# Turbidity Dynamics in Small Streams as a Key Component of

## Water Quality Management

by

Gillian Fuss

H.B.Sc, The University of Toronto, 2014

### A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF

### THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

in

### THE FACULTY OF GRADUATE AND POSTDOCTORAL STUDIES

(Forestry)

### THE UNIVERSITY OF BRITISH COLUMBIA

(Vancouver)

February 2019

© Gillian Fuss, 2019

The following individuals certify that they have read, and recommend to the Faculty of Graduate and Postdoctoral Studies for acceptance, a thesis/dissertation entitled:

Turbidity dynamics in small streams as a key component of water quality management.

submitted by	Gillian Fuss	_ in partial fulfillment of the requirements for			
the degree of	Master of Science				
in	Forestry				
Examining Committee:					
Dr. John Richa	ardson				
Supervisor					
Dr. Marwan H	assan				
Supervisory Committee Member					
Dr. Sarah Gerg	gel				
Supervisory Committee Member					
Dr. Jordi Hone	ey-Roses				
Additional Examiner					
Additional Sur	pervisory Committee Members:				

Wm. Patrick Lucey Supervisory Committee Member

Supervisory Committee Member

### Abstract

Given the reliance of many communities on surface water, and the continued degradation of aquatic ecosystems, understanding the limits and uncertainties of water quality assessment is vital. Turbidity is a common measurement of water quality for public health and ecosystem function. It has been frequently studied in larger water bodies but not in small streams. We characterized turbidity dynamics in two sets of small streams over seasonal and spatial scales, by monitoring. We collected continuous turbidity measures every 15 minutes, for one year, with monthly spot turbidity samples, in two regions in British Columbia with varying degrees of land use for over a year. Three streams were in the University of British Columbia's Malcolm Knapp Research Forest, with forestry as the dominant land use type. Our other study area was the Shawnigan Lake Watershed, located on southern Vancouver Island. We had three sites on each of two creeks, McGee Creek and Van Horne Creek, where Van Horne Creek had higher percentages of industrial and urban land uses determined using a Normalized Difference Vegetation Index. In the Research Forest streams, turbidity maximums were ~16 NTU, whereas McGee Creek reached a maximum of 67 NTU, and Van Horne Creek reached 371 NTU. Using Principle Component Analysis and Linear Mixed Effects Models, we found that both rainfall and discharge were significant drivers of turbidity, particularly during periods of intense precipitation. Turbidity also displayed mostly clockwise hysteresis dynamicss during storm events. Interestingly, turbidity displayed a highly significant seasonal response, where the first-flush response of a few of the highest turbidity events occurred during the spring and summer. Land use was also a significant driver of turbidity, particularly forestry, urban and construction land uses. Our research showed that turbidity was spatially complex, and highly variable over time and space, with individual sites

and streams being significantly different from each other. Our results have important implications for turbidity monitoring and assessment, given that current monitoring schemes may be insufficient to determine changes in turbidity due to land uses and to assess water quality accurately over spatial scales.

### Lay Summary

This thesis analyzed turbidity (water cloudiness) changes in small, forested streams. Small streams provide drinking water to many people worldwide, but the causes of turbidity fluctuation are not well understood. We looked at turbidity change in small streams in a protected research forest to identify ecological causes, of which we identified flow, rainfall, and seasonal organic matter availability. We also looked at turbidity in streams close to human activities, such as agriculture, forestry, construction and industrial development. Study streams showed high levels of turbidity, which we found to potentially be due to local industrial activities. The turbidity levels downstream of these activities were well above provincial guidelines for aquatic life and suggest that these activities may be impairing the drinking water quality and ecosystem functions.

### Preface

This thesis is based on field data collected in the Shawnigan Lake Watershed and the Malcolm Knapp Research Forest (MKRF). Field data was collected in collaboration with Aqua-Tex Scientific Consulting Ltd., and under the supervision of Dr. J.S. Richardson (UBC Forestry) and Wm. Patrick Lucey (Aqua-Tex Scientific Consulting Ltd.). All equipment in MKRF was installed in 2013 by Justin Knudson, and maintained by the Richardson Lab until 2016. Equipment in the Shawnigan Watershed was installed with the support of Aqua-Tex, including Patrick Lucey, Tracy Motyer, Sarah Karkanis, Reid Bentzen, and Steve Voller, using a water quality station set-up designed by Patrick Lucey. Monitoring equipment was from Campbell Scientific, Hach Company, and Yellow Springs Instruments.

All data collection were taken with the aforementioned individuals, in collaboration with the Richardson Lab (particularly Alex Yeung and Dr. L. Kuglerova), and several field and undergraduate assistants (particularly Barb Gass). Laboratory sample processing was done partly by M.B. Labs in Sydney, BC, and partly by the Canadian Forest Service in Sault Ste. Marie, ON.

Aside from the above assistance, I undertook all laboratory sample processing, analysis of data, and writing of the thesis manuscript, though with much input from the aforementioned individuals.

# **Table of Contents**

Abstract	iii
Lay Summ	naryv
Preface	vi
Table of C	Contents vii
List of Ta	blesix
List of Fig	gures xi
List of Syı	mbols xiv
List of Ab	breviationsxv
Acknowle	dgements xvii
Dedication	n xviii
Chapter	1: Introduction
1.1	Turbidity Defined
1.2	Turbidity Dynamics in Streams
1.3	Turbidity and Ecology
1.4	Turbidity and Water Quality Management
1.5	Turbidity in this Thesis
Chapter	2: A characterization of turbidity dynamics in small streams over differing time scales
in the M	alcolm Knapp Research Forest
2.1	Introduction

2.2	Methods	
2.3	Results	
2.4	Discussion	
2.5	Conclusions	
Chapter	3: Variations in turbidity associated with hydrology, seasonality and land	l use in the
Shawnig	an Lake Watershed	51
3.1	Introduction	51
3.2	Methods	54
3.3	Results	61
3.4	Discussion	
3.5	Conclusions	
Chapter	4: Discussion	
4.1	Turbidity as a Measurement	
4.2	Turbidity as an Ecological Parameter	
4.3	Turbidity as a Water Quality Indicator	
4.4	Turbidity Monitoring	101
4.5	Turbidity as a Policy Objective	106
Chapter	5: Conclusion	109
Reference	ces	110
Appendi	х	

# List of Tables

<b>Table 1.1.:</b> The different turbidity-related water quality standards from British Columbia and surrounding regions (EPA 1986; BCMOE 1997; CCME 1999; Yukon Public Health and Safety Act 2007; EPA 2009; Alberta Government 2012; Health Canada 2012a; Health Canada 2012b; EPA 2012; BCMOE 2017a; Health Canada 2017; Washington State Ecology 2017; WHO 2017a; WHO 2017b; Alberta Government 2018; EPA 2018). No data (N/D) does not imply that regulations do not deal in some way with that issue, instead it typically means that authorities may have not set a turbidity-specific regulation, and instead regulate a turbidity-related measure, such as water clarity, suspended particles or even coliforms
<b>Table 2.1.:</b> Summary of hydrological variables for study creeks. Data taken from Kiffney et al.(2002), Kiffney and Richardson (2010) and Leach and Moore (2015)
Table 2.2.: Linear mixed-effects model results for drivers of turbidity, with coefficients as seen in Table 2.3 *Indicates significance
<b>Table 2.3.:</b> Linear mixed-effects model coefficients for drivers of turbidity, with modelcomparisons in Table 3.2. *Indicates significance.39
<b>Table 2.4.:</b> Linear mixed-effects model results for drivers of turbidity during storm periods, withcoefficients as seen in Table 2.5. *Indicates significance.42
<b>Table 2.5.:</b> Linear mixed-effects model coefficients for drivers of turbidity during storm periods,with model comparisons in Table 2.2. *Indicates significance.42
<b>Table 3.1:</b> Turbidity (NTUs) in the Shawnigan Watershed for the six continuously monitoredsites. Table shows that while Van Horne tends to have higher maximum turbidities, all sitesshow a large percentage of samples above 1 NTU.62
<b>Table 3.2:</b> Linear mixed-effects model results for drivers of turbidity in streams in theShawnigan Watershed, with coefficients as seen in Table 3.3. *Indicates significance
<b>Table 3.3:</b> Linear mixed-effects model coefficients for drivers of turbidity in streams in theShawnigan Watershed, with model comparisons in Table 3.2. *Indicates significance
<b>Table 3.4:</b> Linear mixed-effects model results for drivers of turbidity in cases of high discharge events, in streams of the Shawnigan Watershed, with coefficients as seen in Table 3.5. *Indicates significance.   74
<b>Table 3.5:</b> Linear mixed-effects model coefficients for drivers of turbidity in cases of highdischarge events, in streams of the Shawnigan Watershed, with model comparison as seen inTable 3.4. *Indicates significance

<b>Table 3.6:</b> Linear mixed-effects model results for drivers of turbidity using spot sample data, instreams of the Shawnigan Watershed, with coefficients as seen in Table 3.7. *Indicates
significance
<b>Table 3.7:</b> Linear mixed-effects model coefficients for drivers of turbidity using spot sample data, in streams of the Shawnigan Watershed, with model comparison as seen in Table 3.6.
*Indicates significance
<b>Table 3.8:</b> Linear mixed-effects model results for drivers of low turbidity (<5 NTU) using spot sample data, in streams of the Shawnigan Watershed, with coefficients as seen in Table 3.9.   *Indicates significance. 80
<b>Table 3.9:</b> Linear mixed-effects model coefficients for drivers of low turbidity (<5 NTU) usingspot sample data, in streams of the Shawnigan Watershed, with model comparison as seen inTable 3.8. *Indicates significance
<b>Table 4.1:</b> Water quality guidelines for turbidity from British Columbia (BCMOE 2017) 102

### **List of Figures**

**Figure 2.6.:** Principal Components Analysis of the three storms where particles were measured in addition to turbidity. The two principal components explained approximately 84.3% of the variation. Particles included TSS (Total Suspended Solids), TDS (Total Dissolved Solids), organics, inorganics, and fines (< 100  $\mu$ m). Graph A) shows points separated by stream and B) depicts points separated by season. In both, there are overlapping points, but a defined grouping effect is present.

**Figure 2.7.:** Hysteresis diagram from February 13<sup>th</sup> storm from three creeks in the Malcolm Knapp Research Forest. Graph is of turbidity versus discharge. Arrows show hysteresis loop direction. 43

**Figure 3.3:** Rainfall (mm/h) and discharge (m<sup>3</sup>/s) for the full Shawnigan Watershed, and turbidity (NTUs) for the Van Horne Creeks, with Upstream, Middle and Downstream sites. Note the scale differences, where Van Horne Creek has maximum values of around 350 NTU, but McGee has maximum values of below 100 NTU, and very few measurements above 5 NTU. ... 64

**Figure 3.5:** Turbidity differences between spot turbidity samples and continuous turbidity. Continuous samples are from all sites at each creek. A) is McGee Creek, and B) is Van Horne Creek. Both plots show a large difference in the measured turbidity between the two methods..66

**Figure 3.8:** Principal Components Analysis of the Shawnigan Watershed. The two principal components explained approximately 60.8% of the variation. PC1 appeared dominated by weather variables, such as rain and antecedent moisture (AM) at 2 and 5 days, while PC2 appears to demonstrate seasonal flow, with a combination of water temperature and discharge. Our land

**Figure 3.11:** Spot turbidity below 5 NTU and other variables taken from the Shawnigan Watershed, with McGee and Van Horne creeks. TSS refers to total suspended solids, TDS refers to total dissolved solids, and TOC refers to total organic carbon. A shows turbidity against TSS, with an  $R^2$  of 0.06 for McGee Creek and <0.01 for Van Horne Creek. B shows turbidity against TDS turbidity, with an  $R^2$  of 0.05 for McGee Creek and <0.01 for Van Horne Creek. C shows turbidity against TOC, with an  $R^2$  of 0.07 for McGee Creek and 0.10 for Van Horne Creek. 82

# List of Symbols

 $\mu$ : Micrometers

°: Degrees Centigrade

### List of Abbreviations

AB: Alberta
AM: Antecedent Moisture
ANOVA: Analysis of Variance
BC: British Columbia
BMFNLRO: British Columbia Ministry of Forests, Lands, Natural Resources Operations and
Rural Development
BCMOE: British Columbia Ministry of the Environment
BCMOH: British Columbia Ministry of Health
CA: Canada
CVRD: Cowichan Valley Regional District
DWPA: Drinking Water Protection Act
GLS: Generalized Least Squares
LME: Linear Mixed Effects
NDVI: Normalized Difference Vegetation Index
NTU: Nephlometric Turbidity Units
NY: New York
ON: Ontario
PC: Principle Component
PCA: Principle Component Analysis
TDS: Total Dissolved Solids
TSS: Total Suspended Solids
USEPA: United States Environmental Protection Agency

VIHA: Vancouver Island Health Authority

WHO: World Health Organization

### Acknowledgements

I thank my supervisors Dr. John Richardson and Patrick Lucey for their guidance, and the opportunity to undertake this project. I thank Aqua-Tex staff and the Richardson Lab for immeasurable support, guidance and assistance, particularly Lenka Kuglerova, Alex Yeung, Danielle Courcelles, Liam Irwin, Tracy Motyer, Reid Bentzen, and Steve Voller. I also want to thank the many volunteers, summer assistants and field assistants without whom this work would have been impossible, particularly Barb Gass for being my beacon in the dark in some periods. I am also grateful to the staff at UBC and MKRF who assisted me with many tasks. Furthermore, I owe thanks to the Shawnigan Community for allowing us to perform our research in their backyard. I also thank Dr. Dave Kreutzweiser and Scott Capell for processing my samples at the CFS, and the staff at MB labs for processing Shawnigan samples.

For funding, I thank both my supervisors for the funding of the project. I also thank NSERC for funding for my salary, as well as the Faculty of Forestry for several scholarships.

Finally, I thank my family and friends for their words of support and encouragement. Thank you to UBC Karate for giving me an avenue to vent any frustration. I also thank Let's Talk Science UBC for giving me a community and great memories. Most of all, I thank my boyfriend, Oleg Eustace, for everything you've done, I couldn't have done this without you.

# Dedication

"He that plants trees loves others besides himself."

- Dr. Thomas Fuller

For my friends and family (human and non-human).

### **Chapter 1: Introduction**

Streams are a key component to the freshwater (aquatic) ecosystem, which consists of rivers, wetlands, lakes, and ponds. In total, surface sources of fresh water account for a small percentage of global water but provides many critical ecosystem services. The Millennium Ecosystem Assessment (2005) defines an ecosystem service as an ecosystem process that has value for humans (MEA 2005). Palmer and Richardson (2009) listed the following stream-provided ecosystem services: carbon and nitrogen mineralization, flood control, erosion control, temperature regulation, recreation, food (through agriculture or fish production) and clean water (Postel and Carpenter 1997; MEA 2005). Aquatic ecosystems are also unique beyond provisioning for human beings, acting as a critical hotspot for biodiversity and providing unique habitats in transition zones (Wipfli and Richardson 2015). Streams and rivers also play key roles in global landscapes by transporting large amounts of sediments and other particles, such as organic carbon, over long distances (Leopold et al. 1964; Cole et al. 2007). This in turn impacts global biological energy supporting ecosystems and aids in creating the landscape diversity (Cole et al. 2007).

Despite the importance of the aquatic ecosystem, the millennium ecosystem assessment predicted that freshwater water quality would be one of the most pressing issues of the 21<sup>st</sup> century (MEA 2000). Globally, freshwater ecosystems are considered one of the most threatened ecosystems, and biodiversity decline is higher here than in any other ecosystem (MEA 2005; Dudgeon et al. 2006). In the United States, as of 2008, 55% of rivers and streams were considered impaired by pollutants (USEPA 2016). Aquatic ecosystems connect landscapes and are highly dependent on the surrounding terrestrial land for chemical, geological and climatic regulation (Alexander et al. 2007; Freeman et al. 2007; EPA 2015). As such, aquatic ecosystems can be said

to be impacted by most land cover or engineered changes, such as agriculture, damming, or urbanization (Tilman 1999; GWSP 2004; Foley et al. 2005).

Advances have been made in understanding aquatic degradation, but we appear to be far from able to predict and manage consequences. Research continues to identify new stressors on water quality. For example, recent research into pharmaceutical levels in aquatic ecosystems revealed pharmaceuticals can be more pervasive than expected (Metcalfe et al. 2012; Rosi-Marshall et al. 2015). Pollutants can also have complex interactions with each other and other stressors, where even low concentrations can be harmful in degraded ecosystems or in instances of multiple stressors (Ormerod et al. 2010; Chara-Serna and Richardson 2018). Furthermore, increasing human populations and climate change creates additional uncertainties about the future of aquatic ecosystem health (Whitehead et al. 2009).

Equally important to understanding the current and future threats to aquatic ecosystems is understanding how we assess water quality, ecosystem health, and provision of ecosystem services. Differences among streams is immense, where variations in hydrology, climate, and geology can all alter stream structure and processes, to the point where Rosgen's (1994) classification scheme included 41 stream types. Variation can also exist both among streams (Church 1992, 2004) and within them (Angradi 1996). Furthermore, nearby ecosystems can also alter stream processes. For example, Johnston et al. (1990) found relationships between wetlands along streams and several water quality variables, while Connolly et al. (2015) found large differences in nitrogen between upstream and downstream reaches due to increasing agriculture. These aspects will all determine the type and quantity of particles that flow through the streams, ultimately affecting water quality. Therefore, describing water quality is inherently complex. Measurement systems of the different hydrological, chemical and biological variables are the vehicle to assessing water quality, and the mechanics and limitations of each will alter our conclusions about water quality. Increasing technical innovations for the assessment of water quality have given us more precise measurements that can be done automatically, with less personnel; however, precise measurements do not make accurate measurements. As streams are highly heterogeneous in space and in time, water quality assessments may not provide an accurate depiction of background water quality, or water quality changes (due to land use or geology) for decision-making (Robertson and Roerish 1999; Lovell et al. 2001). It is therefore imperative to understand the methods of water quality assessment, and appreciate their limitations, to accurately understand changes to aquatic ecosystems.

This thesis sought to evaluate a major water quality assessment metric, turbidity, which is the inverse of water clarity. In recent years, turbidity has become a staple in water quality assessment, for both human and ecological considerations. Our study seeks to document how turbidity behaves over spatial and temporal scales, and with varying degrees of human activities. We hope to provide a basis for future research and provide insight into water quality management.

#### **1.1 Turbidity Defined**

The definition of turbidity has changed over time, though typically it is defined by the method of measurement. Initially, turbidity was measured using a Jackson Candle Test, where a tube would be filled with water until the depth when one could no longer see a candle flame (Sadar 1998). In this case, turbidity was similar to water clarity, which is the depth at which a human eye can no longer see a Secchi disk (Davies-Colley et al. 2001). With increasing digital sensing

technology, turbidity measurement methods have changed considerably. Most current definitions take a much more literal approach and define turbidity as the amount of light scattered by particles in the water column or sample measured using modern sensors (Sadar 1998). This allows a much more precise measurement of opacity than the Jackson Candle Test but retained the optical nature of turbidity (Sadar 1998).

The different ways of measuring turbidity are defined by the different angles from which light scattering is measured (Figure 1.1). The most common method of turbidity assessment is nephelometry, defined as measuring the amount of light scattered at a 90° angle from a fixed light beam by particles in the water column (Van de Hulst 1957; Sadar 1998; Saraceno et al. 2017). It is thought this angle can be used to estimate all light scattering, assuming an empirical relationship between turbidity and particle concentration (Van de Hulst 1957; Ginting and Mamo 2006). The exact specifications can be found in EPA method 180.1, which is the recommended method by the EPA for drinking water assessment (O'Dell 1993). Many sensors, particularly if following EPA method 180.1, measure water turbidity in Nephelometric Turbidity Units (NTUs), however, turbidity can be measured in Formazin Nephelometric Units (FNUs) (O'Dell 1993; Ankcorn 2003). Formazin refers to a white polymer substance that is commonly used as a standard for turbidity (Ankcorn 2003). Formazin is used for turbidity measurement in both NTUs and FNUs, though the calculation is slightly different and generally the two units are not considered equivalent (Ankcorn 2003). Typically, the nephelometry equation yields accurate measurements for lower turbidities, between 0 and 40 NTUs (Sadar 1998; Sadar 1999).

Over the past few decades, optical backscatter methods have becoming increasingly common, as they can often be deployed remotely, and take continuous measurements (Downing 2006). This can aid in capturing time series and reducing sample bias. Optical backscatter sensors

measure turbidity by the amount of light scattered back at a 45° angle rather than a 90° angle (though the range of sensing angles can be extended), so the sensor can act as a probe, without needing sample cuvettes (Downing 2006). These sensors typically employ infrared light, and work best at high turbidities between 1000 NTU and 4000 NTU due to the smaller angle of light scattering (Sadar 1999; Downing 2006). Other sensors that may be more accurate in lower turbidities are usually not able to measure continuously, which can potentially result in a reduction of resolution in field data.

Finally, the particles causing the backscatter themselves can add an additional layer of complexity into the turbidity measurement. As turbidity is a function of light scattering, turbidity can be caused by nearly any particle in the water column, from small ions to cobbles, and both inorganic or organic particles. However, particles of different compositions and sizes do not cause equal light scattering (Gregory 2006, pp 26). For example, Gregory (1985) found that turbidity response was highest between the 0.7 and 5  $\mu$ m range, peaking at approximately 1.2  $\mu$ m. Gregory (1985) suggested that this was due to the highest ratio of light scattering surface area per volume of particles which occurred at these sizes. In contrast, Yao et al. (2014) found that particles > 5  $\mu$ m contributed more light scattering and therefore higher turbidities than smaller particles. Additionally, Gippel (1995) identified that organic particles, such as organisms, bacteria, or Dissolved Organic Matter (DOM), caused higher turbidity than inorganic particles. This was attributed to the texture of organic particles, which is rougher than inorganic particles and so may cause increased light scattering (Gippel 1995).

In addition to the complexity of the light scattering of the particles themselves, turbidity measurement is also inherently complex. For example, organic particles can also absorb light, which Sadar (2002) determined would prevent light from scattering, and cause artificially low

turbidity measurements. However, Saraceno et al. (2017) found that organic particles can also redirect light as fluorescence, and thus may also cause higher turbidity values. Additionally, organic particles will scatter red light waves more intensely than inorganic particles, which may alter the response of the sensor (Sadar 1998). Concentration of particles can also be important, as Yao et al. (2014) found that turbidity increased exponentially along a gradient of particle concentration until approximately 40 NTU, after which the relationship became linear with a low slope. Downing (2006) also identified that the Optical Backscatter Sensor's peak turbidity response is typically between approximately 300-800 mg/L of suspended sediment concentration, which is what might be expected in a medium to large river. Downing (2006) also mentioned that most particles are more likely to produce a larger amount of forward light scattering (180°) than other angles, meaning most turbidity probes only capture a fraction of the actual scattering. Altogether, it seems that there are complex interactions between particle size, particle concentration, and particle composition, which will influence a particular turbidity measurement.

### **1.2 Turbidity Dynamics in Streams**

Stream ecosystems have strong relationships with the ecosystems surrounding them, and this interplay determines the quality of the water in the stream. The area in which stream-land interaction occurs is termed a drainage basin, or a watershed, and is the area from which the water is collected and ultimately becomes a stream (Horton 1932; Horton 1945; Allan and Castillo 2007 p. 35). As water flows overland through the watershed, it carries a number of particles (such as sediments, organics, or cobbles) from the uplands (the terrestrial areas above the channel) into the stream, which greatly influence the channel and water chemistry. However, streams can also

influence the watershed, by erosion or deposition of particles in the floodplain. Therefore, stream processes are not just unidirectional transport of watershed material, but instead have feedback mechanisms with the surrounding landscapes.

In general, there are two major ways hydrogeology can influence the stream: sediment supply and transport capacity. Sediment supply refers to the availability of sediment and the type of sediment that is transported within a watershed to the stream, which is governed by the sediment type, soil type, vegetation cover, hydrology and climate (Montgomery and Buffington 1997). Transport capacity refers to the ability of the stream to move particles, either by amount or size, which is governed by the slope, discharge rate and climate, where typically steeper slopes and greater discharge rates yield bigger transport capacities (Holden 2008 p. 302). As an example, a mountainous stream, with minimal soil depth and a steep slope, has low sediment supply but high transport capacity, and so streambeds may be areas of exposed bedrock mixed with large boulders or cobbles. In contrast, the delta of a large river may have low slope, but high discharge, and high sediment supply due to the large drainage area. As such, it receives a large amount of sediment that is deposited in braided channels and along the sides of the bank during overbank flow and become the dominant sediment type of the riverbed.

Once in the stream, the various particles can be classified as dissolved (0-0.5  $\mu$ m), wash (0.5-0.0625 mm) and solids (>0.0625 mm) (e.g., Knighton 1998). This can be further broken down by source, where dissolved particles are primarily from weathering/erosion of parent material and consist of free ions and other small particles (Knighton 1998). Similarly, the solid load (also known as the bed load) is determined by the solids available in the river bed and the transport capacity (Knighton 1998). The wash load, however, is typically fine particles like silt or clay, typically from the streambed or from external sources in uplands (Knighton 1998). Alternatively, Church (2002)

7

suggested defining the larger particles by transportation method, suspension, bed or traction, and saltation. Suspension refers to particles that are supported by upward currents in the water column (Church 2002). Bed load (or traction transport) refers to particles that roll, slide or bounce along the bottom, not supported by upwelling currents (Church 2002). Finally, saltation refers to particles that may be launched into the upper part of the water column, then drop back down (Church 2002).

From a more practical standpoint, particles can be defined by measurement techniques. Finer particles that are typically dissolved are measured as Total Dissolved Solids (TDS), which are similar to the dissolved load from Knighton (1998), but is defined by the APHA as any particle that passes through a standard glass fibre filter (APHA 2005), which can be interpreted as 0.45  $\mu$ m, a 0.7  $\mu$ m or 1.2  $\mu$ m based on available filters, though 0.7  $\mu$ m appears to be the standard size for streams (Houser et al. 2006; Rugner et al. 2012; Masese et al. 2014; Carillo et al. 2018). However, larger particles are typically measured as Total Suspended Solids (TSS), defined as particles that are captured by a 0.7  $\mu$ m filter paper, or Suspended Sediments (SS) in many older papers. This usually encompasses the suspended particles defined by Church (2002), and the bed and wash loads of Knighton (1998).

At annual or decadal time scales, particles can have complex dynamics between smaller streams, which can have large implications for hydrogeology. For example, Wolman and Miller (1960) identified that the majority of sediment is moved during smaller, but more frequent, storm events, while Smith et al. (2003) identified that most sediment mobilization occurs in large but infrequent storm events. On the scale of a storm event, sediments may also not behave consistently over the course of a hydrograph. This phenomenon is known as hysteresis, which refers to the relationship between a particle concentration (like sediment) and discharge, such that as discharge increases (as does transport capacity), and sediment supply reaches its maximum, so particle concentration may decrease despite the overall number of particles increasing or as sediment supply is exhausted (Paustian and Beschta 1979; Langois et al. 2005; Landers and Sturm 2015).

As a measurement of light scattering by suspended particles, turbidity dynamics are typically governed by when and how these particles enter the water column. Measuring particle concentrations can be labour intensive, requiring a sediment trap or equivalent, or are simply limited to single samples (Downing 2006). On the other hand turbidity can be measured *in situ*, and can be used to estimate almost all in-stream particles, and therefore is often measured in proxy during monitoring or research activities (Downing 2006). Gippel (1989) identified a method of using turbidity to estimating suspended solid concentrations using a rating curve calibrated to each stream. Later, Lewis (1996) confirmed the linear relationship between suspended sediment and turbidity, though also requiring calibration within the stream. Dodds and Whiles (2004) identified that turbidity levels were 90% correlated with suspended sediment fluctuation. However, some authors (Gregory 1984; Keim and Schoeholtz 1999; Miller et al. 2015; Saraceno et al. 2017) have noted that these relationships can break down, particularly with either high or low turbidities, when the response of turbidity to different particle types is noticeable.

Turbidity itself can also behave in complex ways at short temporal scales. For example, Likens et al. (1969) found that peak turbidities did not occur during the periods of highest runoff, or even in the most disturbed areas. Bilotta and Brazier (2008) also identified that turbidity can be caused by many more particle types than just suspended solids, such as dissolved particles, which may interfere with the estimation of specific particles. Saraceno et al. (2017) identified that at high turbidity levels, the relationship between turbidity and DOM was site-dependent, as specific sizes and composition of these particles were causing different turbidity responses for the same concentrations. Turbidity differences may also be due to the sensitivity of turbidity to errors or

other artifacts that may scatter light. For example, Jordan (1996) found that turbidity overestimated suspended sediment due to sensor errors or turbulence in the stream. Lewis et al. (2007) more recently indicated that this remains an issue despite technological advances.

For our purposes we focused on small streams, though a "small stream" can be defined many ways. One of the most common methods is by Horton (1945), and later Strahler (1952), who developed the stream order classification scheme. It begins with the smallest streams at first order and continues up as streams of the same order come to confluence (Strahler 1952). This can be useful for prediction; however, this classification scheme can vary with scale (Allan and Castillo 2007 p 35). This thesis focuses on the dynamics of small, sub-montane streams. To define these, we draw our definition from Hassan et al. (2005), modifying Church (1992). Church (1992) describes small channels as less than 3-5 m wide, and where bed particles greatly influence channel morphology. Hassan et al. (2005) however, mentions that even this classification can be problematic, as in forested channels large wood can have dramatic influences, even though this was thought to be a characteristic of intermediate channels. Therefore, they defined small mountainous streams as between small to intermediate channels in the Church (1992) classification scheme, with higher slopes, and high sediment caliber, but without a high amount of sediment supply. Using Church (2006), this would make the dominant reach types in our ecosystems steppool cascades or wandering channels.

### **1.3** Turbidity and Ecology

The ecosystem service of the provision of clean water is not only vital for humans, but also for organisms that live in streams. Stream habitat is influenced by many different factors within the stream, including geomorphological, hydrological and biological processes. The base of this habitat is the substrate, i.e. the channel bottom, and provides habitat for many invertebrates, a base for primary production, and often a mating area for both invertebrates and fish (Allan and Castillo 2007 p. 88). Substrates can range from silts or cobbles to root structures or sticks and can be defined by their roughness or the number of crevices, though different species may have their preferences (Allan and Castillo 2007 pg. 88). Another key aspect of stream ecology is the ecology of the uplands. Vegetation in the uplands, for example, can influence the amount of sediment available for runoff during a storm (Jones et al. 2001). Much of this interaction takes place in the riparian area, the area next to the stream characterized by high moisture and high biodiversity due to it being a transition zone between the aquatic and terrestrial ecosystems (Gregory et al. 1991; Naiman et al. 2005; Richardson et al. 2007). The vegetation in the riparian area can impact how much runoff enters the stream, and what vegetation or wood the stream receives (Rivenbark and Jackson 2004; Hoover et al. 2010). It also can influence stream shading, affecting temperature and other dynamics related to light (Beschta et al. 1986; Gomi et al. 2006; Perkin et al. 2011).

Vegetation and the uplands can also influence a stream through the export of organic particles. Organic particles are a key base of stream metabolism and aquatic food webs (Fisher and Likens 1973). Once in the stream, leaf litter and other organic matter are an important food source to many invertebrates and fish (Wipfli et al. 2007). Organic particles are also defined by size. The smaller (dissolved) particles are measured as Dissolved Organic Material (DOM) or Dissolved Organic Carbon (DOC) (Zsolnay 2003; Tank et al. 2010). Similarly, non-dissolved organics are also distinguished as particulates. These are further differentiated by Coarse Particulate Organic Matter (CPOM) and Fine Particulate Organic Matter (FPOM), which are > 1 mm and between 0.45  $\mu$ m - 1 mm, respectively (Tank et al. 2010). The source of the organic particle can also be a

critical aspect of stream ecology. Organic particles are associated with leaf litter (falling leaves) or organic contents in soil, and primary production (Kaplan and Bott 1989; Fiebig et al. 1990; Kaplan and Newbold 1993; Webster et al. 1999). Webster and Meyer (1997) analyzed 37 streams for the source of their organic matter, finding that mountainous streams had low primary production but high litterfall inputs, while low-land streams have high levels of both. As such, in forested areas, organic particles are typically associated with seasonal and climatic variables, such as leaves falling in autumn yielding large inputs of organics (Pozo et al. 1997). Additionally, organic inputs can typically be sourced to the riparian zone, though they may also come from plant breakdown in aquatic areas or wetlands (Webster et al. 1995; Wallace et al. 1995; Palmer et al. 2001).

Turbidity is frequently thought to have no real ecological function in itself, though it can be ecologically significant as a function of particles in the column (Sadar 1998). The exception to this may be that clear waters (low turbidity) are often needed for aquatic animals to find food/mates (Shoup and Lane 2015; Suh 2016), or for light to pass through for photosynthesis. Recently, research has also shown that slight increases in turbidity can also offer predation refuge for certain species (Utne-Palm 2004; Glotzbecker 2015). Additionally, areas of high turbidity can offer unique habitats for specially adapted animals (Webster et al. 2007; Dubey et al. 2012; Moran et al. 2018). Turbidity is also a natural phenomenon, and some level of turbidity is present in every stream as water molecules themselves can create turbidity (Sadar 1998). Turbidity can be caused by any number of increases in particles, such as runoff, landslides, or an increase in leaf litter, and is common in a stream (Bilotta and Brazier 2008). However, artificially high particle concentrations can have dramatic ecosystem effects. For example, large increases in concentrations of either sediments or organic matter can reduce water clarity, reducing photosynthesis (Lloyd et al. 1987), as well as impairing vision of predators, influencing predatorprey dynamics (Glotzbecker 2008). There can also be impacts of the particles themselves, for example, sediments may harm breathing apparatuses (Berg and Northcote 1985; Lloyd et al. 1987), or alter benthic composition by filling in gravel beds, changing mating environments or, in extreme cases, smothering developing eggs (Scrivener and Brownlee 1989). High turbidity has similar consequences. Even small increases in turbidity (10 NTU) can have large impacts on fish and aquatic food webs (Bachmann 1958; Berg 1982; McCabe and O'Brien 1983; Lowe et al. 2015). Therefore, it is important to understand what causes increases in turbidity, and how to manage them adequately.

High particle concentrations, and thus high turbidity, can be a common result of human impacts. An example is runoff from roads, as they allow overland flow to increase speed and potentially carry larger and a higher concentration of particles into a stream (Reid et al. 2016). An example of increased sediment on construction sites may occur if soil erosion is not controlled adequately, rainstorms can pick up any loose soil and transport that to the nearest water body (McCaleb and McLaughlan 2008). Forestry related activities are similar, due to the creation of roads and the soil disturbance associated with logging (Lieberman and Hoover 1946; Greig et al. 2005; Miller et al. 2015). Urban areas are an example of both increased runoff speed and increased availability, but typically the particles transported are organic in composition. Thus, urban turbidity can increase due to runoff from roads or storm-water drainage, which carry hydrocarbons and other pollutants (Metadier and Bertrand-Krajewski 2012). Agricultural areas are also a good example of both, as loose soil can increase sediment entry into the stream, as well as organic-laden runoff due to the highly fertilized soil or manure top layer (Alberto et al. 2016). Agriculture also

will induce turbidity with an increase in flora and organic particles within the water body (Abrantas et al. 2006).

There are many techniques to reduce anthropogenically induced turbidity. Several methods involve simply reducing the amount of solids available to be eroded by run-off at the site itself, such as separating construction into phases to minimize soil disturbance, or stabilizing hillslopes with vegetation (MPCA 2018). Other methods seek to slow down overland flow, and give the suspended material a chance to settle out before runoff reaches a water body, such as water bars, silt fences or settling ponds. However, McCaleb and McLaughlan (2008) found most methods were not effective at trapping sediment during large storm events. Far more common in most industries (forestry, agriculture, construction) is a riparian buffer. Riparian buffers are strips of vegetation on either side of the stream in the riparian zone. These buffers are typically recommended to be at least 30 m wide (Sweeney and Newbold 2014); however, Rivenbark and Jackson (2004) found that even buffers of over 60 m would not prevent all breakthroughs (water carrying suspended material entering the stream). Additionally, buffers are usually applied to streams and rivers that are perennial, with some exceptions (Moore and Richardson 2003). This leaves channelized streams (ditches) and ephemeral streams unprotected (Moore and Richardson 2003).

#### **1.4** Turbidity and Water Quality Management

Arguably one of the more important concerns about turbidity is in the management of drinking water. Beginning in the 1800s, turbidity was used to assess water quality in London, England as it was linked to both the pathogenic aspect of water quality, as well as the aesthetic aspect, and often taste (Allen et al. 2008). Since that time, turbidity has become a staple in water quality monitoring. Turbidity is frequently used to assess raw water for suitability as drinking water, or the type of treatment needed, and is also used for quality control within filtration plants and water distribution systems.

Many authors have identified associations between turbidity and public health concerns (LeChevallier et al. 1991; LeChevallier and Norton 1992; Hrudey et al. 2002; Health Canada 2012; De Roos et al. 2017). Turbidity, particularly high turbidity, can be linked to heavy metal content or pollution (Nasrabadi et al. 2016), in addition to its relationship with TSS. Turbidity increases due to weather events are also frequently followed by an increase in gastrointestinal illness and emergency department visits (Aramini et al. 2000; Tinker et al. 2010). Despite this, there is no defined relationship between turbidity and pathogenic bacteria, viruses or protists (Health Canada 2012). Some sources have identified positive relationships between *Giardia* or *E. coli* and turbidity (Atherholt et al. 1998; Dorner et al. 2007), while others have found no relationship (St. Pierre et al. 2009). Furthermore, there is evidence to suggest that hospital visits are still common during turbidity events in source water, even while treated drinking water remains at low turbidity (Payment et al. 1984; Tinker et al. 2010).

Generally, treatment of water has two stages: filtration and disinfection (WHO 2017a). Filtration is a process of removing harmful particles through a filter, such as sand filtration or membrane filtration (WHO 2017a). Disinfection is the process of deactivating pathogens, using chlorine, iodine, or more advanced methods, such as UV (WHO 2017a). The WHO recommends the use of both stages, known as multi-barrier protection (WHO 2017a,b). Additionally, treatment of high turbidity waters is not always straightforward or effective. As Tinker et al. (2010) suggest, bacteria can pass through treatment during turbidity events. It has also been shown that high turbidity can reduce the effectiveness of filtration and disinfection, by clogging filters, or by sheltering pathogens in particulate clumps (Ulhrich and Bragg 2003). Furthermore, some disinfection techniques, such as chlorine, can increase water toxicity with high turbidities, as the disinfection reaction creates negative by-products (Ulrich and Bragg 2003; WHO 2017b). Ulhrich and Bragg (2003) provide a benchmark of approximately a 20 NTU maximum of source water intake into a treatment facility, beyond which the filtration process may be less effective, may suffer damage, or produce excess harmful particles. However, the WHO recommends turbidities of above 2 NTU may be high enough to interfere with chlorination but not disinfection (WHO 2017b).

Another method of water quality management is to improve protection of source water to keep it at lower turbidities (CCME 2004). This has also been coined Source-to-Tap protection, suggesting that management of water quality should occur before water reaches the filtration plant (Byleveld et al. 2008; Summerscales and McBean 2011). Examples of this include the protected watersheds for drinking water in many areas around British Columbia, Washington and New York. Furthermore, many rural or lower income areas may have small or no community water supplies (Dunn et al. 2014). In such cases the WHO recommends performing the best treatment possible by the resident/community, but also that the local government focus on maintaining low turbidity in the source water (WHO 2017b). However, as Aramini et al. (2000) identified, even protected watersheds such as those of the Greater Vancouver Area, can still result in gastrointestinal illness

outbreaks following turbidity events, indicating that managing water quality for pathogens is highly complex.

Water quality standards for drinking water involve two sets of standards: standards for the treatment of water, maintaining water at potable quality, and the second are standards for the maintenance of surface water, both of which can be found in Table 1 for BC and the surrounding area. Standards for water treatment are set by Health Canada, and are utilized by the provinces health authorities (Health Canada 2012). Health Canada recommends that turbidity post-treatment be at a maximum of 1 NTU in small systems, but <0.3 NTU in most systems (Health Canada 2012a). In the case of British Columbia, drinking water is regulated by the Drinking Water Protection Act (DWPA), which applies to any water supplier (provision of water supply other than a single-family dwelling and systems excluded through the regulation), and their source water (DWPA 2001). The act does not specify a turbidity level but does reference contamination of drinking water sources (DWOA 2001). The act is also enforced by Drinking Water Officers, whose protocols involve using post-treatment turbidity as an indicator for whether a Boil Water Advisory is necessary or not (BCMOE 2017b).

Surface water quality prior to treatment is typically under the jurisdiction of the BC Ministry of the Environment and Climate Change Strategy, who set guidelines and perform monitoring (BCMOE 2017a; Government of BC 2018). These guidelines are for regional goal setting only and are not enforceable. They can be enforceable under the DWPA if there is a direct and immediate threat present (heavy metal contamination or high pathogen levels), but turbidity may not be considered a direct threat (DWPA 2001). Additionally, the Fisheries Act (2018) also enables the enforcement of water quality in fish-bearing waterways, and may take guidance from

provincial standards, but the act itself references no specific turbidity or TSS level (Fisheries Act

2018).

**Table 1.1:** The different turbidity-related water quality standards from British Columbia and surrounding regions (EPA 1986; BCMOE 1997; CCME 1999; Yukon Public Health and Safety Act 2007; EPA 2009; Alberta Government 2012; Health Canada 2012a; Health Canada 2012b; EPA 2012; BCMOE 2017a; Health Canada 2017; Washington State Ecology 2017; WHO 2017a; WHO 2017b; Alberta Government 2018; EPA 2018). No data (N/D) does not imply that regulations do not deal in some way with that issue, instead it typically means that authorities may not have set a turbidity-specific regulation, and instead regulate a turbidity-related measure, such as water clarity, suspended particles or even coliforms.

Jurisdiction	Water after Treatment	Raw Drinking	Recreational	Aquatic Ecosystem
		Water	Water	
Canada	Changes based on type of	N/D	<50 NTU, unless	<u>Clear Flow:</u> Maximum
	filtration. Conventional		users are	increase of 8 NTUs from
	filtration maximum 0.3		dissatisfied	background levels for a
	NTU in 95% of monthly			short-term exposure (e.g.,
	measurements, never			24-h period). Maximum
	above 1.0 NTU. Slow			average increase of 2
	sand and diatomaceous			NTUs from background
	earth filtration maximum			levels for a longer term
	1.0 NTU in 95% of			exposure (e.g., 30-d
	monthly measurements,			period). High flow or
	never above 3.0 NTU.			turbid waters: Maximum
	Membrane filtration			increase of 8 NTUs from
	maximum 0.1 NTU in			background levels at any
	95% of monthly			one time when
	measurements, greater			background levels are
	than 0.1 NTU for more			between 8 and 80 NTUs.
	than 15 minutes should			Should not increase more
	warrant an investigation			than 10% of background
				levels when background
				is > 80 NTUs.
British	Water should be <1 NTU	With Treatment:	Change of	Change from background
Columbia		Change from	background from	by 8 NTU for 24 hours, or
		background of 5	5 NTU when	2 NTU at anytime for a
		NTU when	background is	duration of 30 days, or a 5
		background is	<50 NTU	NTU from a background
		<50 NTU.		of 8-50 NTU during high
		Change from		flows or in turbid waters,
		background of		or a change of 10% in
		10% when		waters <50 NTU.
		background is		
		>50 NTU		
		Without		
		Treatment: Change of 1 NTU when less than 1 NTU, otherwise change of 5 NTU, if water is >50, change of 10% from background		
------------	--	--	--	---
Alberta	Differs with type of filtration, should stay below 0.3 NTU, may reach 1 NTU for 15 minutes per day, sand filtration may be under 3 NTU for short periods of time	N/D	<50 NTU, unless users are dissatisfied	For clear waters: maximum increase of 8 NTU for short term, maximum increase of 2 NTU for longer term exposure. For high flow/turbid water: maximum increase of 8 NTU when background between 8 and 80 NTU. Should not exceed 10% background >80 NTU
Yukon	Same as Canadian Guidelines	N/D	Same as Canadian Guidelines	N/D
US-EPA	Differs by type of filtration. Conventional filtration turbidity cannot exceed 1 NTU, samples must be < 0.3 NTU at least 90% of month. For other filtration types turbidity must not exceed 5 NTU.	N/D	N/D	Guidelines indicate water should not have objectionable turbidity.
Washington	According to EPA Guidelines	N/D	N/D	Varying depending on type of aquatic life, typically a 5-20% increase
WHO	Differs with type of filtration, < 0.3 NTU with no measurements exceeding 1 NTU, or <1 NTU with no measurements exceeding 5 NTU.	N/D	No absolute value, only that there should not be significant change from background	N/D

In general, Canada has more stringent standards for turbidity in water sources, with many standards indicating turbidity may not be safe with changes as small as 1 NTU or 5 NTU (Table 1). There is also text related to the protection of drinking water sources in the BC DWPA, though the act does not reference turbidity specifically, nor is there mention of source water turbidity elsewhere in the legislation or DWPO manual. Additionally, there are a few publications by the federal government referencing source-to-tap protection, and in BC the only publications on community watersheds or source-to-tap information are brochures for water suppliers, not for DWPOs. This indicates that though the standards may be strict, there may be little enforcement of drinking water protection or injection of source water protection in actual land use planning.

## **1.5** Turbidity in this Thesis

Overall, turbidity has the potential to be a highly useful measure for water quality monitoring of many objectives. However, many uncertainties remain as to turbidity dynamics in the field. This thesis addresses four major questions:

- 1. How does turbidity behave at long- and short-time scales?
- 2. What are the major drivers of turbidity fluctuation?
- 3. How does turbidity behave in small, clear streams with limited land uses?
- 4. How does mixed land use impact turbidity dynamics?

To answer these questions, we collected turbidity data to form a time series from two watersheds in coastal British Columbia. In the following chapters, we analyze these separately per watershed. Our second chapter focuses on turbidity through time, analyzing turbidity dynamics in forested sub-montane streams through seasonal and hydrological changes. We present turbidity at long and short time-scales, with a specific focus on monitoring over the course of a rainstorm. The third chapter analyzes turbidity in time and space, tracking the turbidity in the Shawnigan Lake Watershed, a developing area with a variety of land uses and water quality concerns. Here we compare continuous and spot turbidity measurements, along with several other water quality measures. In each of the chapters we discuss the implications of our research for turbidity science, ecology and finally water quality management in BC.

# Chapter 2: A characterization of turbidity dynamics in small streams over differing time scales in the Malcolm Knapp Research Forest

# 2.1 Introduction

Turbidity is one of the most common ways to assess water quality (Bilotta and Brazier 2008). It is a measurement of the amount of light scattered at a specific angle by particles suspended in water (Sadar 1998). Due to the general nature of the turbidity measurement, turbidity is typically positively correlated with concentrations of most particles found in water bodies (Finlayson 1985; Gippel 1995). As high particle concentrations can have negative impacts on aquatic ecosystems, as well as harbouring water-borne pathogens, elevated turbidity can also be an ecosystem and a public health concern (Ryan 1991; Seehausen 1997; Health Canada 2012). Furthermore, both sediment and turbidity have been positively associated with many types of land uses and anthropogenic activities (Ryan 1991; Cunha et al. 2016). However, the major reason for turbidity's increased use in monitoring water quality is that it can be measured *in situ*, continuously, and without complex sampling regimes (Downing 2006; Rasmussen et al. 2009). As such, turbidity has become a staple in water quality monitoring, particularly in the form of assessing movement of particles in aquatic environments.

The dynamics of turbidity and other particles have been characterized by many authors. Turbidity has been found to have positive relationships (typically  $r^2 \ge 0.7$ ) to various particles, including suspended solids, suspended sediments, and organic particles (Finlayson 1985; Gippel 1989; Grayson et al. 1996; Gippel 1995; Lewis 1996; Dodds and Whiles 2004). Exact relationships between turbidity and particles are specific to both the water body and to the particle, for example particles of different sizes (Gregory 1984; Gippel 1995), and between organic (Saraceno et al. 2017) and inorganic particles. It has also been noted that turbidity does not scale directly with discharge. Several studies on rivers (Likens 1969; Jordan 2006) identified that turbidity peaks did not correspond to periods of the highest discharge, but instead occurred during rain events after a dry period. Time of year also appears to be a large contributor, where turbidity increases can occur due to potential organic inputs at certain times of the year (Goni et al. 2014) and hydrological changes (Kiffney et al. 2002; Gomi et al. 2005).

While turbidity dynamics have generally been studied in rivers with a variety of anthropogenic impacts, and not often in small streams. Smaller streams typically have smaller particle loads than rivers, due to the lower discharge, but have more variable responses to weather events (Davies-Colley and Smith 2001; Bilotta and Brazier 2008). Therefore, turbidity response may vary with stream size. For example, Miller et al. (2015) and Keim and Schoeholtz (1999) found little correlation between suspended sediment concentration and turbidity in small, forested streams. Additionally, at lower turbidities, the likelihood of turbidity being influenced by particle size or type is higher. For example, Yao et al. (2014) and Ziegler et al. (2014) identified that turbidity relationships with particle concentrations change dramatically as turbidity increases from 0.5 NTU to 30 NTU. Jordan (1996) identified that streams with high turbulence yielded overestimated turbidity measurements due to the interference of bubbles or sensor obstructions. Small streams have a high potential for turbulence, though at such low turbidities, even small amounts of interference may affect turbidity measurements. Therefore, it is important to understand turbidity dynamics and drivers in small stream ecosystems.

Understanding turbidity fluctuations also requires analysis over small time scales. Due to varying relationships between suspended solids and discharge over the course of a storm, a phenomenon called hysteresis can occur, where sediment concentration decreases despite increasing or sustained discharge rates. Williams (1989) identified small streams as "sediment starved streams", where suspended solid concentrations typically peak before discharge, and a dilution effect occurs as discharge continues to increase while suspended solids may not (Williams 1989; Landers and Sturm 2015; Lloyd et al. 2016).

Our study seeks to understand turbidity fluctuation in small streams, to better comprehend the drivers of turbidity over long- and short-time scales. We measured the turbidity of three forested, sub-montane streams for over a year in the UBC Malcolm Knapp Research Forest, and compared it to hydrological and weather variables. We also analyzed ten peak discharge events for hysteresis dynamics, five hydrologic events and five high turbidity events, and analyzed three storms in detail for particulates contributing to turbidity fluctuation. This was to produce a reference of variability for turbidity. We expected that turbidity would respond most to discharge, as this would increase capacity for transport of suspended particles. We expected that turbidity would be most related to total suspended solids over the course of a storm, and that each of the streams would demonstrate clockwise hysteresis loops, typical for sediment-starved streams.

# 2.2 Methods

#### 2.2.1 Site Description

Data were collected at the UBC Malcolm Knapp Research Forest in Maple Ridge, British Columbia, approximately 45 km from Vancouver (Kiffney et al. 2003; Kiffney and Richardson 2010; Leach and Moore 2015). The research forest is 5,157 ha, all within the sub-montane region (50 to 500 m above sea level) (Miquelajauregui 2008; Hoover et al. 2011). The research forest is a Coastal Western Hemlock Rainforest, and the main vegetation consists of western hemlock (*Tsuga heterophylla*), Douglas-fir (*Pseudotsuga menziesii*) and western redcedar (*Thuja plicata*), along with the deciduous tree red alder (*Alnus rubra*) (Hoover et al. 2006; Hoover et al. 2011; Leach and Moore 2015). Riparian areas are typically dominated by red alder, or salmonberry (*Rubus spectabilis*) (Hoover et al. 2011). The forest has a history of forest harvesting. Most of the forest was logged at the beginning of the 20<sup>th</sup> century, and experienced a large fire in 1931 (Kiffney et al. 2002; Kiffney and Richardson 2010). As such, the forest is primarily second growth with an approximate age of 85 years at the time of the study (Kiffney et al. 2002; Kiffney and Richardson 2010). Since 1931, it has been logged sparingly, for research activities or commercial logging.

Rainfall is high and temperatures are mild. Mean annual precipitation is 2184 mm/year, though this may be an underestimation as measurements are made at a lower altitude (approximately 150m) (Leach and Moore 2015). Approximately 95% of precipitation is rainfall, falling mostly between October and April, and snow accounts for 5% of total precipitation, increasing with elevation (Leach and Moore 2015). Air temperature ranges from a mean monthly high of 22.7°C in July to a mean monthly low of -0.5°C in January, averaging approximately 9.6°C over the year (Hoover et al. 2011). These variables are also significantly impacted by the El Niño Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO), and so can vary highly between years (Kiffney et al. 2002).

Small streams (first to third order), as well as the upper reaches of the North Alouette River, are present in the forest. They range in size from 0.5 to 3.0 m wetted width (Kiffney and Richardson 2010). Most streams drain into this river or the Pitt River, and eventually to the Fraser River. Although salmon are absent, however cutthroat trout are present in the lower areas of East Creek (Kiffney et al. 2002). Most streams drain thin glacial till (from a soil type of coarse-textured humo-ferric podzol) approximately 1 m in thickness, with igneous bedrock underlying (Feller and Kimmins 1979). Stream substrate is mostly composed of gravel and cobbles, with some boulders, as well as sand and gravel and organic detritus at pools or debris jams (Kiffney et al. 2003). Streamflow typically responds rapidly to increasing precipitation, especially in saturated antecedent moisture conditions (Leach and Moore 2015).

Three headwater streams were selected within the forest with slightly different hydrological and land use histories: Griffith, Upper East and South creeks (Table 2.1 and Fig. 2.1), and their water chemistry has been described previously (Feller 1977; Feller and Kimmins 1979). Griffith and South creeks were subjected to selective riparian harvesting studies, in 1998 for South Creek, and Griffith in 2004 (Kiffney et al. 2003; Kiffney et al. 2010). The riparian harvesting studies sought to understand the long-term impacts on stream ecohydrology under different harvesting techniques, including partial harvesting within the riparian zone, or clearcutting with riparian buffers of fixed-widths (Kiffney et al. 2003).

Creek	Catchment Area (Ha, above the weir)	Gradient (%)	Stream Width (m)	Elevation Range (m)	Mean Discharge (m³/s)	Discharge Range 2016 (m³/s)	Forest Harvesting Treatment
Upper East	35	8	1.8-2.2	280-447	0.031	<0.01 – 0.73	None
Griffith	11	15	0.5-0.9	450-520	0.14	0.02 - 0.73	50% riparian tree removal in 2004
South	18.6	10	0.8-2.2	270-320	0.11	<0.01 – 0.11	30 m buffer reserves in 1998

**Table 2.1:** Summary of hydrological variables for Malcolm Knapp watershed study creeks. Data taken from Kiffney et al. (2002), Kiffney and Richardson (2010) and Leach and Moore (2015).

# 2.2.2 In-Stream Turbidity



**Figure 2.1:** Stream sites in the Malcolm Knapp Research Forest, Maple Ridge, British Columbia, Canada. Pictures are taken downstream of the weir and weir house, underneath which is the sensor. A) Griffith Creek, B) South Creek, C) Upper East Creek.

Turbidity was measured in Nephlometric Turbidity Units (NTUs) with Campbell Scientific OBS3+ Turbidity Sensors, with a 0 to 4000 NTU range (Campbell Scientific, Edmonton, AB). Conductivity and temperature measures were obtained using CS547A Conductivity and Temperature Probes, as well as CS451 Pressure Transducers sensors (for water level) installed at each creek at the weir (all from Campbell Scientific, Edmonton, AB). Sensors were attached to a Campbell CR1000 Data Logger (Campbell Scientific, Edmonton, AB). Measurements were taken every 15 seconds, and averaged to achieve 15 minute resolution. Weirs were visited approximately every 1-3 months for all of 2016 and 2017 for maintenance, including sensor cleaning, battery changing and replacing desiccants to remove excess moisture around the electronics. During each visit, a spot turbidity sample was taken using a Hach 200q handheld sensor (Hach Instruments Co, Loveland, CO, NTU range of 0 to 1000 NTU), and pH, specific conductance, total dissolved solids, and temperature were measured using a YSI Water Quality probe for QA/QC of the monitoring stations (YSI Inc./Xylem Inc. Yellow Springs, OH).

Turbidity samples were taken according to USGS guidelines, with the addition of a near zero NTU standard made with Isopropyl alcohol to account for potential cuvette scratches (USGS 2005). In some instances, turbidity measures from the Hach 200q sensor were manually calibrated due to some programming errors in the sensor leading to a failure in the calibration program and the high NTU level of the standards. The manual calibration fitted a linear regression to the values from a standard formazine solution to calculate the slope, and the zero standard of isopropyl alcohol was used to calculate the intercept. This equation was then applied to field measurements. In cases where the calibration program was effective, the same process was applied to the 1.0 NTU standard and the <0.1 NTU standard that were outside the normal calibration range. Turbidity data from the OBS3+ sensor were also cleaned to eliminate possible errors from insects, biofouling,

branches or bubbles, which can result in false turbidity values (Ryan 1991; Gippel 1995; Lewis and Eads 2001). This was done through removing maximum turbidity values resulting from confirmed artifacts, and results with similar signatures.

Discharge was calculated using rating curves for these three weir stations (Jason Leach, Natural Resources Canada, *personal communication*). These are as follows, where H is stage height in cm:

- Upper East  $Q = 0.015 H^{2.59}$ ,
- Griffith  $Q = 0.0202H^{2.286}$
- South  $Q = 0.00617 H^{2.65}$ .

With these rating curves, South Creek has a notably smaller discharge than the other two creeks. Weather data were obtained from the Malcolm Knapp Research Forest Field House, which takes hourly weather measurements of total rainfall, rain rate, and temperature.

# 2.2.3 In-Stream Turbidity

#### 2.2.4 Storm Turbidity

Additional data were collected over 3 storms using an ISCO 6700 (Teledyne ISCO, Lincoln, NE) and a Sigma 400 (Hach Instruments, Loveland, CO) automatic water sampler. Samplers were cleaned with an acid-based detergent Extran prior to deployment and run for 2 full cycles with de-ionized water (Millipore Sigma, Burlington, MT). Each ISCO collected samples every 1 or 2 hours over a 24 or 48 h period, respectively, depending on the duration of the storm. Spot samples were collected at all three creeks prior to the storm and when samples from the

automatic samplers were collected according to USGS guidelines (USGS 2005). Samples taken every 2-4 hours were then vacuum filtered using a 100 µm sieve and pre-ashed 25 mm Whatman-GF/F filters (0.7 µm pore size) within 36 hours of collection, using APHA protocols (Whatman Inc, Little Chalfont, UK.; APHA 2005). Samples were then dried at 60°C, and ashed at 550°C for 1 hour, to get estimates of mass per unit volume of inorganic and organic portions of both TDS and TSS, following protocols of EPA (EPA 1993). Turbidities were also taken of each sample and the resulting filtrate using the 200q Hach handheld turbidity meter.

#### 2.2.5 Model Development

Data analysis for both continuous and storm data was performed using Linear Mixed Effects Models to understand the relationship between hydrology and weather on turbidity using R-Studio (Kutner 2005; R-Studio 2015). Most of the variables were already included in the dataset, with the exception of antecedent moisture conditions, which we calculated based on the sum of precipitation over the previous 2 or 5 days (AM<sub>2</sub> and AM<sub>5</sub> respectively), using the methods in Mukundan et al. (2013). Once all variables were accounted for there were several additional steps prior to model selection. First, to identify potentially important variables, all variables were graphed against turbidity, and individual r<sup>2</sup> terms were calculated. All variables appeared to have some relationship to turbidity, and so we then used a principal components analysis (PCA) to visualize the variation space to understand impactful variables and how variables were grouped (Seeger et al. 2004). Once variables were chosen, we then went ahead with applying our statistical models. We used two linear mixed-effects model methods. First was Generalized Least Squares (*Gls*) (with stream and season as factors, and correlated through time per stream) similar to Gomi

et al. (2006), to understand the importance of individual streams and seasons. We also used a Linear Mixed Effects Regression (*Lmer*) method (with month as a factor, using hierarchical correlation by stream), to better understand the relationships between variables and turbidity across the three streams. We then compared models using AIC and ANOVA. As our variables were highly related in time, we applied an AR1 (autoregressive) error covariance structure to reduce temporal autocorrelation (Pfaff 2008). For the same reasons, we also used a principle component regression to understand the relationship between our variables using orthogonal principle components from our earlier analysis (Zuur et al. 2009).

Storm data followed a similar protocol. These data were first analyzed using basic linear regression, then using a PCA (again to understand groups of variables driving turbidity variation and how the data varied by stream), and then further analyzed using a linear mixed effects model approach for the 3 storms captured. Variables included total inorganic/organic dissolved solids ( $< 0.7 \mu m$ ), total inorganic/organic suspended solids (divided into greater than, or less than 100  $\mu m$ ), and discharge as fixed effects.

### 2.2.6 Hysteresis Analysis

For individual storm dynamics, we followed the methods used by Lloyd et al. (2016), based on Williams (1989). This involved the graphing of discharge and turbidity to manually describe the shape of the hysteresis curve, which determines the sediment availability during the hydrological event. For this, we selected 13 storms. Three were the storms during which the automatic water samples were used, to compare hysteresis patterns to the particle dynamics, 17 were selected from high discharge events, and 3 were high turbidity events.

#### 2.3 Results

We collected turbidity for the entirety of 2016 and 2017, though we only analyzed turbidity for 2016 with the exception of one storm event in March of 2017 (Fig. 2.2). There were a few instances when turbidity sensor data failed to record, and we additionally removed approximately 2 weeks of data due to sensors becoming dirty or being obstructed. We were able to correct some values by modelling the relationship between continuous and spot samples in January for East Creek.

Before cleaning, turbidity ranged from 0.35 NTU to approximately 400 NTU, and after cleaning the range was from 0.35 to 19 NTU. Turbidity hovered around 0.8 NTU during lower flows, and rarely went below 0.7 NTU. Turbidity peaks were generally below 5 NTU, and rarely exceeded 10 NTU. As no spot s samples could confirm turbidity levels over 20 NTU, we estimated that most values over 20 NTU were due to artifacts. As such, all values over 20 NTU were removed, resulting in data gaps throughout the sampling period. We also were able to calculate the slope of biofouling during the month of January using spot samples and were able to transform the data down to remove the biofouling effect. There was also a sustained higher turbidity period (~3 NTU) in South Creek in August and September of 2016, which was confirmed by spot sampling, and was not due to sensor error, however, it could have been due to weir cleaning activities and benthic disruption.

Discharge during 2016 differed by creek as seen in Table 2.1 where South Creek was an order of magnitude smaller than East and Griffith creeks. There were very low flows occurring in the summer, and higher flows in January through April, then low flows again in September and October. The year 2016 was a bit different than most years, in that it experienced both lower and higher temperatures and precipitation. There was lower precipitation and warmer temperatures than average from January through September, with a sustained period of sparse rainfall throughout the summer. In October and November a large storm produced continual precipitation, increasing runoff during this period, followed by colder temperatures, snow and decreasing runoff for the end of the year. This nonetheless made for many interesting hydrological events to capture within the same year.

#### 2.3.1 Continuous Turbidity

Turbidity seemed highly related to both discharge and rainfall though these relationships changes. When graphed, these relationships appeared to differ by creek and by season (Fig. 2.3). For example, the largest peaks were present in the late spring, summer and early fall periods, with few large peaks occurring in the winter. The largest peaks occurred around June and September for all three creeks, though each creek showed slightly different timing for turbidity peaks, and it was around these periods that sustained high base turbidity was also apparent. Overall, there appeared to be a positive relationship between turbidity and discharge, though in South Creek the overall trend was negative. This is possibly due to the high turbidity in South in the summer, when discharge values were quite low. In comparison with Fig. 2.2, it can be seen that many turbidity peaks occurred at periods of high discharge, though not all. Additionally,  $r^2$  values were quite low,

as shown in Fig. 2.3. It should be noted that  $r^2$  values are not accurate with the temporal intercorrelation with these data. Interestingly, several turbidity peaks and periods of high turbidity, particularly those in the late spring/summer months, occurred in the lower range of observed discharge.

In our Principle Component Analysis, the first two PC axes explained approximately 51.2% of the variation (Fig. 2.4). Points in the PCA grouped by both creek and season. Though similar to the turbidity and discharge relationship, these relationships appear to vary by both season and creek. Our first principle component explained 32.8% of variance, and appeared largely related to flow-related variables, such as discharge, as well as temperature and conductivity (which are indirectly related to flow by season or by baseflow conditions/groundwater percentage). The second principle component explained 18.4% of variance, and was higher for variables such as water level, rain and antecedent moisture conditions. It is possible the second component is related to antecedent moisture conditions and the availability of loose sediments/debris in and around the stream, as determined by the length of dry periods between rain events. There are also 5 observations that appear to be high on both PCAs.

We then applied two types of statistical models to understand driving forces of turbidity. We utilized two linear mixed-effects model methods: *Gls* (Generalized Least Squares) analyzing the impact of stream and season and *Lmer* (Linear Mixed Effects Regression), a hierarchical model to understand the contribution of each of the variables. This included conductivity, discharge, antecedent moisture conditions and rain as fixed effects, as well as stream and season. Each model was significant, though a 2-way ANOVA favoured the *Gls* model. All variables were significant except for antecedent moisture conditions over the past 5 days in both models, though rain and winter were also not significant in the *Lmer* model. Discharge had the highest impact on turbidity, with the largest coefficient for both models. The second was a Principle Component Regression, which found that the two components as displayed in Fig. 2.3 were significant (P<0.001).



Sep

Dec

0.03

0.00

Mar

Jun

Date





**Figure 2.3:** Turbidity plotted against rainfall (A) and discharge (b) separated by season for East, Griffith and South creeks. Turbidity is shown on a log scale. Trend lines are basic linear regression for the full year and show overall positive relationships between turbidity and discharge for East and Griffith, but negative relationships with South Creek. Points show that these relationships may vary by season.



Season • winter • spring • summer • fall

**Figure 2.4:** Principal Components Analysis of measured variables related to turbidity in the Malcolm Knapp Research Forest. The two principal components explained approximately 51.2% of the variation. Discharge in m<sup>3</sup>/s, temperature in °C, conductivity in  $\mu$ S/cm, water level is in cm, rain is mm/h, rain rate is rainfall in mm/s, AM<sub>2</sub> and AM<sub>5</sub> are in mm). Graph A) represents points separated by stream and B) depicts points separated by season. In both, there are overlapping points, but a defined grouping effect is present.

Model	AIC	LogLik	Correlation Phi	Anova to Null	Anova between Models		
Null Gls	-22097.48	11051.74	0.934				
Gls	-23173.16	11598.58	0.933	P < 0.01*			
Null Lmer	-22270.37	11140.18	0.929				
Lmer	-23305.84	11661.92	0.917	P < 0.01 **	P < 0.01*		

**Table 2.2:** Linear mixed-effects model results for drivers of turbidity, with coefficients as seen in Table 2.3.

 \*Indicates significance.

**Table 2.3:** Linear mixed-effects model coefficients for drivers of turbidity, with model comparisons in Table 2.2.

 \*Indicates significance.

Gls Mo	odel		Lmer Model		
Coefficient	Value	<b>P-value</b>	Coefficient	Value	<b>P-value</b>
Intercept	-0.50	< 0.01*	Intercept	-0.50	< 0.01*
(log) Conductivity	0.14	< 0.01*	(log) Conductivity	0.14	< 0.01*
(log) Discharge	6.10	< 0.01*	(log) Discharge	6.15	< 0.01*
(log) Rain	0.02	< 0.01*	(log) Rain	0.01	0.06
(log) Antecedent Moisture 2	0.03	< 0.01*	(log) Antecedent Moisture 2	0.06	0.01*
Days			Days		
(log) Antecedent Moisture 5	0.01	0.10	(log) Antecedent Moisture 5	0.00	0.98
Days			Days		
Stream Griffith	-0.37	< 0.01*	Stream Griffith	NA	NA
Stream South	0.24	< 0.01*	Stream South	NA	NA
Season Winter	0.07	0.01*	Season Winter	NA	NA
Season Spring	-0.32	< 0.01*	Season Spring	NA	NA
Season Summer	-0.11	0.01*	Season Summer	NA	NA

# 2.3.2 Storm Turbidity

We captured three storms in detail for each stream using automatic samplers, on Nov. 4 and Dec. 5, 2016 and March 3, 2017. These storms were analyzed for turbidity, but also for mass per unit volume of inorganic and organic dissolved and particulate materials (full results found in Fig. A1). The storms were picked based on rainfall events and lasted for approximately 2 days. Turbidity ranged from 0.7 NTU to 4 NTU, while discharge ranged from a low of 0.02 m<sup>3</sup>/s to 0.17 m<sup>3</sup>/s. Overall, particle mass per unit volume have individual relationships with turbidity, which

differ between streams (Fig. 2.5). Organic particles appear to have positive relationships across the three streams, while inorganic particles tended to have negative relationships.

To quantify this, we performed a PCA and applied Linear Mixed-Effects Models, similar to the continuous turbidity. Our PCA (see Fig. 2.6) explained approximately 84.3% of the variation in the first two components. Our first PC appeared to relate to particle concentration, while the second PC appeared to suggest the makeup of the particle, inorganic or organic. Due to the low sample size, we were unable to perform a principle component regression on these data. Our linear mixed-effects model, however, had no significant variables. Furthermore, our error range for several of these data was also quite high.



**Figure 2.5:** Turbidity and particle mass per unit volume for the three storms analyzed in detail by time for East, Griffith, and South Creeks. Particles include Total Dissolved Solids (TDS), Total Suspended Solids (TSS), Organics, Inorganics, and Fines (organic and inorganic). Particles appear to have little relationship with turbidity



**Figure 2.6:** Principal Components Analysis of the three storms where particles were measured in addition to turbidity. The two principal components explained approximately 84.3% of the variation. Particles included TSS (Total Suspended Solids), TDS (Total Dissolved Solids), organics, inorganics, and fines (< 100  $\mu$ m). Graph A) shows points separated by stream and B) depicts points separated by season. In both, there are overlapping points, but a defined grouping effect is present.

Model	AIC	LogLik	<b>Correlation Phi</b>	Anova to Null	Anova	between
					Models	
Null Gls	-25.45	15.73	0.833			
Gls	-29.67	25.84	0.513	P = 0.01*		
Null Lmer	-29.64	23.82	0.671			
Lmer	-23.46	15.73	0.801	P = 0.01*	P = 0.13	

**Table 2.4:** Linear mixed-effects model results for drivers of turbidity during storm periods, with coefficients as seen in Table 2.5. \*Indicates significance.

**Table 2.5:** Linear mixed-effects model coefficients for drivers of turbidity during storm periods, with model comparisons in Table 2.2. \*Indicates significance.

	Gls Model		L	Lmer Model		
Coefficient	Value	<b>P-value</b>	Coefficient	Value	P-value	
Intercept	0.045	0.60	Intercept	0.086	0.20	
(log) TSS	16.11	0.44	(log) TSS	13.92	0.44	
(log) TDS	19.44	0.43	(log) TDS	18.15	0.40	
(log) Organics	-9.82	0.60	(log) Organics	-11.68	0.49	
(log) Inorganics	-11.41	0.55	(log) Inorganics	-9.40	0.60	
(log) Fines	-5.21	0.68	(log) Fines	-5.62	0.58	
Stream Griffith	0.033	0.75	Stream Griffith	NA	NA	
Stream South	0.18	0.15	Stream South	NA	NA	

# 2.3.3 Hysteresis

Finally, we analyzed hysteresis for three storms (Fig 2.7), as well as 10 other events (Fig. 2.A2). These events varied in time of year and discharge level, and included some high turbidity events. We applied Williams' (1989) method of classifying the shape of the curve as clockwise or counterclockwise, shown in Fig. 2.A2. An example of hysteresis from the February 13<sup>th</sup> storm can be found in Fig. 2.7. The hysteretic loops for all storms can be found in Appendix 1, Fig. 2.A2. The majority of the hysteresis loops were clockwise, though there was one instance of a counterclockwise loop (Fig. 2.A2). In addition, two events could not be classified into loops, due to a lack of discharge in that stream or at that event. Turbidity often followed the pattern of discharge closely, showing two peaks if discharge was bi-modal, however, some events showed multiple

peaks despite a smooth hydrograph, which may be evidence of more complex relationships. An example of a clockwise, bimodal and the counterclockwise loop can be found in Fig. 2.7. Maximum turbidity of over 10 NTU occurred over brief periods in October, September, December, August and May, sometimes without a noticeable increase in discharge, as seen in Fig. 2.A3.





**Figure 2.7:** Hysteresis diagram from February 13<sup>th</sup> storm from three creeks in the Malcolm Knapp Research Forest. Graph is of turbidity versus discharge. Arrows show hysteresis loop direction.

# 2.4 Discussion

# 2.4.1 Turbidity over Long-Time Scales

Our study measured turbidity fluctuation over short- and long-time scales, and in a variety of hydrological and seasonal conditions. We collected continuous turbidity data over a year-long period, and compared it to discharge, temperature, rain, and other variables. We also performed additional analyses on storms, identifying turbidity patterns and particle fluctuations over the course of the storms. Overall, we found that turbidity was generally related to stream discharge and rainfall, with many variables shifting seasonally within these two conditions (water level, conductivity, temperature, etc.). We found that turbidity was instead most affected by season, and in particular responded to "flush" events, the first in a series of storms. We also found that turbidity was not affected by organic particles over the course of a storm, as seen by our model.

The relationship between discharge and turbidity has been documented for some time (Lewis 1996; Lessels and Bishop 2013). However, several authors, such as Lloyd et al. (2016) and Miller et al. (2015) report small relationships between discharge and turbidity. This also fits with Pfannkuche and Schmidt (2003), who identified that suspended sediments and discharge are often related but can have low correlation. Many of our findings were in contrast to the literature. For example, Iglesias et al. (2014) identified that turbidity was best predicted by temperature, while we found that temperature was significant, it was most likely to be confounded by season and discharge. This may be a factor of land use within the watershed, as our streams had limited historical land use, so remained fully shaded and with no wastewater outflow, as was present in

Iglesias et al. (2014). Finally, we found season to be a large contributor to variation in turbidity, with high turbidities occurring in the spring and summer seasons, a result not yet noted.

# 2.4.2 Turbidity at Short-Time Scales

We measured turbidity at short-time scales of a few days during storm events, to better understand the influence of different particles on turbidity and the relationship of turbidity and discharge. The linear relationship between turbidity and suspended solids, sediments or organics have been known to break down at both low and high turbidities due to the impact of the particle sizes at high concentrations (Lewis 1996; Slaets et al. 2014; Yao et al. 2014; Ziegler et al. 2014). For example, Landers and Sturm (2013) identified that small size changes in wash material altered the amount of suspended sediment at a given turbidity. We analyzed three streams in detail, with the hypothesis that turbidity would be highly influenced by suspended solids, due to the large amount of research demonstrating this relationship. However, contrary to our expectations we found that turbidity was not significantly explained by any specific particle type, though our comparisons and PCA did suggest that organic particles may have a positive relationship with turbidity. Slaets et al. (2014) also found that turbidity was influenced by organic matter concentration, however, this varied depending on the degree of particle clumping, and that more dissolved forms had higher turbidity than clumps. This has three potential meanings: that organic particles are a large component of small stream turbidity, that the storms we analyzed simply had too low levels of discharge to mobilize sediments, or that our samples were likely below the limit of our detection, and this result should undergo further testing. If the former, this would interestingly add to the hypothesis that turbidity at low levels would be heavily influenced by many

particle types and estimating a single particle type may be inaccurate at low levels (Bilotta and Brazier 2008; Yao et al. 2014).

We also found that turbidity generally experienced hysteretic relationships with discharge over time. We found many instances of hysteresis in the clockwise direction and several that could not be classified (without full increase in discharge). Clockwise hysteresis curves indicate a dilution effect, where discharge reaches high levels during storm periods, or sediment quickly runs out (Williams 1989; Brasington and Richards 2000; Macdonald et al. 2003; Langlois et al. 2005). This pattern is expected for small streams with limited sediment availability (Williams 1989). Other authors have also identified that clockwise hysteresis indicates that sediments originate from the local area, rather than further away in the watershed (Heidel 1956; Church and Gilbert 1975; Walling and Webb 1981). Additionally, Hughes et al. (2012) found that the watershed with less intensive land use was also associated with a clockwise hysteresis curve.

We also identified more complex hysteresis patterns, such as two peaks, one counterclockwise and one clockwise discharge curve (Fig. 2.A2), as well as two events with no peaks. The variation in hysteresis relationships matches that seen by Reid et al. (2016) and Lloyd et al. (2016), where both found a majority clockwise pattern, but many other patterns as well. The bipeaked curves indicate that turbidity is produced by discharge increases rather than total discharge. As discharge was estimated based on water level, this may indicate that these peaks represent the increase of water level, and so increase in particles from immediately around the stream. This also matches the clockwise hysteresis model of particles originating from close to the stream, such as after a flood (Church and Gilbert 1975; Walling and Webb 1981; Asselman et al. 2003). Our counterclockwise curve would then be the opposite, suggesting that sediment came from further away in the watershed. However, this was the highest discharge we recorded, at nearly 0.8 m<sup>3</sup>/s,

46

and this suggests that perhaps this was due to temporary local flooding, or a stochastic pulse (such as the loosening of large wood or a bank cave in). Finally, Zabaleta et al. (2006) identified a number of events classified as "lined", which were explained as events where the stream responded directly to rainfall in the immediate area. Some events in South Creek could be described this way, indicating that turbidity response to rainfall apart from the increase in discharge. This could also possibly mean that South responds to the surrounding forest more than the other two creeks, due to the type of historical land use or the underdevelopment in understory vegetation around the riparian zone at South Creek.

# 2.4.3 Implications for Turbidity Management

A large motivation for this study was to inform both watershed managers and ecologists about turbidity dynamics in small streams. The current B.C. Ministry of Environment Water Quality Guidelines recommend a surface drinking water source be considered unclean if there is a change of 1 NTU (interpreted as an increase of 1 NTU) with a background of <5 NTU in areas with no particulate treatment, while areas with particulate treatment have higher guidelines of a change of 5 NTU when background is < 50 NTU (BCMOE 2017). The BCMOE also suggests that sustained increases of 2 NTU for 30 days or an increase of 8 NTU for 24 hours may be outside of ideal conditions for aquatic life (BCMOE 2001). The streams in our study showed many small increases of 1 NTU, and sustained seasonal increases of 1 NTU, as well as many instances of changing by 5 NTU, and even one change of 2 NTU for 30 days. Our streams have past land uses, but otherwise show signs of low sediment availability (with the hysteresis direction and dilution effects), and yet may not comply with regulations for both drinking water and protection of aquatic

life. Based on the small increments in turbidity required to affect water quality management, our research has three potential implications.

First, our results suggest that even relatively intact streams are on the border of being considered unsuitable for drinking water supplies or harmful for aquatic life. This suggests that slight increases in sediment runoff in headwater catchments, due to land use or other causes, may result in stream turbidities being higher than recommended levels. This implies that watershed management will therefore be strict. Additionally, the timing of the high turbidity events coincides with lower rainfall and water availability in Southern B.C. (e.g. summer droughts). This may present a further stress on water provisioning, as water quality is degraded at the same time water quantity may be decreasing in lakes or reservoirs. Alternatively, it may mean that using current water quality methods for small streams have many periods over the course of the year where turbidity exceeds guidelines. Additionally, many turbidity increases appeared to be highly seasonal, and related to local hydrology and instream metabolism processes. This suggests that watershed management for the purpose of water quality may be attempting to reach an unattainable goal for surface water, at least for small streams.

Secondly, it was very easy for biofilm to build up or for obstructions to affect the sensors, causing sudden artificial increases in turbidity. Additionally, our hand-held sensor was typically nearly 0.5 NTU higher than our continuous sensor, as both sensors were at the lower end of their detection limits. This could be increased by heterogeneity of particles within the stream, water temperature causing fogging, or simply wet conditions, making it difficult to assess water quality in the range of 0-5 NTU. Finally, even with accurate and well-maintained sensors, we found it difficult to determine background turbidity from seasonal/high flow turbidity and sensor errors. Turbidity appeared to be impacted by many highly localized processes, such as the presence of the

weir pond, the time since the last rain, or the water level, which are specific to each stream, and likely to a particular reach. Therefore, determining a change from background turbidity may be challenging without a sophisticated monitoring regime of continuous turbidity samplers in different areas to define background turbidity, a monitoring method procedure which is currently not in operation.

We suggest that the current guidelines may not adequately serve their purpose for safe drinking water in British Columbia. Whether these guidelines are used for goal-setting or monitoring purposes, they may be setting water managers up for failure. That is, identifying background turbidity correctly may not be possible for many water managers, potentially resulting in inaccuracies in determining the influence of land use and stochastic events. Furthermore, if the guidelines are challenging to adhere to, as evidenced by our streams, it suggests that compliance may be highly difficult, something which has not been reported. This begs the question of what guidelines are currently being used for monitoring and managing of water resources.

#### 2.5 Conclusions

We showed that turbidity has complex dynamics in headwater streams over the course of the year. Our data suggest that turbidity is highest during the spring and summer periods, during low water-level or flush events. We also identified that turbidity shows hysteresis dynamics at low levels and is likely driven by many different particle types. Finally, we discussed the potential concern of water quality guidelines referencing turbidity being unrealistic for current water bodies. This thesis also identified several uncertainties in the measurement of turbidity. That is, due to the degree of localization of turbidity, it is uncertain if turbidity variation is as different between reaches as it is between streams. Additionally, experimental tests of our predictions of what causes turbidity increases may also be necessary. Finally, we would suggest that future research focus on the concentrations of individual particles over the course of several turbidity events in low turbidity streams to understand causal drivers of turbidity`.

# Chapter 3: Variations in turbidity associated with hydrology, seasonality and land use in the Shawnigan Lake Watershed

# 3.1 Introduction

Surface water sources (including lakes, rivers, streams, and reservoirs) provide a large percentage of drinking water supplies worldwide, and approximately 75% of Canadians. These areas are also at high risk for degradation by human activities (Foley et al. 2005; Statistics Canada 2013). For example, estimates suggest that nearly 75% of the rivers in the United States have increased sediment loads due to human activities (USEPA 2006). This can have dramatic consequences for many rural communities which do not have access to municipally managed water supplies and receive their water directly from surface or groundwater sources (Pons et al. 2015).

Turbidity is commonly used to assess drinking water quality (Bilotta and Brazier 2008; WHO 2017 a,b). Turbidity is related to water clarity, and measures the amount of light scattered by different particles in the water column (Gippel 1991). Thus, turbidity can assess the visual aspects of water, such as how desirable water is to drink (Allen et al. 2008). Turbidity also has a linear relationship with suspended solids, and so is often used as a proxy for measuring the amount suspended particles in a water body (Gippel 1989; Dodds and Whiles 2004; Downing 2006; Bilotta and Brazier 2008). Due to this relationship, turbidity is associated with harmful compounds, such as heavy metals (Nasrabadi et al. 2016; Yao et al. 2016), and pathogens, for instance *Escherichia coli* or *Giardia* (LeChevallier et al. 1991; LeChevallier and Norton 1992; De Roos et al. 2017).

Recently, many associations between turbidity and drinking water-caused illness have also been identified, including gastrointestinal illness occurrences in New York, NY (Hseih et al. 2014) and the *E. coli* outbreak in Walkerton, ON (Hrudey et al. 2002). High levels of suspended particles can also have impacts on the aquatic ecosystem (Ryan 1991; Henley et al. 2000).

Despite its usefulness in monitoring drinking water, turbidity displays high variation and localization in its response to environmental drivers. That is, turbidity is understood to be affected by geological, weather and seasonal attributes (Kiffney et al. 2002; Gomi et al. 2005; Goni et al. 2014), as each aspect may alter the base load of suspended particles, as well as their sizes and textures, resulting in slightly different turbidities for each stream (Grayson et al. 1996; Lewis 1996). Moreover, land use generally results in increased concentrations of suspended particulates and higher turbidity (Townsend et al. 1997). These land uses include forestry (Leiberman and Hoover 1948), urbanization (Cunha et al. 2016), agriculture (Lenhart et al. 2010) and construction (Trenouth et al. 2016). However, the ways that these variables combine to cause turbidity increases and variation is not yet predictable. For example, turbidity typically increases as a response to rainfall or discharge, but this does not occur with every event, nor does turbidity always scale with discharge (Mukundan et al. 2013; Miller et al. 2015). These weather-driven turbidity responses are known to be specific to the individual stream (Grayson et al. 1996; Miller et al. 2015), but may also be specific at the reach scale (Lenhart et al. 2010). Understanding this variation is imperative to water quality management, as even small changes in turbidity can quickly degrade drinking water and/or impair aquatic habitat health (Logsdon et al. 1985).

Our study objectives were to assess drivers of turbidity within a watershed with a variety of land use intensities to better inform water quality policy. To do this we monitored turbidity from July 2016 to April 2017 in the Shawnigan Lake Watershed, located just north of Victoria, Canada,

as a case study. The watershed consists of three major streams that flow into a Shawnigan Lake, and a larger stream connecting the lake to the ocean. Shawnigan Lake is a developing watershed, with an increasing industrial sector in the south end, and historical dense residential developments along the lake. Many of these residences draw water from surface (mostly from the lake) or ground sources (wells), making water quality is a primary concern of residents (Rieberger 2007). There are at least three Authorized and Regulated water pruveyors that obtain a "municipal" supply of drinking water from the Lake for residential consumption. The Regional District monitors turbidity at the north end of the lake, and in a few key locations around the perimeter of the lake (Rieberger et al. 2004; CVRD 2018; VIHA 2018). Additionally, a recent study has been performed on South Shawnigan Creek to address concerns about growing industrial development (Rieberger et al. 2004; CVRD 2018; VIHA 2018). Bacterial blooms and phosphorus levels have also been a concern; however, those are outside the scope of this study (Rieberger 2007). Despite water quality concerns, few studies have been performed on the sources of pollutants and turbidity in the Shawnigan Watershed (Aqua-Tex Scientific 2014). We compared three sites along each of two streams with differing land uses, using weather and hydrological variables, to understand turbidity fluctuation and drivers of variation in turbidity. We predicted that turbidity would positively increase with weather-related measures across the watershed and between sites within the same stream, and that turbidity peaks and averages would be higher with a higher proportion of nonvegetated land area.

# 3.2 Methods

# **3.2.1** Site Description

The Shawnigan Lake Watershed (48.653680, -123.623120) is located on southern Vancouver Island, in southwestern Canada (Fig. 3.1). The watershed is small, with a drainage area of 110 km<sup>2</sup> (Rieberger 2007). The hydrological system consists of a small lake (Shawnigan Lake), and four major streams (used interchangeably with creek in this study): Shawnigan Creek, South Shawnigan Creek, Van Horne Creek, and McGee Creek. The outflow creek from the lake (Shawnigan Creek) also has a weir present, making the Lake a regulated reservoir. The coastal climate is typical for southern Vancouver Island, with wet and mild winters, and mild, dry summers. During the study period, the weather ranged from a daily mean of around  $3^{\circ}$ C in the winter (Dec 2016 to March 2017), to a mean of around 17°C during the summer (July to Sept) (Weaver and Wiebe 2018). Rainfall data were retrieved from the Shawnigan Lake Museum weather station, as part of the School Weather Monitoring Program, where rainfall and other weather variables are collected at hourly intervals (Weaver and Wiebe 2018). Unfortunately, weather stations were located in the northern part of the watershed and may not represent the exact weather at each site. The watershed also receives some snow in the winter, though snowfall level was not collected by the weather stations (Bryden and Barr 2002; Rieberger et a. 2007). The area ranges from 610 m at the mountain peaks to 116 m at full pool lake level (Bryden and Barr 2002; Rieberger et a. 2007). The geology is shallow bedrock, and streambeds often consist of coarse gravel or bedrock. The watershed is fish-bearing, i.e. kokanee salmon are native (Oncorhynchus nerka), with rainbow trout (O. mykiss) and cutthroat trout (O. clarkii) are stocked by the Province
(Glova 1987; Best 2001; Rieberger et al. 2007). The biogeoclimatic zone is Coastal Western Hemlock (Jungen 1985; Nault 1988; Aqua-Tex Scientific 2014).

From a land-cover perspective, the Shawnigan Watershed is a site of active development due to its proximity to the Greater Victoria Region. It contains three communities, Shawnigan Lake, Cobble Hill and Mill Bay. Historically, the immediate area around the lake was lightly populated with seasonal-use cottages, and as such, many residents live around the lake and in the area outside the three communities (Worley Parsons 2009). However, after the 1980s, the population have increased, as well as the number of full-time residents replacing the cottagers, increasing residential and municipal infrastructure (Worley Parsons 2009; Statistics Canada 2017). Though residential use is increasing, forestry is by far the largest land use type, with private forest land, crown land, and First Nation land recently logged; the recent is either second or third growth timber harvesting (Worley Parsons 2009). Agriculture is the third most common land use, with nearly 10% of land use belonging in the Agricultural Land Reserve (Rieberger 2007). The remaining land uses are aggregates processing sites, a saw mill, and a small quarry, which make up only 1.5% of the study area.

For our study, we selected two creeks within the Shawnigan Lake Watershed of similar hydrological and geological properties, but differing land uses: McGee and Van Horne Creeks. On each creek, we placed three turbidity sensors bracketing reaches with distinct hydrological and land use features (Fig. 3.1). We also performed spot sampling at these six sites, and an additional site at the mouth of McGee Creek and the mouth of South Shawnigan Creek (which Van Horne flows into). In total, we had 8 turbidity monitoring sites.

### 3.2.2 Land Use Classification

We outlined watersheds using a 30 m digital elevation model provided by the Government of Canada (Natural Resources Canada 2013). Watersheds were delineated using Hydrology Tools in the Spatial Analysis package of ArcMap 10 and coordinates from sampling sites (ESRI 2011). Spatial data were taken from the RapidEye Satellite, where we extracted the red and green bands for our region (Tyc et al. 2005; Planet Labs. 2018). We then calculated a Normalized Differential Vegetation Index (NDVI) for the entire Shawnigan Watershed, a common method for remote sensing (Burrough 1986; Carlson and Arthur 2000; Andrew et al. 2015; Coops and Tooke 2017). The watersheds, satellite imagery and NDVI were then compared to base satellite imagery, additional maps of the site, and previous knowledge of the site to ensure accuracy.



**Figure 3.1:** Map of the Shawnigan Watershed with the sample points, creeks and watersheds identified the calculated NDVI spectrum with classifications from RapidEye Satellite Sensor (Planet Labs Inc. 2018).

Land use classification based on the NDVI was done using the trained classification in the Spatial Analysis package (ESRI 2011), using methods as modified by Kuglerová et al. (*in prep*). Trained pixels were developed using knowledge from previous visits to the different areas of the Shawnigan Watershed. We loosely followed the classification of Duan et al. (2015), but classified water, forest, agriculture/vegetation, built, industrial, and clearcut. The percentage of each was then calculated for the combined watershed of each sampling point, as in Kuglerová et al. (*in prep*). Overall, all sites had a high percentage of forestry, though McGee Creek also showed some agricultural land use, and Van Horne had a larger amount of other land uses (approximately 5% compared to <1%).

### 3.2.3 Field Data Collection

Turbidity was measured in Nephelometric Turbidity Units (NTUs) with Campbell OBS3+ Turbidity Sensors, with a 0 to 4000 NTU range (Campbell Scientific Edmonton, AB). Temperature measures were obtained using Campbell T109 Temperature Sensors installed at each site as shown in Fig. 3.2 (Campbell Scientific Edmonton, AB). Sensors were attached to a Campbell CR1000 Data Logger (Campbell Scientific Edmonton, AB). Measurements were taken every 15 seconds, and averaged to achieve 15-minute resolution. Sites were visited approximately once every three weeks from July 2016 to April 2017 for maintenance, including sensor cleaning, battery changing, downloading data and replacing desiccants.

At each visit, a spot turbidity sample was taken using a Hach 200q handheld sensor (Hach Instruments Co, Loveland, CO) (NTU range of 0 to 1000 NTU), and pH, specific conductivity, total dissolved solids, and temperature were measured using a YSI Water Quality probe (YSI Inc./Xylem Inc. Yellow Springs, OH). Samples of total organic carbon (TOC) were taken during two field days, and were sent to Natural Resources Canada in Sault Ste. Marie, Ontario for analysis. Additional samples of turbidity, total suspended solids and total dissolved solids were also taken, and sent to M.B. Laboratories in Sidney, B.C. for analysis. Total suspended solids (TSS) refer to suspended particles that are captured by a filter of 0.7  $\mu$ m, such as sediments or clay particles (Church 2002). In comparison, total dissolved solids (TDS) refer to particles that pass through the filter and may include dissolved ions (Knighton 1998). Turbidity samples from the Hach 200q

58

sensor were taken according to USGS guidelines, with the addition of a zero NTU standard taken with Isopropyl alcohol (USGS 2005). These samples were also manually calibrated due to some programming errors in the sensor leading to a failure in the calibration program and the high NTU level of the standards. The manual calibration fitted a linear regression to the values from standard formazine solution (provided by Hach) to calculate the slope, and the zero standard was used to calculate the intercept. This equation was then applied to all field measurements. In cases where the calibration program was effective, the same process was applied to the 1.0 NTU standard and the < 0.1 NTU standard that were outside the normal calibration range.

Discharge data were not collected in either of the creeks we worked in, though it was historically collected at the mouth of the lake. Instead, we used discharge from a monitoring station approximately 16 km away at Bings Creek Hydrological Station (08HA016, Lat: 48.79, Long: - 123.73.), north of the watershed (Environment Canada 2017). Of the available data, Bings Creek best matched our streams in terms of size and weather patterns.

## 3.2.4 Data Analysis

Turbidity data from the OBS3+ sensors were initially going to be cleaned to eliminate possible artifacts from insects, biofouling, branches or bubbles, which can cause false and large turbidity values (Gippel 1991, Lewis and Eads 2001). This was to be done through a smoothing process where data gaps of more than 20% of the average of the 10 observations before and after, or a 50 NTU jump between observations. However, it could not be confirmed if using this method would remove actual measurements, and so data were cleaned manually, using field notes and examining turbidity peaks as compared to discharge.

All data analyses were performed using R-Studio (Kutner 2005; R-Studio 2015). Data analysis consisted of three steps which we performed 3 times in total, first data selection, then a Principle Component Analysis, from which we could determine key variables, and finally a Linear Mixed Effects model, as well as an additional linear mixed effects model. Most of the variables were already included in the dataset, with the exception of antecedent moisture conditions. To simulate antecedent moisture, we calculated the average precipitation over the previous 2 and 5 days and attached those as other variables in the model. Once all variables were accounted for, there were several steps before model selection. First, to identify potentially important variables, turbidity was graphed against all variables, and individual  $r^2$  terms were calculated. Turbidity appeared to have some relationship to all of our variables, and so we then used a principal components analysis to visualize the variation space and further prune potential contributors to turbidity (Seeger et al. 2004). Once variables were chosen, we then went ahead with model selection. The models were applied using a Generalized Least Squares (Gls) fit (with stream and season as factors, and correlated through time per stream) similar to Gomi et al. (2006), and Linear Mixed Effects Regression (Lmer) models (with month as a factor, using hierarchical correlation by stream). We then compared models using AIC and ANOVA. Due to the high correlation between variables, we used an AR1 (autoregressive) error covariance structure (Pfaff 2008). For the same reason, we also applied a principle component regression (Zuur et al. 2009).

The first analysis was on continuous turbidity and external variables, including discharge, rainfall, and temperature. We also estimated antecedent moisture by averaging rainfall over 2 and 5 days (AM<sub>2</sub> and AM<sub>5</sub>, respectively). Our models also included site, stream and seasonal variables. A land-use variable was also included, determined by the PCA. Our second analysis (only the linear mixed effects model) was done on the high discharge data alone, to determine if base

turbidity and high flow turbidity had different drivers. Our third analysis was of the spot sample data, which included total dissolved solids (TDS), total suspended solids (TSS), and water temperature, as well as land use. Finally, we repeated the same analysis from the spot samples, but for low turbidity (< 3 NTU), to determine drivers of low turbidity. For this analysis, we included a comparison with the total organic carbon, as well as TDS and TSS.

### 3.3 Results

#### 3.3.1 Turbidity

We collected turbidity data from July of 2016 to April of 2017. Overall, for our continuous turbidity, we removed 1600 turbidity measurements out of approximately 46,000 (3.5%), due to sensor obstruction or biofouling, or in one case due to the stream drying, leaving the probe out of water. This was slightly different between sites, with Van Horne Upstream having the lowest number of removals at 66 measurements, and McGee Upstream with the highest at 540. We also had a number of high turbidity points (between 100 NTU and 400 NTU) not recorded by the sensor due to a programming error. Such high jumps (from 50 NTU to >200 NTU) are typically indicative of sensor obstruction, however, some values above 200 NTU were confirmed by spot sampling.

As can be seen in Table 3.1, the maximum turbidity value was 371 NTU from Middle Van Horne, and the minimum turbidity was 0.56 from Upstream McGee, though the two creeks showed similar median turbidity values. Turbidity response was very different at each of the different sites, and between the two creeks, as can be seen in the coefficients of variation, in Fig. 3.2, Fig. 3.3, and Fig. 3.4. Turbidity peaks occurred in Van Horne Creek during high discharge or after rainfall events. McGee Creek had some turbidity peaks during periods of high discharge and rainfall, but not during every event. In comparison, Van Horne Creek regularly went to 100 NTU during many discharge and rain events. Turbidity also displayed seasonal patterns (Fig. 3.2 and 3.3). Additionally, several turbidity events did not transcend between sites, and were diluted or absorbed before the next site. The highest turbidity was found in Van Horne Creek, during the spring months. A landslide occurred during this period that may have increased the turbidity in this area, as after this time turbidity did not return to the previous base load.

**Table 3.1:** Turbidity (NTUs) in the Shawnigan Watershed for the six continuously monitored sites. Table shows that while Van Horne tends to have higher maximum turbidities, all sites show a large percentage of samples above 1 NTU.

Site	Min	Max	%	%	%	Median	Coefficient
	Turbidity	Recorded	Samples	Samples	Samples	Turbidity	of
		Turbidity	above	above 5	above 1		Variation
			20 NTU	NTU	NTU		
Van Horne Upstream	0.67	106.28	1	6	63	1.13	205.91
Van Horne Middle	0.79	371.68	9	26	73	2.87	193.83
Van Horne Downstream	0.64	119.55	12	33	70	1.90	183.40
McGee Upstream	0.56	40.2	0.1	1	33	0.86	79.24
McGee Middle	0.70	67.95	0.1	3	70	0.89	119.35
McGee Downstream	0.57	12.4	0	2	19	1.46	95.89



**Figure 3.2:** Rainfall (mm/h) and discharge (m<sup>3</sup>/s) for the full Shawnigan Watershed, and turbidity (NTUs) for the McGee Creeks, with Upstream, Middle and Downstream sites. McGee has maximum values of below 100 NTU, and very few measurements above 5 NTU. Note the relationship with discharge and scale differences.



**Figure 3.3:** Rainfall (mm/h) and discharge (m<sup>3</sup>/s) for the full Shawnigan Watershed, and turbidity (NTUs) for the Van Horne Creeks, with Upstream, Middle and Downstream sites. Note the scale differences, where Van Horne Creek has maximum values of around 350 NTU, but McGee has maximum values of below 100 NTU, and very few measurements above 5 NTU. Note the relationship with discharge and the scale differences.

For our spot analysis, turbidity behaved similarly to the continuous turbidity. Most of our samples for both creeks were below 5 NTU, except for one or two high flow events which were captured in the spot data (Fig. 3.5. However, we did capture samples of above 400 NTU in Van Horne Creek. We also were able to use spot sampling to compare between turbidity measurement methods (continuous, handheld, laboratory). Differences range from 0 to 23 NTU between measurement methods, shown in Fig. 3.6, though generally differences were ~1 NTU. However, the standard error range for each technique is reported as approximately 0.5 NTU.



0

Mar 01

Mar 15

Date

Apr 01

(Sept), B) fall discharge related events, and C) spring turbidity after the landslide. These plots show a number of instances where turbidity increases by 1 NTU for more than 48 hours, and increases above 20 NTU on many occasions. Note different scales on the y-axes.



**Figure 3.5:** Turbidity differences between spot turbidity samples and continuous turbidity. Continuous samples are from all sites at each creek. A) is McGee Creek, and B) is Van Horne Creek. Both plots show a large difference in the measured turbidity between the two methods.



**Figure 3.6:** Turbidity differences between different measurement methods. TurbDiff1 refers to the difference between the handheld sensor (Hach) and the continuous sensor (OBS3+). TurbDiff 2 refers to the difference between the continuous sensor and the lab analysis. TurbDiff 3 refers to the difference between the handheld sensor and the lab analysis.

# 3.3.2 Hydrology and Weather

Discharge ranged from unmeasurable (dry creeks) to 6.66 m<sup>3</sup>/s, while rainfall ranged from 0 to 7.8 mm/hr, both peaks in October and March, respectively (Fig. 3.2 and 3.3). The study period experienced slightly abnormal weather for this region. That is, we experienced heavy and consistent rainfall in October, November and March. However, much of December, January and February was cold enough for snow and snow accumulation on the ground. This reduced discharge

and rainfall from normal levels for these months and resulted in rain-on-snow events in December, late February and March. We also graphed turbidity against discharge and rainfall, found in Fig. 3.7. Both revealed positive relationships with turbidity, though  $R^2$  values were generally quite low, with the highest being Van Horne Creek at 0.22.



Downstream Middle Upstream

Figure 3.7: Turbidity compared to rain and discharge for Van Horne and McGee creeks. Turbidity is shown on a log scale. A and C show Van Horne Creek, while B and D show McGee Creek.

### 3.3.3 Land use

We used a total of six land use categories: water, forest, industry, agriculture/vegetation, built and clearcut. These were used to develop a trained classification, then to identify the percentages of each land use for each of our watersheds. In some cases, industrial areas were classified as water as they may have had small ponds or pooling on the properties; however, water made up a small percentage of the watershed (~1%), so this was accepted. Van Horne Creek sites had much higher percentages of non-forest cover, while McGee Creek sites showed higher forested areas (Fig 3.7). All sites had a minimum of 47% forested cover, however, the maximum was 76.21% at Downstream Van Horne. In our statistical models, however, only one land use variable could be used, so we calculated the % non-vegetated from the industry, built, and clearcut land uses. Van Horne Creek had non-vegetated of above 5%, with the downstream reaches at around 8%, but McGee Creek remained at only around 1% non-vegetated.

#### 3.3.4 Continuous Turbidity PCA and Models

We performed two principle component analyses. As seen in Fig. 3.8 our first PC appeared to be largely driven by weather, including Rain and Antecedent Moisture, while our second had hydrological and seasonal variables, with Discharge and Temperature. Land Use (% Non-Vegetated) appeared very slightly on PC1 but was actually the driver of a third principle component. Overall, it appeared that the two streams overlapped in variation space, though they

shared similar weather and hydrology. Season did, however, appear to have some grouping effect. It is unclear if the streams would overlap if the third principle component was mapped.

For our model, we used a *Gls* and an *Lmer* model to identify the largest drivers of turbidity. We chose the percentage of non-vegetated landscape as a variable that encompassed most land use types that represent land use modification beyond forestry or agriculture, and we could not confidently differentiate between forest, other vegetation and agriculture using the simple methods we used. We therefore chose percentage non-vegetated over the percentage of non-forest to keep land use estimates as accurate as possible. We compared these models with AIC, log likelihood and ANOVA (Table 3.2). The significant variables differed between the models (Table 3.3), of which the LME model was found to be a significantly better fit (p<0.001). Both models found rainfall, temperature, and AM5 significant. The Gls model found that discharge was not significant, but AM<sub>2</sub> and the percent non-vegetated were significant. Also significant were the upstream sites, the stream term (differentiating between the two streams), and the spring and fall seasons. The *Lmer* model found that discharge was significant, but antecedent moisture of 2 days and the percent non-vegetated non-significant. In both models, turbidity had a negative relationship with antecedent moisture of 2 days, and in the Gls model, turbidity was negatively associated with the percentage of non-vegetated land cover, site and season. All other variables were positive.

Land use was only significant in one of our models, and this was counter to our hypothesis that turbidity would be positively associated with both antecedent moisture and land use. However, this may be an artefact of our analysis for two reasons. First, land use consisted of one value per site, compared to turbidity with thousands of values and high variation. This would mean our model might underestimate the amount of variation due to land use, as land use is not changing. Secondly, turbidity in Van Horne Creek was not sensed beyond approximately 120 NTU, meaning the majority of the large peaks (turbidity in this stream could reach 400 NTU) were not captured by this analysis. This possibly could have masked the differences between the creeks and the impact of the land use. However, in our analysis of all data, site was significant, and in our analysis of high-flow data only, we found that land use was significant in both models. Alternatively, land use may also have been found significant in our second model due to a type II error.



**Figure 3.8:** Principal Components Analysis of the Shawnigan Watershed. The two principal components explained approximately 60.8% of the variation. PC1 appeared dominated by weather variables, such as rain and antecedent moisture (AM) at 2 and 5 days, while PC2 appears to demonstrate seasonal flow, with a combination of water temperature and discharge. Our land use variable, % Non-vegetated also appeared on PC2, though dominated the third PC. Graph A) shows points separated by stream and B) shows points separated by season. In both, there are overlapping points, but a small grouping effect is present.

**Table 3.2:** Linear mixed-effects model results for drivers of turbidity in streams in the Shawnigan Watershed, with coefficients as seen in Table 3.3. \*Indicates significance.

Model	AIC	LogLik	<b>Correlation Phi</b>	ANOVA to Null	ANOVA
					between Models
Null Gls	-22089.90	11047.95	0.52		
Gls	-23266.04	11648.02	0.97	P < 0.01*	
Null Lmer	-22358.23	11185.12	0.96		
Lmer	-23452.76	11738.38	0.96	P < 0.01*	P < 0.01*

**Table 3.3:** Linear mixed-effects model coefficients for drivers of turbidity in streams in the Shawnigan Watershed,

 with model comparisons in Table 3.2. \*Indicates significance.

Gls Model		Lmer Model			
Coefficient	Value	<b>P-value</b>	Coefficient	Value	<b>P-value</b>
Intercept	0.12	0.79	Intercept	-0.74	0.01*
(log) Discharge	0.03	< 0.01*	(log) Discharge	0.12	0.01*
(log) Rain	0.02	< 0.01*	(log) Rain	0.02	< 0.1*
(log) Antecedent Moisture 2	-0.01	< 0.01*	(log) Antecedent Moisture 2	-0.01	0.68
Days			Days		
(log) Antecedent Moisture 5	0.38	< 0.01*	(log) Antecedent Moisture 5	0.34	< 0.01*
Days			Days		
(log) Temperature	0.26	< 0.01*	(log) Temperature	0.27	< 0.01*
(log) Percent non-vegetated	-1.17	0.03*	(log) Percent non-vegetated	0.43	0.15
Site Middle	0.09	0.27	Site Middle	NA	NA
Site Upstream	-1.17	< 0.01*	Site Upstream	NA	NA
Stream Van Horne	2.38	0.03*	Stream Van Horne	NA	NA
Season Spring	1.26	0.00*	Season Spring	NA	NA
Season Summer	-0.11	0.39	Season Summer	NA	NA
Season Winter	0.67	< 0.01*	Season Winter	NA	NA

For our definition of high discharge, we selected a threshold of 1 m<sup>3</sup>/s, to capture only higher-flow periods, and some sustained high-flow periods, based on our estimated discharge levels found shown in Fig. 3.2. We compared the same variables and parameters (Tables 3.4 and 3.5). The significant variables were quite similar, and there was no significant difference between the two model types. All variables were found to be significant. In the GLS model site and stream were significant, but season was not.

Model	AIC	LogLik	Correlation Phi	ANOVA to Null	ANOVA between Models
Null GLS	-15.73	10.86	0.98		
GLS	-1245.71	636.83	0.94	P < 0.01*	
Null LME	-102.21	57.12	0.94		
LME	-1248.47	636.24	0.93	P < 0.01*	P = 0.54

**Table 3.4:** Linear mixed-effects model results for drivers of turbidity in cases of high discharge events, in streams of the Shawnigan Watershed, with coefficients as seen in Table 3.5. \*Indicates significance.

**Table 3.5:** Linear mixed-effects model coefficients for drivers of turbidity in cases of high discharge events, in streams of the Shawnigan Watershed, with model comparison as seen in Table 3.4. \*Indicates significance.

GLS Model			LME Model			
Coefficient	Value	<b>P-value</b>	Coefficient	Value	<b>P-value</b>	
Intercept	3.16	< 0.01*	Intercept	-0.61	0.07*	
(log) Discharge	0.54	< 0.01*	(log) Discharge	0.54	< 0.01*	
(log) Rain	0.09	< 0.01*	(log) Rain	-0.09	< 0.01*	
(log) Antecedent Moisture 2	-0.06	< 0.01*	(log) Antecedent Moisture 2	-0.06	< 0.01*	
Days			Days			
(log) Antecedent Moisture 5	1.01	0.015*	(log) Antecedent Moisture 5	1.01	0.02*	
Days			Days			
(log) Temperature	-0.11	< 0.01*	(log) Temperature	-0.11	< 0.01*	
(log) Percent non-vegetated	-3.98	< 0.01*	(log) Percent non-vegetated	0.85	0.05*	
Site Middle	-0.03	0.27	Site Middle	NA	NA	
Site Upstream	-3.27	< 0.01*	Site Upstream	NA	NA	
Stream Van Horne	7.17	< 0.01*	Stream Van Horne	NA	NA	
Season Spring	0.32	0.04	Season Spring	NA	NA	
Season Winter	-0.04	0.81	Season Winter	NA	NA	

# 3.3.5 Spot Turbidity PCA and Models

For in-stream particles, using spot samples, we measured TDS, TSS and TOC (Fig. 3.9). When comparing graphically, we found some relationship between turbidity and TSS for Van Horne Creek (R<sup>2</sup> of 0.8) but not TDS, and neither TSS or TDS had a high relationship to turbidity in McGee Creek. To determine the most influential variables, we performed another PCA with the land use data included, shown in Fig. 3.9. This led to similar results as above, with the land use

appearing in the first PC, and the hydrological variables on the second PC. TSS and water temperature were both strong on PC2, while TDS appeared to be more associated with PC1, specifically percent forest. Our model results again were counter to our hypotheses, as seen in Tables 3.4 and 3.5. Neither model indicated that TDS, percent non-vegetated or stream were significant. This suggests a link between turbidity and TSS, as well as reiterates the importance of site in explaining turbidity over stream (Fig. 3.10). We could not compare TOC in this part, as we did not have enough samples to include in the final model.



**Figure 3.9:** Spot turbidity and other variables taken from the Shawnigan Watershed, with McGee and Van Horne creeks. TSS refers to total suspended solids, and TDS refers to total dissolved solids. Turbidity and TSS are shown on a log scale. A, B, and C are turbidity, TSS and TDS over time, respectively, with different y-axis scales. D shows TSS against turbidity, with an  $R^2$  of 0.06 for McGee Creek and 0.80 for Van Horne Creek. E shows TDS against turbidity, with an  $R^2$  of 0.05 for McGee Creek and <0.01 for Van Horne Creek.

Model	AIC	LogLik	Correlation Phi	ANOVA to Null	ANOVA between Models
Null Gls	347.41	-170.86	-0.063		
Gls	324.02	-154.01	-0.11	P < 0.01*	
Null Lmer	351.63	-170.86	-0.063		
Lmer	324.63	-154.32	0.96	P < 0.01*	NA

**Table 3.6:** Linear mixed-effects model results for drivers of turbidity using spot sample data, in streams of the Shawnigan Watershed, with coefficients as seen in Table 3.7. \*Indicates significance.

**Table 3.7:** Linear mixed-effects model coefficients for drivers of turbidity using spot sample data, in streams of the Shawnigan Watershed, with model comparison as seen in Table 3.6. \*Indicates significance.

Gls	Lmei	Lmer Model			
Coefficient	Value	<b>P-value</b>	Coefficient	Value	<b>P-value</b>
Intercept	-319.88	0.48	Intercept	-288.91	0.21
(log) TDS	59.69	0.61	(log) TDS	47.60	0.68
(log) TSS	62.94	< 0.01*	(log) TSS	62.68	< 0.01*
(log) Temperature	11.70	0.58	(log) Temperature	12.12	0.57
(log) Percent non-vegetated	10.41	0.81	(log) Percent non-	36.41	0.15
			vegetated		
Stream Van Horne	44.19	0.52	Stream Van Horne	NA	NA



**Figure 3.10:** Principal Components Analysis of spot samples from the Shawnigan Watershed. The two principal components explained approximately 69.8% of the variation. PC1 appeared dominated by the land use variable, %Non-Vegetated (summary of industry, built, clearcut, and industry), but also included TDS (total dissolved solids). PC2 was dominated by hydrological variables, such as temperature and TSS (total suspended solids). Graph A) shows points separated by stream and B) represents points separated by season. In both, there are overlapping points, but a defined grouping effect is present.



**Figure 3.11:** Spot turbidity below 5 NTU and other variables taken from the Shawnigan Watershed, with McGee and Van Horne creeks. TSS refers to total suspended solids, TDS refers to total dissolved solids, and TOC refers to total organic carbon. A shows turbidity against TSS, with an  $R^2$  of 0.06 for McGee Creek and <0.01 for Van Horne Creek. B shows turbidity against TDS turbidity, with an  $R^2$  of 0.05 for McGee Creek and <0.01 for Van Horne Creek. C shows turbidity against TOC, with an  $R^2$  of 0.07 for McGee Creek and 0.10 for Van Horne Creek

We also compared the same variables under low turbidity (<5 NTU) to determine if the same relationships occurred during low turbidity, and if there was a relationship to TOC (Fig. 3.11). In general, there was no consistent relationship between low turbidities and any of the three measurements. Turbidity showed low  $R^2$  values with all three variables, and our LME model (which again did not contain TOC) showed no variables to be significant at low turbidity, though the LME model was not significantly different from the null model. However, our GLS model did

find that TDS and TSS were significant at this level, and was significantly different from the null model. This suggests that there is perhaps a more complex relationship between particles at different turbidity levels.

sucams of the 2	treams of the Shawingan watershed, with coefficients as seen in Fable 5.9. Indicates significance.								
Model	AIC	LogLik	<b>Correlation Phi</b>	ANOVA to Null	ANOVA between				
					Models				
Null Gls	73.07	-28.53	0.67						
Gls	77.39	-35.70	0.77	P = 0.01					
Null Lmer	80.85	-35.42	0.28						
Lmer	80.60	-32.30	0.48	P = 0.10	NA				

**Table 3.8:** Linear mixed-effects model results for drivers of low turbidity (<5 NTU) using spot sample data, in streams of the Shawnigan Watershed, with coefficients as seen in Table 3.9. \*Indicates significance.

**Table 3.9:** Linear mixed-effects model coefficients for drivers of low turbidity (<5 NTU) using spot sample data, in streams of the Shawnigan Watershed, with model comparison as seen in Table 3.8. \*Indicates significance.

Gls	Lmei	Lmer Model			
Coefficient	Value	<b>P-value</b>	Coefficient	Value	P-value
Intercept	15.14	0.01*	Intercept	4.15	0.52
(log) TDS	-3.92	0.01*	(log) TDS	-1.10	0.50
(log) TSS	0.35	0.05*	(log) TSS	0.40	0.06*
(log) Temperature	-0.15	0.55	(log) Temperature	0.042	0.89
(log) Percent non-vegetated	1.00	0.49	(log) Percent non-	0.88	0.08*
			vegetated		
Stream Van Horne	-1.27	0.58	Stream Van Horne	NA	NA

### 3.4 Discussion

The purpose of this study was to better understand drivers of turbidity fluctuation over time and space, to inform water quality policy. Our results suggest that turbidity has relationships with hydrological and weather variables, and potentially with land-use activities. We also showed that turbidity may respond in complex ways to particle types in streams. These results have implications for current management strategies in British Columbia, which will be discussed in this section.

### 3.4.1 Turbidity over Time

For drivers of turbidity over time we considered discharge, rainfall, antecedent moisture conditions and water temperature. When utilized in our statistical model, we found they turbidity had positive associations with most variables, except for AM<sub>2</sub>. Additionally, our model with better fit found discharge was significant, but AM<sub>2</sub> was not. In either case, turbidity appears to be influenced by rainfall (including antecedent moisture) and discharge, both graphically and according to our models.

Many of the higher turbidity peaks appear to be driven by discharge, particularly in Van Horne Creek and this positive relationship has been demonstrated before (Lewis 1996; Lessels and Bishop 2013). Additionally, discharge-driven sediment and organics release have also been noted (e.g. Smith et al. 2003; Raymond and Saiers 2010). That discharge was less important as a driver of turbidity in McGee creek may suggest that it simply had low sediment availability, and discharge events quickly became diluted, as noted in Chapter 2. The maximum TSS level was much higher in Van Horne Creek, which also had a much higher R2 between TSS and turbidity than McGee Creek. Alternatively, Smith et al. (2003) and Miller et al. (2015) identified that very high sediment/turbidity events can occur during overbank flow. This, however, could also suggest that in-stream sediment availability in McGee Creek may also be low.

Despite the strong relationship with discharge, however, our models suggested turbidity had a stronger relationship to rainfall. This could potentially be that the Shawnigan Lake Watershed is more responsive to rainfall. However, when compared graphically, it appears that turbidity was more influenced by discharge than rainfall, and in Van Horne Creek the  $R^2$  was higher between discharge and turbidity than rainfall and turbidity. Alternatively, our discharge was estimated from a nearby creek, while rainfall was a local measurement. This means discharge measures could be slightly different from our creeks, and therefore its importance could be potentially underestimated.

Rainfall could also have been a highly significant predictor of turbidity due to the effect of antecedent moisture. In the full model, the relationship between turbidity and AM<sub>5</sub> was significant and positive, while turbidity only had a negative and non-significant relationship to AM<sub>2</sub>, and our high-flow model found the opposite. This could be similar to the mechanism described by Bača (2008), who found that more days of continuous rain would increase erosion, which was certainly the case of the March landslide. However, several others have also identified that low antecedent moisture can lead to higher sediments or turbidities as it provides time for material to accumulate in and around the stream (Zabaleta et al. 2007; Mukundan et al. 2013). In our previous study (Chapter 2), we found that AM<sub>2</sub> was significant, while AM<sub>5</sub> was not. This difference may be a factor of the watershed size for the individual creeks, where the creeks studied in Chapter 2 were much smaller than the creeks analyzed in this study (first order compared to third or fourth order). The larger watershed size in Shawnigan may have resulted in a delay from a rainfall event to when water reached the creek, resulting in the number of days for antecedent moisture to have an impact. Meanwhile, during heavy rainfalls, precipitation could generate overland flow, and enter the creek much quicker, resulting in the change from  $AM_5$  to  $AM_2$ . This was found by Penna et al. (2010), where overland flow was much greater in periods of high antecedent moisture.

Our study also indicated that there was some seasonal component to turbidity. Though not the model of best fit, our GLS model indicated that the fall, spring and winter seasons were significant explanatory variables for turbidity, as was temperature. Turbidity dynamics show very few peaks occurring in the summer period, while many turbidity instances occur during the fall, winter and spring, particularly in Van Horne Creek. Temperature, however, showed an opposite effect, where temperature had a positive relationship to turbidity, indicating turbidity should be higher with higher temperatures, such as the summer. This may be due to the colder winter experienced during the study period, where snow fell for a large portion (there was snow on the ground from about Dec. 1 to Feb. 1, though it rained intermittently through that period), and so warmer weather associated with rain and storms may have driven this result.

As with our previous study, a few turbidity peaks appeared with moderate rainfall and discharge. Van Horne Middle and Upstream sites showed the opposite, with turbidity peaks accompanying nearly each rise in discharge. Miller et al. (2015) identified a similar effect, where the largest turbidity events occurred during periods of flood, but many turbidity events occurred with unremarkable levels of discharge. In the fall, we attributed these events to "the first flush" (the first rainfall after a period of no rain) at a few sites, which has previously been found to produce high levels of turbidity (Grayson 1996; Jordan 2006). These turbidity responses to rain events could also be similar to what was discussed in Chapter 2 and by Mukundan et al. (2013), where rain events that were further spaced apart in time produced higher turbidity. The January event in Upstream and Downstream McGee appeared to have occurred after a rain event with snow on the ground, creating a rain-on-snow event, which was also found to cause turbidity by Jordan (2006). The September event in McGee Middle may have also been a first flush, as rain fell on a dry creek, likely creating a highly turbid mixture, or muddy puddles. However, the extent of

turbidity peaks were not recorded due to sensor error (as explained above, the sensor did not record between 200 NTU and 400 NTU). This resulted in all high turbidity peaks for these sites recorded at approximately 200 NTU or 400 NTU, and no resolution within this range to determine turbidity drivers, meaning it is impossible to determine if turbidity was scaling with discharge, or reach a threshold. Finally, the April high turbidity events of over 400 NTU in Middle and Downstream Van Horne were likely due to a landslide that occurred just upstream of Middle Van Horne made primarily of clay. At this point, the toe of the slope had entered the stream, and was likely delivering large amounts of clay to the stream with any amount of rainfall. While it is possible that these events are artifacts, due to sensor obstruction, they usually occurred in more than one area of the stream or were confirmed with spot sampling.

### 3.4.2 Turbidity over Space

According to the results of this study, turbidity response clearly varied over the landscape. This was shown in the difference of turbidity responses between the two creeks, and between sites within each creek, to the same rainfall patterns. This indicates that turbidity is highly susceptible to local changes in the watershed, on the reach scale, such as small differences in bed type or slope. Miller et al. (2013) monitored continuous turbidity over three small, forested streams, and found turbidity responses to be significantly different despite being nearby to each other. Mukundan et al. (2013) also found that turbidity was highly influenced by individual streams, and predictive models were significantly better with the inclusion of tributary, though they attributed these differences to environmental factors (such as season, soil characteristics and rainfall).

The significant difference between the two streams, and the sites within streams, could also be explained by land use. Many authors have also found a relationship between declining stream condition and declining forest and vegetation cover (Sutherland et al. 2002; Cunha et al. 2016). Additionally, many authors have suggested associations between turbidity and the following land use types: clearcut/forestry (Reiter et al. 2009; Klein et al. 2012), built/urban (Roy et al. 2003; Hasenmueller et al. 2016), and agriculture (Townsend et al. 1997; also see review by Allan et al. 2004). Van Horne Creek showed many more turbidity peaks (a peak for almost each rain/discharge event) than McGee Creek, and the two downstream monitoring sites reached much higher turbidities, which may suggest that there is more available sediment in the Van Horne catchment. Streams in this area we would expect to be sediment-starved, due to the sub-mountainous terrain and shallow top soil, and so would require long periods between rain to build up enough sediment around the stream or behind large wood to cause a turbidity event. Large turbidity responses in small creeks, however, are indicative of modified land uses. For example, Nelson and Booth (2002) identified urbanizing land use as a major driver of sediment entering their water through increased runoff over roads, and Henderson and Golding (1983) found that hydrophobic soils in clearcut areas produced more overland flow. Dalzell et al. (2004) identified that agricultural areas produced sediment, increasing total maximum daily sediment loads. Alternatively, Hughes et al. (2012) found that eroding hillslopes could be a major source of sediment, even without land-use change.

Furthermore, a landslide at an industrial lot occurred in March of 2017 between the upstream and middle sites of Van Horne Creek. This is clear in our results, where base turbidity remained above 10 NTU for the remainder of the study period, and turbidity became increasingly flashy. Landslides are frequently attributed to land use, where Sullivan (1985) found a relationship

between forestry and bank failures, and Gurnell and Warburton (1990) identified landslides would cause sediment pulses thereafter. After the land slide, Van Horne closely matches Sutherland et al. (2002), who found turbidity of a disturbed stream was consistently above 10 NTU, while the reference streams rarely ventured that high. Interestingly, this also suggests that there may be a disturbance gradient, where high flow turbidity may first increase, and as disturbance increases, base turbidity increases.

In this study, we did not distinguish between the various types of land use in our analysis, aside from broadly grouping uses into vegetated (forest, ag/veg) and non-vegetated (industry, clearcut and built). We can potentially narrow down by analyzing turbidity dynamics between the streams and sites in our watershed. For example, Van Horne Creek showed much higher turbidities than McGee Creek, despite both having similar agricultural land use percentages. Additionally, our upstream site in Van Horne was just downstream of a small amount of industrial land use. The confluence just above this site was fed by two creeks, one coming off a forested mountain, and one coming from the industrial area. These creeks exhibited dramatically different turbidities during storm periods, similar to the difference between McGee and Van Horne creeks overall. Additionally, this upstream site showed a much more muted turbidity response compared to middle and downstream Van Horne, which had higher percentages of industrial and forestry land uses. This similarity itself is different from all other sites, which showed more turbidity pulses, indicating there is a consistent source of sediment, aside from agriculture, that overshadows stochastic and seasonal turbidity.

The major land use types on the downstream reaches of Van Horne Creek are urban, forestry and industrial, all of which have been known to increase turbidity. DeGasperi et al. (2009) found that aquatic environments near urban landscapes had high pulses of both discharge and sediments. Additionally, Webb and Haywood (2005) found that even more modern forestry techniques still caused excessive turbidities. The industrial land use type in Van Horne Creek consisted of new construction and aggregates processing plants, something not as often found in the literature, though highly impactful on aquatic systems. Harbour (1999) identified that erosion can increase by up to 40,000 times from construction areas. McCaleb and McLaughlin (2008) showed that construction areas in particular can be large sources of sediment, even with a variety of sediment control mechanisms. Additionally, Duan et al. (2015) identified that in a watershed with a higher percentage of agricultural land cover than "developing lands", the "developing lands" contributed more sediment. This suggests that even though industry/construction makes up a small percent of watershed land use, it has a disproportionate, large impact, something that may be missed at larger spatial scales.

#### 3.4.3 Internal Turbidity Drivers

In this study, we sought to understand drivers of turbidity both over time and space. To support this, a number of spot measurements were taken of stream particles to determine the particles contributing to the turbidity measurement. We analyzed turbidity with TSS and TDS for most of our spot samples. Supporting our hypothesis, TSS was the only significant variable in both models. This positive relationship between turbidity and TSS has been found by many authors (Dodds and Whiles 2004; Downing 2006). However, counter to our hypothesis, the relationship between turbidity and TSS did not appear to be significantly different between streams or land uses. This is also contrary to findings in Finlayson (1985), Grayson et al. (1996), and Dodds and Whiles (2004), who suggested that both geology and land use can change the type of suspended

particle, altering the relationship between turbidity and TSS to be specific to each stream (Finlayson 1985; Grayson et al. 1996; Dodds and Whiles 2004). This could possibly be interpreted that both streams had a similar relationship between turbidity and TSS, although Van Horne Creek had greater sediment supply.

We also compared turbidity at lower values (< 5 NTU) to TDS, TSS and TOC. This was due to evidence suggesting that at lower turbidities, smaller particles, and the nature of the individual particles (particle size, material) may impact the turbidity measurement. That is, several authors (Gregory 1985; Gippel 1998; Yao et al. 2014) found that the linear relationship between turbidity and particle concentrations broke down at very low turbidities, and that turbidity measurement becomes less reliable due to the mechanics of light scattering. Our model indicates that turbidity at lower values is likely being influenced less by larger particles, such as TSS, and more so by dissolved ions. It also may suggest that the relationship between TSS and turbidity may differ with land use at low levels, but perhaps not with higher turbidities. We did not have enough samples for TOC to be compared in the model; however, graphically it showed no relationship. Overall, this analysis points to the idea that turbidity at low levels can be influenced by many particles, and caution should be taken when using turbidity as a proxy at these levels. We also suggest that further work be performed in this area, determining the in-stream relationships between particles and turbidity at various turbidity levels, and with the influence of land use.

### 3.4.4 Water Quality Impacts

As a small community, most residents of Shawnigan Lake have little access to municipal water supplies, and many use surface water or wells to obtain their water, though few residences

rely solely on streams. The Shawnigan Watershed is also a functional aquatic habitat for many species of fish, among other animals and plants. Therefore, not maintaining surface water quality may pose a public health risk to this community, as well as threaten the ecosystem well-being. As we have demonstrated, land use has impacted water quality in the two creeks. In this section, we review the degree to which surface water has been affected, and what levels of risk/degradation we would expect in this watershed.

In British Columbia, turbidity regulations are in the form of guidelines set by the Ministry of the Environment (BCMOE 2017; BC 2018). These guidelines differ depending on the type of treatment available and the background turbidity level of the water. For the Shawnigan region, with clear water and limited water treatment for most areas, water is considered unsafe if turbidity increases by 1 NTU (for any amount of time) provided the background is <5 NTU, or an increase of 5 NTU when the background is <50 NTU provided municipal supply system users had some filtration (BCMOE 1997; BCMOE 2017). For aquatic life, the B.C. Guidelines indicate turbidity is considered unsafe if rising more than 2 NTU above background at any point for 30 days (interpreted as in a 30 day window), or more than 8 NTU at any point for 48 hours (BCMOE 1997; BCMOE 2017). This does not include high flows, however, which was when we saw the majority of our turbidity events. In our research we observed a number of potential instances of noncompliance with water quality guidelines. We present three specific examples in Fig. 3.3 showing McGee Creek increasing by approximately 2 NTU or more over several occasions during low flows within a 30 day window, and Fig. 3.4 shows Van Horne Creek increasing by more than 8 NTU on several instances, and plot C shows Van Horne Creek increasing by more than 8 NTU for the majority of March and April. All panels in Fig. 3.3 and Fig. 3.4 show increases in turbidity of at least 1 NTU on many occasions. This study showed that both study streams are slightly above

recommended guidelines in some cases, and Middle and Downstream Van Horne Creek are well above after the landslide in March.

For drinking water, there is mixed evidence around the relationship between turbidity and source water pathogenicity, Most publications refer to turbidity levels after treatment, though recently there has been a number of publications indicating source water turbidity can impact pathogenicity of water (De Roos et al. 2014). A study out of New York (Hsieh et al. 2014), an unfiltered water supply, showed a link between turbidity in source water and diarrhea hospitalization, with turbidities reaching a maximum of 3 NTU. Turbidity can also impact drinking water systems with filtration and disinfection, meaning treatment cannot always remove the risk from turbid waters. Tinker et al. (2010) identified that increases in source water turbidity in Atlanta (with filtration and disinfection treatment) were correlated with gastrointestinal illness, with turbidities ranging from a mean daily maximum of 1.5 NTU to 55 NTU, though the maximum recorded turbidity was above 1000 NTU. Alternatively, a study from Edmonton (with filtration and disinfection) showed no link between gastrointestinal illness and turbidity (Lim et al. 2002). Finally, turbidity can also interfere with treatment processes, for example, Ulrich and Bragg (2003) identified that turbidity levels of above 20 NTU can clog filters, or may have excessive concentrations of particulates, and may cause harmful byproducts when reacting with chlorine disinfection. As we saw in Table 3.1, Middle and Downstream Van Horne Creek were above 20 NTU approximately 10% of the time. Together these suggest that turbidity levels may be impacting drinking water, or at least treatment effectiveness, though turbidity may not be a reliable source of pathogenicity.

For aquatic biota, the negative impacts of turbidity are also not straightforward. Many impacts of turbid water are due to increased levels of suspended solids and sediments. These
particles can impact a variety of life stages and trophic levels (Berg and Northcote 1984; Sigler et al. 1984; Van Nieuwenhuyse and LaPerriere 1986; Servizi and Mertens 1990; Ryan 1991). It can also be lethal to many organisms (Scrivener and Brownlee 1989; Robinson et al. 2009). Additionally, suspended sediments have been indicated by several authors to be the most detrimental aspect in cumulative effects research (Ormerod et al. 2010; Chará-Serna and Richardson 2018). The exact effects will depend on the duration of exposure and the species' individual tolerance range, though generally impacts will increase as both concentration and exposure increase, above a threshold (Newcombe and McDonald 1991; Bilotta and Brazier 2008; Diehl and Wolfe 2010). Diehl and Wolfe (2010) identified that moderate impacts occur in normally clear streams within a range of longer exposure of lower concentration (< 55 mg/L for <24 hours) and short exposure of high concentration (< 1097 mg/L for <3 hours) but see Bilotta and Brazier (2008) for a more complete summary. However, Shaw and Richardson (2001) found that both invertebrate abundance and trout mass gain decreased with sediment pulses (704 mg/L) of increasing duration (1-6 hours). Our total suspended solids concentration was typically around 10, but could reach 100 mg/L at high flows in Van Horne Creek, which frequently lasted 24 hours or more. This also does not include the > 400 NTU turbidity events seen after the land slide, which likely had much higher levels of TSS. These suggest that suspended sediment is having a large impact on the aquatic wildlife in Van Horne Creek, and possibly downstream in the Shawnigan Watershed.

Turbidity, apart from the impacts of suspended sediment, has been known to impact predator-prey behaviour (Ferrari et al. 2010; VanLendeghem et al. 2011) and impact migration behaviour (Richardson et al. 2001), as well as reducing photosynthetic abilities (Lloyd et al. 1987). Most studies indicate a turbidity level of between 20 and 30 NTU as the turbidity at which 91 ecosystem impacts become noticeable. This includes avoidance behaviour (Boubee et al. 1997), migration behaviour (Richardson et al. 2001), feeding behaviour (Hecht and Van der Lingen 1992; Gregory 1993), and weight gain (Shaw and Richardson 2001). Though others have found lower turbidity to alter behaviour, such as 5-10 NTU (Abrahams and Kattenfeld 1997; De Robertis et al. 2003), and much higher, such as 120 NTU (Gregory and Levings 1998). It is possible that this range is an underestimation of the true tolerance of stream ecosystems, given that most studies occur in a laboratory, and so cannot take into account heterogeneity of natural streams (including potential hiding spots). For example, Shoup and Lane (2015) found negative correlations between largemouth bass daily consumption with turbidity in the lab, but no correlation in the field. However, even if these are an underestimation, Van Horne Creek's downstream reaches again had turbidity of above 20 NTU nearly continuously after the March landslide. This further emphasizes the potential degradation of Van Horne Creek.

As suggested based on the literature, aquatic life is likely affected by the high turbidity and sediment levels of Van Horne Creek. However, the turbidity seen in this creek prior to the landslide, as shown in Fig. 3.4, may not be captured by the guidelines. This is because the guidelines indicate that measurement should occur during "clear flows", or during winter periods without rainfall or discharge. However, according to our analysis these are the periods that produced the highest turbidity and suspended sediment levels, which can be as lethal as moderate turbidity over long periods (Diehl and Wolfe 2010). Bolstad and Swank (1997) found a similar phenomenon in the Coweeta Watershed, where water quality changes, effected by land use were larger during high flows than baseflow. This did change after the landslide in March, when "clear flow" turbidity increased, however the stream was already at risk before the landslide. In any case,

our results indicate a potential issue in the current guidelines, where the guidelines do take variation and duration of turbidity levels into account, but not during typical turbidity conditions (i.e. high discharge or after a rain event).

Finally, no amount of turbidity monitoring will be of aid if the correct legislation is not present to prosecute non-compliance when turbidity measurements exceed guidelines. Source-totap protection has been cited as a critical aspect of maintaining high quality drinking water, particularly in rural areas where water treatment may be less reliable (CCME 1999; CCME 2004; Dunn et al. 2013). To this effect, the Drinking Water Protection Act does have a clause that drinking water sources should be protected, though the act does not specifically reference turbidity, but does include turbidity in the Drinking Water Compliance Officer turbidity decision-tree (DWPA 2001; BCMH 2004). Additionally, several acts have been put into place to ensure water quality is maintained for aquatic life. This includes the Riparian Areas Protection Act (2016), suggesting that there should be fixed-width riparian buffers on new construction, which should reduce sediment entering stream tributaries (including some ditches) (RAPA 1997). All land and waterways are also subject to the Federal Fisheries Act, where there is a viable fish population, which regulates that it is an offense to allow substances deleterious to fish in waterways, of which sediments are included (Fisheries Act 2018). All of these regulations indicate that increased turbidity loads in the stream are not permissible by law; however, our study indicates that land use may still be a driver of turbidity, and source-to-tap protection is not currently effective in the Shawnigan Lake watershed.

#### 3.4.5 Study Limitations

Many study limitations have been mentioned: discharge was estimated from a nearby creek, we experienced many issues with turbidity sampling, and our land use methods had some inaccuracies in estimation. The largest limitation was the error with our turbidity sensors, as mentioned above, where turbidites exceeding 120 NTU were not recorded, despite on-site spot turbidity recorded turbidities of over 400 NTU in middle and downstream Van Horne Creek. This suggests that some of our results are underestimated, as the true peaks of Van Horne Creek were typically not recorded by the continuous turbidity sensors. Additionally, we performed our study over a nine-month period of unusual weather, with both warmer and cooler temperatures, and more rain earlier in year (Sept.-Oct.), and less during December-January. This would have led to much more hydrologic activity than would be seen in an average year, and may have over-estimated our results.

#### 3.5 Conclusions

Our research analyzed the patterns of turbidity over 8 months in small streams in the Pacific Northwest, in a watershed with many types of land-use. We found turbidity was driven by hydrology and weather and exhibited seasonal changes. Land use was also a driver of turbidity, with the two streams being significantly different in turbidity responses. This indicated that the Shawnigan Watershed was likely highly degraded in terms of water quality for both drinking water and aquatic life. We identified that contributing land use was forestry and built, as well as industry/construction, which is less well documented as a contributor to water quality degradation. Furthermore, we found that turbidity response to land use was better seen in high flows, and so may not be captured by current water quality standards.

## **Chapter 4: Discussion**

Turbidity is a common metric in water quality management. Despite its usefulness as a measure for water quality monitoring for many objectives, however, many uncertainties remain as to turbidity dynamics in the field. We explored many of these questions for turbidity in small streams. We monitored turbidity from two watersheds in coastal British Columbia for 2016 and 2017. Between our two monitoring areas, we identified several major themes: turbidity was highly influenced by local stream dynamics (discharge, rainfall, seasonal organic inputs), and was linked to land use. We develop these ideas further in the following sections, and discuss their implications for the measurement of turbidity, ecology and finally water quality management.

### 4.1 Turbidity as a Measurement

The first, and most obvious, conclusion we found was that turbidity as a measurement could be said to be simple in its collection, but difficult in its interpretation. That is, sensors deployed in the field were sensitive to biofouling, particularly in the summer months, as well as obstructions, such as sand, leaves, branches or other objects, which could produce turbidities ranging from 2 to 800 NTU. At the lower NTUs, these obstructions can be difficult to distinguish from actual turbidity. We estimate that to maintain maximum accuracy in the field, turbidity probes should be maintained and cleaned at least once every two weeks. Additionally, the error range for these sensors is 0.5 NTU; however, with the added risk of cuvette/lens scratching, or small instances of biofouling, we would suggest that the field range of error might be larger. Others have noted that turbidity sensors easily became fouled, resulting in artificially higher measures of

turbidity (e.g. Olive and Rieger 1988; Ryan 1991, 2006; Lewis 2001). Wagner et al. (2006) also identified that most sensors have an accuracy of +/- 5% or 2 NTU, whichever is larger. Finally, Soler et al. (2012) identified that different sensor types can be influenced by size/particle composition differently, further increasing the error range.

An additional concern of turbidity interpretation is that we found the relationship between particles and measured turbidity differed depending on the concentration of the particles and the particle type. For example, we identified that at low turbidities TDS may be a large component of a turbidity measurement, and therefore using turbidity to estimate TSS may not be reliable at low turbidities. Yao et al. (2014) and Ziegler et al. (2014) identified a similar phenomenon, where the relationship between turbidity and various particles differed as turbidity increased. Furthermore, Saraceno et al. (2017) found that relationships for total suspended solids had to be re-calibrated at high turbidities, due to high levels of organic material. Our research suggests that this may be required for low turbidities as well. We found evidence to suggest that the types of particles influencing turbidity is dependent upon season and weather. For example, lower turbidities often occurred during winter base flows, while in one stream, summer base flow had higher turbidity, suggesting higher levels of organic particles in and around the stream (Lewis 1996; Hudson 2001; Lenhart et al. 2010)

## 4.2 Turbidity as an Ecological Parameter

Despite the ecological significance of turbidity, its dynamics has not been well studied in small streams. Our results were somewhat surprising, in terms of what caused turbidity events, and how localized these events could be. The majority of particles typically are exported during periods of high discharge, during flooding or overbank flow (e.g. Smith et al. 2003; Raymond and Saiers 2010). Smith et al. (2003) found that approximately 70% of the total sediment mobilized over their study year were exported during three of such events. Miller et al. (2015) also identified that the highest turbidity events took place during overbank flow periods. We therefore expect that these events would produce the highest turbidities and would be consistent along the stream. We did find that high discharge was associated with turbidity events, though it did not always produce a turbidity event or the largest turbidity events.

Decoupling between turbidity and discharge has also been identified by Likens (1969), and Jordan (2006). The location of our study may have contributed as well, where the research forest described in our second chapter was on a mountainside with steeper channels than those in the Shawnigan Watershed and may require extreme events to cause flooding. This geology can produce sediment-starved streams due to shallow soil depth, which would mean that increasing discharge would quickly cause dilution, causing turbidity levels to decrease (Williams 1989; Church 1992; Brasington and Richards 2000; Langlois et al. 2005). The low sediment-availability has been demonstrated by other research in the region (Gomi et al. 2005) and was shown by our hysteresis analysis. Furthermore, our studies took place in a pluvial region, meaning that most hydrology is influenced by rain, particularly in the winter months. This can further induce dilution, as found by Bilotta et al. (2009), who identified discharge was a poor predictor of sediment concentration. Additionally, Van Horne Creek, with the highest total suspended sediment rates, had an  $r^2$  of 0.22 between discharge and turbidity, while McGee Creek had an  $r^2$  of only 0.054. These differences suggest that turbidity's relationship with discharge may be low in areas of low sediment availability, and it may increase with sediment availability.

We were able to identify two other ways discharge and rainfall can potentially influence turbidity: rainfall or discharge events, even small events, during low flows, which appear to act at a reach-scale. For example, we found the first flush effect in most of our study streams, where the first discharge/rain event causes a large turbidity event due to the amount of material having built up over the spring and summer. We also identified several turbidity events that appeared to occur with rainstorms on low discharge or even dry creeks in some cases. In McGee Creek Middle site, which we attributed to a first flush, as the creek bed dried and the rainfall likely created a turbid puddle. However, the turbidity events described in Chapter 2 and Chapter 3 may have similar mechanism. As this rainfall occurred during low discharge, it is possible that these events caused local mixing and high turbidity, similar to what was found by James and Barko (1993) and Atherhold et al. (1998). Conditions in and around the weir ponds or in dry creeks would also facilitate this interaction, due to the high storage of organic material on the ground, and in the streambed, which would be mobilized during a rain event. Furthermore, with low discharge, transport capacity would likely be low, meaning particles resuspended due to local disturbance may not be washed away, nor would a dilution effect take place.

Reach-scale turbidity was not limited solely to rain events. We were able to characterize base load turbidity, showing that turbidity has a range to which it returns when between discharge peaks. We found that turbidity had a higher base load in the spring and summer. This effect could be due to organic matter, such as in Lenhart et al. (2010) where turbidity was higher during low

flow periods due to organic matter in the (though it was not mentioned if this was not during the summer). Organic material can also increase in pool areas in small streams (Speaker et al. 1984; Clifford 1993; Mermillod-Blondin et al. 2000). Though artificial, the weir ponds that contained our sensors would likely have a similar effect, creating the period of high turbidity during the summer seen at South Creek. Together our results suggest that low turbidities in small streams is reflective of suspended sediments and dissolved solids, both inorganic and organic. This suggests that using turbidity as a proxy for suspended sediments may be less accurate or require a separate calculation at lower turbidities and in clearer waters.

## 4.3 Turbidity as a Water Quality Indicator

In our research, we identified the impacts of land-use modifications on turbidity, suggesting that turbidity is also driven by the landscape context. We had three streams in MKRF with historical land use (forestry and roads), and McGee Creek with more intense land use in the form of forestry and agriculture, which showed similar turbidity patterns. In comparison, Van Horne Creek, with higher percentages of non-forest land use (such as industrial, forestry and built) showed higher turbidities overall, as well as experiencing a turbidity event with almost every rain events. This stream also experienced a land-slide in March, which increased base load turbidity to approximately 20 NTU (from 0.7 NTU). These levels (the short pulses of high turbidity prior to the landslide, and the continuous turbidity after the landslide) likely have or will have large ecosystem impacts (Lloyd 1987; Gregory 1993; Boubée et al. 1997; Robinson et al. 2009; Shoup and Wahl 2009; Lowe et al. 2015). We identified that forestry, urban/built and construction/industry were likely contributors to turbidity, which have all been previously

described as sources of turbidity (Harbour 1999; Roy et al. 2005; McCaleb and McLaughlan 2008; Klein et al. 2012). Finally, what is very concerning about Van Horne Creek is the small percentage of land use (less than 10% for forestry, built and urban) and the large impact minor land use had on water quality.

When comparing both sub-watersheds to each other there were a number of differences. McGee Creek had much higher turbidities than the Malcolm Knapp Research Forest creeks was much more responsive to high discharge events. We were unable to perform a hysteresis analysis on either of the Shawnigan Watershed creeks due to using discharge values from a nearby creek outside of the watershed, resulting in a slight time lag. Despite this, McGee Creek showed only small signs of dilution at high discharge. This may suggest that there is increased sediment availability as compared to the streams in MKRF (Walling and Webb 1982). McGee Creek has a large amount of recent forestry, agricultural areas, and roads/ATV trails, as well as reaches with riparian zone vegetation, all of which have also been known to increase sediment availability and turbidity (Dalzell et al. 2004; Reiter et al. 2009; Klein et al. 2012). This evidence may therefore counter our previous conclusion that agriculture was not a driver of turbidity, as we only compared between the two creeks in the watershed. This also suggests caution in identifying background water quality measures, where finding reference sites unimpacted by land use is likely a challenge.

In addition to viewing turbidity as a water quality concern, we argue that some level of turbidity is also associated with a healthy, productive ecosystem. That is, base load turbidity appeared to be linked to seasonal (organic matter contributions and microbial activity) and hydrological (sediment release) variation. In this way, turbidity is likely a product of stream metabolism and hydrology, and therefore turbidity may be thought of as a water quality parameter associated with stream function, similar to TDS. This was also found by Lenhart et al. (2010),

100

where seasonal turbidity was linked to allochthonous organic carbon inputs, though an exact turbidity range was not given. Furthermore, several studies (Hech and Van Der Lingen 1992; Utne-Palm 2002; De Robertis et al. 2003) have reviewed the impact of turbidity on feeding behaviour and predator-prey dynamics. Though many focus on the decrease of the rate of prey capture resulting from small turbidity increases (less than 50 NTU) (Hech and Van Der Lingen 1992; Abrahams and Kattenfeld 1997; De Robertis et al. 2003), more recent literature has identified these same increases in turbidity (typically around a maximum of 40 NTU) can increase prey survivability (Utne-Palm 2002; Figueiredo et al. 2016; Ward et al. 2016). Carter et al. (2010) hypothesized that turbidity is as important as stream cover as an ecological parameter affecting predator-prey dynamics. Together, the results of these studies suggest that short term or seasonal turbidity may be a part of a healthy ecosystem, though long-term turbidity may have deleterious consequences. However, further research would be needed to determine the level/exposure time at which turbidity stops alleviating predation, and begins to have adverse cascading effects on habitat.

#### 4.4 Turbidity Monitoring

We monitored turbidity during base load and high flows in two catchments, the Malcolm Knapp Research Forest and the Shawnigan Watershed, which provided a small short sequence of turbidity dynamics in small streams, which we can use to compare against current water quality guidelines. In B.C., turbidity is recommended to be monitored by water suppliers at their water intake and after treatment, and a target turbidity level is specified. For surface water, the B.C. Ministry of Environment sets water quality guidelines, which include turbidity, and performs some monitoring along with the individual municipalities (Table 4.1). We identified that all streams studied in this research frequently had turbidity increases of 1 NTU, generally for a few hours, with a maximum duration of about a month, and had periods of being above guidelines for aquatic life as well. This could suggest that these streams are currently not in good condition as per these guidelines, and that their ability to provide clean drinking water and functional aquatic habitat is periodically compromised, which is likely the case for Van Horne Creek. Alternatively, it is also possible that guidelines are not capturing stream turbidity dynamics. In the following paragraphs we address these concerns.

Water after Treatment	Raw Drinking Water	Recreational Water	Aquatic Ecosystem
Water should be <1 NTU	<u>With Treatment:</u> Change from background of 5 NTU when background is <50 NTU. Change from background of 10% when background is >50 NTU <u>Without Treatment:</u> Change of 1 NTU when less than 1 NTU, otherwise change of 5 NTU, if water is >50, change	Change of background from 5 NTU when background is <50 NTU	Change from background by 8 NTU at anytime one time for a duration of 24 hours, or 2 NTU at anytime for a duration of 30 days, or a 5 NTU from a background of 8-50 NTU during high flows or in turbid waters, or a change of 10% in waters <50 NTU.
	of 10% from background		

**Table 4.1:** Water quality guidelines for turbidity from British Columbia (BCMOE 2017).

Our first concern is that guidelines are not structured for accurate determination of turbidity. Based on our experience measuring turbidity in the field, these standards are quite strict considering the sensors' range of error. B.C. guidelines for drinking water imply that any water is unclean if there is an increase of 1 NTU in clear water. Determining these numbers would require highly accurate instruments and consistent sampling regimes, such as dual-beam sensors, not the typical sensors used in research (Wagner et al. 2006). In our study and others (Ryan 1991; Gippel 102

1995; Lewis and Eads 2001), turbidity appeared to be challenging to measure accurately. We estimated above that although the manufacturer's sensor error range is ~0.5 NTU, it is likely higher with field measurements. Wager et al. (2006) estimated the true error range to be 2 NTU or 5%, whichever is larger. This suggests that the current methods of looking for turbidity changes of 1 NTU in the field may be inaccurate, though changes of 2NTU to 5 NTU may be possible.

Our results, and those of others (Lenhart et al. 2010, Miller et al. 2015), show that turbidity is highly specific to the individual stream, and even to the stream reach, due to the varying effects of geology, land use, and lakes or wetlands. Several other water quality parameters also vary with these parameters, such as organic carbon and wetland area (Creed et al. 2008). Furthermore, even within streams, it has been identified that pool structures differ from riffles in their particle fluctuation and dynamics, something that would also affect turbidity (Clifford 1993; Hassan et al. 2005; Thompson 2011). This suggests that samples taken in one area may not be informative of other reaches in the same creek, and guidelines should reflect this uncertainty. This may mean that water quality assessments made in one location are only accurate for the immediate area. Even the use of "background" turbidity with which standards are compared, defined by the province as an upstream or nearby site, may not be a reliable measure for comparison if taken too far away. For drinking water, in some situations, sensors are in a fixed and sheltered location near the intake pipe, so can typically be monitored with accuracy. However, many residences in the Shawnigan Watershed, and in other parts of B.C., are monitored at select locations regularly (but not continuously) and those measurements are used to estimate the drinking water quality for the whole area. Given the frequency and range of turbidity fluctuations (1 NTU fluctuations can happen quickly for many reasons and be highly local), anything less than high resolution,

continuous monitoring will not provide an accurate estimate of turbidity to meet guidelines or regulatory levels.

In addition to inaccurately sensing turbidity, standards also appear to not be well suited for seasonal or anthropogenically-induced variation. Raymond and Saiers (2010) found that organic carbon increased significantly during summer periods, which we also observed in streams with minimal or historical land use. Furthermore, it has been noted that large turbidity events can occur from snow-melt or rain-on-snow events(Jordan 2006). That there is no mention of this in the B.C. turbidity guidelines suggests that perhaps seasonal variability is not taken into account, which may lead to incorrect interpretations of anthropogenic impacts during seasonal turbidity periods. Furthermore, turbidity guidelines are applied at "clear flows", defined as low flows during the winter and fall. However, it has been shown by Lenhart et al. (2010) that turbidity behaves quite differently at different flow levels, and others have found that high sediment pulses are common in anthropogenically affected streams (Bolstad and Swank 2007; Gasperi et al. 2009). We identified that before the landslide, Van Horne returned to a similar base load turbidity as McGee Creek, but experienced much higher peaks during each rainstorm (of ~100 NTU, though this may be an underestimate), sometimes for up to 24 hours. Both of these states, continuously high turbidity and short duration bursts (of a few hours) of high turbidities (or suspended sediments) can be harmful to aquatic life (Newcombe and Macdonald 1991; Robinson et al. 2009; Lowe et al. 2015). Furthermore, Shaw and Richardson (2001) found frequent but short pulses of sediments at a turbidity of only 23 NTU altered invertebrate drift and reduced trout weight gain, while we identified turbidities of up to 100 NTU regularly for hours at a time. Therefore, capturing turbidity only at clear flows may fail to detect some impacts of land use.

Finally, appropriate monitoring is not complete without adequate enforcement. There are several Regulatory Acts that are applicable to the Shawnigan Area, though few specify turbidity or suspended solids. The Drinking Water Protection Act (DWPA, 2001) applies to all water sources and water users in B.C. above single family residences and provides protection for drinking water sources. Associated with the DWPA are a Boil Water Advisory Tree and the Drinking Water Protection Officers Guide, which mention turbidity, and could apply to the Shawnigan Watershed surface water, though the act mostly focuses on *E. coli* levels (BCMOH 2017a; BCMOH 2017b; BC 2018). The Environmental Management Act (2003) applies to most ecosystems, not specifying regulations for aquatic areas, but does mention the stabilization/control of sediments on property. The Riparian Areas Protection Act (2016), formerly the Fish Protection Act, outlines streamside management, though does not mention roadside ditches or sediment control outside of these areas.

Finally, the federal Fisheries Act (2018) is a strong Regulatory Act for the protection of aquatic ecosystems and is relevant to many of these streams as they are fish-bearing (Hutchings and Post 2013). However, the Fisheries Act only mentions that no activity should occur that results in serious harm to commercial or recreational fish, and Aboriginal Fisheries, but does not specify what activities would cause this or what water quality guidelines are followed. The Fisheries Act could utilize the B.C. Ambient Water Quality Guidelines, though the CCME has produced similar guidelines (CCME 2001; BCMOE 2017).

Despite active legislation, water quality in the Shawnigan Watershed has continued to degrade, even over the course of this thesis. This is also not a case of a lack of reporting or unconcerned citizens, as the Shawnigan area has an active Residents' Association, and the citizens are passionate about their water, even pursuing legal action against industrial development. Instead, it is possible that the Shawnigan Watershed has fallen into a legislation gap, where decentralized water management can lead to a separation between water planning and regulatory enforcement (Bakker and Cook 2011; Hill et al. 2013; Dunn et al. 2014). For example, Dunn et al. (2015) noted that water managers in B.C. typically feel left out of land-use planning and can feel powerless to prevent water quality degradation. Adding insult to injury, the province appears to make assumptions that if single-family residences and small water suppliers do not have adequate treatment, their water source must be of good quality (BCMOH 2017a; BCMOH 2017b; BC 2018).

#### 4.5 Turbidity as a Policy Objective

So where does one go from here? It is clear that a number of improvements are needed for water quality management in B.C. We have identified several concerns with current guidelines. Therefore, we would suggest that guidelines incorporate high flow and seasonal components, as well as duration records for turbidity, which could be determined with high quality monitoring. However, improving guidelines is not as simple as altering numbers. Magner and Brooks (2007) identified that standards for ambient water quality and pollution identification should be slightly different, while B.C. currently appears to have a combined set of guidelines. For example, the recommendation that water quality should not increase by more than 2 NTU for 30 days implies a general range within which turbidity should remain, rather than providing a gradient of turbidities and exposure times at which various amounts of ecosystem degradation will occur. Our suggestion, therefore, is to separate the ambient water quality recommendations from maximum acceptable turbidity limits, which could be more specific to turbidity levels/exposure duration that have large ecosystem impacts.

In addition to improved standards, the most obvious recommendation is to improve monitoring capacity, which will improve the province's ability to apply current water quality standards. One of the benefits of turbidity as a water quality measure is that it can (and arguably should) be monitored continuously (Magner and Brooks 2007). Many have found that regular sampling (but not continuous) can underestimate sediment loads (Walling and Webb 1981; Ferguson 1987; Foster et al. 1990), and similarly to our study, Magner and Brooks (2007) found that the effects of land use were difficult to determine due to low-resolution monitoring. Lenhart et al. (2010) suggested spatially distributed monitoring to ensure that all landforms are taken into consideration. This line of thinking has produced the sentinel method, of forming a spatially complex monitoring system to determine changes in water quality (Wiersma 2005; Magner and Brooks 2007). Though these authors imply that a monitoring system is applicable to most water bodies, this could be limited to drinking water sources, or areas where development or industry is currently active.

Monitoring turbidity alone may be insufficient. Although turbidity is useful, it is difficult to pinpoint a turbidity source, and manage land use accordingly. Furthermore, many of the environmental impacts associated with turbidity are due to suspended sediments and anthropogenic changes that are known can alter more water quality parameters than turbidity. Therefore, using one measurement may not capture the entire story. Several authors (Allan et al. 1997; Nelson and Booth 2002; Dalzell et al. 2004; McCaleb and McLaughlin 2008; Duan et al. 2015) have identified TSS as a measurement that increases with land use, and by extension, substrate samples may also be an indicator (Quinn et al. 1997). Gasperi et al. (2009) found several metrics that could be quite informative for urbanized watersheds, such as measuring Cr, Ni, NP, and PAH concentrations, while Tong and Chen (2002) found NP as a highly important measure

for both agriculture and urbanizing catchments. Additionally, DeGasperi et al. (2009) identified the hydrological indicators High Pulse Count and High Pulse Range as being indicative of urbanized catchments. Alternatively, Bilotta et al. (2012) recommended developing environmentspecific water quality guidelines, using suspended particulate ranges, which could potentially provide more accurate comparisons with less intensive monitoring strategies.

However, even if the above recommendations regarding monitoring are addressed, there are still additional concerns regarding how standards are applied/enforced, where even accurate and improved regulations are not helpful in managing aquatic resources if they are not enforced. A common concept is termed Source-to-Tap Protection, or the multi-barrier approach. This concept implies that protection of water at its source relies less on filtration, and therefore decreases risk in areas of limited treatment (Byveld et al. 2008; Summerscales and McBean 2011; WHO 2017). Watershed management is a similar concept, having been adopted in Ontario through the Conservation Authorities Act (2018). It has also been cited as an effective strategy for environmental management (Porter et al. 2013). It has also been noted that managing water quality for drinking supplies can have beneficial impacts on aquatic life, meaning that protecting surface water can have two-fold benefits (Serveiss and Ohlson 2007). True recommendations for this would require a much more in-depth discussion of water governance, which is well beyond the scope of this thesis. Instead, we simply wish to point out the complexities of monitoring turbidity in the context of governance regulations.

## **Chapter 5: Conclusion**

In summary, our work studied turbidity dynamics and drivers in small, forested streams in coastal British Columbia, with continuous measurements performed over a year period. We found that turbidity behaved in complex ways; however, some patterns were shown to be somewhat predictable, and ecologically significant. For example, turbidity responded to high discharge events, and to small discharge events during low flows. Additionally, turbidity appeared to be caused by organic matter activity during summer periods. We also found turbidity to be a product of discharge, rain and certain land uses. Turbidity in the Shawnigan Lake Watershed streams studied may be well above safe levels for aquatic life. Finally, we addressed concerns regarding turbidity standards and management, and provided several recommendations. Overall, we determined that turbidity is a useful measure of water quality that has the potential to be predictable, though turbidity guidelines may require review for monitoring accuracy and decision-making.

# References

Abrahams, M.V., and Kattenfeld, M.G. 1997. The role of turbidity as a constraint on predatorprey interactions in aquatic environments. *Behavioral Ecology and Sociobiology*, 40: 169-174.

Alexander, R.B., Boyer, E.W., Smith, R.A., Schwarz, G.E. and Moore, RB. 2007. The Role of Headwater Streams in Downstream Water Quality. *Journal of the American Water Resources Association*, *43*: 41-59.

Abrantes, N., Antunes, S.C., Pereira, M.J., & Gonçalves, F. 2006. Seasonal succession of cladocerans and phytoplankton and their interactions in a shallow eutrophic lake (Lake Vela, Portugal). *Acta Oecologica*, *29*: 54-64.

Alberta Government. 2012. Part 1: Standards for Municipal Waterworks of a Total of 5 Parts. ISBN: 978-0-7785-9634-9

Alberta Government. 2018. Environmental Quality Guidelines for Alberta Surface Waters. ISBN 978-1-4601-3873-1

Alberto, A., St-Hilaire, A., Courtenay, S.C., & Ven den Heuvel, H.R. 2016. Monitoring stream sediment loads in response to agriculture in Prince Edward Island, Canada. *Environmental Monitoring Assessment*, 188: 415

Alexander, R.B., Boyer, E.W., Smith, R.A., Schwarz, G.E., & Moore, R.B. 2007. The role of headwater streams in downstream water quality. *Journal of the American Water Resources Association, 43:* 41-59

Allan, J.D., & Castillo, M.M. 2007. *Stream Ecology: structure and function of running waters*. Dordrecht, The Netherlands: Springer.

Allen, M.J., Brecher, R.W., Copes, R., Hrudey, S.E., & Payment, P. 2008. Turbidity and Microbial Risk in Drinking Water. *British Columbia Ministry of Health.* <u>http://www.wsabc.ca/wp-content/uploads/2011/04/TAC-Turbidity-Report.pdf</u> Accessed Oct 3, 2015

Ahearn, D.S., Sheibley, R. W., Dahlgren, R.A., Anderson, M., Johnson, J., & Tate, K. W. 2005. Land use and land cover influence on water quality in the last free-flowing river draining the western Sierra Nevada, California. *Journal of Hydrology*, *313*: 234-247.

Andrew, M.E., Wulder, M.A., Nelson, T.A., & Coops, N.C. 2015. Spatial data, analysis approaches, and information needs for spatial ecosystem service assessments: a review. *GIScience & Remote Sensing*, *52*: 344-373

Angradi, T.R. 1996. Inter-habitat variation in benthic community structure, function, and organic matter storage in 3 Appalachian headwater streams. *Journal of the North American Benthological Society*, *15*: 42-63.

Ankcorn, P. 2003. Clarifying Turbidity – The Potential and Limitations of Turbidity as a Surrogate for Water-Quality Monitoring. *Proceedings of the 2003 Georgia Water Resources Conference*: 1-4

Aqua-Tex Scientific Consulting Ltd. 2014. McGee Creek Proper Functioning Condition Assessment. pp 1-277

Aramini, J., McLean, M., Wilson, J., Holt, J., Copes, R., Allen, B., & Sears, W. 2000. Drinking water quality and health-care utilization for gastrointestinal illness in greater Vancouver. *Canada Communicable Disease Report*, *26*: 211-214.

APHA. 2005. Standard methods for the examination of water and wastewater, 21<sup>st</sup> ed. Washington, DC: American Public Health Association

Asselman, N.E. 1999. Suspended sediment dynamics in a large drainage basin: the River Rhine. *Hydrological Processes*, *13*: 1437-1450.

Asselman, N.E., Middelkoop, H. & Van Dijk, P. M., 2003. The impact of changes in climate and land use on soil erosion, transport and deposition of suspended sediment in the River Rhine. *Hydrological Processes*, *17*: 3225-3244.

Atherholt, T.B., LeChevallier, M.W., Norton, W.D., & Rosen, J.S. 1998. Effect of rainfall on Giardia and Crypto. *Journal of the American Water Works Association*, *90*: 66-80.

Bača, P. 2008. Hysteresis effect in suspended sediment concentration in the Rybárik basin, Slovakia/Effet d'hystérèse dans la concentration des sédiments en suspension dans le bassin versant de Rybárik (Slovaquie). *Hydrological Sciences Journal*, *53*: 224-235.

Bachmann, R.W. 1958. *The ecology of four north Idaho trout streams with references to the influence of forest road construction*. Moscow, Idaho: University of Idaho, Thesis.

Bakker, K., & Cook, C. 2011. Water governance in Canada: Innovation and fragmentation. *Water Resources Development*, 27: 275-289.

Barrett, G., & Slaymaker, O. 1989. Identification, characterization, and hydrological implications of water repellency in mountain soils, southern British Columbia. *Catena*, *16*: 477-489.

Beaudeau, P., Le Tertre, A., Zeghnoun, A., Zanobetti, A., & Schwartz, J. 2012. A time series study of drug sales and turbidity of tap water in Le Havre, France. *Journal of Water and Health*, *10*: 221-235.

Benda, L., Hassan, M.A., Church, M., & May, C.L. 2005. Geomorphology of Steepland Headwaters: The transition from Hillslopes to Channels. *Journal of the American Water Resources Association*, *41*: 835-851

Berg, L. 1982. The effect of exposure to short-term pulses of suspended sediment on the behavior of juvenile salmonids. *Proceedings of the Carnation Creek Workshop: a ten-year review. Department of Fisheries and Oceans, Pacific Biological Station, Nanaimo, Canada.* pp. 177-196.

Berg, L., & Northcote, T. G. 1985. Changes in territorial, gill-flaring, and feeding behavior in juvenile coho salmon (Oncorhynchus kisutch) following short-term pulses of suspended sediment. *Canadian Journal of Fisheries and Aquatic Sciences*, *42*, 1410-1417.

Beschta, R.L., Bilby, R.E., Brown, G.W., Hotldby, L.B., & Hofstra, T.D. 1986. Stream temperatures an aquatic habitat: fisheries and forestry interactions. In: Salo, E.O. & Vcundy, T. W. (eds) *Symposium on Streamside Management Forest and Fishery Interactions*. Seattle, Washington: University of Washington, pp. 191-232.

Best, R. 2001. Shawnigan watershed – a fisheries perspective. http://www.websitings.net/shawnigan\_watershed/download.shtml

Bilotta, G.S., Brazier, R.E., Haygarth, P.M., Macleod, C.J., Butler, P., Granger, S., Krueger, T., Freer, J. & Quinton, J., 2008. Rethinking the contribution of drained and undrained grasslands to sediment-related water quality problems. *Journal of Environmental Quality*, *37*: 906-914.

Bilotta, G.S., Burnside, N.G., Cheek, L., Dunbar, M.J., Grove, M.K., Harrison, C., Joyce, C., Peacock, C. and Davy-Bowker, J., 2012. Developing environment-specific water quality guidelines for suspended particulate matter. *Water Research*, *46*: 2324-2332.

Bilotta, G.S., & Brazier, R.E. 2008. Understanding the influence of suspended solids on water quality and aquatic biota. *Water Research*, *42*: 2849-2861.

Bisson, P.A., & Bilby, R.E. 1982. Avoidance of suspended sediment by juvenile coho salmon. *North American Journal of Fisheries Management*, 2: 371-374.

Bolstad, P.V., & Swank, W.T. 1997. Cumulative Impacts of Landuse on Water Quality in a Southern Appalachian Watershed. *Journal of the American Water Resources Association*, *33*: 519-533.

Boubée, J.A., Dean, T.L., West, D.W., and Barrier, R.F. 1997. Avoidance of suspended sediment by the juvenile migratory stage of six New Zealand native fish species. *New Zealand Journal of Marine and Freshwater Research*, *31*: 61-69.

Brasington, J., & Richards, K. 2000. Turbidity and suspended sediment dynamics in small catchments in the Nepal Middle Hills. *Hydrological processes*, *14*: 2559-2574

British Columbia. 2018. Shared Roles & Responsibilities in Protecting B.C. Drinking Water. *Victoria BC (CA): BC Ministry of Health. Accessed online at* <u>https://www2.gov.bc.ca/gov/content/environment/air-land-water/water/water-quality/drinking-water-quality/shared-roles-and-responsibilities</u> Accessed July 14, 2018

British Columbia Ministry of Environment. 1997. British Columbia Ambient Water Quality Guidelines (Criteria) for Turbidity, Suspended and Benthic Sediments. Technical Appendix. Victoria, B.C. Accessed online from <u>http://www2.gov.bc.ca/assets/gov/environment/air-landwater/water/waterquality/wqgs-wqos/approvedwqgs/turbidity-tech.pdf</u> Accessed August 28, 2018

British Columbia Ministry of Environment. 2001. Water Quality: Ambient Water Quality Guidelines (Cirteria) for Turbidity, Suspended and Benthic Sediments Overview Report. Prov. B.C., Victoria B.C. <u>https://www2.gov.bc.ca/assets/gov/environment/air-land-</u> water/water/waterquality/wqgs-wqos/approved-wqgs/turbitity-or.pdf Accessed August 28, 2018

British Columbia Ministry of Environment. 2017. B.C. Source Drinking Water Quality Guidelines: Guideline Summary. Water Quality Guideline Series, WQG-01. Prov. B.C., Victoria B.C. <u>https://www2.gov.bc.ca/assets/gov/environment/air-land-water/water/waterquality/wqgswqos/approved-wqgs/source\_drinking\_water\_quality\_guidelines\_bcenv.pdf</u> Accessed August 28, 2018

British Columbia Ministry of Health and Ministry of Water, Land and Air Protection. 2004. Drinking Water Source-to-Tap Screening Tool. *Victoria BC (CA): BC Ministry of Health. Accessed online at <u>https://www2.gov.bc.ca/assets/gov/environment/air-land-</u> <u>water/water/documents/bc\_drinking\_water\_screening\_tool.pdf</u> Accessed August 28, 2018* 

British Columbia Ministry of Health. 2017a. Drinking Water Officers' Guide. *Victoria BC (CA):* BC Ministry of Health. Accessed online at <u>https://www2.gov.bc.ca/gov/content/environment/air-</u>

<u>land-water/water-quality/drinking-water-quality/how-drinking-water-is-protected-in-</u> <u>bc#Drinking%20Water%20Officers%20Guide</u> Accessed July 14, 2018

British Columbia Ministry of Health. 201b7. Small Water System Guidebook. *Victoria BC (CA):* BC Ministry of Health. Accessed online at <u>https://www2.gov.bc.ca/assets/gov/environment/air-</u> land-water/water/waterquality/monitoring-water-quality/small\_water\_system\_guidebook\_january\_2017.pdf Accessed July 14, 2018

Bryden, G., & Barr, L. 2002. Shawnigan Lake Community Water Supply. Land and Water British Columbia Inc.

Burrough, P.A. 1986. Principals of geographical information systems and land resources assessment. Oxford, UK: Clerandon Press

Byleveld, P.M., Deere, D., & Davison, A. 2008. Water safety plans: planning for adverse events and communicating with consumers. *Journal of Water and Health*, 6: 1-9.

Canadian Council of Ministers of the Environment. 1999. Canadian water quality guidelines for the protection of aquatic life. In: *Canadian Environmental Quality Guidelines*, 1999, Winnipeg, Manitoba: Canadian Counsil of Ministers of the Environment.

Canadian Council of Ministers of the Environment. 2004. From Source to Tap: Guidance on the Multi-Barrier Approach to Safe Drinking Water. Winnipeg, Manitoba: Canadian Counsil of Ministers of the Environment.

Carlson, T.N., & Arthur, S.T. 2000. The impact of land use—land cover changes due to urbanization on surface microclimate and hydrology: a satellite perspective. *Global and Planetary Change*, 25: 49-65.

Carrillo, U., Díaz-Villanueva, V. & Modenutti, B., 2018. Sustained effects of volcanic ash on biofilm stoichiometry, enzyme activity and community composition in North-Patagonia streams. *Science of the Total Environment*, *621*: 235-244.

Carter, M.W., Shoup, D.E., Dettmers, J.M., & Wahl, D.H. 2010. Effects of turbidity and cover on prey selectivity of adult smallmouth bass. *Transactions of the American Fisheries Society*, *139*: 353-361.

Chará-Serna, A.M. 2017. *Cumulative effects of multiple agricultural stressors on freshwater ecosystems.* PhD Thesis UBC.

Chará-Serna, A.M., & Richardson, J.S. 2018. Chlorpyrifos interacts with other agricultural stressors to alter stream communities in laboratory microcosms. *Ecological Applications*, *28*, 162-176.

Church, M. 1992. Channel Morphology & Typology. *In:* Calow, C., & Petts, G. (eds), *The Rivers Handbook*. Oxford, United Kingdom: Blackwell, pp. 126-143

Church, M. 2002. Geomorphic thresholds in riverine landscapes. *Freshwater Biology*, 47: 541-557.

Church, M. 2006. Bed material transport and the morphology of alluvial river channels. *Annual Reviews in Earth and Planetary Science*, *34*: 325-354.

Church, M., & Gilbert, R. 1975. Proglacial fluvial and lacustrine sediments. In Jopling, A.V. & McDonald, B.C. (Eds.), Glaciofluvial and glaciolacustrine sedimentation. Tulsa, OK: Society of Economic Paleontologists and Mineralogists, pp. 23.

Clifford, N.J. 1993. Differential bed sedimentology and the maintenance of riffle-pool sequences. *Catena*, 20: 447-468.

Cole, J.J., Prairie, Y.T., Caraco, N.F., McDowell, W.H., Tranvik, L.J., Striegl, R.G., Duarte, C.M., Kortelainen, P., Downing, J.A., Middelburg, J.J. & Melack, J. 2007. Plumbing the global carbon cycle: integrating inland waters into the terrestrial carbon budget. *Ecosystems*, *10*: 172-185.

Connolly, N.M., Pearson, R.G., Loong, D., Maughan, M., & Brodie, J. 2015. Water quality variation along streams with similar agricultural development but contrasting riparian vegetation. *Agriculture, Ecosystems & Environment, 213:* 11-20.

Coops, N.C., & Tooke, T.R. 2017. Introduction to Remote Sensing. In *Learning Landscape Ecology*. New York, NY: Springer, pp. 3-19.

Cowichan Valley Regional District. 2018. Water Samples. *Available from* <u>www.cvrd.bc.ca/2708/Shawnigan-Lake-North</u> Accessed August 20, 2018

Creed, I.F., Beall, F.D., Clair, T.A., Dillon, P.J., & Hesslein, R.H. 2008. Predicting export of dissolved organic carbon from forested catchments in glaciated landscapes with shallow soils. *Global Biogeochemical Cycles*, 22: GB4024

Cunha, D.G. F., Sabogal-Paz, L.P., & Dodds, W.K. 2016. Land use influence on raw surface water quality and treatment costs for drinking supply in São Paulo State (Brazil). *Ecological Engineering*, *94*: 516-524.

Dalzell, B.J., Gowda, P.H., & Mulla, D.J. 2004. Modelling sediment and phosphorus losses in an agricultural watershed to meet TMDLs. *Journal of the American Water Resources Association*, *40*, 533-543.

Davies-Colley, R.J., & Smith, D.G. 2001. Turbidity, suspended sediment, and water clarity: a review. *Journal of the American Water Resources Association*, *37*, 1085-1101.

DeGasperi, C.L., Berge, H.B., Whiting, K.R., Burkey, J.J., Cassin, J.L., & Fuerstenberg, R.R. 2009. Linking Hydrologic Alteration to Biological Impairment in Urbanizing Streams of the Puget Lowland, Washington, USA 1. *Journal of the American Water Resources Association*, *45*: 512-533.

De Robertis, A., Ryer, C. H., Veloza, A., and Brodeur, R. D. 2003. Differential effects of turbidity on prey consumption of piscivorous and planktivorous fish. *Canadian Journal of Fisheries and Aquatic Sciences*, 60: 1517-1526.

De Roos, A.J., Gurian, P.L., Robinson, L.F., Rai, A., Zakeri, I., & Kondo, M. C. 2017. Review of epidemiological studies of drinking-water turbidity in relation to acute gastrointestinal illness. *Environmental Health Perspectives*, *125*: 086003.

Diehl, T.H., and Wolfe, W.J. 2010. Suspended-Sediment Concentration Regimes for Two Biological Reference Streams in Middle Tennessee 1. *Journal of the American Water Resources Association*, 46: 824-837.

Dodds, W.K., & Whiles, M.R. 2004. Quality and quantity of suspended particles in rivers: continent-scale patterns in the United States. *Environmental Management, 33:* 355-367.

Dorner, S.M., Anderson, W.B., Gaulin, T., Candon, H.L., Slawson, R.M., Payment, P., & Huck, P.M. 2007. Pathogen and indicator variability in a heavily impacted watershed. *Journal of Water and Health*, *5*: 241-257.

Downing, J. 2006. Twenty-five years with OBS sensors: The good, the bad, and the ugly. *Continental Shelf Research*, 26: 2299-2318.

Drinking Water Protection Act. 2001. SBC c. 9.

Duan, W.L., He, B., Takara, K., Luo, P.P., Nover, D., & Hu, M.C. 2014. Modeling suspended sediment sources and transport in the Ishikari River basin, Japan, using SPARROW. *Hydrological Earth Systems Science*, *19*: 1293-1306

Dubey, V.K., Sarkar, U.K., Pandey, A., Sani, R. & Lakra, W.S., 2012. The influence of habitat on the spatial variation in fish assemblage composition in an unimpacted tropical River of Ganga basin, India. *Aquatic Ecology*, *46*: 165-174.

Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z.I., Knowler, D.J., Lévêque, C., Naiman, R.J., Prieur-Richard, A.H., Soto, D., Stiassny, M.L. & Sullivan, C.A. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews*, *81:* 163-182.

Dunn, G., Harris, L., Cook, C., and Prystajecky, N. 2014. A comparative analysis of current microbial water quality risk assessment and management practices in British Columbia and Ontario, Canada. *Science of the Total Environment*, *468*: 544-552.

Ekness, P., & Timothy R., 2007. Effects of Riparian Areas, Stream Order, and Land Use Disturbance on Watershed-Scale Habitat Potential: An Ecohydrologic Approach to Policy. *Journal of the American Water Resources Association*, *43*: 1468-1482.

Environment Canada. 2017. Real-Time Hydrometric Data. *Accessed from* https://wateroffice.ec.gc.ca/mainmenu/real\_time\_data\_index\_e.html accessed on May 20, 2017

EPA. 1986. Quality Criteria for Water. EPA 440/5-86-001

EPA. 1993. *Method 180.1: Determination of Turbidity by Nephlometry*. Cincinnati, Ohio: Environmental Monitoring Systems Laboratory Office of Research and Development U.S. Environmental Protection Agency.

EPA. 2009. National Primary Drinking Water Regulations Complete Table. EPA 816-F-09-004

EPA. 2012. 2012 Recreational Water Quality Criteria. EPA 820-F-12-061

EPA. 2015. Connectivity of Streams and Wetlands to Downstream Waters: A Review and Synthesis of the Scientific Evidence (Final Report). U.S. Washington, DC: U.S. Environmental Protection Agency, EPA/600/R-14/475F

EPA. 2016. Office of Water and Office of Research and Development. National Rivers and Streams Assessment 2008-2009: A Collaborative Survey (EPA/841/R-16/007). Washington, DC: U.S. Environmental Protection Agency <u>http://www.epa.gov/national-aquatic-resource-surveys/nrsa</u> accessed on: July 2, 2018

EPA. 2018. National Recommended Aquatic Life Criteria Table. *Accessed at* <u>https://www.epa.gov/wqc/national-recommended-water-quality-criteria-aquatic-life-criteria-table#a</u> accessed on: July 2, 2018 ESRI. 2011. ArcGIS Desktop. Release 10. Redlands, CA: Environmental Systems Research Institute.

Feller, M.C. 1977. Nutrient movement through western hemlock-western redcedar ecosystems in southwestern British Columbia. *Ecology*, *58*: 1269-1283.

Feller, M.C., & Kimmins, J.P. 1979. Chemical characteristics of small streams near Haney in southwestern British Columbia. *Water Resources Research*, *15*: 247-258

Ferguson, R.I. 1987. Accuracy and precision of methods for estimating river loads. *Earth Surface Processes and Landforms*, *12*: 95-104.

Fiebig, D.M., Lock, M.A., & Neal, C. 1990. Soil water in the riparian zone as a source of carbon for a headwater stream. *Journal of Hydrology*, *116:* 217-237.

Figueiredo, B.R., Mormul, R.P., Chapman, B.B., Lolis, L.A., Fiori, L.F., & Benedito, E. 2016. Turbidity amplifies the non-lethal effects of predation and affects the foraging success of characid fish shoals. *Freshwater Biology*, *61*: 293-300.

Finlayson, B.L. 1985. Field calibration of a recording turbidity meter. Catena, 12: 141-147.

Fisher, S.G., & Likens, G.E. 1973. Energy flow in Bear Brook, New Hampshire: an integrative approach to stream ecosystem metabolism. *Ecological Monographs*, *43*: 421-439.

Fisheries Act. R.S.C., 2018, c. F-14.

Fondriest Environmental Inc. 2016. Measuring Turbidity, TSS and Water Clarity. Accessed at <u>https://www.fondriest.com/environmental-measurements/equipment/measuring-water-</u> guality/turbidity-sensors-meters-and-methods/ accessed on June 5, 2018

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K. & Helkowski, J.H. 2005. Global consequences of land use. *Science*, *309*: 570-574.

Foster, I.D.L., Millington, R., & Grew, R.G. 1992. The impact of particle size controls on stream turbidity measurement; some implications for suspended sediment yield estimation. *Erosion and Sediment Transport Monitoring Programmes in River Basins*, 210: 51-62.

Freeman, M.C., Pringle, C.M. & Jackson, C.R. 2007. Hydrologic Connectivity and the Contribution of Stream Headwaters to Ecological Integrity at Regional Scales. *Journal of the American Water Resources Association*, *43:* 5-14.

Frameworking Committee of the GWSP. 2004. The Global Water System Project: Science Framework and Implementation Activities. *Earth System Science Partnership*. (Draft: February 6, 2004)

Gasperi, J., Garnaud, S., Rocher, V., & Moilleron, R. 2009. Priority pollutants in surface waters and settleable particles within a densely urbanised area: Case study of Paris (France). *Science of the Total Environment*, 407: 2900-2908.

Gasperi, J., Sebastian, C., Ruban, V., Delamain, M., Percot, S., Wiest, L., Mirande, C., Caupos, E., Demare, D., Kessoo, M.D.K. & Saad, M., 2014. Micropollutants in urban stormwater: occurrence, concentrations, and atmospheric contributions for a wide range of contaminants in three French catchments. *Environmental Science and Pollution Research*, *21*: 5267-5281.

Ginting, D., & Mamo, M. 2006. Measuring runoff-suspended solids using an improved turbidometer method. *Journal of Environmental Quality*, *35*: 815-823.

Gippel, C.J. 1989. The use of turbidimeters in suspended sediment research. *Hydrobiologia*, *176:* 465-480.

Gippel, C.J. 1995. Environmental hydraulics of large woody debris in streams and rivers. *Journal of Environmental Engineering*, *121*: 388-395.

Glotzbecker, G.J., Ward, L.W., Waters, D.M., & Blum, M.J. 2015. Turbidity alters pre-mating social interactions between native and invasive stream fishes. *Freshwater Biology*, 60: *1784-1793* 

Glova, G.J. 1987. Comparison of allopatric cutthroat trout stocks with those sympatric with coho salmon and sculpins in small streams. *Environmental Biology of Fishes*, 20: 275-284.

Gomi, T., Dan Moore, R., & Hassan, M.A. 2005. Suspended sediment dynamics in small forest streams of the pacific northwest. *Journal of the American Water Resources Association*, *41*: 877-898.

Goni, M.A., Moore, E., Kurtz, A., Portier, E., Alleau, Y. & Merrell, D., 2014. Organic matter compositions and loadings in soils and sediments along the Fly River, Papua New Guinea. *Geochimica et Cosmochimica Acta*, *140*: 275-296.

Greig, S.M., Sear, D.A., & Carling, P.A. 2005. The impact of fine sediment accumulation on the survival of incubating salmon progeny: Implications for sediment management. *Science of the Total Environment, 344:* 241-258.

Grayson, R.B., Finlayson, B.L., Gippel, C.J., & Hart, B.T. The potential of field turbidity measurements for the computation of total phosphorus and suspended solids loads. *Journal of Environmental Management*, 47: 257-267.

Gregory, J. 1985. Turbidity fluctuations in flowing suspensions. *Journal of Colloid and Interface Science*, 105: 357-371.

Gregory, R. S. 1993. Effect of turbidity on the predator avoidance dynamics of juvenile chinook salmon (Oncorhynchus tshawytscha). *Canadian Journal of Fisheries and Aquatic Sciences*, *50*: 241-246.

Gregory, S.V., Swanson, F.J., McKee, W.A., & Cummins, K.W. 1991. An ecosystem perspective of riparian zones. *BioScience*, *41*: 540-551.

Gurnell, A.M., & Warburton, J. 1990. The significance of suspended sediment pulses for estimating suspended sediment load and identifying suspended sediment sources in Alpine glacier basins. *Hydrology of Mountainous Regions*, *1*: 463-470.

Harbor, J. 1999. Engineering geomorphology at the cutting edge of land disturbance: erosion and sediment control on construction sites. *Geomorphology*, *31*: 247-263.

Hasenmueller, E.A., Criss, R.E., Winston, W.E., & Shaughnessy, A.R. 2017. Stream hydrology and geochemistry along a rural to urban land use gradient. *Applied Geochemistry*, 83: 136-149.

Hassan, M.A., Hogan, D.A., Bird, S.A., May, C.L., Gomi, T., & Campbell, D. Spatial and temporal dynamics of wood in headwater streams of the pacific northwest. *Journal of the American Water Resources Association*, *41*: 899-919.

Hassan, M.A., Church, M., Lisle, T.E., Bardinoni, F., Benda, L., & Grant, G.E. 2005. Sediment Transport and Channel Morphology of Small, Forested Streams. *Journal of the American Water Resources Association*, *41*: 853-876.

Hassan, M.A., Hogan, D.L., Bird, S.A., May, C.L., Gomi, T., & Campbell, D. 2005. Spatial and the temporal spatial and temporal dynamics of wood in headwater streams of the pacific northwest. *Journal of the American Water Resources Association*, *41*: 899-919.

Health Canada 2012a. Guidelines for Canadian Drinking Water Quality: Guideline Technical Document — Turbidity. Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario

Health Canada. 2012b. Guidelines for Canadian Recreational Water Quality, Third Edition. Water, Air and Climate Change Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario. (Catalogue No H129-15/2012E).

Health Canada. 2017. Guidelines for Canadian Drinking Water Quality- Summary Table. Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario.

Hecht, T., & Van der Lingen, C.D. 1992. Turbidity-induced changes in feeding strategies of fish in estuaries. *African Zoology*, 27: 95-107.

Heidel, S.G., 1956. The progressive lag of sediment concentration with flood waves. *Eos, Transactions American Geophysical Union*, *37*: 56-66.

Henley, W.F., Patterson, M.A., Neves, R.J., & Lemly, A.D. 2000. Effects of sedimentation and turbidity on lotic food webs: a concise review for natural resource managers. *Reviews in Fisheries Science*, 8: 125-139.

Henderson, G.S., & Golding, D.L. 1983. The effect of slash burning on the water repellency of forest soils at Vancouver, British Columbia. *Canadian Journal of Forest Research*, *13*: 353-355.

Hill, C., Furlong, K., Bakker, K., & Cohen, A. 2008. Harmonization versus subsidiarity in water governance: A review of water governance and legislation in the Canadian provinces and territories. *Canadian Water Resources Journal*, *33*: 315-332

Holden, J. (Ed.). 2005. An Introduction to Physical Geography and the Environment. Harlow, UK: Pearson Education

Hoover, T.M., Richardson, J.S., & Yonemitsu, N. 2006. Flow0substrate interactions create and mediate leaf littler resource patches in streams. *Freshwater Biology*, *51*: 435-447

Hoover, T.M., Marczak, L.B., Richardson, J.S., & Yonemitsu, N. 2010. Transport and settlement of organic matter in small streams. *Freshwater Biology*, 55: 436-449.

Hoover, T.M., Pinto, X., & Richardson, J.S. 2011. Riparian canopy type, management history, and successional stage control fluxes of plant litter to streams. *Canadian Journal of Forest Research*, *41*: 1394-1404.

Horton, R.E. 1932. Drainage-basin characteristics. *Eos, Transactions American Geophysical Union*, 13: 250-361

Horton, R.E. 1945. Erosional development of streams and their drainage basins: hydrophysical approach to quantitative morphology. *Bulletin of the Geological Society of America*, *56*: 275-350.

Houser, J.N., Mulholland, P.J. & Maloney, K.O., 2006. Upland disturbance affects headwater stream nutrients and suspended sediments during baseflow and stormflow. *Journal of Environmental Quality*, *35*: 352-365.

Hughes, A.O., Quinn, J.M. & McKergow, L.A., 2012. Land use influences on suspended sediment yields and event sediment dynamics within two headwater catchments, Waikato, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 46: 315-333.

Hrudey S.E., and Hrudey E.J. 2002. Walkerton & North Battleford—key lessons for public health professionals. *Canadian Journal of Public Health*. *93*: 332–3.

Hsieh, J.L., Nguyen, T.Q., Matte, T., and Ito, K. 2015. Drinking water turbidity and emergency department visits for gastrointestinal illness in New York City, 2002-2009. *PloS one*, *10*: e0125

Hudson, R.O. 2001. Interpreting turbidity and suspended-sediment measurements in high-energy streams in coastal British Columbia. [Research Section], Vancouver Forest Region, pp. 1-16

Iglesias, C., Torres, J.M., Nieto, P.G., Fernández, J.A., Muñiz, C.D., Piñeiro, J.I., & Taboada, J. 2014. Turbidity prediction in a river basin by using artificial neural networks: a case study in northern Spain. *Water Resources Management*, 28: 319-331.

James, W.F., & Barko, J.W. 1993. Sediment resuspension, redeposition, and focusing in a small dimictic reservoir. *Canadian Journal of Fisheries and Aquatic Sciences*, *50*: 1023-1028.

Johnston, C.A., Detenbeck, N.E., & Niemi, G.J. 1990. The cumulative effect of wetlands on stream water quality and quantity. A landscape approach. *Biogeochemistry*, *10*: 105-141.

Jones, K.B., Neale, A.C., Nash, M.S., Van Remortel, R.D., Wickham, J.D., Ritters, K.H., & O'Neill, R.V. 2001. Predicting nutrient and sediment loadings to streams from landscape metrics: a multiple watershed study from the United States Mid-Atlantic Region. *Landscape Ecology*, *16*: 301-312.

Jordan, P. 1996. Turbidity and suspended sediment measurements using OBS meters, West Arm Demonstration Forest sediment budget study. *Proceedings, The Second Automatic Water Quality Monitoring Workshop*, Richmond, B.C., Feb. 12-13. Environment Canada and British Columbia, Ministry of Environment, Lands and Parks, pp. 1-10

Jordan, P. 2006. The use of sediment budget concepts to assess the impact on watersheds of forestry operations in the southern interior of British Columbia. *Geomorphology*, *79*: 27-44.

Jungen, J.R. 1985. *Soils of southern Vancouver Island. BC Soil Survey Rep.* 44. MOE Tech. Rep. 17. BC Ministry of Environment, Victoria, BC.

Kaplan, L.A., & Bott, T.L. 1989. Diel fluctuations in bacterial activity on streambed substrata during vernal algal blooms: effects of temperature, water chemistry, and habitat. *Limnology and Oceanography*, 34: 718-733.

Kaplan, LA., & Newbold, D. 1995. Measurement of stream-water biodegradable dissolved organic carbon with a plug-flow bioreactor. *Water Resources*, 29: 2696-2706.

Keim, R.F., & Schoenholtz, S.H. 1999. Functions and effectiveness of silvicultural streamside management zones in loessial bluff forests. *Forest Ecology and Management*, *118*: 197-209.

Kiffney, P.M., Bull, J.P., & Feller, M.C. 2002. Climate and hydrologic variability in a coastal watershed if southwestern British Columbia. *Journal of the American Water Resources Association, 38*: 1437-1451.

Kiffney, P.M., Richardson, J.S., & Bull, J.P. 2003. Responses of periphyton and insects to experimental manipulation of riparian buffer width along forest streams. *Journal of Applied Ecology*, *40*: 1060-1076.

Kiffney, P.M., & Richardson, J.S. 2010. Organic matter inputs into headwater streams of southwestern British Columbia as a function of riparian reserves and time since harvesting. *Forest Ecology and Management*, 260: 1931-1942.

Klein, R.D., Lewis, J., & Buffleben, M.S. 2012. Logging and turbidity in the coastal watersheds of northern California. *Geomorphology*, *139*: 136-144.

Knighton, A.D. 1998. Fluvial Forms and Processes. New York, NY: Routledge

Kuglerová et al. (*in prep*). Importance of scale, land-use, and stream network properties for riparian plant communities along an urban gradient.

Kutner, M.H. 2005. Applied Linear Statistical Models. Boston, MA: McGraw-Hill Irwin

Lajtha, K. & Jones, J., 2018. Forest harvest legacies control dissolved organic carbon export in small watersheds, western Oregon. *Biogeochemistry*, *140*: 1-17.

Landers, M.N. & Sturm, T.W. 2013. Hysteresis in suspended sediment to turbidity relations due to changing particle size distributions. *Water Resources Research*, *49*: 5487-5500.

Langlois, J.L., Johnson, D.W., & Mehuys, G.R. 2005. Suspended sediment dynamics associated with snowmelt runoff in a small mountain stream of Lake Tahoe (Nevada). *Hydrological Processes: An International Journal, 19:* 3569-3580.

Leach, J.A., & Moore, R.D. 2015. Observations and modeling of hillslope throughflow temperatures in a coastal forested catchment. *Water Resources Research*, *51*: 3770-3795.

LeChevallier, M.W., Norton, W.D., & Lee, R.G. 1991. Occurrence of Giardia and Cryptosporidium spp. in surface water supplies. *Applied and Environmental Microbiology*, *57*: 2610-2616.

LeChevallier, M.W., & Norton, W.D. 1992. Examining relationships between particle counts and Giardia, Cryptosporidium, and turbidity. *Journal-American Water Works Association*, 84: 54-60.

Lenhart, C.F., Brooks, K.N., Heneley, D., & Magner, J.A. 2010. Spatial and temporal variation in suspended sediment, organic matter, and turbidity in a Minnesota prairie river: implications for TMDLs. *Environmental Monitoring and Assessment*, *165*: 435-447.

Leopold, L.B., Wolman, M.G., & Miller, J.P. 1964. *Fluvial Processes in Geomorphology*. New York, NY: Dover Publications Inc.

Lessels, J.S., & Bishop, T.F.A. 2013. Estimating water quality using linear mixed models with stream discharge and turbidity. *Journal of Hydrology*, 498: 13-22.

Lewis, J. 1996. Turbidity-controlled suspended sediment sampling for runoff-event load estimation. *Water Resources Research, 32:* 2299-2310.

Lewis, J., & Eads, R. 2001. Turbidity threshold sampling for suspended sediment load estimation. In *Pages III-110-III-117, in: Proceedings of the Seventh Federal Interagency Sedimentation Conference, 25-29 March 2001, Reno, Nevada. Federal Interagency Project, Technical Committee of the Subcommittee on Sedimentation.* 

Lieberman, J.A. & Hoover, M.D. 1948. The effect of uncontrolled logging on stream turbidity. *Water and Sewage Works*, 95: 255-258.

Likens, G.E., Bormann, F.H., & Johnson, N.M. 1969. Nitrification: importance to nutrient losses from a cutover forested ecosystem. *Science*, *163*: 1205-1206.

Lim G., Aramini J., Felury M., Ibarra R., & Meyers R. 2002. Investigating the relationship between drinking water and gastroenteritis in Edmonton: 1993–1998. In: Division of Enteric FaWD, editor. Ottawa: Health Canada

Lloyd, D.S. 1987. Turbidity as a water quality standard for salmonid habitats in Alaska. *North American Journal of Fisheries Management*, 7: 34-45.

Lloyd, C.E.M., Freer, J.E., Johnes, P.J., & Collins, A.L. 2016. Using hysteresis analysis of high-resolution water quality monitoring data, including uncertainty, to infer controls on nutrient and sediment transfer in catchments. *Science of the Total Environment*, *543*: 388-404.

Logsdon, G.S., Thurman, V.C., Frindt, E.S., & Stoecker, J.G. 1985. Evaluating sedimentation and various filter media for removal of Giardia cysts. *Journal-American Water Works Association*, 77: 61-66.

Lovell, B., McKelvie, I.D., & Nash, D. 2001. Sampling design for total and filterable reactive phosphorus monitoring in a lowland stream: considerations of spatial variability, measurement uncertainty and statistical power. *Journal of Environmental Monitoring*, 3: 463-468.

Lowe, M.L., Morrison, M.A., and Taylor, R.B. 2015. Harmful effects of sediment-induced turbidity on juvenile fish in estuaries. *Marine Ecology Progress Series*, *539*: 241-254.

Magner, J.A., & Brooks, K.N. 2008. Integrating sentinel watershed-systems into the monitoring and assessment of Minnesota's (USA) waters quality. *Environmental Monitoring and Assessment*, *138*, 149-158.

McCaleb, M.M. & McLaughlin, R.A. 2008. Sediment trapping by five different sediment detention devices on construction sites. *Transactions of the ASABE*, *51*: 1613-1621.

McCabe. G.D., & O'Brien, W.J. 1983. The effects of suspended silt on feeding and reproduction of *Daphnia pulex*. *American Midland Naturalist*, 110: 324-337.

Masese, F.O., Kitaka, N., Kipkemboi, J., Gettel, G.M., Irvine, K. & McClain, M.E., 2014. Litter processing and shredder distribution as indicators of riparian and catchment influences on ecological health of tropical streams. *Ecological Indicators*, *46*: 23-37.

Mather, A.L., & Johnson, R.L. 2015. Event-based prediction of stream turbidity using a combined cluster analysis and classification tree approach. *Journal of Hydrology*, *530*: 751-761.

McCaleb, M.M. & McLaughlin, R.A. 2008. Sediment trapping by five different sediment detention devices on construction sites. *Transactions of the ASABE*, *51*: 1613-1621.

Mermillod-Blondin, F., Creuzé des Châtelliers, M., Marmonier, P., & Dole-Olivier, M.J. 2000. Distribution of solutes, microbes and invertebrates in river sediments along a riffle-pool-riffle sequence. *Freshwater Biology*, *44*: 255-26

Métadier, M. & Bertrand-Krajewski, J.L. 2012. The use of long-term on-line turbidity measurements for the calculation of urban stormwater pollutant concentrations, loads, pollutographs and intra-event fluxes. *Water Research*, *46:* 6836-6856.

Metcalfe, C.D., Chu, S., Judt, C., Li, H., Oakes, K.D., Servos, M.R., & Andrews, D.M. 2010. Antidepressants and their metabolites in municipal wastewater, and downstream exposure in an urban watershed. *Environmental Toxicology and Chemistry*, 29: 79-89.

Millennium Ecosystem Assessment. 2005. *Ecosystems and Human Well-Being: Biodiversity Synthesis*, Washington, DC.: World Resources Institute

Miller, J.R., Sinclair, J.T., & Walsh, D. 2015. Controls on suspended sediment concentrations and turbidity within a reforested, Southern Appalachian headwater basin. *Water*, 7: 3123-3148.

Minnesota Pollution Control Agency. 2018. *Site stabilization: NPDES/SDS Construction Stormwater permit requirements. Available from:* <u>https://www.pca.state.mn.us/sites/default/files/wq-strm2-25.pdf</u> Accessed on Sept 1, 2018

Miquelajauregui, Y. 2008. Vegetation community development eight years after harvesting in small streams buffers at the Malcolm Knapp Research Forest (Doctoral dissertation, University of British Columbia)

Montgomery, D.R., & Buffinton, J.M. 1997. Channel-reach morphology in mountain drainage basins. *GSA Bulletin*, 109: 596-611.

Moore, R.D., & Richardson, J.S. 2003. Progress towards understanding the structure, function and ecological significance of small stream channels and their riparian zones. *Canadian Journal of Forest Research*, 33: 1349-1351.

Moran, C.J., Ward, D.L. & Gibb, A.C., 2018. Key Morphological Features Favor the Success of Nonnative Fish Species under Reduced Turbidity Conditions in the Lower Colorado River Basin. *Transactions of the American Fisheries Society*, *147:* 948-958.

Mukundan, R., Pradhanang, S.M., Schneiderman, E.M., Pierson, D.C., Anandhi, A., Zion, M.S., Matonse, A.H., Lounsbury, D.G. & Steenhuis, T.S., 2013. Suspended sediment source areas and future climate impact on soil erosion and sediment yield in a New York City water supply watershed, USA. *Geomorphology*, *183*: 110-119
Naiman, R.J., Bechtold, J.S., Drake, D.C., Latterell, J.J., O'keefe, T.C., & Balian, E.V. 2005. Origins, patterns, and importance of heterogeneity in riparian systems. In Weather, K.C. (Eds) *Ecosystem Function in Heterogeneous Landscapes*. New York, NY: Springer, pp. 279-309.

Nasrabadi, T., Ruegner, H., Sirdari, Z.Z., Schwientek, M., & Grathwohl, P. 2016. Using total suspended solids (TSS) and turbidity as proxies for evaluation of metal transport in river water. *Applied Geochemistry*, 68: 1-9.

Natural Resources Canada. 2013. Canadian Digital Surface Model Product Specifications, Edition 1.1. GeoGratis Client Services.

Newcombe, C.P., and MacDonald, D.D. 1991. Effects of suspended sediments on aquatic ecosystems. *National American Journal of Fisheries Management*, *11*: 72–82.

O'Dell, J.W. 1993. Method 180.1 Determination of Turbidity by Nephelometry. United Sates Environmental Protection Agency. Office of Research and Development. Cincinnati, Ohio: U.S. Environmental Protection Agency

Olive, L.J., & Rieger, W.A. 1988. An examination of the role of sampling strategies in the study of suspended sediment transport. *IN:* Bordas, M.P., and Walling, D.E. *Sediment Budgets*. Wallingford, UK: *IAHS Publication*, pp. 174

Ormerod, S.J., Dobson, M., Hildrew, A.G., & Townsend, C. 2010. Multiple stressors in freshwater ecosystems. *Freshwater Biology*, 55: 1-4.

Osei, N.A., Gurnell, A.M. & Harvey, G.L., 2015. The role of large wood in retaining fine sediment, organic matter and plant propagules in a small, single-thread forest river. *Geomorphology*, 235: 77-87.

Palmer, S.M., Hope, D., Billett, M.F., Dawson, J.J., & Bryant, C.L. 2001. Sources of organic and inorganic carbon in a headwater stream: evidence from carbon isotope studies. *Biogeochemistry*, 52: 321-338.

Palmer, M.A., & Richardson, D.C. 2009. Provisioning services: a focus on freshwater. p 625. In Levin, S.A., Carpenter, S.R., Godfray, H.C., Kinzig, A.O., Loreau, M., Losos, J.B., Walker, B., and Wilcove, D.S. (Eds). *The Princeton Guide to Ecology*. Princton, NJ: Princeton University Press

Paustian, S.J. & Beschta, R.L. 1979. The suspended sediment regime of an Oregon coast range stream. *Journal of the American Water Resources Association*, *15*: 144-154.

Payment, P., Trudel, M., Satar, S.A., Sprinthorpe, V.S., Subrahmanyan, T.P., Gregory, B.E., Vajdic, A.H., Blaskovic, P., Guglielmi, I.J., & Kudfrewko, O. 1984. Virological examination of drinking water: a Canadian collaborative study. *Canadian Journal of Microbiology*, 30: 104-112.

Penna, D., Borga, M.A. Sangati, M.A., & Gobbi, A.L. 2010. Dynamics of soil moisture, subsurface flow and runoff in a small alpine basin. *IAHS Publications, Red Book Series*, *336*: 96-102

Perkin, E.K., Hölker, F., Richardson, J.S., Sadler, J. P., Wolter, C., & Tockner, K. 2011. The influence of artificial light on stream and riparian ecosystems: questions, challenges, and perspectives. *Ecosphere*, 2: 1-16.

Pettorelli, N., Vik, J.O., Mysterud, A., Gaillard, J.M., Tucker, C.J., & Stenseth, N.C. 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. *Trends in Ecology & Evolution*, 20: 503-510.

Pfaff, B. 2008. *Analysis of integrated and cointegrated time series with R*. New York, NY: Springer Science and Business Media.

Pfannkuche, J. & Schmidt, A., 2003. Determination of suspended particulate matter concentration from turbidity measurements: particle size effects and calibration procedures. *Hydrological Processes*, *17*: 1951-1963.

Planet Labs Inc. 2018. Planet World Imagery. Licensed under Creative Commons BY NC SA 4.0 International License on <u>www.planet.com</u>

Pons, W., Young, I., Truong, J., Jones-Bitton, A., McEwen, S., Pintar, K., & Papadopoulos, A. 2015. A systematic review of waterborne disease outbreaks associated with small non-community drinking water systems in Canada and the United States. *PLoS One*, *10*: e0141646.

Postel, S., & Carpenter, S. 1997. Freshwater ecosystem services p. *195*. In Daily, G.C. (Eds). *Nature's services: Societal dependence on natural ecosystems*. Washington, DC: Island Press.

Public Health and Safety Act. 2007. SYT c. 176

Pozo, J., González, E., Díez, J.R., Molinero, J., & Elósegui, A. 1997. Inputs of particulate organic matter to streams with different riparian vegetation. *Journal of the North American Benthological Society*, *16*: 602-611.

Quinn, J.M., Cooper, A.B., Davies-Colley, R.J., Rutherford, J.C., & Williamson, R.B. 1997. Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. *New Zealand Journal of Marine and Freshwater Research*, 31: 579-597.

R Studio Team. 2015. RStudio: Integrated Development for R. RStudio, Inc., Boston, MA URL <u>http://www.rstudio.com/</u>.

Rasmussen, E.K., Petersen, O.S., Thompson, J.R., Flower, R.J., & Ahmed, M.H. 2009. Hydrodynamic-ecological model analyses of the water quality of Lake Manzala (Nile Delta, Northern Egypt). *Hydrobiologia*, 622: 195.

Raymond, P.A., & Saiers, J.E. 2010. Event controlled DOC export from forested watersheds. *Biogeochemistry*, *100*, 197-209.

Reid, D.A., Hassan, M.A., & Floyd, W. 2016. Reach-scale contributions of road-surface sediment to the Honna River, Haida Gwaii, BC. *Hydrological Processes*, *30*: 3450-3465.

Richardson, D.M., Holmes, P.M., Esler, K.J., Galatowitsch, S.M., Stromberg, J.C., Kirkman, S.P., Pyšek, P. & Hobbs, R.J., 2007. Riparian vegetation: degradation, alien plant invasions, and restoration prospects. *Diversity and Distributions*, *13*: 126-139.

Rieberger, R. 2007. Water quality assessment and objectives for Shawnigan Lake: overview report. Prepared in pursuant to Section 5(e) of the Environment Management Act, 2003. ISBN 978-0-7726-5848-7

Rieberger, R, Epps, D., & Wilson, J. 2004. *Shawnigan Lake water quality assessment, 1976-2004.* ISBN 0-7726-5269-4

Riedel, M.S., Vose, J.M. & Bolstad, P.V., 2004. Characterizing hysteretic water quality in southern Appalachian streams. In *In Proc. 2004 National Water Quality Monitoring Conference, United States Advisory Committee on Water Information–National Water Quality Monitoring Council, Chattanooga, TN, May 17th–20rd.* 

Reiter, M., Heffner, J.T., Beech, S., Turner, T., & Bilby, R.E. 2009. Temporal and Spatial Turbidity Patterns Over 30 Years in a Managed Forest of Western Washington 1. *Journal of the American Water Resources Association*, *45*: 793-808.

Riparian Areas Protection Act. 1997. SBC C. 21

Rivenbark, B.L., & Jackson, C.R. 2004. Concentrated flow breakthroughs moving through silviculture streamside management zones: Southeastern Piedmont, USA. *Journal of the American Water Resources Association*, 40: 1043-1052.

Roberston, D.M., & Roerish, E.D. 1999. Influence of various water quality sampling strategies on load estimates for small streams. *Water Resources Research*, 35: 3747-3759.

Robinson, S. E., Capper, N. A., and Klaine, S. J. 2010. The effects of continuous and pulsed exposures of suspended clay on the survival, growth, and reproduction of Daphnia magna. *Environmental Toxicology and Chemistry*, *29*: 168-175.

Rosi-Marshall, E.J., Snow, D., Bartelt-Hunt, S.L., Paspalof, A., & Tank, J.L. 2015. A review of ecological effects and environmental fate of illicit drugs in aquatic ecosystems. *Journal of Hazardous Materials*, 282: 18-25.

Roy, A.H., Rosemond, A.D., Paul, M.J., Leigh, D.S., & Wallace, J.B. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, USA). *Freshwater Biology*, *48*: 329-346.

Rugner, H., Schwientek, M., Beckingham, B., Bertram, K, & Grathwohl, P. 2012. Turbidity as a proxy for total suspended solids (TSS) and particle facilitated pollutant transport in catchments. *Environmental Earth Science*, *69*: 373-380.

Ryan, P.A. 1991. Environmental effects of sediment on New Zealand streams: a review. *New Zealand Journal of Marine and Freshwater Research*, 25: 207-221.

Sadar, M J. 1998. Turbidity Science. Hach Company.

Sadar, M.J. 1999. Turbidimeter instrument comparison: low-level sample measurements. *Hach Technical Information Series*, pp. 1- 55.

Sadar, M.J. 2002. Turbidimeter Instrumentation Comparison: Low-level Sample Measurements. *Hach Technical Information Series*, pp. 7-10

Saraceno, J.F., Shanley, J.B., Downing, B.D., & Pellerin, B.A. 2017. Clearing the waters: Evaluating the need for site-specific field fluorescence corrections based on turbidity measurements. *Limnology and Oceanography: Methods*, *15*: 408-416.

Scrivener, J.C., & Brownlee, M.J. 1989. Effects of forest harvesting on spawning gravel and incubation survival of chum (Oncorhynchus keta) and coho salmon (O. kisutch) in Carnation Creek, British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences*, 46: 681-696.

Seeger, M., Errea, M.P., Begueria, S., Arnáez, J., Marti, C. & Garcia-Ruiz, J.M. 2004. Catchment soil moisture and rainfall characteristics as determinant factors for discharge/suspended sediment hysteretic loops in a small headwater catchment in the Spanish Pyrenees. *Journal of Hydrology*, 288: 299-311. Seehausen, O., Van Alphen, J.J. & Witte, F., 1997. Cichlid fish diversity threatened by eutrophication that curbs sexual selection. *Science*, *277*: 1808-1811.

Serveiss, V.B., and Ohlson, D.W. 2007. Using ecological risk assessment principles in a source water protection assessment. *Human and Ecological Risk Assessment*, *13*: 402-417.

Shoup, D.E., and Drew Lane, W. 2015. Effects of turbidity on prey selection and foraging return of adult largemouth bass in reservoirs. *North American Journal of Fisheries Management*, *35*: 913-924.

Silva, L., & Williams, D.D. 2001. Buffer zone versus whole catchment approaches to studying land use impact on river water quality. *Water Research*, *35*: 3462-3472.

Simon, A. 1989. The discharge of sediment in channelized alluvial streams. *Journal of the American Water Resources Association*, 25: 1177-1188.

Slaets, J.I., Schmitter, P., Hilger, T., Lamers, M., Piepho, H.P., Vien, T.D. & Cadisch, G., 2014. A turbidity-based method to continuously monitor sediment, carbon and nitrogen flows in mountainous watersheds. *Journal of Hydrology*, *513*: 45-57.

Smith, B.P.G., Naden, P.S., Leeks, G.J.L. & Wass, P.D., 2003. The influence of storm events on fine sediment transport, erosion and deposition within a reach of the River Swale, Yorkshire, UK. *Science of the Total Environment*, *314*: 451-474.

Soler, M., Nord, G., Catari, G., & Gallart, F. 2012. Assessment of suspended sediment concentration measurement error in relation to particle size, using continuous sensors in a small mountain stream (Vallcebre catchments, Eastern Pyrenees). *Zeitschrift für Geomorphologie, Supplementary Issues*, *56*: 99-113.

Speaker, R., Moore, K., & Gregory, S. 1984. Analysis of the process of retention of organic matter in stream ecosystems: With 3 figures and 1 table in the text. *Internationale Vereinigung für theoretische und angewandte Limnologie: Verhandlungen*, 22: 1835-1841.

St. Pierre, K., Levesque, S., Frost, E., Carrier, N., Arbeit, R.D., & Michaud, S. 2009. Thermotolerant coliforms are not a good surrogate for *Campylobacter* spp. in environmental water. *Applied Environmental Microbiology*, *75:* 6736-6744.

Statistics Canada. 2017. *Cowichan Valley, RD [Census division], British Columbia and Saskatchewan [Province]* (table). *Census Profile*. 2016 Census. Statistics Canada Catalogue no. 98-316-X2016001. Ottawa. Released November 29, 2017. <u>https://www12.statcan.gc.ca/census-recensement/2016/dp-pd/prof/index.cfm?Lang=E</u> accessed May 17, 2017). Suh, D., 2016. The Impact of Turbidity on the Predator-Prey Relationship between Red Swamp Crayfish (Procambarus clarkii) and Pacific Tree Frog (Pseudacris regilla) Tadpoles.

Sullivan, K. 1985. Long-term patterns of water quality in a managed watershed in Oregon: Suspended sediment. *Journal of the American Water Resources Association*, *21*: 977-987.

Summerscales, I.M., & McBean, E.A. 2011. Incorporation of the Multiple Barrier Approach in drinking water risk assessment tools. *Journal of Water and Health*, 9: 349-360.

Sutherland, A., Meyer, J., & Gardiner, E. 2002. Effects of land cover on sediment regime and fish assemblage structure in four southern Appalachian streams. *Freshwater Biology* 47: 1791-1805

Sweeney, B.W., & Newbold, J.D. 2014. Streamside forest buffer width needed to protect stream water quality, habitat and organisms: a literature review. *Journal of the American Water Resources Association*, 50: 560-584.

Tananaev, N.I. & Debolskiy, M.V., 2014. Turbidity observations in sediment flux studies: Examples from Russian rivers in cold environments. *Geomorphology*, *218*: 63-71.

Tank, J.L., Rosi-Marshall, E.J., Griffiths, N.A., Entrekin, S.A., & Stephen, M.L. 2010). A review of allochthonous organic matter dynamics and metabolism in streams. *Journal of the North American Benthological Society*, 29: 118-146.

Tilman, D. 1999. Global environmental impacts of agricultural expansion: the need for sustainable and efficient practices. *Proceedings of the National Academy of Sciences, 96:* 5995-6000.

Tinker, S.C., Moe, C.L., Klein, M., Flanders, W.D., Uber, J., Amirtharajah, A., Singer, P. & Tolbert, P.E., 2010. Drinking water turbidity and emergency department visits for gastrointestinal illness in Atlanta, 1993–2004. *Journal of Exposure Science and Environmental Epidemiology*, *20*, 19-28.

Tong, S.T., & Chen, W. 2002. Modeling the relationship between land use and surface water quality. *Journal of Environmental Management*, 66: 377-393

Townsend, C., Arbuckle, C., Crowl, T., & Scarsbrook, M. 1997. The relationship between land use and physicochemistry, food resources and macroinvertebrate communities in tributaries of the Taieri River, New Zealand: a hierarchically scaled approach. *Freshwater Biology*, *37*: 177-191.

Trenouth, W.R., Gharabaghi, B., MacMillan, G., & Bradford, A. 2013. Better management of construction sites to protect inland waters. *Inland Waters*, *3*: 167-178.

Tyc, G., Tulip, J., Schulten, D., Krischke, M., & Oxfort, M. 2005. The RapidEye mission design. *Acta Astronautica*, *56*: 213-219.

Uhrich, M.A., & Bragg, H.M. 2003. Monitoring instream turbidity to estimate continuous suspended-sediment loads and yields and clay-water volumes in the Upper North Santiam River Basin, Oregon, 1998-2000. U.S. Geological Survey Water-Resources Investigations Report 03-4098, pp. 43.

Utne-Palm, A.C., 1999. The effect of prey mobility, prey contrast, turbidity and spectral composition on the reaction distance of Gobiusculus flavescens to its planktonic prey. *Journal of Fish Biology*, *54*: 1244-1258.

Van de Hulst, H.C. 1957. *Light Scattering from Small Particles*. New York, NY: John Wiley and Sons Inc.

Vancouver Island Health Authority. 2018. Water Samples. Available from www.healthspace.ca/Clients/VIHA/VIHA\_Website.nsf/ Accessed August 20, 2018

Wagner, R.J., Boulger Jr, R.W., Oblinger, C.J., & Smith, B.A. 2006. *Guidelines and standard procedures for continuous water-quality monitors: station operation, record computation, and data reporting* (No. 1-D3).

Wallace, J.B., Whiles, M.R., Eggert, S., Cuffney, T.F., Lugthart, G.J., & Chung, K. 1995. Longterm dynamics of coarse particulate organic matter in three Appalachian Mountain streams. *Journal of the North American Benthological Society*, *14*: 217-232.

Walling, D.E. & Webb, B.W., 1981. The reliability of suspended sediment load data. *Erosion and Sediment Transport Measurement* (Proceedings of the Florence Symposium, June 1981). IAHS 133.

Ward, D.L., Morton-Starner, R., & Vaage, B. 2016. Effects of turbidity on predation vulnerability of juvenile humpback chub to rainbow trout and brown trout. *Journal of Fish and Wildlife Management*, 7: 205-212.

Washington State Department of Ecology. 2017. Water Quality Standards for Surface Waters of the State of Washington. Chapter 173-201A WAC

Weaver, A., & Wiebe, E. 2018. School-Based Weather Station Network. Licensed under Creative Commons BY NC SA 4.0 International License on <u>www.victoriaweather.ca/</u>

Webster, M.M., Atton, N., Ward, A.J.W. & Hart, P.J.B., 2007. Turbidity and foraging rate in threespine sticklebacks: the importance of visual and chemical prey cues. *Dynamics*, *144*: 1347-1360.

Webster, J.R. & Benfield, E.F., 1986. Vascular plant breakdown in freshwater ecosystems. *Annual Review of Ecology and Systematics*, 17: 567-594.

Webster, J.R., Benfield, E.F., Ehrman, T.P., Schaeffer, M.A., Tank, J.L., Hutchens, J.J. & D'angelo, D.J., 1999. What happens to allochthonous material that falls into streams? A synthesis of new and published information from Coweeta. *Freshwater Biology*, *41*: 687-705.

Webster, J.R., & Meyer, J.L. 1997. Organic matter budgets for streams: a synthesis. *Journal of the North American Benthological Society*, *16*: 141-161.

Webster, J.R., Wallace, J.B., and Benfield, E.F. 1995. Organic processes in streams of the eastern United States. Pp. 117-187. In Cushing, C.E., Cummins, K.W., & Minshall, G.W. *River and Stream Ecosystems of the World*. Los Angeles, CA: University of California Press.

Whitehead, P.G., Wilby, R.L., Battarbee, R.W., Kernan, M., & Wade, A.J. 2009. A review of the potential impacts of climate change on surface water quality. *Hydrological Sciences Journal*, 54: 101-123

WHO. 2017a. Guidelines for drinking-water quality, 4<sup>th</sup> edition, incorporating the 1<sup>st</sup> addendum. CC BY-NC-SA 3.0 IGO

WHO. 2017b. Water Quality and Health: Review of Turbidity, Information for Regulators and Water Suppliers. Technical Brief 17.01

Williams, G.P., 1989. Sediment concentration versus water discharge during single hydrologic events in rivers. *Journal of Hydrology*, *111*: 89-106.

Wipfli, M.S., & Richardson, J.S. 2015. Riparian management and the conservation of stream ecosystems and fishes. Pp. 270-291. In Closs, G.P., Krkosek, M., & Olden, J.D. (Eds) *Conservation of Freshwater Fishes*. Cambridge, UK: Cambridge University Press

Wipfli, M.S., Richardson, J.S., & Naiman, R.J. 2007. Ecological Linkages Between Headwaters and Downstream Ecosystems: Transport of Organic Matter, Invertebrates, and Wood Down Headwater Channels 1. *Journal of the American Water Resources Association*, *43*: 72-85.

Wolman, M. G., & Miller, J. P. 1960. Magnitude and frequency of forces in geomorphic process. *Journal of Geology*, 68: 54-74.

Worley Parsons Consulting Ltd. 2009. South Cowichan Water Plan Study: A preliminary assessment of water supply and needs within the South Cowichan Region, pp. 1-30

Wotton, R.S. & Malmqvist, B., 2001. Feces in Aquatic Ecosystems: Feeding animals transform organic matter into fecal pellets, which sink or are transported horizontally by currents; these fluxes relocate organic matter in aquatic ecosystems. *AIBS Bulletin*, *51*: 537-544.

Yao, M, Nan, J., & Chen, T. 2014. Effect of particle size distribution on turbidity under various water quality levels during flocculation processes. *Desalination*, 354: 116-124.

Zabaleta, A., Martínez, M., Uriarte, J.A. & Antigüedad, I., 2007. Factors controlling suspended sediment yield during runoff events in small headwater catchments of the Basque Country. *Catena*, *71*: 179-190.

Ziegler, A.D., Benner, S.G., Tantasirin, C., Wood, S.H., Sutherland, R.A., Sidle, R.C., Jachowski, N., Nullet, M.A., Xi, L.X., Snidvongs, A. & Giambelluca, T.W., 2014. Turbiditybased sediment monitoring in northern Thailand: Hysteresis, variability, and uncertainty. *Journal of Hydrology*, *519*: 2020-2039.

Zsolnay, A. 2003. Dissolved organic matter: artefacts, definitions, and functions. *Geoderma*, *113*: 187-209.

Zuur, A., Ieno, E.N., Savaliev, A.A., & Smith, G.M. 2009. *Mixed Effects Models and Extensions in Ecology with R.* New York, NY: Springer-Verlag

## Appendix





**Figure A2:** Hysteresis diagram from storms with high discharge over the 2016 year from three creeks in the Malcolm Knapp Research Forest. Graph is between turbidity and discharge (Q). Arrows show hysteresis loop direction.



**Figure A3:** Hysteresis diagram from storms with high turbidity over the 2016 year from three creeks in the Malcolm Knapp Research Forest. Graph is between turbidity and discharge (Q). Arrows show hysteresis loop direction.