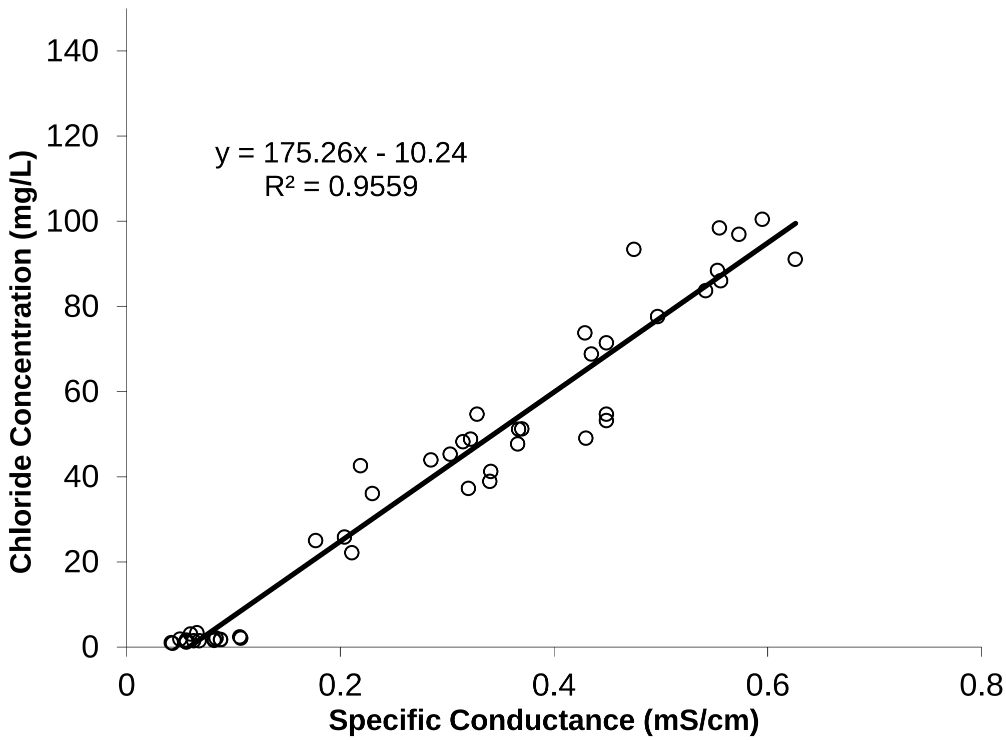

FIGURE 1: SpC and chloride concentration regression for all developed watersheds.

The strength of the individual and combined regression lines supports the calculation of chloride concentrations over the entire monitoring period. The highest SpC readings for each year and therefore the highest chloride concentrations were observed at all four stations during the late summer period when stream flows are low and relative groundwater contribution is greatest. Based on the continuous record of SpC in each watershed and the individual regressions between SpC and concentration of chloride, we are able to identify potential chronic or acute exposure exceedances of the recommended chloride thresholds on dates other than those on which we collected grab samples. Table 3 describes the predicted specific conductance value at the chronic exposure level (230mg/L) for each of the developed watersheds based on the individual watershed regressions. The combined dataset predicts a chloride concentration of 230 mg/L at a specific conductance of 1.028 mS/cm. Based on our daily mean conductance data, Potash Brook was in exceedance of the 4-day average chronic exposure for approximately half of our monitoring period. Muddy Brook exceeded the 4-day average concentration six times based on the Muddy Brook regression; however no exceedances were observed based on the combined dataset regression.

TABLE 3: Predicted specific conductance at the chronic exposure threshold.

Watershed	**Predicted SpC at 230mg/L Cl (mS/cm)**	**Peak Measured SpC (mS/cm)**	**Date**
Potash	1.064	2.770	11/25/08
Muddy	0.898	1.270	8/19/09
Allen	1.194	0.675	9/14/09
Alder	1.267	0.674	9/11/09

Muddy Brook specific conductance values and the highest winter baseflow grab sample chloride concentration from Alder Brook suggest that these watersheds are very close to the threshold and may meet the requirements for an impairment listing. We tested the regression line for Muddy Brook against all of the other independent regressions and against a regression of the combined dataset without Muddy. Using an ANCOVA analysis the slope of the Muddy regression was not significantly different ($p=0.87$ for Muddy against developed watersheds excluding Muddy). A plot of this regression including 95% confidence intervals for the individual points and the mean of the combined regression is shown in Figure 2. While not significant, a visible shift near the 230 mg/L chronic criteria is evident in the Muddy data. This suggests that more sampling is necessary for sites that are near the threshold concentrations.

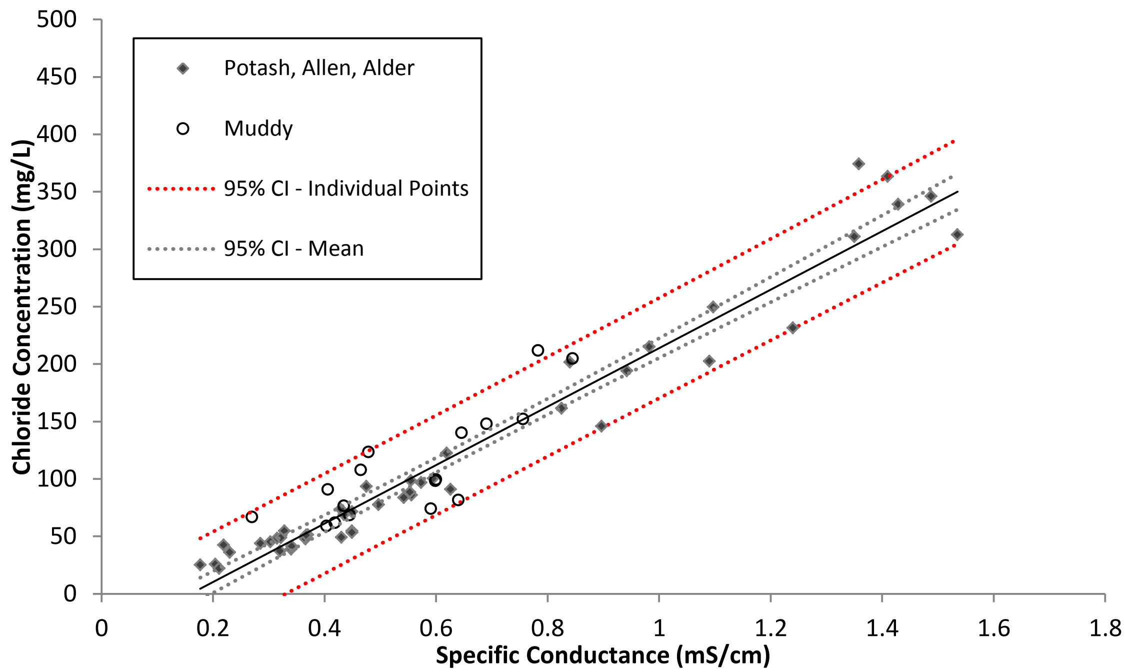

FIGURE 2: Regression plot for grab sample chloride and specific conductance data from Potash, Allen, and Alder Brook with 95% confidence intervals of the individual points and the mean. Muddy Brook data is overlaid to indicate a possible shift near 230mg/L.

The peak observed chloride concentration from Alder Brook was collected during a winter baseflow. This value (222 mg/L) is very close to the chronic threshold but only represents a "snapshot" of the stream. This highlights the importance of high resolution continuous data for assessing chloride impairment. The establishment of a statewide chloride listing process has far-reaching implications for municipalities, residents, and commercial property owners; therefore it is critical that well informed stream science is utilized to establish watershed chloride conditions and inform river scientists.

APPENDIX B

Macroinvertebrate Community Summary

Snipe Island Brook and Mill Brook were assessed using the Small High Gradient (SHG) and Medium High Gradient (MHG) criteria respectively; the remaining four sites were classified as Warm Water Medium Gradient (WWMG). VTDEC assesses streams as either non-supporting or supporting of aquatic life using eight calculated metrics to describe the macroinvertebrate community. A range of values for biocriteria metrics is defined for the three levels of aquatic life use support: A1 (green), B-WMT 1 (Yellow), B-WMT 2&3 (Orange), and non-supporting (Red). A qualitative community assessment rating is assigned based on these scores and the observed community composition. Additional years of macroinvertebrate and fish community assessments and water quality monitoring are typically conducted before a stream is officially listed as non-supporting of aquatic life uses. A large storm event occurred prior to sampling at Muddy and Potash, likely affecting the community composition. Chironomidae, Oligochaeta and silt ratings were all reduced in the VTDEC samples compared to historic data. Further community sampling is required at these sites to characterize the benthic community and assess the level of impairment.

Stream	Community Assessment	Density	Richness	EPT Richness	PMA-O	HBI	Oligo%	EPT/ EPT+C	PPCS
Snipe	Excellent	738	46.0	27.5	73.1	3.14	1.9	0.92	0.56
Mill	Excellent	593	47.5	29.0	89.2	2.80	0.3	0.91	0.68
Alder	Good	3520	40.0	19.0	64.5	3.88	7.4	0.95	0.51
Allen	Very Good	1438	42.5	19.0	73.2	4.02	0.1	0.82	0.71
Muddy	Good	1976	41.0	16.0	63.7	4.52	0.0	0.94	0.54
Potash	Fair	5024	39.0	13.0	56.1	5.45	0.0	0.92	0.39

Stream	Community Description
Snipe	Abundance and HBI are low for SHG stream types but very high taxa and EPT richness suggest a low productivity stream with excellent community composition. Elevated Oligochaeta and the presence of Lumbriculidae indicate some response to sediment stress.
Mill	Abundance is low and high proportion of nutrient sensitive taxa suggest that this stream is unproductive or oligrotophic. Community composition is excellent with high taxa and EPT richness and is very similar to the reference community. Slightly elevated Oligochaeta suggest potential sediment impacts, however observed embeddedness and silt rating were low.
Alder	Good taxa richness and EPT/EPT+C, bio index is slightly elevated as expected for a somewhat developed watershed. Oligochaeta are elevated indicating sediment stress on the community; this was also evident in the pebble count, embeddedness, and silt rating.
Allen	Overall community is very good with high richness and low density of nutrient/pollution tolerant taxa. Community composition and functional feeding groups are similar to reference WWMG streams.
Muddy	Overall community was assessed as good due to high richness and low proportion of Oligochaeta and Chironomidae (likely influenced by large rain event 1 week before sampling). Increased Coleoptera richness and a high proportion of scraper feeding group organisms and the high macro algae coverage noted in the pebble count indicate nutrient enrichment.
Potash	Overall community was assessed as fair due to very high abundance but low EPT richness and low abundance of sensitive taxa. High HBI indicates stress from nutrient enrichment. Very high density of Hydropsychid caddisflies indicate a community dominated by collector filterers.