Contemporary Forest Harvesting Impacts on Drinking Water Quality and Treatability

by

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AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Forested watersheds supply ~75% of global accessible freshwater resources and serve as important sources of drinking water. Both natural and anthropogenic landscape disturbances in these regions can deteriorate water quality in downstream environments. Climate change-exacerbated increases in the frequency and severity of disturbances such as wildfires, floods, and hurricanes threaten global drinking water security because the deterioration and/or increased variability of drinking water source quality can challenge treatment plants beyond their design and/or operational capacity, resulting in increased infrastructure and operating costs, servies disruptions or potentially catatrophic service outages. The water quality impacts of such events can last for decades; thus, risk reduction startegies must be developed to ensure the uninterrupted provision of adequate amounts of safe drinking water.

Althgouh forests have not been historically managed for water, forest harvesting has been proposed for pre-emptive mitigation of severe natural disturbance effects on source water quality and treatability; Depending on how it is implemented, however, it can also deteriorate source quality. Critically, the impacts of forest harvesting on drinking water treatability have not been investigated. Thus, the focus of this research was to investigate the impacts of contemporary forest harvesting on drinking water source quality and treatability. Three types of contemporary forest harvesting (clear-cut with patch retention, strip-shelterwood cut, and partial cut) were investigated at the watershed-scale over a range of flow conditions in the eastern slopes of the Rocky Mountains of southwestern Alberta, Canada. Reference and harvested stream water turbidity and various water quality metrics related to NOM (and associated DBP formation potentials) were evaluated during and over the first three post-harvest years. The correlations between THM- and HAA-FPs and several proxy indicators (particularly, aromaticity) were also investigated. Reported pre-disturbance data from the study watersheds were included in this analysis. Notably, no appreciable impacts of forest harvesting on water quality and treatability were observed during the harvest and three post-harvest years. Thus, contemporary forest harvesting approaches coupled with state-of-the-art BMPs for erosion control show promise as “green” source water protection (SWP) technologies for mitigating severe disturbance risks to drinking water treatability, even in source water regions rich with glacially-derived fine sediments (e.g., many parts of western North America). To maximize the impacts of forest management-based approaches to SWP and to develop climate change adaptation strategies—in lieu of traditional landscape-level, time series trend monitoring—focused paired catchment investigations that are designed as before-after-control-impact (BACI) studies are urgently needed.

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Dedication

To Kayhan; hoping when you are old enough to read this we have already overcome the climate change battle.

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List of Abbreviations

AB	Alberta
AESRD	Alberta Environment & Sustainable Resource Development
AO	Aesthetic Objectives
APHA	American Public Health Association
ASL	Above Sea Level
AWWA	American Water Works Association
BC	Britich Columbia
BDCM	Bromodichromethane
BMPs	Best Management Practices
CA	California
CCME	Canadian Council of Ministers of the Environment
CDC	Centers for disease control and prevention
DBCM	Dibromochloromethane
DBP	Disinfection By-Products
DOC	Dissolved Organic Carbon
DWI	Drinking Water Inspectorate
FAO	Food and Agriculture Organization
HAA	Haloacetic Acids
IC	Inorganic Carbon
IARC	International Agency for Research on Cancer
IPCC	Intergovernmental Panel on Climate Change

JAMA	The Journal of the American Medical Association
LC-OCD	Liquid Chromatography- Organic Carbon Detention
LMW	Low Molecular Weight
MA	Massachusetts
MAC	Maximum Acceptable Concentration
MB	Manitoba
MCL	Maximum Contaminant Level
MS	Mississippi
NB	New Brunswick
NDMA	N-Nitrosodimethylamine
NHMRC	National Health and Medical Research Council
NL	Newfoundland and Labrador
NOM	Natural Organic Matter
NPOC	Non-Purgeable Organic Carbon
NS	Nova Scotia
NTU	Nephelometric Turbidity Units
NV	Nevada
NWT CIMP	The Northwest Territories Cumulative Impact Monitoring Program
NY	New York
OCD	Organic Carbon Detector
OG	Operational Guidance
ON	Ontario
OND	Organic Nitrogen Detector
PEI	Prince Edward Island

POC	Particulate Organic Carbon
QC	Quebec
SD	Standard Deviation
SEC	Size-Exclusion Chromatography
SK	Saskatchewan
SUVA	Specific Ultraviolet Absorbance
SWP	Source Water Protection
THM	Trihalomethanes
TOC	Total Organic Carbon
UNFF	United Nations Forum on Forests
US	United States
USEPA	U.S. Environmental Protection Agency
USGCRP	U.S. Global Change Research Program
UV254	Ultraviolet Absorbance at the Wavelength of 254 nm
UVA	Ultraviolet Absorbance
UVD	Ultraviolet Absorbance at the Wavelength of 254 nm Detector
WHO	World Health Organization

1 Introduction

1.1 Research Motivation

Forested watersheds supply ~75% of global accessible freshwater resources and serve as important sources of drinking water (UNFF, 2016; FAO, 2018). In North America, over two-thirds of drinking water supplies originate in forested watersheds (Stein & Butler, 2004; Natural Resources Canada, 2015). Both natural and anthropogenic landscape disturbances in these regions can deteriorate the quality of water in downstream environments, however (Janetos *et al.*, 1997; Christensen *et al.*, 2004; Huntington *et al.*, 2009; Whitehead *et al.*, 2009; Watts *et al*., 2015). Deterioration and/or increased variability of source water quality can challenge treatment plants beyond their design and/or operational capacity and consequently increase the cost of drinking water treatment (Emelko *et al*., 2011; Emelko and Sham, 2014; Emelko *et al*., 2016).

Natural organic matter (NOM) is a complex mixture of organic compounds that affects particle (including pathogen) charge in natural systems (Yun *et al.*, 2011). It can affect the taste, odor, and color of water (Greeson, 1981), as well as its reactivity and amenability to treatment (Aiken *et al.*, 1996; AWWA, 1999). Dissolved organic carbon (DOC) is the most widely used indicator of aqueous NOM. Landscape disturbances can lead to increases in the concentration of receiving stream DOC, and change its aromaticity and structure. These changes can challenge drinking water treatment by driving coagulant dose, leading to membrane fouling, exerting oxidant demand, and promoting microbial instability/proliferation in distribution systems (Gallard *et al.*, 2002; Westerhoff *et al.,* 2004; Zularisam *et al.*, 2006; Guo *et al.,* 2012; Sillanpää *et al.*, 2018). Moreover, DOC can react with disinfectants (e.g., chlorine, chloramine, chlorine dioxide, ozone) to form disinfection by-products (DBPs), some of which are probable carcinogens (Edzwald *et al.*, 1985; Kitis *et al.*, 2002; Anderson *et al.*, 2010). Consequently, periodically elevated or more frequently fluctuating concentrations and character of DOC resulting from landscape disturbances can result in the need to use more advanced and expensive treatment technologies to effectively treat water (Emelko *et al*., 2011).

Due to climate change, the average global surface temperature has risen by ~1°C (±0.2°C) since the 19th century and continues to rise ~0.2°C (±0.1°C) every 10 years (IPCC, 2018). This level of temperature change is enough to alter the hydrological cycle and affect the frequency and intensity of extreme natural events, such as wildfires, floods, and hurricanes (Dale *et al*., 2001; Schelhaas *et al.*,

2003; Seidl *et al*., 2017). Notably, the profound impacts of such events on water quality can last for decades and lead to significant long-term drinking water treatment challenges (Emelko *et al.*, 2016). The potential for increased surface water quality variability and the challenges it can pose for drinking water treatment necessitates the development and implementation of risk reduction startegies to ensure the uninterrupted provision of adequate amounts of safe drinking water (Emelko *et al.*, 2011; Bladon *et al.*, 2014; Khan *et al.*, 2015).

Forest harvesting has been proposed as one approach for pre-emptive mitigation of landscape disturbance effects on source water quality and treatability in some jurisdictions (Larsen, 1995; Drever *et al.*, 2006; Millar *et al.*, 2007). While forest management-based approaches can be effective in mitigating the severity of natural disturbance effects on the landscape (Nakamura, 1996; Stephans & Moghaddas, 2005; Waltz *et al*., 2014), some of these approaches, such as forest harvesting, also can lead to deteriorated water quality (Nitschke, 2005; Rodríguez & Kouki, 2015)—this is not necessarily surprising because some types of harvesting are specifically designed to emulate natural disturbance regimes on the landscape, to maintain forest structure and function by stimulating regeneration and growth (Nitschke, 2005; Rodríguez & Kouki, 2015). Notably, forests have not been historically managed for water; although there is interest in managing them to mitigate risks to drinking water supplies, the impacts of forest harvesting on drinking water treatability have not been investigated.

1.2 Research Objectives

The overall goal of this research was to evaluate the impacts of contemporary forest harvesting on water quality and drinking water treatability. The specific objectives of this investigation were to:

1. Evaluate the impacts of contemporary forest harvesting on drinking water source quality and treatability,
2. Compare the impacts of several contemporary forest harvesting (clear cut, strip-shelterwood cut, and partial cut) strategies on drinking water source quality and treatability,
3. Identify the NOM concentration and/or characterization metrics that are the best proxy indicators for evaluating drinking water treatability impacts of forest harvesting, and
4. Assess the potential application of contemporary forest harvesting as a source water protection (SWP) technology to mitigate climate change-exacerbated, disturbance-associated threats to drinking water treatability.

1.3 General Research Approach

Contemporary forest harvesting impacts on drinking water source quality and treatability were investigated at the watershed-scale in the eastern slopes of the Rocky Mountains of southwestern Alberta. This work was conducted as part of an ongoing watershed-scale study of wildfire, post-fire salvage logging, and contemporary forest harvesting impacts on hydrology, water quality, and aquatic ecology (i.e., the Southern Rockies Watershed Project [SRWP]), which commenced in 2004 (Silins *et al.*, 2016). The SRWP team studied the undisturbed Star and York Creek watersheds as part of that study. The present investigation began in 2015, when three types of contemporary forest harvesting (clear-cut with patch retention, strip-shelterwood cut, and partial cut) were applied in the previously undisturbed watersheds. Specifically, one approach was implemented in each of three sub-watersheds of Star Creek in 2015. York Creek was not harvested and served as a reference watershed for comparison. During harvesting, best management practices (BMPs) to minimize surface erosion/runoff were utilized. Reference and harvested stream water turbidity and various water quality metrics related to NOM (and associated DBP formation potentials) were evaluated during and over the first three years after forest harvesting. To the extent possible, additional pre-disturbance data (i.e., collected prior to harvesting and reported in the literature) from the study watersheds also were included in this analysis.

1.4 Thesis Organization

This thesis consists of five chapters. Chapter 2 provides a review of landscape disturbance impacts on water quality. Key aspects of water quality and their relationship to drinking water treatability are discussed. The chapter concludes with an overview of source water protection strategies and research needs. Chapter 3 contains a description of the study area, laboratory methods, and approaches for statistical analysis of the data evaluated in this research. The results and discussion are presented in Chapter 4. Chapter 5 includes the conclusions, implications, and recommendations drawn from this research. A list of cited references follows. Appendix A contains the detailed statistical analyses conducted during this study.

2 Literature Review

2.1 Landscape Disturbance Impacts on Water Quality

Forested watersheds supply ~75% of global freshwater supplies (UNFF, 2016). Healthy forests are critical sources of drinking water because they grow in regions with high annual precipitation, produce large quantities of runoff with relatively low contaminant concentrations, and store large volumes of water (Gartner *et al.*, 2014). The value of natural storage and filtration of water by global forests has been estimated at $4.1 trillion (US) (Costanza *et al*., 2014). The forest canopy, floor vegetation, and root systems all contribute to regulating soil erosion and reducing sediment transport to receiving waters; they also promote infiltration and groundwater storage. These natural treatment processes frequently produce high quality water (Figure 2-1). Thus, maintaining the health of forested watersheds is critical to minimizing drinking water treatment costs (Ernst, 2004; Freeman *et al*., 2008). Ernst (2004) found that the cost of drinking water treatment decreases by almost 20% for every 10% increase in forest cover, up to 60% cover (Figure 2-2). As might be expected, the removal of land cover can directly impact the quantity and quality of water in forested watersheds (Kirshen *et al*., 2008; Butman & Raymond, 2011; Evans *et al*., 2012). Climate change-exacerbated landscape disturbances, such as wildfires and hurricanes, can further threaten surface water quality in these environments (Bladon *et al*., 2014; IPCC, 2018).

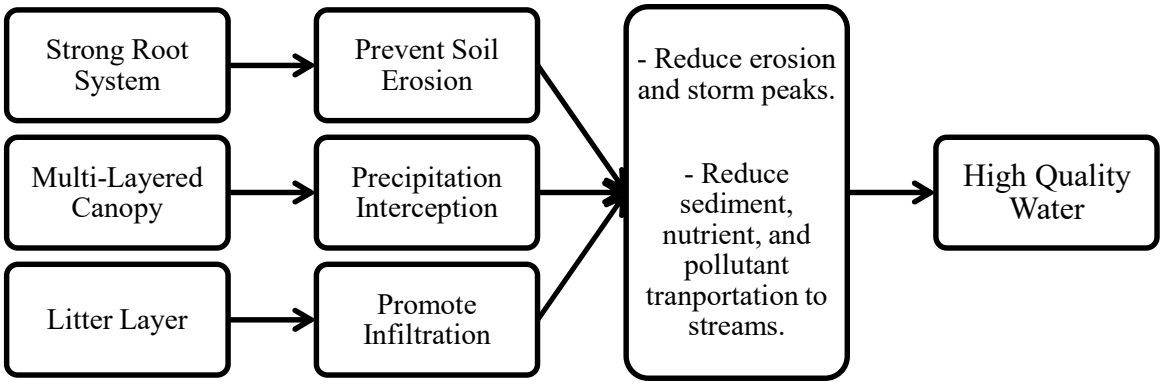

Figure 2-1 The role of forests as natural water treatment infrastructure (Adapted from: Briggs & Smithson, 1985; Kennedy *et al*., 1987; Beeson & Doyle, 1995; Brooks *et al*., 2003; Dudley & Stolton, 2003; de la Crétaz & Barten, 2007; Smith *et al*., 2011; Menese *et al*., 2015).

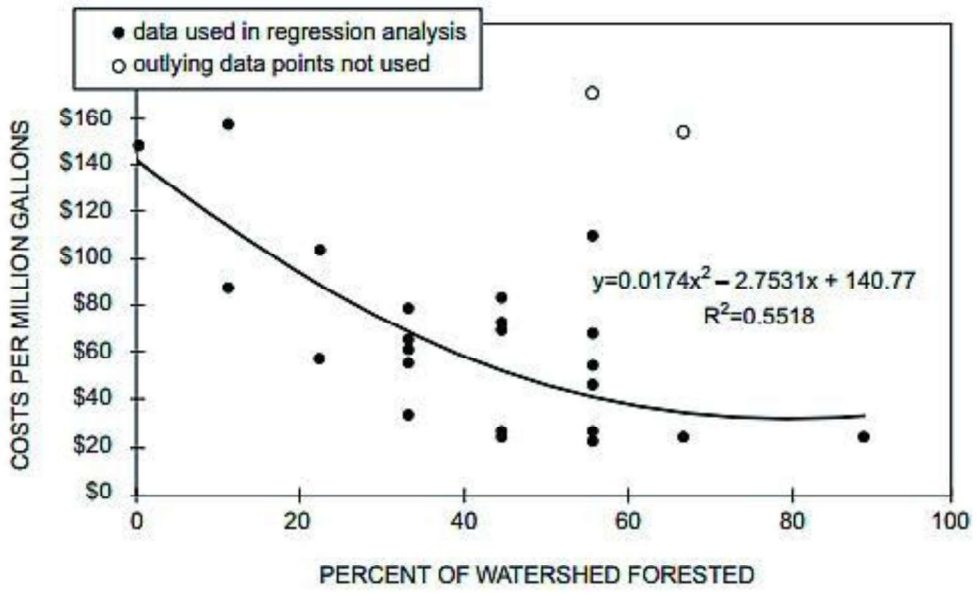

Figure 2-2 Relationship between watershed forested area and drinking water treatment costs (Reproduced from Ernst, 2004 with permission).

Climate change has profoundly impacted the hydrological cycle in many regions of the world (IPCC, 2018). For example, dramatic changes in the timing and quantity of precipitation in some regions have increased rainfall and flooding (Mirza, 2011; IPCC SREX, 2012; IPCC 2018), while other regions are experiencing drought (IPCC SREX, 2012; Cook, 2018; IPCC, 2018). Climate change also results in hotter and drier atmospheric conditions in some forested areas, and has contributed to the increased frequency of larger, more severe wildfires (EPA, 2016). Ironically, the high quality and quantity of water from forested, snowmelt-dominated watersheds is amongst the most vulnerable to the deleterious impacts of such disturbances (Dale *et al*., 2001; Kaufmann *et al*., 2009; Logan & Powell, 2009; Emelko *et al*., 2011; Loehman *et al.,* 2017).

Wildfires consume ~2.5 million hectares of forests in Canada every year (Natural Resources Canada, 2017). They can affect watershed hydrology by changing the timing of snow melt and increasing net precipitation (Williams *et al*., 2019). Stream temperatures can also increase post-fire (Wagner *et al.*, 2014) and significant amounts of sediment (Kunze and Stednick, 2006; Silins *et al.*, 2009; Bladon *et al*., 2014), nutrients (Ranalli, 2004; Bladon *et al.*, 2008; Aiken et al., 2011; Emelko and Sham, 2014), heavy metals (Wolf *et al.*, 2008; Kelly *et al*., 2006), and other contaminants (Kalabokidis, 2000; Crouch *et al.*, 2006) can be released to receiving waters. These impacts can propagate downstream and last for decades in some cases (Stone *et al.*, 2014; Emelko *et al.*, 2016), leading to more variable water quality

(Stone *et al.*, 2011) and cascading impacts on stream ecology (Silins *et al.*, 2009; 2014; Martens *et al.*, 2019). Severe wildfire can change the structure of soils and create a hydrophobic layer, thereby leading to decreased infiltration capacity and increased runoff (DeBano, 1991). These conditions increase the transport of sediments and associated contaminants (including nutrients) from hillslopes to receiving streams (Hauer & Spencer, 1998; Rhoades *et al*., 2018). The resulting deterioration and greater variability in source water quality can substantially challenge water treatment operations (Emelko & Sham, 2014; Shams, 2017) and increase treatment costs (Emelko *et al.*, 2011; Price *et al.*, 2016). The most significant challenges for drinking water treatment are increased variability in turbidity and aqueous natural organic matter (NOM) (Emelko *et al.*, 2011). The potential effects of wildfire on various surface water quality parameters and their maximum acceptable concentrations (MACs) and maximum contaminant level (MCL) based on Health Canada and EPA guidelines, respectively, in *treated* drinking water are listed in Table 2-1. Critically, most of these parameters can be readily removed by conventional drinking water treatment processes—the greatest treatment challenges occur when water quality changes quickly (precluding adequate response in a timely matter) or deteriorates beyond key thresholds, thereby necessitating additional treatment infrastructure (Emelko *et al.*, 2011).

Table 2-1 Water quality parameters that can change as a result of wildfire and their maximum concentrations in treated drinking water based on Health Canada and EPA guidelines (Adapted from Smith *et al.*, 2011; Health Canada, 1978; 1979; 1987; 1991; 1994; 1998; 2006; 2008; 2012; 2013; 2016; 2018; 2019; EPA, 2017; 2018; 2019).

	Parameter	**Potential Human Health Risks**	**Natural Source**	**Health Canada Guideline MAC (mg/L)**	**EPA MCL (mg/L)**
Heavy Metals	Aluminum (Al)	Neurotoxic	Soils and rocks	OG[a]: 0.1 (conventional) 0.2 (other types of treatment)	0.05 to 0.2[c]
	Arsenic (As)	Carcinogenic	Dissolution of minerals, industrial and mining effluent	0.01	0.01
	Chromium (Cr)	Carcinogenic, diffuse hyperplasia of the small intestine	Soils and rocks	0.05	0.1
	Copper (Cu)	Aesthetic, and gastrointestinal tract, and liver and kidney issues	Soils and rocks	2 (AO[b]: 1)	1.3
	Iron (Fe)	Aesthetic, and staining of pipes and fittings	Soils and rocks	AO: 0.3	0.3[c]
	Lead (Pb)	Toxic (affects the central nervous system)	Dissolution from natural sources	0.005	0.015

	Parameter	Potential Human Health Risks	Natural Source	Health Canada Guideline MAC (mg/L)	EPA MCL (mg/L)
Heavy Metals	Manganese (Mn)	Aesthetic	Soils and rocks	0.12 (AO: 0.02)	0.05[c]
	Mercury (Hg)	Toxicity (kidneys, neurological disorders and mental disability	Atmospheric deposition from natural (e.g. volcanoes) sources	0.001	0.002
	Zinc (Zn)	Aesthetic, and gastrointestinal effects at high concentration	Widely distributed in rocks	AO: 5	5[c] (secondary standard)
Other in-organics	Ammonia (NH_3)/ Ammonium (NH_4^+)	Corrosion of copper pipes and fittings; food source for some microorganisms	Microbial metabolism and animal waste	none	30[d]
	Barium (Ba)	Vasoconstriction and peristalsis, convulsions and temporary paralysis	Soils and rocks	2	2
	Chloride (Cl)	Aesthetic and corrosion of pipes and fittings	Dissolution of salt deposits	AO: 250	250[c]
	Cyanide	Highly toxic; thyroid gland and nervous system issues	Biomass burning, natural decomposition of some plants, and production by some microorganisms	0.2	0.2

	Parameter	Potential Human Health Risks	Natural Source	Health Canada Guideline MAC (mg/L)	EPA MCL (mg/L)
Other in-organics	Nitrate (NO_3^-) and Nitrite (NO_2^-)	Thyroid gland issues and methaemoglobinemia in children	Nitrate: oxidation of organic waste, nitrogen-fixing bacteria in soils, or lighting strikes. Nitrite (unstable): reduction of nitrate in low oxygen water	45 (for nitrate) 3 (for nitrite)	45 (for nitrate) 3 (for nitrite)
	Sodium (Na^+)	Aesthetic (taste)	Dissolution of salt deposits	AO: 200	30-60[d]
	Sulfate (SO_4^{2-})	Aesthetic; Purgative effects and corrosion of pipes at high concentrations.	Minerals	AO: 500	250[e]
Organics	Total organic carbon (TOC)	TOC can carry heavy metals and other pollutants to water bodies, and lead to formation of disinfection by-products	allochthonous and autochthonous sources	none	none

	Parameter	Potential Human Health Risks	Natural Source	Health Canada Guideline MAC (mg/L)	EPA MCL (mg/L)
	Polychlorinated dibenzodioxins and dibenzofurans (PCDD/Fs)	Toxic and Carcinogenic	May be produced during forest fires (0.5–28 µg/tonne of fuel)	none	0.0005
Polycyclic aromatic hydrocarbons (PAHs)	Benzo[a]pyrene	Carcinogen to humans	Formed during incomplete combustion of organic matter (eg., wildfire)	0.00004	0.0002
	Benzo[b]fluoranthene	Probably carcinogen to human		none	none
	benzo[j]fluoranthene			none	none
	benzo[k]fluoranthene			none	none
	indeno[1,2,3-cd] pyrene			none	none

	Parameter	Potential Human Health Risks	Natural Source	Health Canada Guideline MAC (mg/L)	EPA MCL (mg/L)
	Total nitrogen (TN)	Excessive algal growth and cyanobacterial blooms (eutrophication)	Soil- and sediment-associated N, organic matter	none	none
	Total phosphorous (TP)	Algal/cyanobacterial blooms (eutrophication)	Soil- and sediment-associated P, organic matter	none	none
	Total dissolved solids (TDS)	Aesthetic; High TDS may also result in excessive corrosion of pipes and fittings	Inorganic salts, small amounts of organic material, clay particles, colloidal iron, manganese oxides, and silica	AO: 500	500[c]
	Color	Aesthetic	Nutrients and organics	AO: 15 TCU	15 TCU[c]
	Turbidity/total suspended solids (TSS)	Used to monitor treatment process performance; May carry adsorbed contaminants	Landscape and in-stream erosion; runoff	0.1 NTU	0.3 NTU

Notes:

a: OG: operational guidance, established based on operational guidance.

b: AO: aesthetic objectives, established based on aesthetic considerations.

c: Secondary Drinking Water Regulations, these values are non-mandatory and non-enforceable standards.

d: Taste threshold: Concentration at which the majority of consumers do not notice an adverse taste in drinking water; it is recognized that some sensitive individuals may detect a chemical at levels below this threshold.

2.1.1 Forest Harvesting Impacts on Water Quality

The benefits of investment in forest management (i.e., utilization of "green infrastructure") compared to building or upgrading water treatment equipment (i.e., "grey infrastructure") have been widely discussed (Ernst, 2004; Gartner *et al*., 2014; Warziniack *et al.*, 2016). Consequently, state-of-the-art forest health protection approaches, such as forest harvesting, mechanical thinning, and prescribed burns, have been proposed (and implemented to some extent) to increase forest resistance and resilience to various landscape disturbances (Steenberg *et al.*, 2011), and reduce risks to drinking water supply (Gartner *et al*., 2014). Several watershed-scale investigations have shown that natural/green infrastructure options for achieving water management objectives can be competitive with gray infrastructure alternatives by reducing/eliminating the need for operating or upfront capital costs and increasing resilience/resistance to natural hazards (Schmidt and Mulligan, 2013). The examples in Figure 2-3 illustrate the potential benefits of natural infrastructure investment for communities of all sizes in a variety of forested physiographic regions (Gartner *et al*., 2014). Despite this recognition, the impacts of forest management approaches such as forest harvesting on water quality and treatability have not been described or quantified.

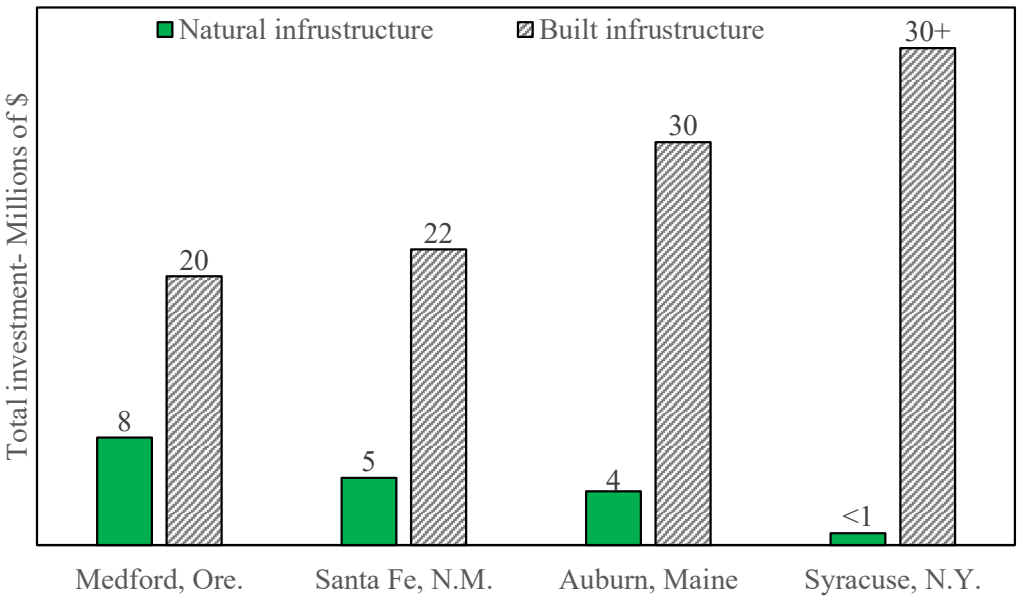

Figure 2-3 Comparison of financial merits of integrated natural and built infrastructure alternatives for desired ecological outcomes (Reproduced from Gartner *et al*., 2014 with permission).

The removal of forest cover as a result of harvesting can affect stream water quality in a manner similar to that of other natural disturbances (Likens *et al*., 1970; Meyer & Tate, 1983; Adamson and Hornung, 1990; Neal *et al*., 1992; Reynolds *et al*., 1992; Nieminen, 2004; Laudon *et al.*, 2009; Schelker *et al.*, 2014). For example, forest harvetsing effects on sediment and turbidity production in aquatic systems have been widely reported—they are highly variable and depend on watershed conditions such as soils and surficial materials, ground cover, hydroclimatic conditions, and topography (Crooke & Hairsine, 2001; Aust & Blinn, 2004; Neary *et al.*, 2010; Anderson & Lockaby, 2011; Lewis *et al.*, 2019). Elevated levels of sediment and turbidity may be harmful for some aquatic organisms, alter habitats, and render water supplies unacceptable for recreational uses—they also can lead to increased drinking water treatment costs and challenges (Emelko *et al*., 2011). Sediment and turbidity yields often increase after harvesting as a result of greater soil erosion by water, ice, and wind; however these increases are usually transient because of vegetation regrowth and the implementation of erosion control measures (e.g., buffer strips, slash retention, silt fence installation, mulch application, etc.) (Rice *et al.*, 1972; Hatten *et al*., 2018). As would be expected, recovery to pre-disturbance conditions is widely variable and may last for years, even decades (Beschta, 1978; Lynch & Corbett, 1990; Madoui *et al.*, 2015; Bartels *et al.*, 2016). Road networks required for forest harvesting operations are generally recognized as the major contributors of sediment/turbidity to receiving streams (Megahan *et al.*, 2001; Sidle *et al*., 2004; Baird *et al*., 2012; Wang *et al.*, 2013) even though they comprise a small fraction (typically <0.5%) of total watershed area (Ziegler & Giambelluca, 1997; Ziegler *et al*., 2001; Arismendi *et al*., 2017).

Corrigan (2017) evaluated the initial impacts of rapid road and road-stream crossing decommissioning for minimization of sediment and turbidity impacts on high value headwater streams after a short duration (10-month) forest harvesting operation in three headwater sub-catchments in the southwestern Rocky Mountains of Alberta, Canada. This work was conducted as part of an ongoing watershed-scale study of wildfire, post-fire salvage logging, and contemporary forest harvesting impacts on hydrology, water quality, and aquatic ecology (i.e., the Southern Rockies Watershed Project [SRWP]) (Silins *et al.*, 2016). Notably, both total suspended solids (TSS) concentrations and turbidity remained low and within the range of natural variability in that investigation. Specifically, the impact of the combined disturbance of rapid harvest (2015) and subsequent road decommissioning (2016) on total suspended solids, wash load concentrations, and sediment ingress was largely negligible, and turbidity was often significantly higher at locations just upstream of forest harvesting during both years of post-disturbance monitoring (Figure 2-4). These observations were attributed to the collective impacts of 1) implementation of secondary erosion control Best Management Practices (BMPs) (e.g., silt fences)

to roads and bridge crossings, 2) rapid decommissioning of roads and crossings to limit exposure of those surfaces to erosion processes, and 3) drier El Niño climatic conditions during the study.

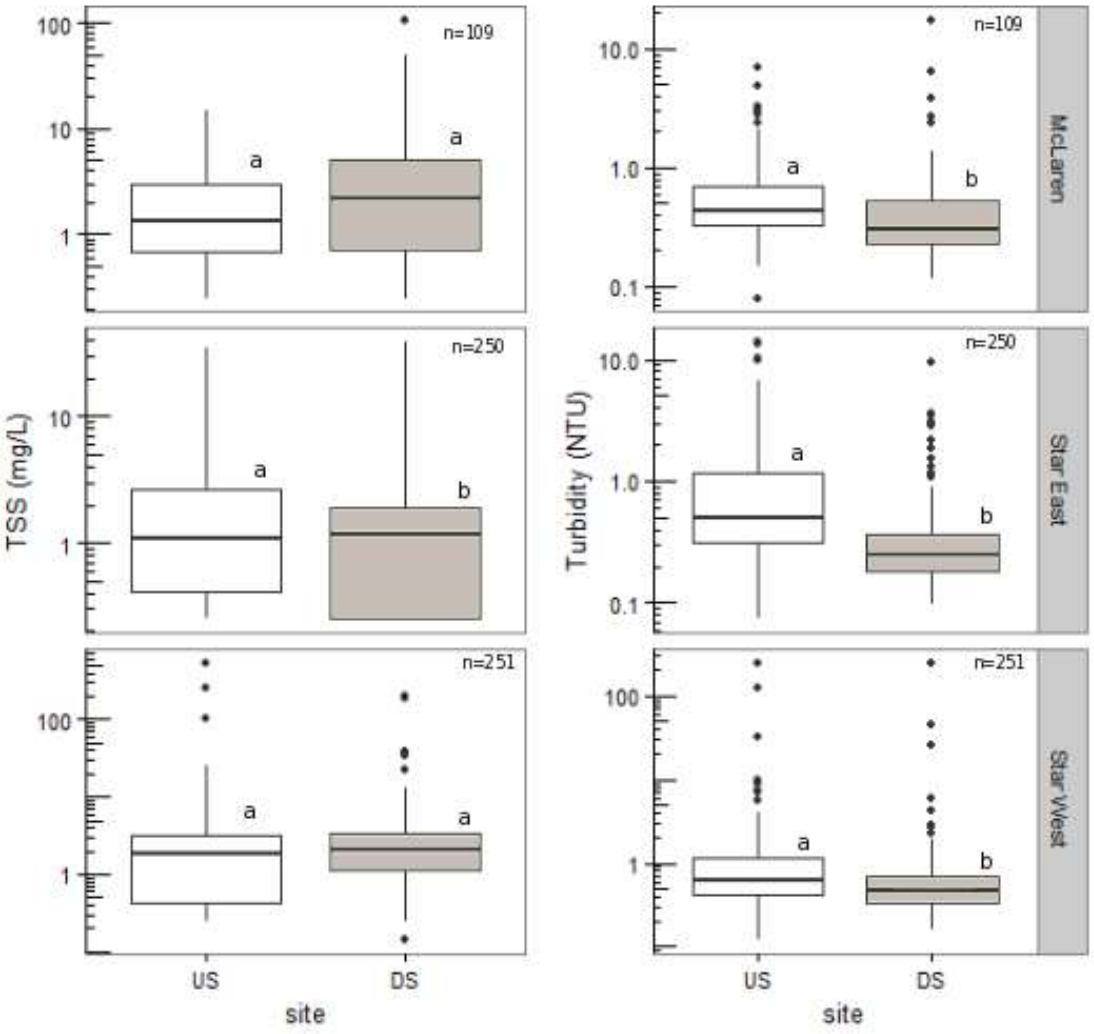

Figure 2-4 Distribution of total suspended solids concentrations (TSS; mg/L), and turbidity (NTU) upstream (US; white) and downstream (DS; grey) at three stream crossings: McLaren (top), Star East (middle), and Star West (bottom). Horizontal lines represent median, while upper and lower limits of boxplots indicate 75th and 25th percentile, whiskers indicate the 95th and 5th percentile, solid dots indicate outliers. Different letters are significantly different (Wilcoxon-sign ranked, $p < 0.016$) (Reproduced from Corrigan, 2017 with permission).

DOC concentrations in streams before and after several types of natural disturbances (i.e., forest harvesting, storm/flood, wildfire, and insect infestation) in forested watersheds are summarized in (Figure 2-5). It should be noted that forest harvesting and storm/flood impacts on stream DOC have been reported for systems with a relatively wider range of pre-disturbance DOC concentrations than those that experienced wildfire or insect infestation. While the reported post-disturbance DOC concentrations for forest harvesting- and storm/flood- impacted rivers appear higher than for wildfire- and insect infestation-impacted rivers, the relative increase from pre-disturbance concentrations was higher for the latter two disturbance types (Figure 2-6). *Critically*, the data presented in Figure 2-5 were compiled to provide a brief perspective on the observations that have been reported in the literature—they do not speak to the frequency and type (e.g., mean values/raw data, collection over a range of flow conditions, watershed conditions, etc.) of data collected during any given investigation. Thus, they are not intended as a rigorous comparison of disturbance effects on stream DOC, but rather as an overview of what has been reported. A more in depth analysis of these numerous investigations was beyond the scope of this research.

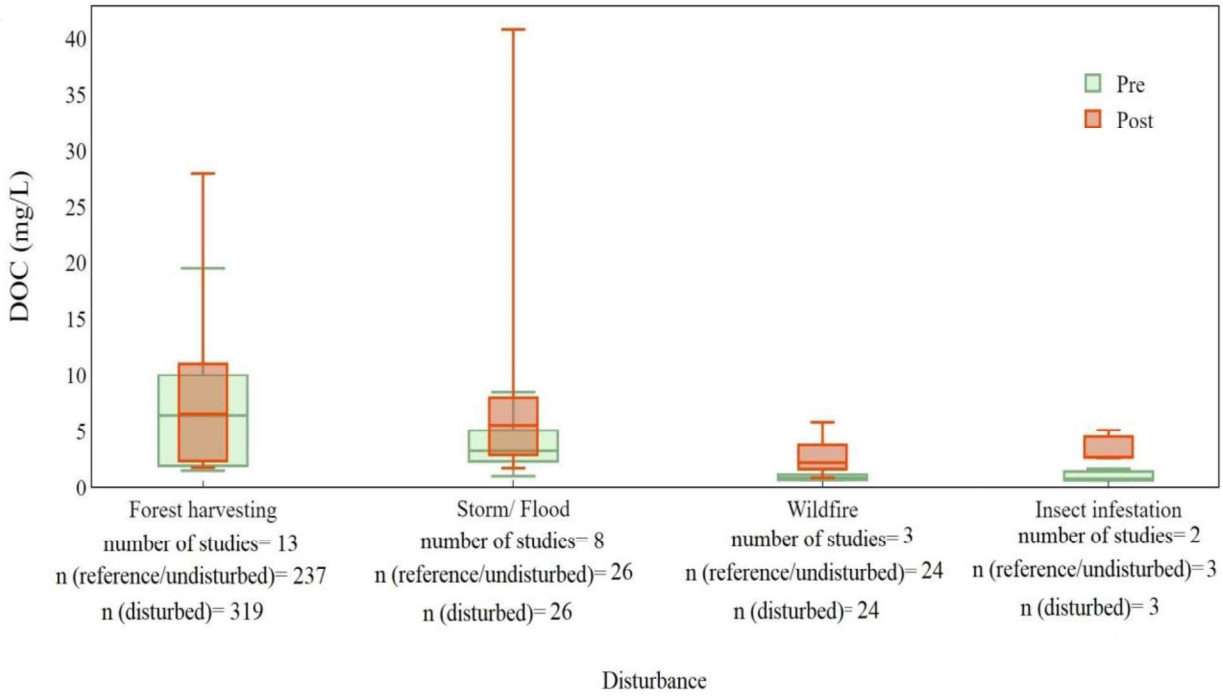

Figure 2-5 DOC concentrations in streams before and after forest harvesting, storm/flood, wildfire, and insect infestation disturbances in forested watersheds (Adapted from Moore, 1989; Neal *et al*., 1992; Hinton *et al*., 1998; Buffam *et al*., 2001; Inamdar *et al*., 2004; Nieminen, 2004; Mladenov *et al*., 2005; Buffam *et al*., 2007; Tetzlaff *et al*., 2007; Mast & Clow, 2008; Morel *et al*., 2009; Clow *et al*.,2011; Emelko *et al.,* 2011; Inamdar *et al*., 2011; Jeong *et al*., 2012; Mikkelson *et al*., 2012; Cawley *et al*., 2014; Musetta-Lambert *et al*., 2017; Shams, 2018).

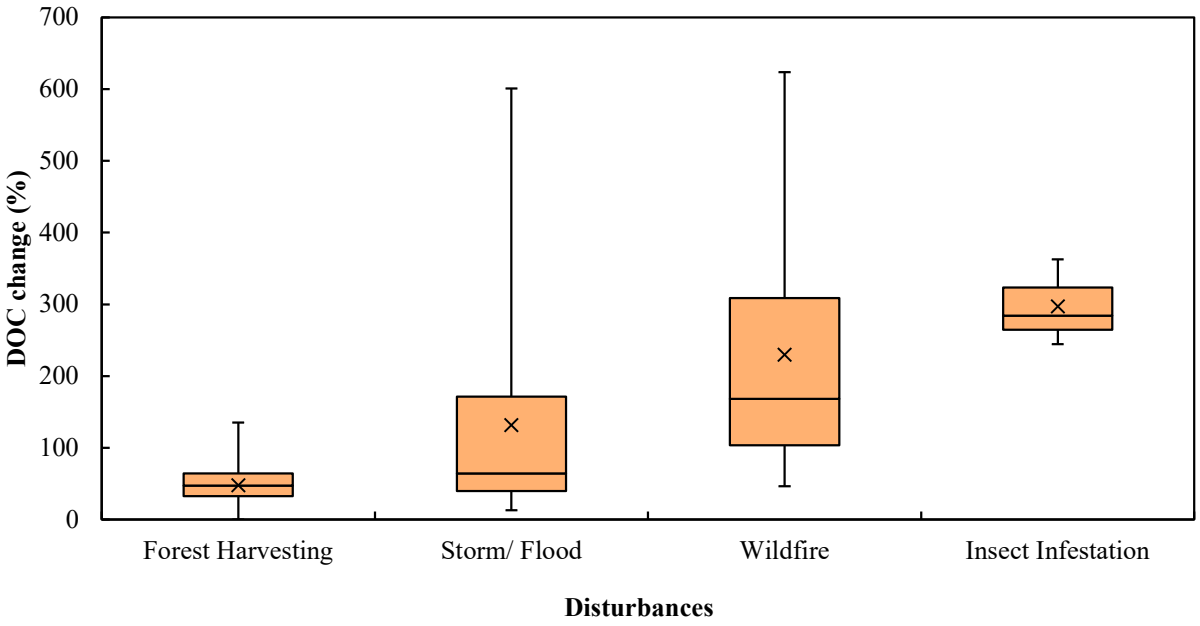

Figure 2-6 percentage change in DOC concentrations in streams as a result of different landscape disturbances in forested watersheds (Adapted from Moore, 1989; Neal *et al*., 1992; Hinton *et al*., 1998; Buffam *et al*., 2001; Inamdar *et al*., 2004; Nieminen, 2004; Mladenov *et al*., 2005; Buffam *et al*., 2007; Tetzlaff *et al*., 2007; Mast & Clow, 2008; Morel *et al*., 2009; Clow *et al*.,2011; Emelko *et al.,* 2011; Inamdar *et al*., 2011; Jeong *et al*., 2012; Mikkelson *et al*., 2012; Cawley *et al*., 2014; Musetta-Lambert *et al*., 2017; Shams, 2018).

The currently available, reported investigations of forest harvesting effects on stream DOC concentrations are summarized in Table 2-2 and Table 2-3. Notably, it has been suggested that forest harvesting impacts on water quality will depend on watershed geology and hydro-climatology (Kiikkilä e*t al*., 2014), as well as the duration and intensity of harvesting activities (Neal *et al*., 1992; Nieminen, 2004; Kreutzweiser *et al*., 2008; Laudon *et al*., 2009; Bolan *et al*., 2011), and applied best management practices (e.g., erosion control) (Wynn *et al*., 2000). While many studies have reported deleterious effects of forest harvesting on soil moisture, temperature, and infiltration capacity (e.g., Standish *et al*., 1988; Wynn *et al*., 2000; Schelker *et al*., 2012), others have reported increased soil infiltration capacity that enabled DOC penetration to deeper soil layers after harvesting, thereby delaying delivery to receiving streams and resulting in decreased stream DOC concentrations (Boyer *et al*. 1996; Boyer *et al*. 1997; Glaz *et al*., 2015). Overall, these data show that DOC concentrations are highly variable across landscapes and disturbance types. **Notably, no investigations reported to date have linked forest harvesting impacts on DOC to drinking water treatability.**

Table 2-2 Summary of studies on forest harvesting impacts on water quality.

Study	Study Area	Type of Stream	Vegetation Type	Average Watershed Elevation a.s.l. (m)	Average Watershed Area (ha)	Annual Precipitation (mm)	Average annual temperature of the Watershed (°C)	Number of sites
Meyer & Tate, 1983 and Webster *et al.*, 1983	North Carolina, US	Second order	Deciduous	675	60	180	10	2 (1 reference, 1 disturbed)
Moore, 1989	New Zealand	N/A	Beech (evergreen species at study location)	300	4.2	2400	12	8 (2 reference, 6 disturbed)
Moore & Jackson, 1989	New Zealand	First order	Coniferous shrubs, replanted with Coniferous trees prior to sampling period.	300	10.5	1559	12.5	3 (1 reference, 2 impacted)
Neal *et al.*, 1992	mid-Wales, UK	Tributary	On average 58% Coniferous and 42% Peat and moorland cover.	530	341	2310	7.5	3 (1 reference, 2 impacted)
Cummins & Farrell, 2003a and b	(western) Ireland	Permanent Streams	Coniferous	118	153	1000	10	3 (1 reference, 2 disturbed)
Nieminen, 2004	(southern) Finland	N/A (data collected from ditch)	Coniferous	87	6	635	4	5 (2 control, 3 disturbed)

Study	Study Area	Type of Stream	Vegetation Type	Average Watershed Elevation a.s.l. (m)	Average Watershed Area (ha)	Annual Precipitation (mm)	Average annual temperature of the Watershed (°C)	Number of sites
Tetzlaff *et al.*, 2007	(central) Scotland	Third order	Coniferous	167	115	1978	N/A	2 (disturbed)
Löfgren *et al.*, 2009	(northern) Sweden	First order (headwaters)	Coniferous	281	76	500	0.6	4 (2 reference, 2 disturbed)
Laudon *et al.*, 2009	(northern) Sweden	First order (headwaters)	Coniferous	281	76	500	0.6	4 (2 reference, 2 disturbed)
Schelker *et al*, 2012	(northern) Sweden	First order (headwaters)	Coniferous	281	51	500	0.6	4 (2 reference, 2 disturbed)
Cawley *et al.*, 2014	New Hampshire, US	First order	Deciduous	497	29.5	1400	4.5	9 (5 references, 4 disturbed)
Eklöf *et al.*, 2014	(northern) Sweden	N/A	Coniferous and Deciduous (near streams)	300	35	631	1.8	3 (1 reference, 2 disturbed)
Kiikkilä *et al.*, 2014	(eastern) Finland	N/A (data collected from ditch)	Coniferous	200	3.1	654	2.1	8 (2 reference, 6 impacted)
Palviainen *et al.*, 2014	(eastern) Finland	First order	84-90% Coniferous and 10-16% Deciduous	192	93.3	564	1.9	4 (1 reference, 3 disturbed)

Study	Study Area	Type of Stream	Vegetation Type	Average Watershed Elevation a.s.l. (m)	Average Watershed Area (ha)	Annual Precipitation (mm)	Average annual temperature of the Watershed (°C)	Number of sites
Schelker *et al.*, 2014	(northern) Sweden	Both sampling locations were parts of Balan River	Coniferous and Deciduous (near streams)	265	1580	700	6	2 (downstream sites)
Nieminen *et al.*, 2015	(southern, eastern, and south-central) Finland	N/A (data collected for ditch)	Coniferous	130	4.4	630	3.5	22 (5 reference, 17 disturbed)
Musetta-Lambert *et al.*, 2017	Ontario, Canada (Boreal Shield ecozone)	Frist order (headwaters)	Deciduous and Coniferous	396	345	1000	2	25 (8 reference, 17 disturbed)

Table 2-3 Summary of forest harvesting impacts on stream DOC concentrations.

Study	Forest Management Practices	Sampling Period	Historical Land Disturbance	Type of Treatment	Years of impact	% of Impact	Regeneration	disturbed DOC range (ppm)	undisturbed DOC range (ppm)	Average DOC change (%)
Moore & Jackson, 1989	Yes	Jan- Dec 1986	Deciduous forests were clear cut for farming in 19 century then left and became scrub dominated forest.	Reference	N/A	0	N/A	30-50	30-50	N/A
				Crushed+ burned	Oct 1981+ Mar 1982	68	Apr-Jun 1982	25-45		-13
				Crushed+ burned	Oct 1984+ Feb 1985	>80	Aug 1985	10-40		-38
Neal *et al.*, 1992	Yes	May 1983-1989	The Watersheds were replanted forest from 1937 to 1964	Reference	N/A	0	N/A	0.8-2.8	0.8-2.2	N/A
				Clear cut	1985-1988	>80	after 1989	1-2		0
				Clear cut	1985-1988	>80	after 1989	1-1.9		-3
Cummins & Farrell, 2003a and b	Yes	1996-2000	Trees on study locations were planted from 1955-1977	Reference	N/A	0	N/A	50-70	50-70	N/A
				Clear cut	1999	33	after 1999	40-100		6.5
				Clear cut+ Fertilizing	1995	67	after 1997	80-100		29

Study	Forest Management Practices	Sampling Period	Historical Land Disturbance	Type of Treatment	Years of impact	% of Impact	Regeneration	Disturbed DOC range (mg/L)	Undisturbed DOC range (mg/L)	Average DOC change (%)
Nieminen, 2004	Yes	Nov 1992-Dec 1999	Peatland drainage on all sites during early 20th Century	Reference	N/A	0	N/A	7-49	7-49	N/A
				Clear cut 1	1994	40	Naturally	12-21		60
				Clear cut 2	1994	72	Artificially (1998-1999)	6.5-16.5		85
				Clear cut 3	1996	40	Artificially (1998-1999)	4-6.5		47
Tetzlaff *et al.*, 2007	Yes	1988-2005	39% of Watershed was cut and replanted	Partial cut 1	2003-2005	37	No	6-8	4-6	50
			23% of Watershed was partially cut and	Partial cut 2	2003-2005	23	No	10-16	5-14	35
Löfgren *et al.*, 2009	Yes	Jun to Oct 2007	N/A	Reference	N/A	0	N/A	15-19	15-19	N/A
				Clear cut	2006	30	No	17-20		13.5
				Clear cut + Riparian	2006	73	No	17-19		10.5
Laudon *et al.*, 2009	Yes	Jun to Oct 2007	N/A	Reference	N/A	0	N/A	7-47	7-47	N/A
				Clear cut	2006	30	No	10-52		23
				Clear cut + Riparian	2006	73	No	10-52		23

Study	Forest Management Practices	Sampling Period	Historical Land Disturbance	Type of Treatment	Years of impact	% of Impact	Regeneration	Disturbed DOC range (mg/L)	Undisturbed DOC range (mg/L)	Average DOC change (%)
Schelker *et al*, 2012	Yes	2004-2009	N/A	Reference	N/A	0	N/A	18-27	18-27	N/A
				Clear cut	2006	64	No	16-21		3.7- 30.4
				Clear cut + Riparian	2006	88	No	18-32		18- 72.1
Cawley *et al.*, 2014	Yes	May 2010-Apr 2011	Second generation trees (locations clear-cut in 1900-17 and affected by hurricane in 1938)	Reference	N/A	0				
				Devegitated	1965-1968	N/A	No	1-2.5	2-10	-23
				Progressive Strip-cut	1970-1974	N/A	No	1-2	1.5-2.5	-10
				Whole tree harvest	1983-1984	N/A	No	1.5-3	1.5-5	-9
				Calcium addition	1999	N/A	No	2-3.5	1.5-3	25
Eklöf *et al.*, 2014	Yes	Mar 2005-2011	N/A	Reference	N/A	0	N/A	N/A	N/A	N/A
				Clear cut	2006	64	No	18-21	15-16	14- 27

Study	Forest Management Practices	Sampling Period	Historical Land Disturbance	Type of Treatment	Years of impact	% of Impact	Regeneration	Disturbed DOC range (mg/L)	Undisturbed DOC range (mg/L)	Average DOC change (%)
Kiikkilä *et al.*, 2014	No	2008-2010	Drainage of peatlands in 1960s	Reference	N/A	N/A	N/A	7-29	11-32	N/A
				WTH+ stumps 1	2009	100	2011	39-59	31-51	17
				WTH+ stumps 2	2009	31	2011	25-46	18-26	68
				WTH+ stumps 3	2009	71	2011	14-44	13-23	33
				WTH+ stumps 4	2009	41	2011	29-69	15-37	108
				Stem only harvest 1	2009	23	2011	27-57	15-35	48
				Stem only harvest 2	2009	35	2011	29-85	16-64	38
Palviainen *et al.*, 2014	Yes	Oct 1991-Dec 2010	N/A	Reference		0				
		Oct 1991-Dec 2010		Clear-cut+ soil Prep	1996	34	2002	5-6.5	5-7	-17
		Oct 1991-Dec 2002		Clear-cut+ soil Prep	1996	11	2002	27-28	27-28	0.4 to 1
		Oct 1991-Dec 2010		Clear-cut+ soil Prep	2000	8	2002	18-24	18-24	N.S

Study	Forest Management Practices	Sampling Period	Historical Land Disturbance	Type of Treatment	Years of impact	% of Impact	Regeneration	Disturbed DOC range (mg/L)	Undisturbed DOC range (mg/L)	Average DOC change (%)
Schelker *et al.*, 2014	Yes	2004-2012	N/A	Smaller Watershed	2006	18	No	5-47	17-24	13.5
				Bigger Watershed	2006	11		5-36		-6
Nieminen *et al.*, 2015	Yes	2007-2011	N/A	Reference 1	N/A	0		23-32	26-48	N/A
		2007-2011		WTH 1	2009-2010	46		25-62	17-48	53
		2007-2011		Stem only harvest 1	2009-2010	81		33-61	25-38	41
		2007-2012		Reference 2	N/A	0		14-25	19-23	N/A
		2007-2012		Stem only harvest 2	2009-2010	23	2011	30-44	23-27	37
		2007-2012		WTH+ Stump 2	2009-2010	66		17-57	16-44	44
		2008-2012		Reference 3	N/A	0		65-75	65-75	N/A
		2008-2012		WTH 3	2009-2010	100	2011	65-120	57-89	26
		2008-2012		Stem only harvest 3	2009-2010	100	2011	55-110	52-89	24
		2008-2012		WTH+ Stump 3	2009-2010	100		63-112	62-72	16

Study	Forest Management Practices	Sampling Period	Historical Land Disturbance	Type of Treatment	Years of impact	% of Impact	Regeneration	Disturbed DOC range (mg/L)	Undisturbed DOC range (mg/L)	Average DOC change (%)
Musetta-Lambert *et al.*, 2017	Yes	Fall 2010 and Summer 2011	No disturbance at least since 50 years ago except for harvested sites that were clear cut 7-17 years prior to the study.	Reference	N/A	0	N/A	10-12	10-12	
				Burned	1999	84	No	7-10		-27
				Clear cut+ Riparian (30m)	1993-2003	48	No	8-11		-20

2.2 Natural Organic Matter

Natural organic matter (NOM) is a complex, heterogeneous mixture of organic compounds present in all aquatic systems (Egeberg *et al.*, 1999). Aqueous NOM structure is influenced by its source, which may be either allochthonous or autochthonous (Thurman, 1985; McDowell & Likens, 1988; Aiken & Cotsaris, 1995). While the former originates from decomposition of soil organic matter and plants, autochthonous NOM arises from photosynthetic activities and biological processes (Aiken and Cotsaris, 1995; Donahue *et al.*, 1998; McKnight *et al*., 2001). The availability and characteristics of NOM in aquatic systems are influenced by natural and anthropogenic landscape disturbances (Meyer & Tate, 1983; Williams *et al*., 2010; Emelko *et al*., 2011; Schelker *et al.*, 2013; Pagano *et al.*, 2014; Mann *et al*., 2014). Changes in the quantity and chemical properties of NOM also can significantly challenge drinking water treatment by affecting suspended particle surface charge and increasing coagulant demand, producing unpleasant taste and odor compounds, and mobilizing heavy metals and organic pollutants (Stumm & Morgan, 1981; Sugai & Burrell, 1984; Baun & Christensen, 2004; Sharp *et al.*, 2006; Matilainen *et al.,*, 2010). NOM also includes precursor materials required for the formation of disinfection by-products (DBPs) of potential health significance and can lead to conditions that promote bacterial regrowth in drinking water distribution systems (Edzwald and Tobiason, 1999; Kitis *et al*., 2002; Leenheer and Croué, 2003; Liang and Singer, 2003; Ates *et al*., 2007; Matilainen *et al.*,2010; Emelko *et al.*, 2011; Zhang *et al.*, 2015; Sillanpää *et al.*, 2018). Drinking water treatment challenges associated with increases in source water NOM are illustrated in Figure 2-7.

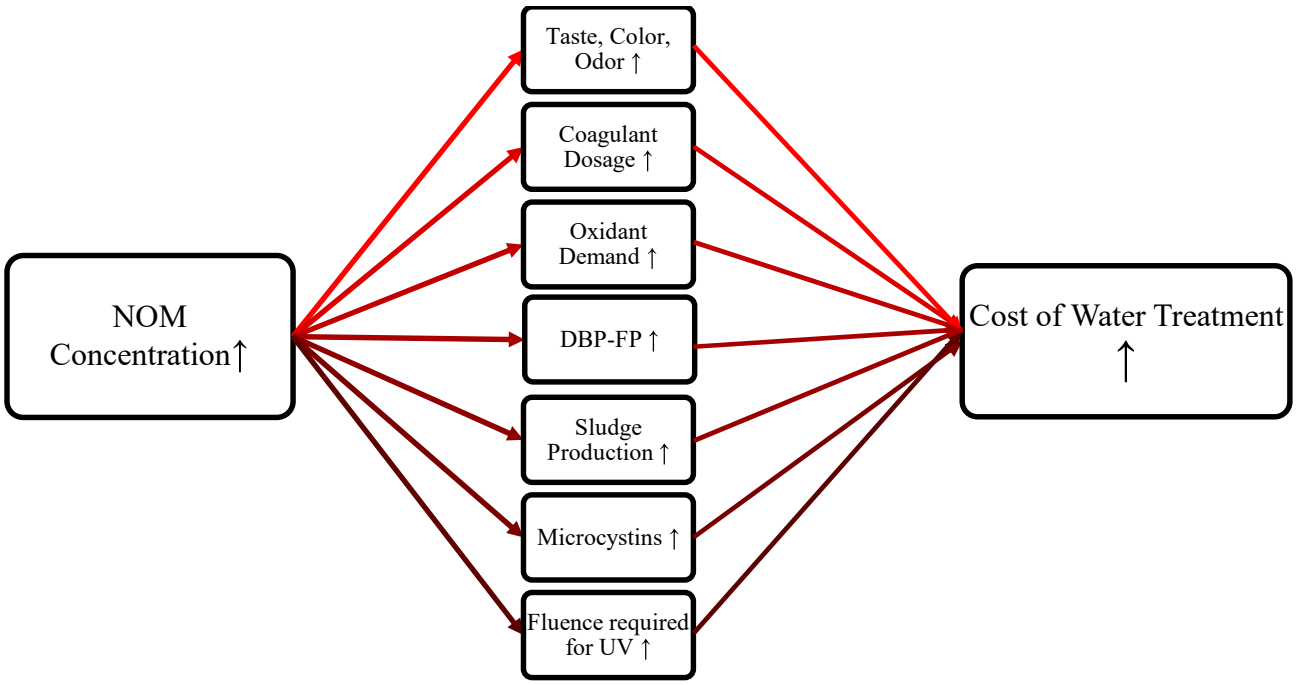

Figure 2-7 Potential implications of changes in NOM concentration to drinking water treatability (Adapted from Emelko *et al.,* 2011).

Aqueous NOM is most frequently quantified as dissolved organic carbon (DOC) concentration. DOC can be classified into two operationally defined fractions: hydrophilic and hydrophobic (Leenheer, 1981; Giabbai *et al.*, 1983; Kitis *et al.*, 2002; Croué, 2004; Panyapinyopol *et al.*, 2005; Sharp *et al.*, 2006; Chong Soh *et al.*, 2008; Filella, 2009; Matilainen *et al*., 2011; Pan *et al.*, 2016). Hydrophilic DOC is described as organic compounds with aliphatic carbon with lower molecular weight, while hydrophobic fractions of NOM are defined as non-biodegradable aromatic compounds (Leenheer & Huffman, 1979; Aiken & Cotsaris, 1995; Krasner *et al.*, 1996). A DOC classification scheme is presented in Figure 2-8.

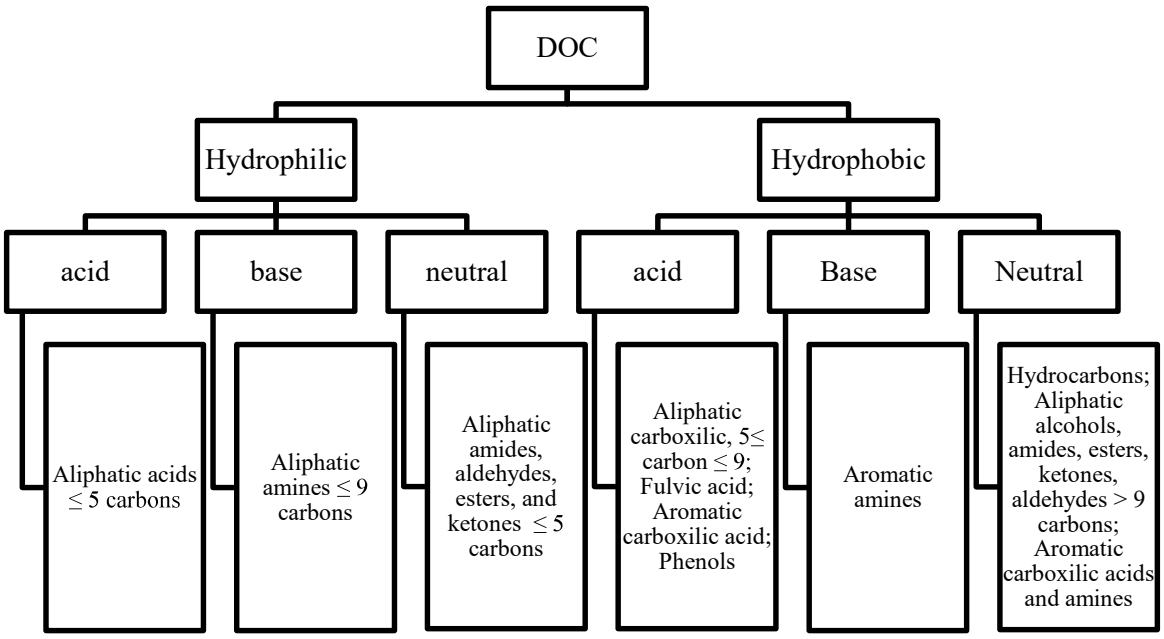

Figure 2-8 Classification of dissolved organic compounds in water (Adapted from Leenheer & Huffman, 1979).

NOM removal during drinking water treatment is highly dependent on its chemical structure and composition (Edzwald, 1993; Matilainen *et al*., 2011; Ji Won *et al*., 2016). Different techniques have been developed to categorize and characterize NOM based on structural similarities such as size, weight, or aromaticity. Shams (2018) reviewed these methods comprehensively. DOC concentration, ultraviolet absorbance, specific ultraviolet absorbance, and fractionation based on size using liquid chromatography-organic carbon detection (LC-OCD) are among the most common aqueous NOM characterization techniques—a brief discussion of each of these follows. More detailed reviews of NOM characterization methods are available in Leenheer *et al.* (2003) and Shams (2018).

2.2.1 TOC and DOC

Total organic carbon (TOC) refers to the sum of particulate and dissolved forms of organic matter in water; therefore, it is often used interchangeably with NOM (Croue *et al.*, 1999). Dissolved organic

carbon (DOC) is operationally defined as the fraction of total organic carbon (TOC) that passes through a 0.45 um filter; the fraction remaining on the filter is referred to as particulate organic carbon (POC) (Kolka *et al*., 2008; APHA, 2017). In natural waters, the concentration of DOC is usually higher than POC; often DOC comprises up to 90% of TOC or more (Thurman, 1985). DOC is also more challenging to remove during drinking water treatment. The concentration of DOC in drinking water sources can range from less than 0.1 mg/L to well above 25 mg/L (APHA, 2017). DOC analyzers oxidize organic carbon to CO_2 and DOC concentration is quantified using either a high-temperature combustion method (HTCM) (Method 5310 B) or persulfate-ultraviolet or heated-persulfate oxidation method (Method 5310 C) (APHA, 2017). In the HTCM (Method 5310 B), inorganic carbon (IC) is removed from water after acidification by purging with an inert gas. The non-purgable organic carbon (NPOC) is then oxidized and the produced CO_2 is measured. While this method is effective in oxidizing resistant compounds, nonvolatile residues can accumulate in the analyzer (Wangersky, 1993; Bolan *et al.*, 1996; APHA, 2017). In contrast, in Method 5310 C, samples are heated and a chemical catalyst, which may be used in combination with UV light, is utilized to oxidize organic carbon (APHA, 2017). High concentrations of aqueous IC can interfere when using this method of DOC analysis, however (Wangersky, 1993; Bolan *et al.*, 1996; APHA, 2017).

2.2.1 UV_{254}

Organic compounds can absorb light over a wide range of visible and ultraviolet wavelengths from 190 to 800 nm (Edzwald *et al*., 1985; Korshin *et al*., 1997; Spencer *et al*., 2007). Aromatic organic compounds are of particular environmental concern because they are highly reactive with disinfectants and other oxidants typically used during drinking water treatment (EPA, 2012). Specifically, they are the main precursors of carbonaceous DBPs of potential health significance, including trihalomethanes (THMs) and haloacetic acids (HAAs) (EPA, 2012). Aromatic and unsaturated organic compounds absorb ultraviolet light at a wavelength of 253.7 nm; thus, UV_{254} is used as a simple indicator of NOM and its character. Based on Beer-Lambert`s law (Equation 2-1), the UV absorbance of any component in water is directly related to the concentration of that compound in water:

$$UVA = a * b * C \qquad \text{Equation 2-1}$$

where: a is the absorption coefficient,

b is the length of cuvette, and

C is the concentration of any compound.

The simplicity and low cost of UV measurement has made UV_{254}, expressed as either UV absorbance (UVA) per cm (cm^{-1}) or UV transmittance (UVT) percent (%) a routine method for monitoring NOM changes in natural systems and treatment plants (Edzwald *et al*., 1985; Korshin *et al*., 1997; Korshin *et al*., 2009).

2.2.2 SUVA

The nature and chemical composition of DOC generally dictates the chemicals and treatment processes that are required for its removal (Edzwald, 1993; Edzwald & Tobiason, 1999). Specific ultraviolet absorbance (SUVA; expressed as L/mg.m) is calculated by dividing UV_{254} by DOC concentration. Thus, it is an indicator of aqueous NOM aromaticity and reactivity (Edzwald, 1993) and can inform strategies for NOM removal by coagulation during drinking water treatment (Edzwald & Tobiason, 1999). Table 2-4 represents the generalized relationship between SUVA and the potential for aqueous NOM removal by coagulation. Generally, NOM in waters with high SUVA (>4 L/mg.m) is considered to be mostly composed of hydrophobic compounds; therefore, coagulation is expected to be an effective method for NOM removal (MWH, 2012). Despite this generalized trend, hydrophilic fractions of aqueous NOM, which are not readily removed by coagulation, also can result in high SUVA (Edzwald, 1993). It is likely for such reasons that a good correlation between SUVA and DBP formation potential (DBP-FP) has been reported in some investigations (Kitis *et al*., 2002; Wassink *et al*., 2011), but not others (Goslan *et al.*, 2002; Ates *et al.*, 2007; Bougeard *et al.*, 2010). Thus, SUVA is not an ideal measure of NOM reactivity, especially as related to DBP formation.

Table 2-4 Relationship between SUVA and NOM removal through coagulation (Adapted from Edzwald & Tobiason, 1999).

SUVA (L/mg.m)	NOM composition	Coagulation	Potential NOM removal
<2	Mostly non-humics, low hydrophobicity, and low molecular weight compounds	NOM has little control	Poor NOM removal (<25% for alum, slightly greater for ferric)
2-4	Mix of aquatic humics and non-humics, mixture of hydrophilic and hydrophobic compounds, mixture of molecular weights	NOM has some influence	Fair to good NOM removal (25-50% for alum, slightly higher for ferric)
>4	Mostly aquatic humics, highly hydrophobic, and high molecular weight compounds	NOM controls coagulation	Good NOM removal (>50% for alum, slightly greater for ferric)

2.2.3 Carbon fractionation by size using LC-OCD

Carbon fractionation using liquid chromatography-organic carbon detection (LC-OCD) is used to characterize aqueous NOM according to different size fractions (Huber *et al.*, 2011; Wassink *et al.*, 2011; Peiris *et al.*, 2013; Mckie *et al.*, 2015; Rutlidge *et al.*, 2015; Chen *et al*., 2016). This occurs via size exclusion, ion interaction, and hydrophobic interaction; however, size exclusion is considered the dominant mechanism of separation. In size-exclusion chromatography, different sizes of molecules have different retention times due to variation in the degree of steric interactions and ability to diffuse within the pores in the stationary phase of a chromatography column (Barth & Boyes, 1992). The mass of organic carbon is determined using both UV light and an organic carbon detector (Huber *et al.*, 2011) (Figure 2-9). The LC-OCD separates NOM into five method-defined groups that include biopolymers, humic substances, building blocks, low molecular weight (LMW) acids, and LMW neutrals (associated with distribution system instability) (Huber *et al.*, 2011) (Figure 2-10). During drinking water treatment, biopolymers are associated with membrane fouling ((Hallé *et al.*, 2009; Kimura *et al.*, 2014),

and humic substances are reported to be the main precursors of disinfection by-products (Nikolaou *et al.*, 2004; Nie *et al*., 2010). Moreover, the presence of low molecular weight neutrals is associated with bacterial regrowth, and the formation of biofilm in distribution system (Hem & Efraimsen, 2001; Hammes *et al*., 2007).

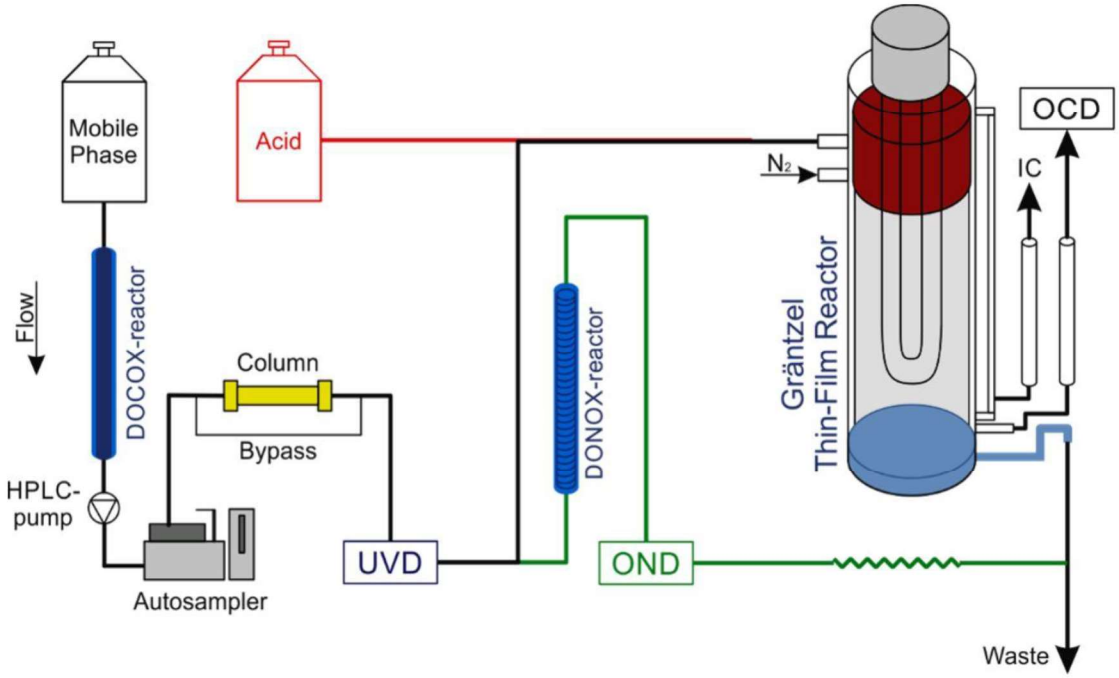

Figure 2-9 LC-OCD flow scheme of Model 8 (Reproduced from Huber *et al.*, 2011 with permission).

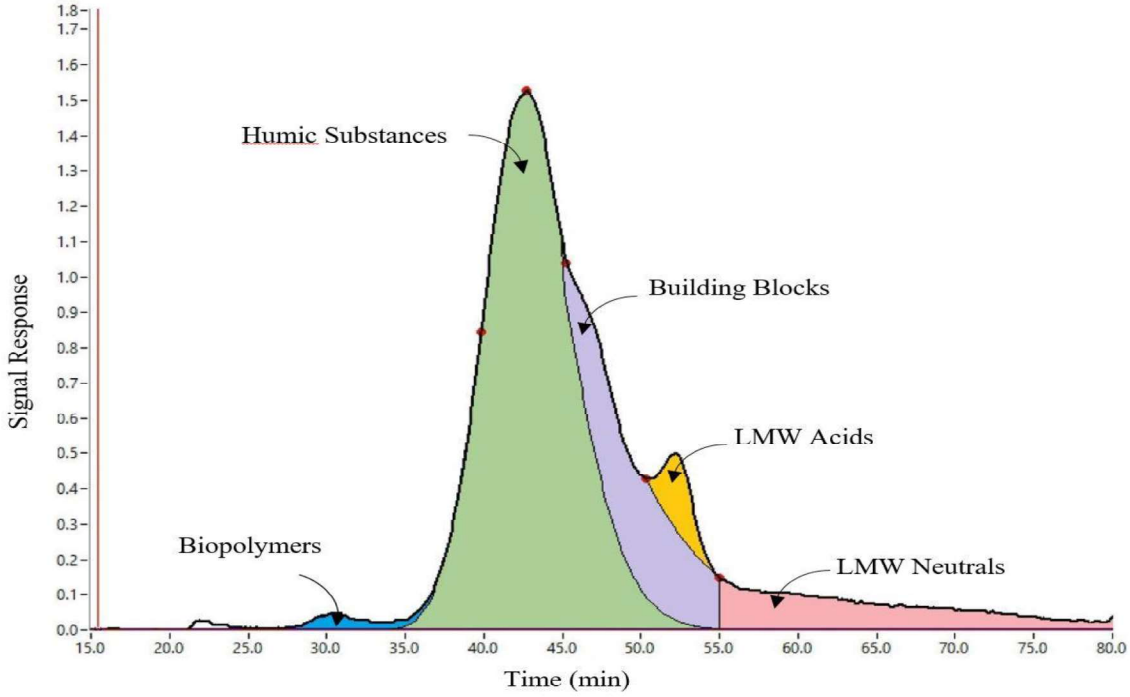

Figure 2-10 Chromatograph of pre-defined NOM fractions measured by LC-OCD following the method of Huber (2011).

2.3 Disinfection By-Product Formation Potentials (DBP-FPs)

Disinfection is a component of drinking water treatment that is critical to the protection of public health from waterborne diseases (Carrell, 1971). While, chemical disinfectants (chlorine, chloramines, chlorine dioxide, and ozone) inactivate waterborne pathogens, their reactivity with different types of organic matter and/or anthropogenic contaminants found in source waters regularly results in the formation of disinfection by-products (DBPs), some of which can be cytotoxic or possibly carcinogenic (Singer, 1999; Plewa *et al*., 2002; Richardson & Postig, 2012). The concentration and type of organic matter or other precursors in water, pH, temperature, type of disinfectant, disinfectant dose, and contact time affect DBP formation (Rodriguez & Sérodes, 2001; Kim *et al*., 2003; Grellier *et al.*, 2015; Tsitsifli & Kanakoudis, 2018). To date, over 600 DBPs have been identified (Richardson *et al*., 2007); however, their diverse structure has hindered the identification of new DBPs and the evaluation of associated toxicological risks to human health (Weinberg, 1999; DWI, 2012). Regulatory agencies around the

world have imposed regulations to limit the formation of some of the known DBPs in drinking water (DWI, 2012; Health Canada, 2009, and 2017; USEPA, 2015; NHMRC, 2018). Table 2-6 lists the maximum concentrations of regulated DBPs based on Health Canada and EPA guidelines. While some studies found a positive correlation between the exposure to DBPs in drinking water and cancer (Lynch *et al.*, 1989; Morris *et al.*, 1992; King & Marrett, 1996), others have reported no evidence of this association (Ranmuthugala *et al*., 2003; Villanueva *et al*., 2004; Rahman *et al*., 2010). Notably, most of the investigations that have concluded carcinogenicity of certain DBPs have neglected other influential factors (e.g., dietary and smoking habits, sex, ethnicity, type 2 diabetes, arsenic, aromatic amines, and occupation)—these methodological limitations have resulted in the rejection of characterization of the use of chlorinated drinking water as carcinogenic to humans (IARC, 1991). A recent investigation of trends in the incidence of bladder cancer in eight countries over the 45 years since THMs were detected in chlorinated water concluded that smoking is a predominant risk factor and that "drinking water-related bladder cancer risks remain questionable and likely small compared to other factors" (Cotruvo & Amato, 2019). Nonetheless, THMs and HAAs are considered indicators of potentially harmful compounds that can be formed in chlorinated water that contains NOM precursors (WHO, 2017); therefore, they are regulated in finished (i.e., treated) water (Health Canada, 2009, and 2017; USEPA, 2015). Several methods have been developed to assess DBP formation potential in various situations. Discussed below, these methods inform different aspects of drinking water treatment; thus, they do not yield equivalent results.

DBP formation potential (DBP-FP) tests are designed to measure the maximum formation of DBPs that can occur in a given water matrix (APHA, 2017). In this test, excessive amounts of chlorine (i.e., hyper-chlorination at levels that would not be typically encountered during regular drinking water treatment) are added to water samples at pH 7, and the concentration of DBPs is measured after 7 days (APHA, 2017). This analysis is especially relevant to source water evaluation as, relative to other methods, it better indicates the maximum DBP formation that could occur from water quality changes resulting from shifts in biogeochemical processes, such as those that may be associated with landscape disturbances (Wei *et al*., 2008; Karapinar *et al.*, 2014; Uyak & Demirbas, 2014; Sutherland *et al.*, 2015). In streams and rivers that serves as drinking water sources, total DBP-FPs ranging from less than 20 µg/L to above 700 µg/L have been reported (Table 2-5). Notably, the efficacy of different water treatment processes in removing DBP precursors is highly dependant on the type of NOM and its molecular weight (Collins *et al*., 1985; Chadik & Amy, 1987). Ahmad & Husain (2015) reported ranges

of FPs for regulated DBPs in various drinking water sources across the provinces of Canada (Table 2-7). When considering these values (and the associated treatment needs that they suggest), it is critical to recognize that DBP-FP analyses are conducted at conditions that are not typically encountered during regular drinking water treatment; thus, they inform the types of treatment needs and DBP formation risks (especially if shock chlorination is required), but not typical DBP formation during drinking water treatment. Other methods (discussed below) are better predictors of DBP formation in plants at typical operating conditions.

Table 2-5 Trihalomethanes formation potentials (THM-FPs) reported for streams and rivers that serve as drinking water sources.

Study	Location	Source	DOC	UV_{254}	THM-FP	HAA-FP	Test conditions
Collins *et al.*, 1985	Canton, NY, US	Grasse River	7.71	31.4	716	-	pH= 7; reaction time= 7 days; temperature= 20 °C
	Las Vegas, NV, US	Colorado River	3.02	4.5	167	-	
	LaVerne, CA, US	Colorado River	3.15	4.3	164	-	
	Orange Co., CA, US	Colorado River	3.13	4.4	167	-	
	Anaheim, CA, US	Colorado River	3.42	4.5	152	-	
	Oceanside, CA, US	Colorado River	3.28	4.5	142	-	
Chadik & Amy, 1987	MS, US	Pearl River	5.62	13.6	284	-	pH= 7; reaction time= 7 days; temperature= 20 °C
	NY, US	Grasse River	6.56	28.8	475	-	
Rathbun, 1996	US	Mississipi River	5.9		576	-	pH= 7; reaction time= 7 days; temperature= 25 °C
		Missouri River	5.3		468	-	
		Ohio River	5		460	-	

Study	Location	Source	DOC	UV254	THM-FP	HAA-FP	Test conditions
Martin-Mousset *et al*., 1997	France	Charente	2.8	5.7	109	-	pH= 7.5; reaction time= 3 days; temperature= 20 °C
		Loire	3.9	8.4	211	-	
		Mayenne	4.2	14	155	-	
		Sevre Nantaise	5.3	15	170	-	
Garvey & Tobiason, 2003	Boston, MA, US	Cadwell Creek	2.05	6.1	188	-	pH= 7; reaction time= 7 days; temperature= 20 °C
		Purgee Brook	1.73	4.3	131	-	
		Atherton Brook	2.63	8.6	246	-	
		West Branch Swift River	2.92	9.4	257	-	
		Dickey Brook	3.86	14.5	365	-	
		Prescott Breek	2.89	9.8	256	-	
		Underhill Brook	3.08	10.5	291	-	
		Hop Brook	3.18	10.8	251	-	
		Middle Branch Swift River	4.93	17.3	436	-	
		West Branch Fever Brook	7.74	28.3	659	-	

Study	Location	Source	DOC	UV_{254}	THM-FP	HAA-FP	Test conditions
Garvey & Tobiason, 2003		East Branch Fever Brook	5.8	20.9	537	-	
		East Branch Swift River	4.72	16.1	447	-	
Kim *et al*., 2003	South Korea	Han River	2.35	7.3	56	11	pH= 5.5, 7, raw water pH; reaction time= 2 days; temperature= 20 °C
		Youngsan River	2.35	7.2	59	19	
		Nackdong River	5.12	11.6	103	9	
van Leeuwen *et al*., 2005	Australia	Middle River	13.77	-	99	-	pH= 5 (HAA-FP), 7 (THM-FP); reaction time= 7 days; temperature= 25 °C
Xu et al., 2007	Shanghai, China	Huangpu River	6.45	14.55	433	312	pH= 7; reaction time= 7 days; temperature= 25 °C
Hong *et al*., 2008	China	Dongjiang River	3.82	4.44	15	17	pH= 5 (HAA-FP), 7 (THM-FP); reaction time= 7 days; temperature= 20 °C

Study	Location	Source	DOC	UV	THM-FP	HAA-FP	Test conditions
Jung & Son, 2008	South Korea	Nakdong River	2.86	6.35	111	112	pH= 8; reaction time= 1 days; temperature= 20 °C
Lantagne *et al*., 2008	Kenya	Not specified	3	-	92	-	pH= 7; reaction time= 7 days; temperature= 20 °C
Chen & Westerhoff, 2010	US	11 Rivers	6.98	13	244	282	pH= 8.2; reaction time= 1 days; temperature= 25 °C
Bush, 2008; Chowdhury *et al.*, 2008	Kamloops, BC, Ca	South Thompson River	2.6	2.6	26	65	pH= 5 (HAA-FP), 7 (THM-FP); reaction time= 7 days; temperature= 25 °C
Zhao *et al*., 2013	China	Songhua River	4.1	12.3	164	382	pH= 7; reaction time= 7 days; temperature= 25 °C

Table 2-6 Maximum concentration of regulated DBPs in drinking water (Adapted from Health Canada, 2009; 2017; EPA, 2016; 2019).

Main Group	Including Compounds	Disinfectant	Health Effect	Health Canada MAC (µg/L)	EPA MCL (µg/L)	Comments
Trihalomethanes (THMs)	Chloroform	Chlorine	Liver problems, kidney and colorectal cancers	100	80	Organic DBP
	Bromodichloromethane	Chlorine				
	Bromoform*	Chlorine/ Ozone				
	Dibromochloromethane	Chlorine				
Haloacetic Acids (HAA)	Bromochloroacetic Acid	Chlorine	Liver cancer, other organ cancer, liver, body, kidney and testes weights effects	80	60	Organic DBP
	Bromodichloroacetic Acid	Chlorine				
	Chlorodibromoacetic Acid	Chlorine				
	Dibromoacetic Acid	Chlorine				
	Monobromoacetic Acid	Chlorine				
	Tribromoacetic Acid	Chlorine				
N-Nitrosodimethylamine (NDMA)		Chloramine	Liver cancer	0.04	none	Organic DBP
Bromate		Ozone	Renal cell tumor	10	10	Inorganic DBP
Chlorate		Chlorine dioxide	Thyroid gland effects	1000	none	Inorganic DBP
Chlorite		Chlorine dioxide	Neurobehavioural effects, decreased absolute liver weights, altered liver weights	1000	1000	Inorganic DBP

* Health Canada has a MAC level of 16 µg/L for bromoform.

Table 2-7 Ranges of THM- and HAA-FP in Canadian drinking water sources (data collected from the inlet of drinking water treatment plants, adapted from Ahmad & Husain, 2015).

Province	Period	Number of Treatment Plants	THM-FP (µg/L)			HAA-FP (µg/L)		
			mean	Range	SD	mean	Range	SD
AB	2000-05	449	61.5	0.6-447	66	38.4	3-141	39
ON	2000-04	179	40.9	0.5-343	39.9	28.6	0.4-244	28.9
QC	2002-06	622	42.5	0-565	53.3	41.2	3.9-166	36.2
MB	2001-06	74	164.9	0.7-640	110.9	72.4	12-249	76.1
SK	2002-06	204	95.3	4-445	71.8	51.8	1-238	70.8
BC	2001-05	13	38.4	9-116	22.7	54.4	11-117	21
NS	1999-04	24	110.2	2-640	84.9	116.2	8-602	119
NL	2001-07	467	77.3	0-470	79.5	107.8	0-507.5	103
NB	1993	4	62.1	4.1-146	45	85.7	10-398	96.1
PEI	2003-06	-	3.5	1.4-5.9	0.96			

The uniform formation conditions (UFC) test for DBP formation was introduced as a variation of the FP test in which a much lower chlorine dose is applied to be more representative of typical chlorination conditions that occur in drinking water treatment plants (Summers *et al*., 1996). Thus, this test (DBP-UFC) enables comparison of DBP formation between water treatment plants at consistent and realistic chlorination conditions. The incubation time during this test is 24±1 h, samples are buffered at pH 8±0.2 and they are stored at 20±1°C. As would be expected, DBP concentrations obtained from the DBP-UFC test are typically lower values than those obtained using the DBP-FP test, due to the higher initial chlorine dose and longer incubation time in FP test (Baribeau, 2006; DiCicco, 2015).

To predict DBP formation in distribution systems, a simulated distribution system (SDS) method also was developed (Koch *et al.*, 1991; APHA, 2017). The conditions of this test (free chlorine residual, incubation temperature, and contact time) are chosen to simulate an actual distribution system and estimate the concentration of potentially formed DBPs at the consumers tap (Koch *et al.*, 1991; APHA, 2017). Thus, this test delivers DBP formation information that is the most specific to a given system.

Notably, DBP-SDS is evaluated to represent a point in the distribution system with the highest probability of DBP formation, and the measured values should be below the regulatory or suggested guideline values (Table 2-6) (Health Canada, 2009; APHA, 2017).

2.4 Source Water Protection Plans

To safeguard public health from waterborne diseases, it is critical to protect drinking water from contamination from its source to the tap. The multi-barrier approach was developed to achieve this goal—it includes: source protection, treatment, distribution, monitoring, and response (CCME, 2004) (Figure 2-11). Source water protection (SWP) plans are designed to protect current and future drinking water sources from contamination and overuse to protect human and ecosystem health (Blundell *et al.*, 2004). SWP strategies involve identifying risks to source water, recognizing the most vulnerable areas to contamination, and planning to minimize the discharge of contaminants to water sources (CCME, 2004; Ivey *et al.*, 2006). In addition to protecting the quality and quantity of water, the implementation of SWP strategies can significantly decrease the costs of drinkng water treatment (NRC (U.S.), 2000). It has been suggested that SWP costs six to more than 20 times less than treatment of contaminated water (Timmer *et al.*, 2007; Patrick, 2011). However, increasing pressures from climate change-exacerbated, severe landscape disturbances pose new risks to water quality and quantity, which need to be addressed through exapnded implementation of SWP strategies (Emelko & Sham, 2014; Robinne, 2019).

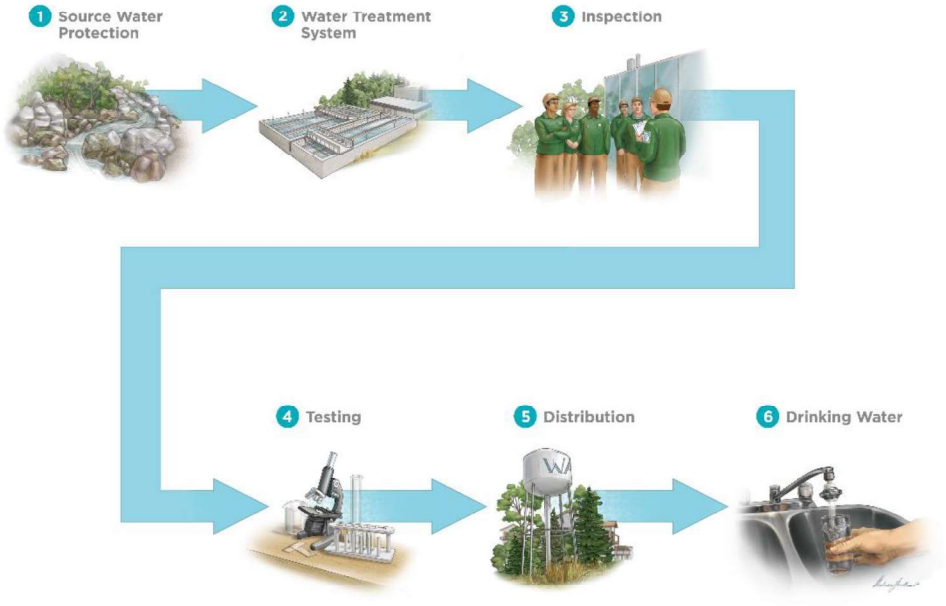

Figure 2-11 Multi-barrier approach to safe drinking water (Reproduced from https://conservationontario.ca/conservation-authorities/source-water-protection/).

2.5 Research Needs

Over the past 30 years, the impacts of various forest harvesting practices on water quality have been widely reported (Table 2-2 and Table 2-3). These investigations have largely focused on linking water quality impacts from the disturbances to specific landscape processes. To date, drinking water treatability implications of forest harvesting practices have not been reported. This likely is in part because of the complexities associated with attributing water quality fluctuations to disturbances (here, forest harvesting)—as opposed to natural variability—and then connecting them to downstream drinking water treatability challenges (Emelko *et at*., 2011). Notably, conducting the requisite long-term and large-scale watershed research required to informing these connections is often logistically and financially prohibitive, especially when the collection of an adequate number of samples is required to account for hydroclimatic variability—this often requires multiple years of data collection with reasonable frequency at a fully representative range of stream flow conditions. Linking source water

quality changes to drinking water treatment impacts is further complicated by the lack of available metrics to generally describe treatment capacity in absence of specific infrastructure and operational practices that may be present in a given water treatment plant. Notably, the potentially catastrophic impacts of natural disturbances, such as wildfire, on water quality and treatability emphasize the need to find a balance between landscape management strategies (e.g., forest harvesting) that mitigate potential risks from natural disturbances and the risk of creating new challenges as a result of those management strategies, which are also landscape disturbances that may have deleterious impacts on water quality and treatability.

3 Materials and Methods

3.1 Research Approach

Contemporary forest harvesting impacts on drinking water source quality and treatability were investigated at the watershed-scale in the eastern slopes of the Rocky Mountains of southwestern Alberta, Canada (Figure 3-1 and Figure 3-2). This work was conducted as part of an ongoing watershed-scale study of wildfire, post-fire salvage logging, and contemporary forest harvesting impacts on hydrology, water quality, and aquatic ecology (i.e., the Southern Rockies Watershed Project [SRWP]), which commenced in 2004 (Silins *et al.*, 2016). The SRWP team studied the undisturbed Star and York Creek watersheds as part of that study. The present investigation began in 2015 when three types of contemporary forest harvesting (clear-cut with patch retention, strip-shelterwood cut, and partial cut) were applied in the previously undisturbed watersheds. Specifically, one approach was implemented in each of three sub-watersheds of Star Creek in 2015. York Creek was not harvested and served as a reference watershed for comparison. During harvesting, best management practices (BMPs) to minimize surface erosion/runoff were utilized. Reference and harvested stream water turbidity, DOC, aqueous NOM proxies (UV_{254}, SUVA, and NOM fractions evaluated by LC-OCD), and DBP-FPs were evaluated during and over the first three years after forest harvesting. To the extent possible, additional pre-disturbance data (i.e., collected prior to harvesting and reported in the literature) from the study watersheds were also included in this analysis.

3.2 Study Sites

The study was conducted in two headwater basins of the Crowsnest River; namely, Star Creek and York Creek (Figure 3-2), which drain an area of 1035 and 865 ha, respectively. These predominantly snow melt dominated headwaters of the basin originate in the eastern slopes of the Rocky Mountains (Silins et al., 2014) and represent an imortant source water region for many municipalities in Southern Alberta (Oldman Watershed Council, 2010). The surficial geology of the basin is complex and

dominated by Cretaceous shales and sandstones (Hamilton et al, 1998; Jackson et al., 2008). These formations are overlain by an array of pre-glacial, glacial and recent alluvial deposits (Landenberg et al., 2006). The elevation of the study sites ranges from 1432 to 2635 m above sea level and the annual precipitation varies from 800 to 1360 mm. The mean annual summer temperature is 16.4°C and the mean winter temperature is -5.3°C (Silins *et al*., 2016). Vegetation in the study watersheds is characterized by Engelman spruce (*Picea engelmanii*), lodgepole pine (*Pinus contorta var. latifolia*), and subalpine fir (*Abies lasoicarpa*) (Silins *et al*., 2016). Because this snowmelt dominated area is predominantly forested, it is source of high quality water in southern Alberta (Oldman Watershed Council, 2010).

Figure 3-1 The Oldman River Basin (Reproduced from Oldman Watershed Council, 2007 with permission).

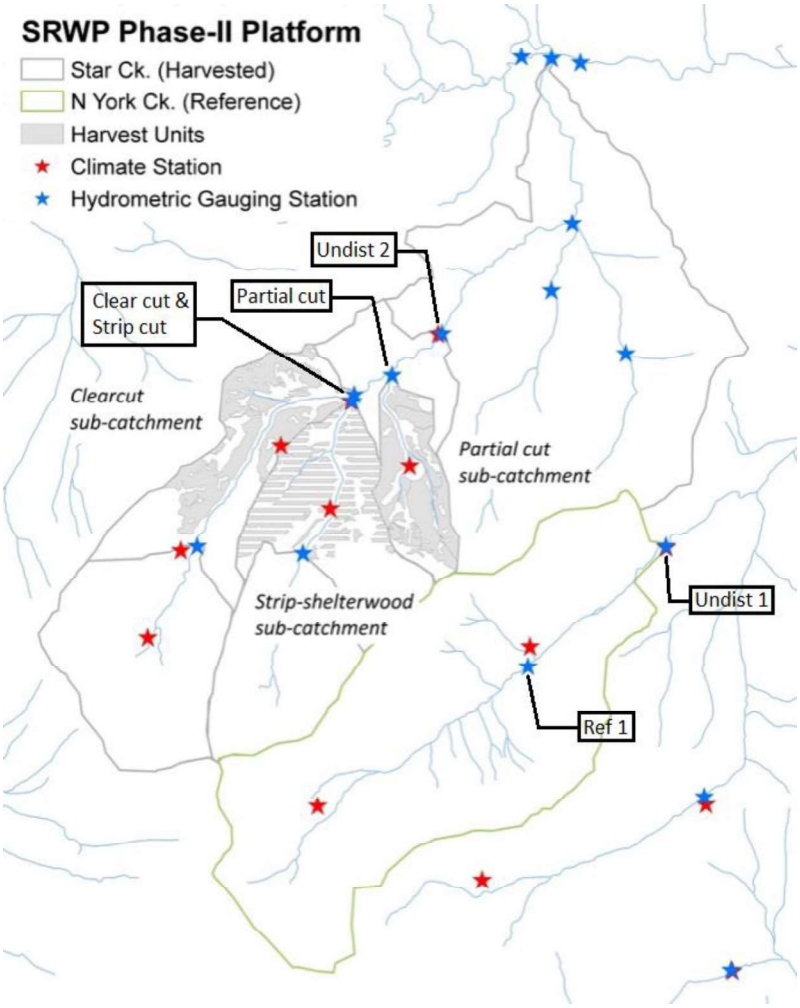

Figure 3-2 Map of the Southern Rockies Watershed Project Phase II research watersheds. From west to east: Star and North York Creeks. Three alternative types of harvesting treatments were performed in three sub-watershed of Star Creek Watershed in 2015.

3.3 Sample Collection

In 2015, three harvest treatments (clear cut, strip cut and partial cut) were applied in three headwater sub-catchments of Star Creek watershed (Figure 3-2). Study sites were located above and below each harvesting treatment in Star Creek and in an unharvested adjacent watershed York Creek (Table 3-1). All water samples were collected to provide a representative assessment of water quality during a range

of streamflow conditions (freshet, stormflow and baseflow). The total number of samples used in this study is presented in Table 3-2. Depth-integrated water samples were collected in acid-washed triple-rinsed high-density polyethylene bottles that were refrigerated at 4°C until analysis, which occurred within four days after collection.

Table 3-1 Description of the study locations (SRWP Phase II: Harvested Watersheds).

Site Name	**Description**	**Treatment**
Ref 1	Headwaters reference	Reference
Undist 1	Undisturbed (not harvested)	Second order stream downstream of reference; published data collected prior to 2015 are included for discussion of natural variability in the region
Undist 2	Undisturbed prior to harvesting, downstream of all harvesting	Second order stream downstream of harvesting; published data collected prior to 2015 are included for discussion of natural variability in the region
Clear Cut	Headwaters harvested	Clear cut with patch retention
Strip Cut	Headwaters harvested	Strip-shelterwood cut
Partial Cut	Headwaters harvested	Partial cut

Table 3-2 Number of samples previously reported or processed during the study.

Location	Period	Turbidity	DOC	UV_{254}	LC-OCD	THM	HAA
Ref 1	2009-2010 (Geng, 2018)	-	18	-	-	-	-
	2013-2016 (Shams, 2018)	7	7	7	2	7	7
	2017-2018	8	8	8	3	7	6
	Total Ref 1	15	33	15	5	14	13
Undist 1	2013-2016 (Shams, 2018)	9	9	9	6	9	1
	2017-2018	-	-	-	-	-	-
	Total Undist 1	9	9	9	6	9	1
Undist 2	2013-2016 (Shams, 2018)	9	10	10	6	9	1
	2017-2018	-	-	-	-	-	-
	Total Undist 2	9	10	10	6	9	1
Clear cut	2013-2016 (Shams, 2018)	8	8	8	3	8	8
	2017-2018	8	8	8	3	7	7
	Total Clear cut	16	16	16	6	15	15
Strip cut	2013-2016 (Shams, 2018)	8	8	8	3	8	8
	2017-2018	8	8	8	3	7	7
	Total Strip cut	16	16	16	6	15	15
Partial cut	2013-2016 (Shams, 2018)	5	5	5	3	5	5
	2017-2018	3	3	3	1	2	2
	Total Partial cut	8	8	8	4	7	7
Total		**73**	**92**	**74**	**33**	**69**	**52**

3.4 Water Quality Analysis

Water samples were stored at 4°C in the dark until analysis. Several NOM characterization and water quality analyses were conducted. The comprehensive raw data are provided in Appendix A.

3.4.1 Turbidity

Turbidity was determined on unfiltered samples based on Standard Method 2130B using a HACH 2100Q (China) low range turbidimeter with a detection limit (DL) of 0.02 NTU (Table 3-4) (Figure 3-7b) (APHA, 2017). The device was calibrated prior to each sets of measurements and the calibration

was checked on every 10th samples, using a standard formazin solution. Moreover, the measurement vial was filled slowly, to prevent the formation of air bubbles. Each sample was analyzed three times and the average of the readings was used for statistical analysis.

3.4.2 DOC

Water samples were filtered through pre-rinsed 0.45 µm nylon filters (ZAPCAP-CR, Sanford, USA). DOC concentrations were analyzed on a Shimadzu TOC-VCPH TOC analyzer (Jiangsu, China;Figure 3-5) using Standard Method 5310B (APHA, 2017). The device was calibrated, with multiple calibration points, using a 10 mg/L solution of potassium hydrogen phthalate that was diluted from a 1000 mg/L stock. A sample calibration curve is presented in Figure 3-3. The detection limit (DL) on this instrument was calculated as 0.11 mg/L (DL= SD*3) (Table 3-4). The samples were analyzed in triplicate with three injections per sample vial; therefore, there were nine points representing each sample. To ensure that the use of the average of these nine points in the comparative statistical analyses was valid, a single factor ANOVA test was performed for each set of data (9 points) with significance level of 5% (α=0.05). The detailed results from this analysis can be found in Appendix B.

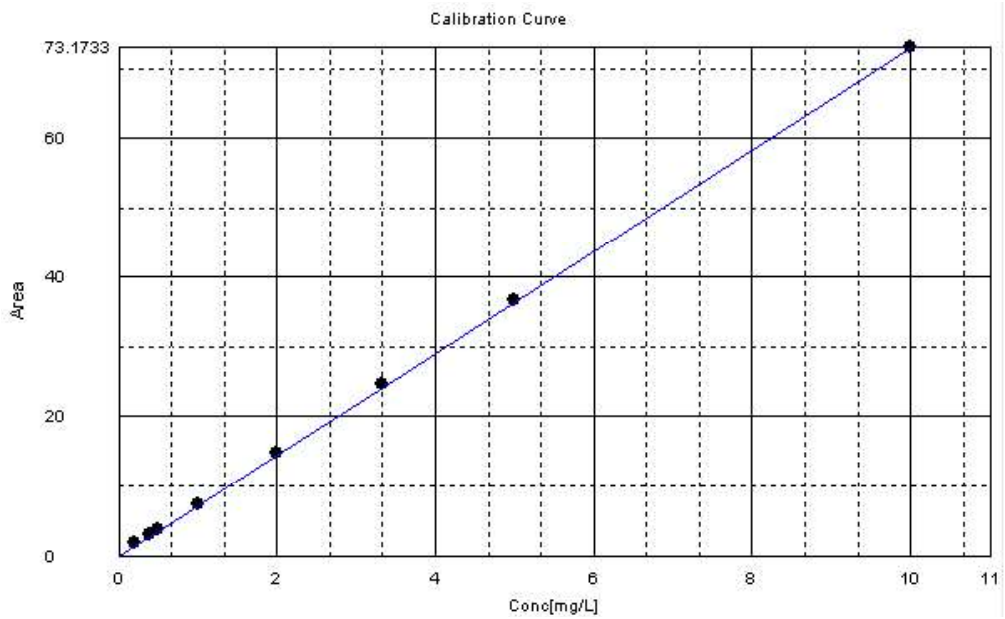

Figure 3-3 Example calibration curve for TOC analyzer (n=9; R^2 = 0.9754).

3.4.3 UV_{254} and SUVA

UV254 absorbance was analyzed using a Hewlett-Packard 8453 (USA) and a Cary 100 Series UV-Vis Spectrometer (Malaysia), both with a 1 cm quartz cell (Method 5910 B; APHA, 2017) (Figure 3-6). Prior to each measurement, the device was calibrated with a blank sample (cuvette filled with Milli-Q™ (MQ) water). The calibration was repeated on every 10^{th} sample and each sample were measured in triplicate. The detection limit for the device was 0.01 m^{-1} (Table 3-4), and the reproducibility of wavelength was equal to 0.02 nm. Specific ultraviolet absorbance (SUVA, in units of mg/L·m) was then calculated as the obtained UV_{254} divided by the DOC concentration (Edzwald *et al.*, 1985).

To check the stability of water samples between collection and analysis, a storage study was conducted. In this study, UV_{254} and DOC values for Ref 1, Clear cut, and Strip cut locations, collected on 26 June 2017, were measured and monitored periodically from the day of arrival until October 2017, (Table 3-3). The samples were stored in the dark at 4°C, without preservative addition. None of the samples showed any significant change in UV_{254} or DOC concentration during the study period (*p*-value >> 0.05); therefore, it was concluded that the shipping period did not degrade DOC concentrations or UV_{254} absorbance.

Table 3-3 UV_{254} stability test results for water samples collected on June 26, 2017.

	28-Jun	29-Jun	30-Jun	02-Jul	05-Jul	12-Jul	21-Jul	15-Aug	04-Sep	04-Oct
Ref 1	1.11	1.14	1.16	1.12	1.13	1.17	1.10	1.14	1.13	1.16
Clear cut	1.94	1.96	1.89	1.94	1.96	1.97	1.99	1.94	1.96	1.95
Strip cut	1.78	1.75	1.77	1.76	1.78	1.75	1.74	1.79	1.76	1.77

3.4.4 Carbon fractionation by size using LC-OCD

NOM was fractionated by size using an LC-OCD (Model 8, DOC-LABOR, Karlsruhe, Germany) (Figure 3-3a). This device employs a technique that uses a weak cation exchange column (250 mm × 20 mm, TSK HW 50S, 3000 theoretical plates) followed by a UV_{254} detector (UVD), an organic carbon detector (OCD), and an organic nitrogen detector (OND). The resulting chromatographs were then

evaluated using ChromCALC, DOC-LABOR data processing software (Huber *et al.*, 2011). The LC-OCD was calibrated for OCD, OND, and UVD every 6 months (Figure 3-4), and the calibration was controlled each time prior to running samples.

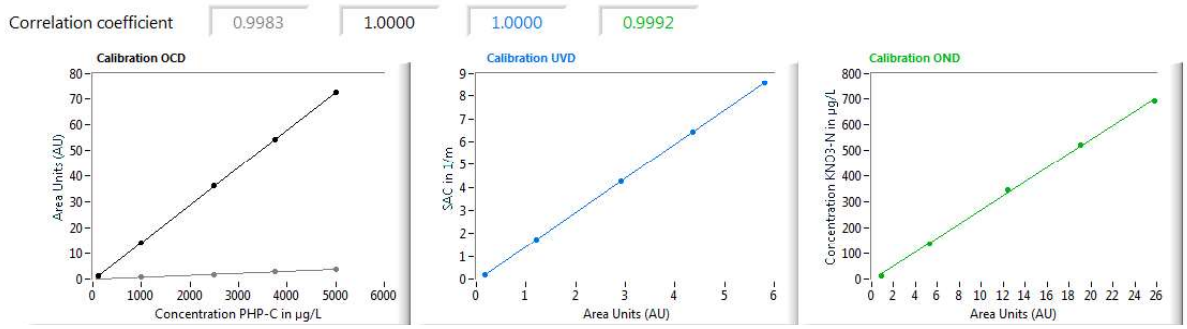

Figure 3-4 Example calibration results for LC-OCD Model 8.

3.4.5 THM- and HAA-FP

Total THM and HAA formation potentials (FPs) were evaluated. THM-FP was evaluated using Standard Method 5710B (APHA, 2017) and an Agilent Technologies 7890B -MS/5977A GC/MS with purge and trap) HAA-FP were analyzed on a Varian CP3800-MS/MS2000 (Saturn MS Ion Trap) GC/MS/MS/CI analyzer. The method utilized for HAA-FP analysis was Method 5710D (APHA, 2017). The DL for each method is presented in Table 3-4.

Table 3-4 Detection limits (DLs) for water quality and treatability analytes.

Analyte	Detection Limit
Turbidity (NTU)	0.02
DOC (mg/L)	0.11
UV_{254} (m^{-1})	0.01
THM-FP (µg/L)	0.37
HAA-FP (µg/L)	5.3

Figure 3-5 Shimadzu TOC-VCPH TOC analyzer used to measure DOC concentration.

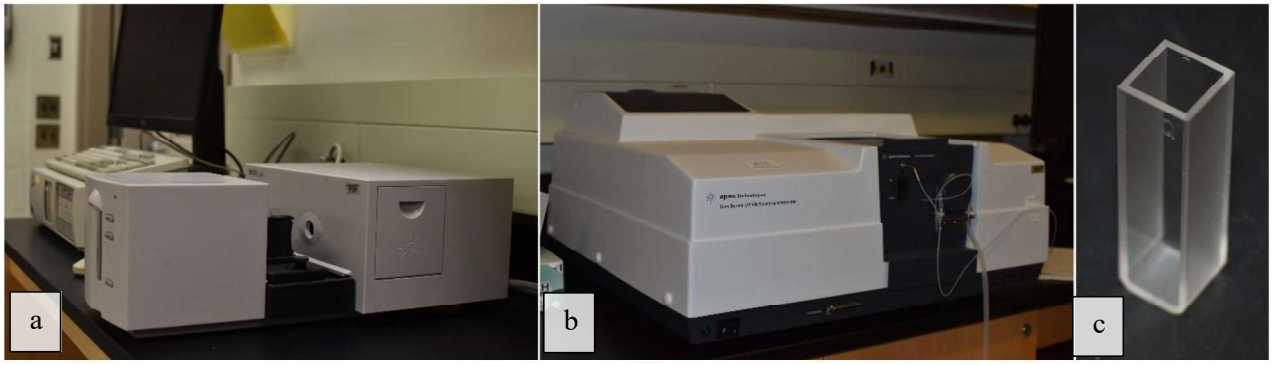

Figure 3-6 UV_{254} analyzer, a) Hewlett-Packard 8453 spectrophotometer, b) Cary Series UV-Vis spectrometer, c) 1 cm quartz cell.

Figure 3-7 a) LC-OCD Model 8 and b) Turbidimeter- HACH 2100Q.

3.5 Statistical Analyses

All of the water quality and treatability data collected herein were tested for normality using the Shapiro-Wilk test. As some of the datasets were not normally distributed, a non-parametric statistical analysis of the data was conducted. While a Wilcoxon signed-rank test would have been desirable to treat the samples collected from the various watersheds as paired because of their concurrent collection across a range of stream flow conditions, the relatively small sample size and the desire to include additional (non-paired) samples in the analysis precluded the use of this test. Specifically, a power analysis conducted to determine the sufficient sample size—using a significance level of 0.05, a power of 0.80, a small effect size (dz = 0.2), and two tails (Faul et al., 2013)—indicated that at least 208 samples would be required to demonstrate a small effect; at least 35 samples would be required to demonstrate a medium effect (dz = 0.5) (Faul *et al.*, 2009). As the required number of samples was not

available, the Mann-Whitney U test (i.e., the Wilcoxon rank sum test, which differs the Wilcoxon signed-rank test) that compares two unpaired groups was used to evaluate whether the investigated stream water quality and treatability metrics differed between the reference and harvested watersheds. A 5% significance level was utilized. Finally, to characterize simple correlations between parameters, coefficients of determination (R^2) were determined for least squares linear regression.

4 Results and Discussion

Drinking water quality and treatability at three different harvested watersheds (clear cut, strip cut, and partial cut) were investigated and compared to that at three reference sites (Ref 1, Undist 1Undist 1, and Undist 2). Below, the turbidity data are discussed first. DOC, UV_{254}, SUVA, and NOM fractions (humic substances, building blocks, and biopolymers measured by LC-OCD) data are presented next, followed by the DBP-FP data.

4.1 Turbidity

Turbidity is critical to optimizing and evaluating overall treatment system performance, especially in conventional surface water treatment plants (MWH, 2012). Turbidity levels observed in the streams draining the study watersheds were all very low (Table 4-1 and Figure 4-1) and typical of high quality forested headwaters regions of the eastern slopes of the Rocky Mountains (Silins *et al*., 2009; Emelko *et al.*, 2011). Forest harvesting disturbances have been widely reported to increase turbidity in streams draining impacted watersheds (Yusop & Suki, 1994; Webb & Haywood, 2004; Basher *et al*., 2011; Lewis *et al*., 2019). The turbidities observed over a range of flow conditions in the streams draining strip cut-, and partial cut-impacted watersheds were not statistically different (i.e., not ranked differently) than those draining the reference watershed (U = 112, p = 1; and U = 41.5, p = 0.238 respectively). Stream turbidities in the undisturbed watersheds at locations downstream of those used in the present investigation indicated that turbidities in those streams also were generally similar to one another in the years prior to harvesting (Appendix C). Although the statistical test results for clear cut-impacted and reference watershed indicated a statistically significant difference (U = 57.5, p = 0.012), the low and relatively consistent stream turbidities that were observed across all watersheds would not pose challenges to conventional surface water treatment (MWH, 2012). Thus, the results from the landscape level monitoring (i.e., synoptic sampling) presented herein suggest that the contemporary forest harvesting that was conducted with BMPs for erosion control did not have appreciable impacts on stream turbidity (Figure 4-1). Comparison of post-harvesting data from streams draining impacted watersheds to downstream pre-disturbance data is neither ideal nor useful for comparison in absence of other data; however, the general similarities in these data further support the observations from the

synoptic sampling. Moreover, these results are notably consistent with the high frequency, focused suspended sediment and turbidity data reported in the study catchments during the same period by Corrigan (2017). Thus, these collective data are compelling and suggest no meaningful impact of forest harvesting on receiving stream turbidity in the present investigation.

Table 4-1 Stream turbidity values in study watersheds.

Location	Turbidity (NTU)		
	Median	mean	SD
Ref 1	0.30	0.37	0.23
Undist 1	1.05	2.12	3.21
Undist 2	1.03	1.07	0.42
Clear cut	0.51	0.58	0.26
Strip cut	0.30	0.33	0.2
Partial cut	0.37	0.8	0.9

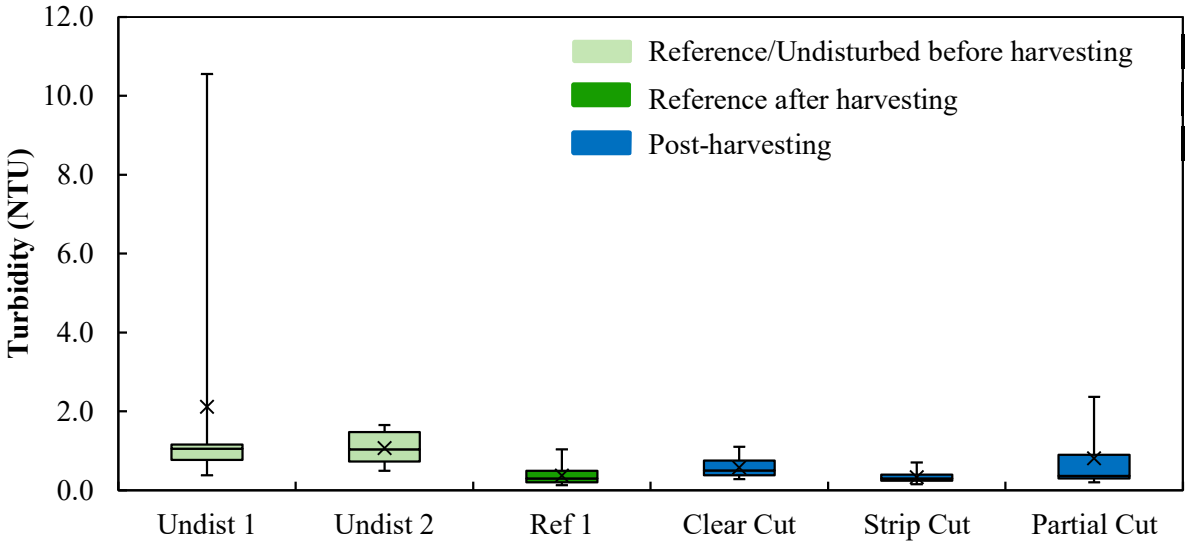

Figure 4-1 Turbidity in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed or reference watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations. The horizontal bar within the boxes is the median value, the bottom and top of each box indicates the 25th and 75th percentiles respectively. The crosses indicate mean values, and the top and bottom of whiskers respectively represent the maximum and minimum values observed.

4.2 DOC

DOC is an indicator of NOM in source waters and is one of the key drivers in designing drinking water treatment plants (Emelko *et al.*, 2011; MWH, 2012). DOC concentrations in the streams draining the study watersheds were all very low (Table 4-2 and Figure 4-2,) and typical of high quality, low DOC forested headwaters regions of the eastern slopes of the Rocky Mountains (Alberta Environment, 2007; Emelko *et al.*, 2011; Chung *et al*., 2019) (Figure 4-3). As discussed in Section 2.1.1, forest harvesting has frequently resulted in elevated DOC concentrations in streams draining impacted watersheds (Moore, 1989; Neal *et al*., 1992; Cummins & Farrell, 2003a and b; Nieminen, 2004; Tetzlaff *et al*., 2007; Löfgren *et al*., 2009; Laudon *et al*., 2009; Schelker *et al*, 2012; Eklöf *et al*., 2014; Kiikkilä *et al*., 2014; Nieminen *et al*., 2015), though it also has resulted in decreased stream DOC concentrations due to increased soil infiltration capacity and associated DOC penetration to deeper soil layers in some cases (Moore & Jackson, 1989; Cawley *et al*., 2014; Palviainen *et al*., 2014; Musetta-Lambert *et al*., 2017). In the present investigation, the DOC concentrations observed over a range of flow conditions in the stream draining the clear cut-impacted watersheds were not statistically different from those draining the reference watershed (U = 82.5, p = 0.140). The results, however, were statistically significant for streams draining the strip cut- and partial cut-impacted watersheds (U = 63.5, p = 0.041; and U = 7, $p < 0.001$ respectively). Consistent with the turbidity results reported above, stream DOC concentrations in the undisturbed watersheds at locations downstream of those used in the present investigation indicated that DOC concentrations in those streams also were generally similar to one another in the years prior to harvesting. The low and relatively consistent stream DOC concentrations that were observed herein would not pose any challenges to conventional surface water treatment (MWH, 2012). Thus, the results from the landscape level monitoring (i.e., synoptic sampling) presented herein suggest that the contemporary forest harvesting that was conducted with BMPs for erosion control did not have appreciable impacts on stream DOC concentrations.

Table 4-2 Stream DOC concentrations in study watersheds.

Location	DOC (mg/L)		
	Median	**mean**	**SD**
Ref 1	0.7	0.84	0.53
Undist 1	0.79	0.86	0.34
Undist 2	0.98	1.04	0.4
Clear cut	0.81	1.19	0.81
Strip cut	0.98	1.38	0.98
Partial cut	1.67	2.33	1.91

During May 2018, stream DOC values were relatively elevated at 3.45, 4.15, 4.79, and 7.38 mg/L in Ref 1, clear cut, strip cut, and partial cut watersheds, respectively (the points are highlighted in red on Figure 4-4)—samples were not collected in the Undist 1 and Undist 2 watersheds during this period. While there is no reason to exclude these data from the overall analysis, this dataset was excluded from the statistical analysis to evaluate its relative impact on the results. Eliminating these values from the data set resulted in slightly lower mean stream DOC concentrations of 0.80±0.26 mg/L, 0.99±0.28 mg/L, 1.14±0.37 mg/L, and 1.61±0.09 mg/L in the Ref 1, clear cut, strip cut, and partial cut watersheds, respectively. Notably, the exclusion of these data from the analysis did not impact the conclusions drawn from the data (Table 4-3 and Table 4-4).

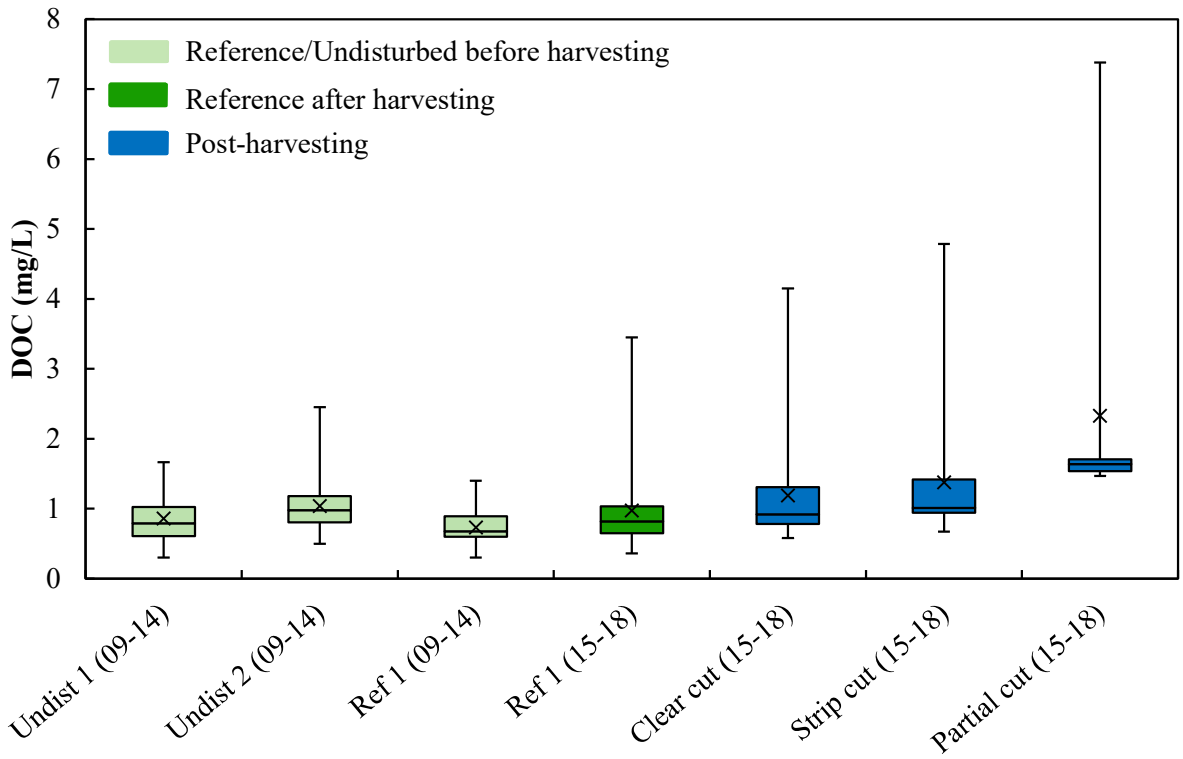

Figure 4-2 DOC concentrations in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed or reference watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations. The horizontal bar within the boxes is the median value, the bottom and top of each box indicates the 25th and 75th percentiles respectively. The crosses indicate mean values, and the top and bottom of whiskers respectively represent the maximum and minimum values observed.

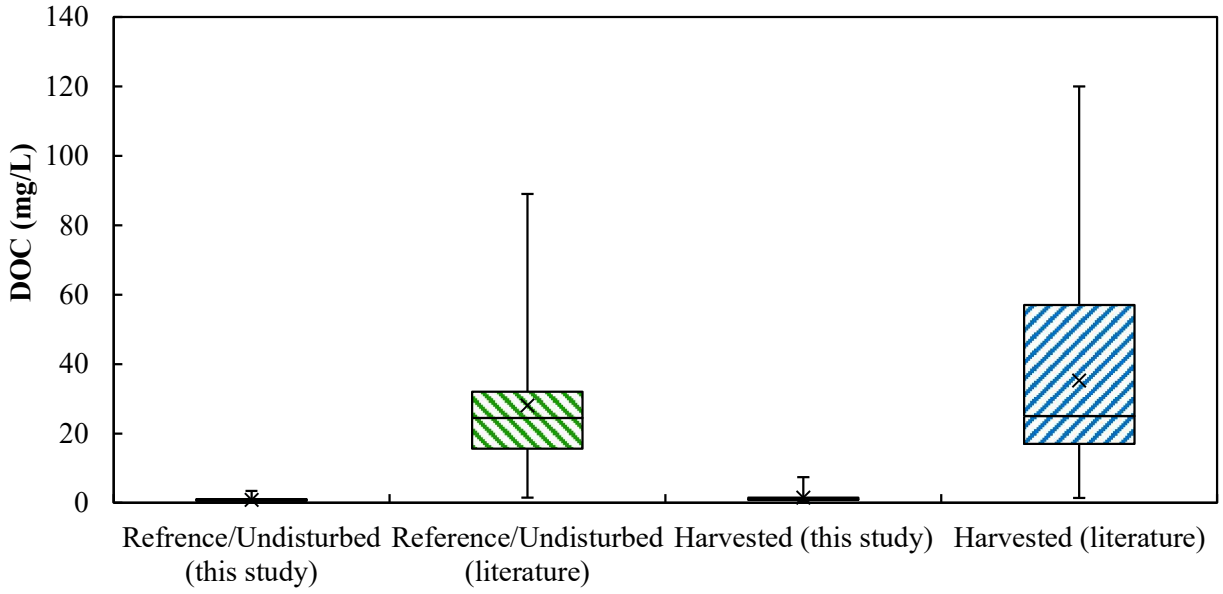

Figure 4-3 Comparison between DOC concentrations in streams in reference/undisturbed and harvested sites from this study and values reported in literature (Adapted from Table 2-2 and Table 2-3). Green represents reference/undisturbed locations and blue represents harvested locations.

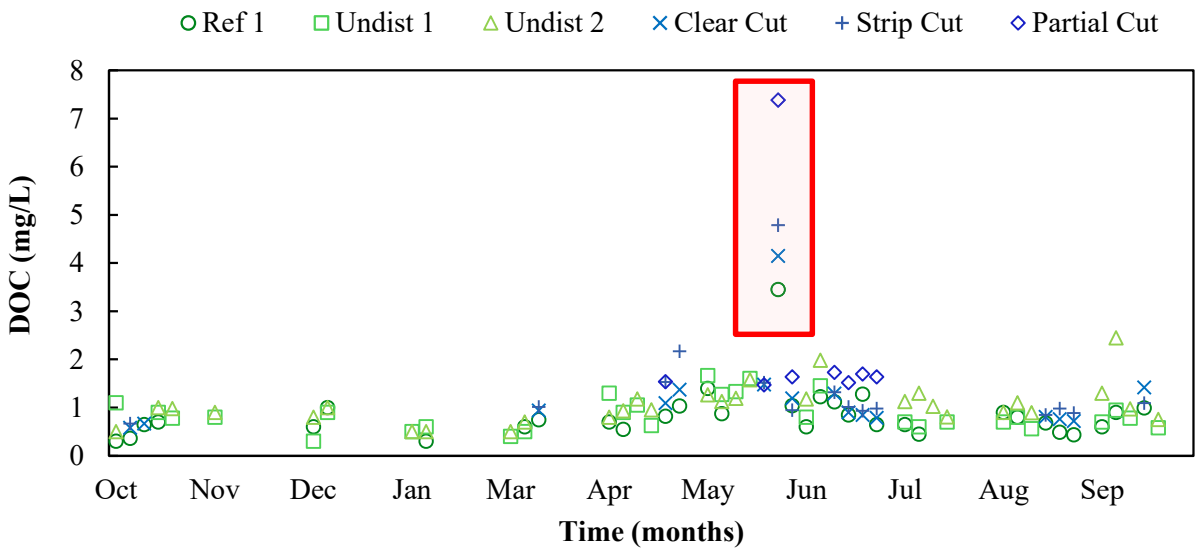

Figure 4-4 DOC concentration time series data from October 2017 to September 2018 for reference, undisturbed and harvested locations. Green represents reference/undisturbed locations and blue represents harvested locations.

Table 4-3 Comparison of stream DOC concentrations in reference/undisturbed and harvested watersheds using all of the collected DOC data.

	U	*p*-value
Ref 1 (2015-18) vs. Clear cut (2015-18)	82.5	0.14
Ref 1 (2015-18) vs. Partial cut (2015-18)	7	0.000
Ref 1 (2015-18) vs. Strip cut (2015-18)	63.5	0.04
Undist 1 (2013-14) vs. Clear cut (2015-18)	47	0.17
Undist 1 (2013-14) vs. Partial cut (2015-18)	3	0.001
Undist 1 (2013-14) vs. Strip cut (2015-18)	36	0.06
Undist 2 (2013-14) vs. Clear cut (2015-18)	73	0.74
Undist 2 (2013-14) vs. Partial cut (2015-18)	3	0.000
Undist 2 (2013-14) vs. Strip cut (2015-18)	62	0.50

Table 4-4 Comparison of stream DOC concentrations in reference/undisturbed and harvested watersheds excluding the May 2018 data.

	U	*p*-value
Ref 1 (2015-18) vs. Clear cut (2015-18)	67.5	0.10
Ref 1 (2015-18) vs. Partial cut (2015-18)	0	0.000
Ref 1 (2015-18) vs. Strip cut (2015-18)	49.5	0.02
Undist 1 (2013-14) vs. Clear cut (2015-18)	47	0.24
Undist 1 (2013-14) vs. Partial cut (2015-18)	3	0.001
Undist 1 (2013-14) vs. Strip cut (2015-18)	36	0.10
Undist 2 (2013-14) vs. Clear cut (2015-18)	63	0.53
Undist 2 (2013-14) vs. Partial cut (2015-18)	3	0.001
Undist 2 (2013-14) vs. Strip cut (2015-18)	62	0.67

4.3 UV_{254} and SUVA

UV absorption by organic compounds is one of the simplest and most useful methods that enable real-time monitoring of organic matter in water (Sławomir *et al.*, 2017). The adverse impacts of various landscape disturbances on receiving stream UV_{254} and DOC aromaticity have been reported (*e.g.*, Elbag, 2006; Wade *et al.*, 2013; Writer *et al.*, 2014; Hohner *et al.*, 2016; Shams, 2018) (Figure 4-6). Although, significant differences between Ref 1 and each of harvested sites ($U = 33.5$, $p < 0.001$; $U = 31$, $p = 0.000$; and $U = 4$, $p < 0.001$ for clear cut, strip cut, and partial cut watersheds, respectively) were observed, the UV_{254} values were low and consistent with high quality source waters in undisturbed watersheds in the region (Table 4-5 and Figure 4-5). The UV_{254} data from the present investigation are compared to those reported in the available literature and demonstrate the high quality of surface water in the study watersheds (Figure 4-6).

Table 4-5 Stream UV_{254} values in the study watersheds.

Location	UV_{254} (m^{-1})		
	Median	**mean**	**SD**
Ref 1	1.09	1.22	0.63
Undist 1	1.35	1.76	1.26
Undist 2	1.68	2.02	0.97
Clear cut	1.62	1.77	0.45
Strip cut	1.85	1.98	0.66
Partial cut	3.44	3.48	1.93

Notably, although stream DOC concentrations and UV_{254} values in the partial cut watershed were higher than those observed in the other watersheds, specific ultraviolet absorbance (SUVA) was similar across all of the study watersheds (Figure 4-7). Stream SUVA values were not significantly different between Ref 1 and each of the clear cut, strip cut, and partial cut watersheds ($U = 95$, $p = 0.22$; $U = 85$, $p = 0.22$; and $U = 33.5$, $p = 0.08$; respectively). The SUVA data collected during this investigation are summarized in Table 4-6. The generally low SUVA values observed in all of the study watersheds (Table 4-8), suggest that stream DOC concentrations in these watersheds would not be a key driver of coagulant dosing requirements during conventional drinking water treatment during which only a small fraction of DOC would likely be removed during coagulation (Edzwald & Benschoten, 1990; Sohn *et al.*, 2007). Furthurmore, the poor correlation between stream DOC concentration and SUVA (R^2=0.44) (Table 4-17) demonstrates that SUVA is not an informative indicator of DOC chemical composition in the Star Creek study sites. This observation is consistent with Weishaar *et al*. (2003), who suggested that the reactivity od DOC molecules with similar properties and compositions can vary considerably.

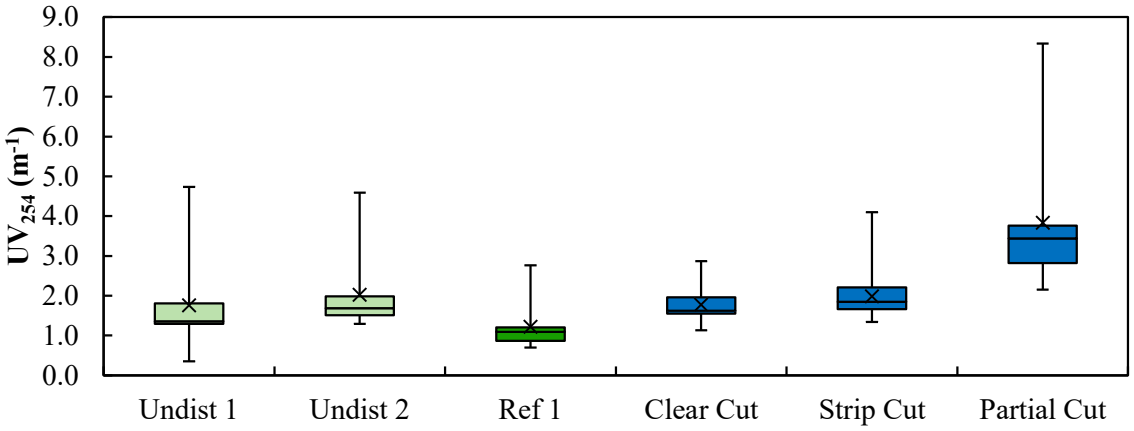

Figure 4-5 UV_{254} absorbance in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations. The horizontal bar within the boxes is the median value, the bottom and top of each box indicates the 25th and 75th percentiles respectively. The crosses indicate mean values, and the top and bottom of whiskers respectively represent the maximum and minimum values observed.

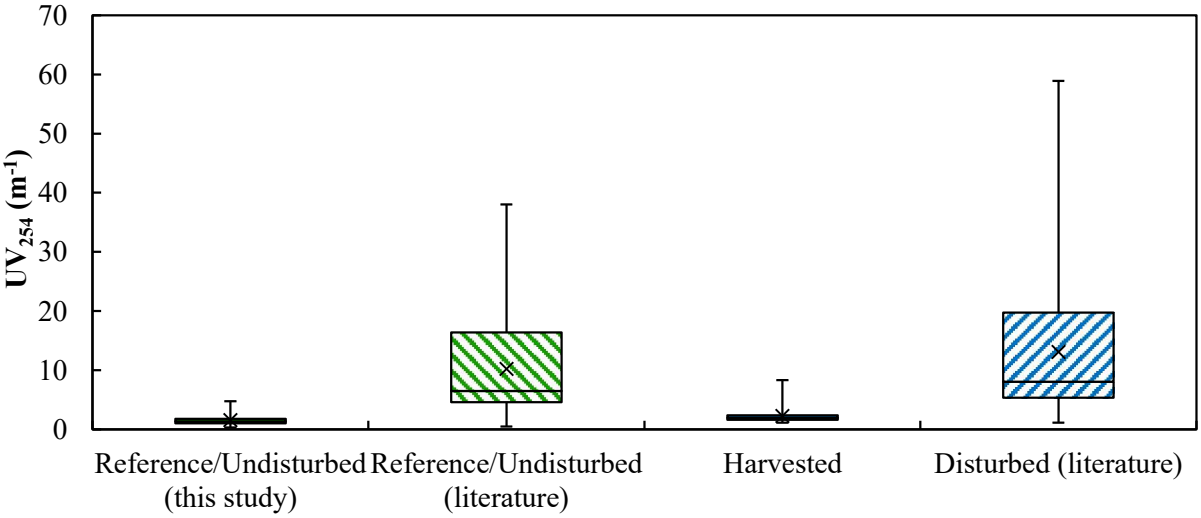

Figure 4-6 Comparison between stream UV_{254} in reference/undisturbed and harvested watersheds in this study and those reported in literature (Adapted from Table 2-5).

Table 4-6 Stream SUVA values in the study watersheds.

Location	SUVA (L/mg.m)		
	Median	mean	SD

Ref 1	1.12	1.4	0.6
Undist 1	1.88	1.88	0.67
Undist 2	1.89	1.93	0.51
Clear cut	1.71	1.62	0.59
Strip cut	1.58	1.6	0.57
Partial cut	1.67	1.76	0.49

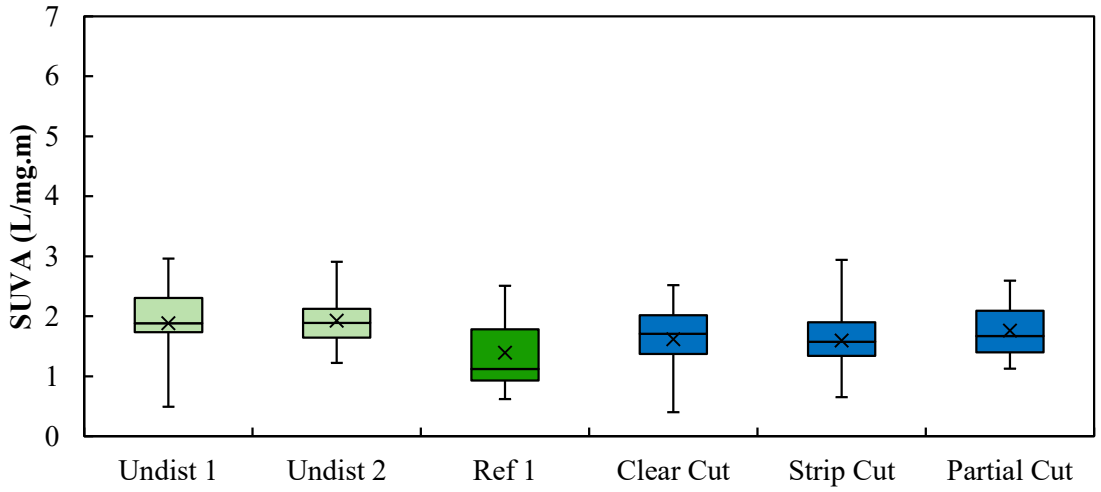

Figure 4-7 SUVA levels in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations. The horizontal bar within the boxes is the median value, the bottom and top of each box indicates the 25th and 75th percentiles respectively. The crosses indicate mean values, and the top and bottom of whiskers respectively represent the maximum and minimum values observed.

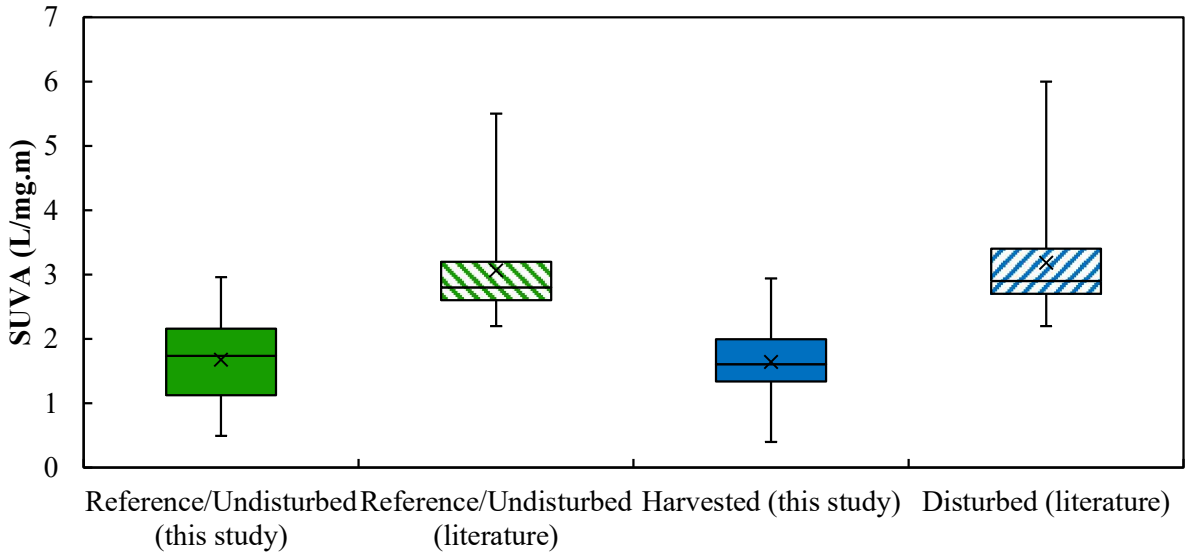

Figure 4-8 Comparison between stream SUVA levels in reference/undisturbed and harvested sites in this study and those reported in literature (Adapted from Table 2-5). Green represents reference/undisturbed locations and blue represents harvested locations.

4.4 Carbon fractions as measured by LC-OCD

The observed stream concentrations of the humic substances, biopolymers, building blocks, LMW acids, and LMW neutrals fractions of DOC in the study watersheds are summarized in Table 4-8 to Table 4-12, as well as shown in Figure 4-9 to Figure 4-13. The concentrations of these fractions of DOC observed over a range of flow conditions in the streams draining the clear cut- and strip-cut impacted watersheds were not statistically different from those draining the reference watershed (Table 4-7). In contrast, the observed concentrations of the humic substances, building blocks, LMW acids, and LMW neutrals fractions of DOC in the partial-cut watershed were statistically different from those in the reference/unharvested watershed (Table 4-7). As mentioned above, the low and relatively consistent stream DOC concentrations that were observed herein would not be expected to pose challenges to conventional surface water treatment (MWH, 2012). Although key concentration threshold values for the various fractions of DOC have not been identified, the observed concentrations of each of these fractions were presumably also low. Notably, reporting the DOC fractions (e.g., humic substances) as a percentage of the overall DOC concentration shows that they were generally consistent

across all of study watersheds (Figure 4-14), which would be expected across similar sub-watersheds within a given basin. Collectively, these data suggest that DOC character and composition did not vary greatly between the study watersheds.

Table 4-7 Comparison of stream concentrations of humic substances, biopolymers, and building blocks fractions of DOC in reference/undisturbed and harvested watersheds.

		Clear cut		Strip cut		Partial cut	
		U	*p*-value	*U*	*p*-value	*U*	*p*-value
Ref 1	Humic substances	8	0.25	9	0.33	0	0.02
	Biopolymers	8	0.48	9	0.90	7	0.89
	Building Blocks	6	0.13	10	0.43	1	0.03
	LMW Acids	2	0.11	4	0.19	NA	NA
	LMW neutrals	16	0.81	7	0.09	3	0.16

Table 4-8 Stream concentrations of the humic substances fraction of DOC in the study watersheds.

Location	**Humics (mg/L)**		
	Median	**mean**	**SD**
Ref 1	0.31	0.38	0.3
Undist 1	0.36	0.51	0.38
Undist 2	0.53	0.57	0.17
Clear cut	0.67	0.66	0.27
Strip cut	0.70	0.65	0.31
Partial cut	1.14	1.14	0.1

Table 4-9 Stream concentrations of the biopolymers fraction of DOC in the study watersheds.

Location	**Biopolymers (mg/L)**		
	Median	**mean**	**SD**
Ref 1	0.005	0.006	0.003
Undist 1	0.010	0.012	0.007
Undist 2	0.010	0.010	0.002
Clear cut	0.007	0.009	0.007
Strip cut	0.005	0.005	0.002
Partial cut	0.005	0.006	0.003

Table 4-10 Stream concentrations of the building blocks fraction of DOC in the study watersheds.

Location	Building blocks (mg/L)		
	Median	**mean**	**SD**
Ref 1	0.05	0.09	0.08
Undist 1	0.07	0.09	0.05
Undist 2	0.10	0.11	0.04
Clear cut	0.15	0.15	0.07
Strip cut	0.16	0.16	0.09
Partial cut	0.27	0.27	0.07

Table 4-11 Stream concentrations of the LMW acids fraction of DOC in the study watersheds.

Location	LMW Acids (mg/L)		
	Median	**mean**	**SD**
Ref 1	0.011	0.012	0.002
Undist 1	0.016	0.016	0.020
Undist 2	0.009	0.010	0.008
Clear cut	0.025	0.021	0.012
Strip cut	0.017	0.048	0.069
Partial cut	0.031	0.031	0.00

Table 4-12 Stream concentrations of the LMW neutrals fraction of DOC in the study watersheds.

Location	LMW Neutrals (mg/L)		
	Median	**mean**	**SD**
Ref 1	0.11	o.15	0.18
Undist 1	0.14	0.17	0.07
Undist 2	0.14	0.23	0.18
Clear cut	0.11	0.12	0.07
Strip cut	0.24	0.28	0.20
Partial cut	0.20	0.20	0.05

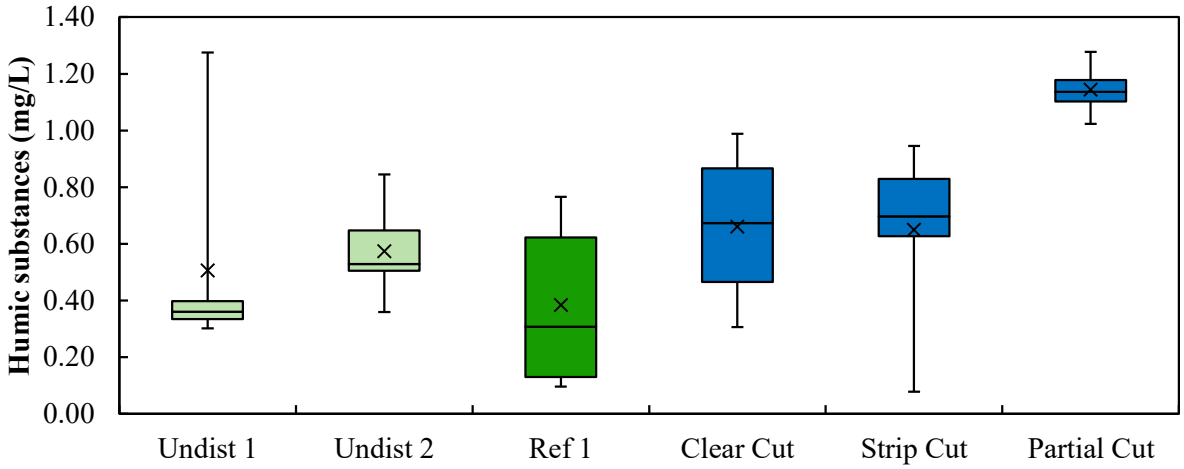

Figure 4-9 Humic substances fraction of DOC in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed or reference watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations. The horizontal bar within the boxes is the median value, the bottom and top of each box indicates the 25th and 75th percentiles respectively. The crosses indicate mean values, and the top and bottom of whiskers respectively represent the maximum and minimum values observed.

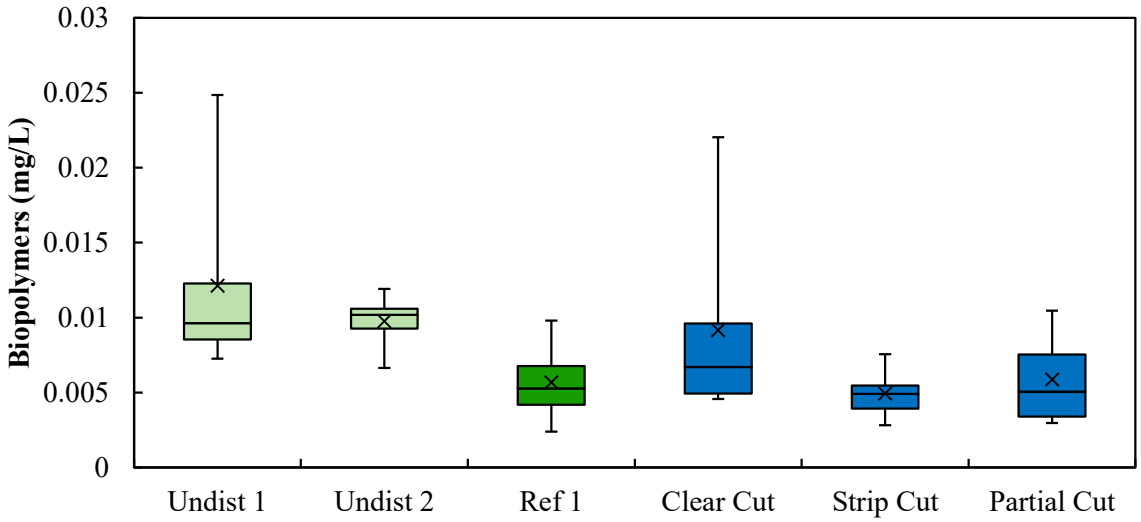

Figure 4-10 Biopolymers fraction of DOC in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed or reference watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations. The horizontal bar within the boxes is the median value, the bottom and top of each box indicates the 25th and 75th

percentiles respectively. The crosses indicate mean values, and the top and bottom of whiskers respectively represent the maximum and minimum values observed.

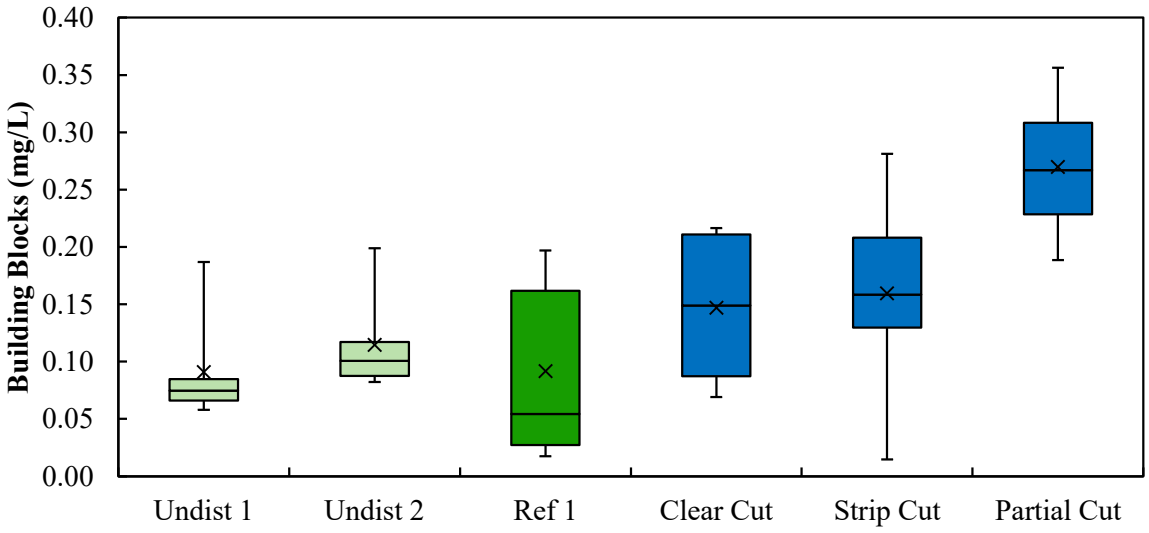

Figure 4-11 Building blocks fraction of DOC in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed or reference watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations. The horizontal bar within the boxes is the median value, the bottom and top of each box indicates the 25th and 75th percentiles respectively. The crosses indicate mean values, and the top and bottom of whiskers respectively represent the maximum and minimum values observed.

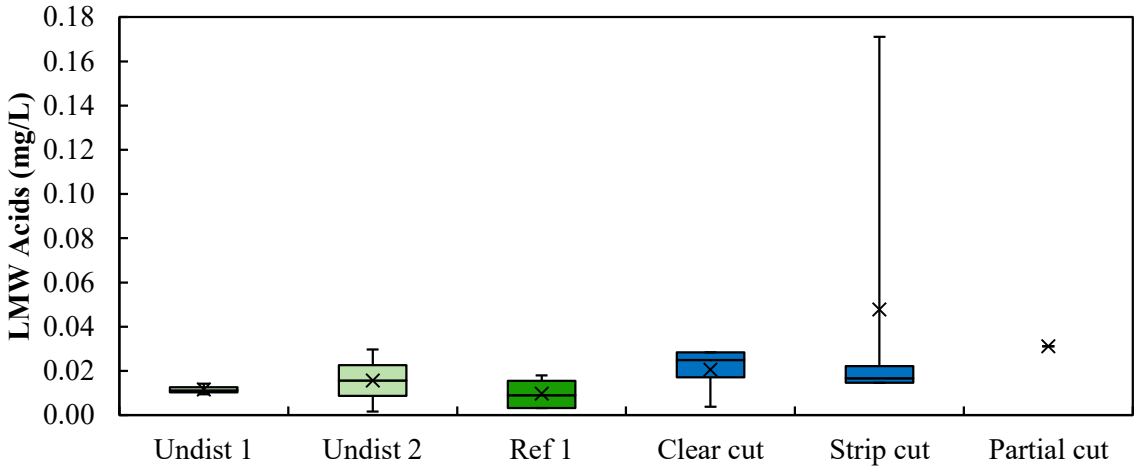

Figure 4-12 LMW acids fraction of DOC in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed or reference watersheds prior to harvesting (2015).

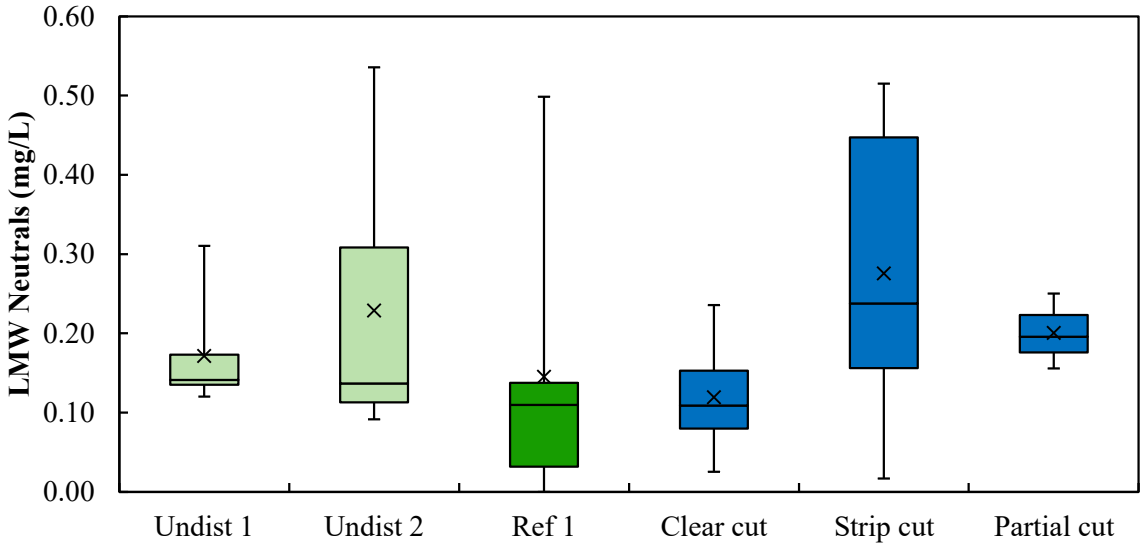

Figure 4-13 LMW neutrals fraction of DOC in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed or reference watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations. The horizontal bar within the boxes is the median value, the bottom and top of each box indicates the 25th and 75th percentiles respectively. The crosses indicate mean values, and the top and bottom of whiskers respectively represent the maximum and minimum values observed.

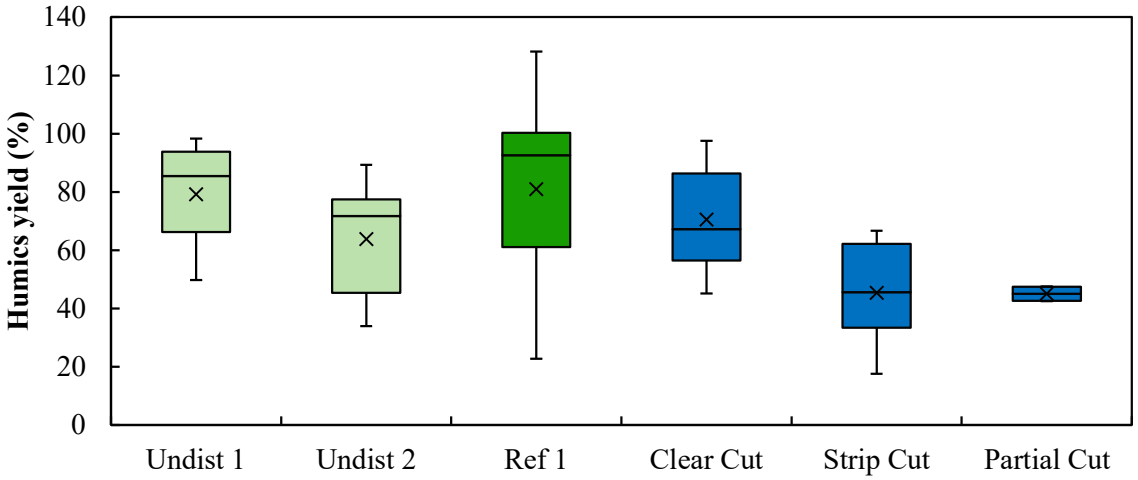

Figure 4-14 Normalized humic substances fraction of DOC in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed or reference watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations.

4.5 Disinfection By-Product Formation Potential

The mean total THM-FP and HAA-FP concentrations in the source waters are presented in Table 4-13, and Figure 4-15; and Table 4-14 and Figure 4-16, respectively. Similar to UV_{254}, significant differences between Ref 1 and each of the disturbed (i.e., clear cut, strip cut, and partial cut) watersheds were observed, with $U = 43$, $p = 0.01$; and $U = 20$, $p = 0.000$, and $U = 1$, $p = 0.000$, respectively. In contrast, The HAA-FP results were similar to those observed for DOC, with $U = 57$, $p = 0.065$; and $U = 43$, $p < 0.001$, and $U = 0$, $p = 0.01$, respectively (similar to DOC). Although some of these differences between the disturbed and reference watersheds were significant, it is critical to recognize that all of the DBP-FP observations were relatively low and not of practical concern. Moreover, especially recalling the hyperchlorination associated with the FP analysis that would result in greater DBP formation than what would be observed at operational relevant applied chlorine doses, the data herein suggest that forest harvesting did not result in any practically relevant changes in DBP-FPs. Indeed, DBP formation was also examined using the simulated distribution system (SDS) method (Table 2-6). These results demonstrate that forest harvesting, as implemented in this study, would not pose any challenges to drinking water treatment. Moreover, Figure 4-17 and Figure 4-18 compare the concentrations of THM- and HAA-FP measured in this study relative to concentrations of these compounds that have been

reported in literature. While the first figure illustrates THM-FP yields from this study relative to rivers and streams that are drinking water sources globally, the latter represents the regulated DBP-FP of this study relative to all Canadians drinking water sources, including lakes and groundwater aquifers. It is evident from these two figures that the study sites investigated herein represent systems with the highest quality of source water; thus, it can be concluded that forest harvesting did not meaningfully impact drinking water treatability in any of harvested watersheds.

The poor correlations between SUVA and THM- and HAA-FP observed herein (R^2= 0.60, and 0.52, respectively) were in a general agreement with the reported litrature (e.g. Weishaar *et al.*, 2003; Hua *et al.*, 2015), suggesting that SUVA is a weak indicator of regulated DBP-FP. In contrast, the fluctuations in THM- and HAA-FP concentrations correlated well with the changes in UV_{254} values (R^2=0.91 and 0.90, respectively) (Table 4-17). This observation is consistent with the widely reported literature on utilizing UV_{254} as a predictor for regulated DBP-FPs (Singer *et al.*, 1981; Edzwald *et al.*, 1985; Reckhow *et al*., 1990; Wassink *et al*., 2011; Awad *et al*., 2016).

Table 4-13 Stream THM-FPs in the study watersheds.

Location	**THM-FP (µg/L)**		
	Median	**mean**	**SD**
Ref 1	16.50	19.31	7.30
Undist 1	32.00	36.33	35.83
Undist 2	27.00	23.58	13.80
Clear cut	26.00	28.33	8.73
Strip cut	31.00	32.67	10.10
Partial cut	50.00	65.86	45.90

Table 4-14 Stream HAA-FPs in the study watersheds.

Location	**HAA-FP (µg/L)**		
	Median	**mean**	**SD**
Ref 1	24.40	29	11.68
Undist 1	42.00	42	-
Undist 2	39.00	39	-
Clear cut	35.20	39.59	17.95
Strip cut	34.00	42.10	15.71
Partial cut	64.00	92.89	73.20

Chloroform, bromodichromethane (BDCM), dibromochloromethane (DBCM), and bromoform are the most abundant groups of THMs in drinking water. As only trace concentrations of bromide were found in the study watersheds, THMs primarily consisted of chloroform across the study sites and. concentrations of DBCM and bromoform were typicallybelow detection limits. The mean percentage of formation potentials of chloroform and BDCM for study locations were equal to 98±3%, and 5±2%, respectively (Table 4-15). Similarly, brominated HAAs were not formed, and HAAs were comprised of 63±7% of trichloroacetic acids and 35±2% of dichloroacetic acids (Table 4-16).

The results of this study have shown no impact of forest harvesting on water quality and treatability during and 3 years post-harvesting in the eastern slopes of the Rocky Mountains, southwestern Alberta. Current contemporary forestry practices range from the creation and expansion of protected areas, where any type of anthropogenic disturbance including forest harvesting with the goal of mitigating potential wildfire or other disturbance risks is prohibited, to integrated forest management employing BMPs to mitigate impacts on water. While, the impacts of employing SWP strategies on water quality and treatability is not very well understood, this work suggests that contemporary forest harvesting with the careful implementation of BMPs will not have any impact on water quality and treatability.

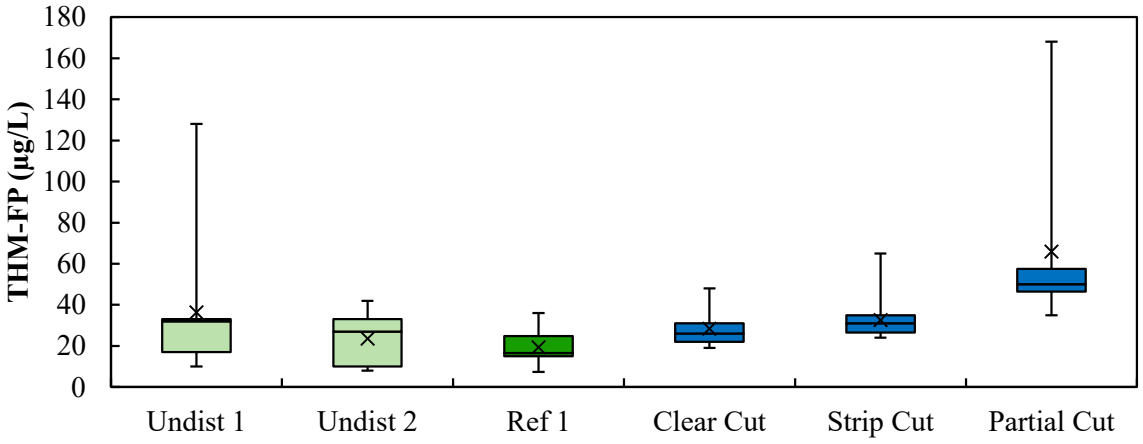

Figure 4-15 THM-FPs concentration in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed or reference watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations. The horizontal bar within the boxes is the median value, the bottom and top of each box indicates the 25th and 75th percentiles respectively. The crosses indicate mean values, and the top and bottom of whiskers respectively represent the maximum and minimum values observed.

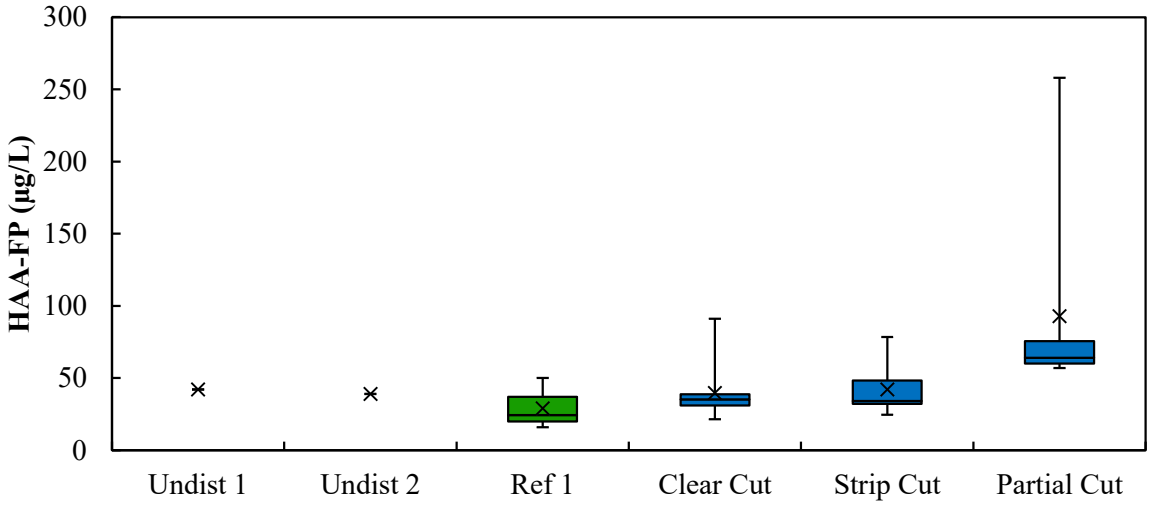

Figure 4-16 HAA-FPs concentrations in streams draining adjacent undisturbed, reference and harvested watersheds. Light green represents data from undisturbed or reference watersheds prior to harvesting (2015); dark green represents data collected after 2015. Blue shading represents data from harvested locations. The horizontal bar within the boxes is the median value, the bottom and top of each box indicates the 25th and 75th percentiles respectively. The crosses indicate mean values, and the top and bottom of whiskers respectively represent the maximum and minimum values observed.

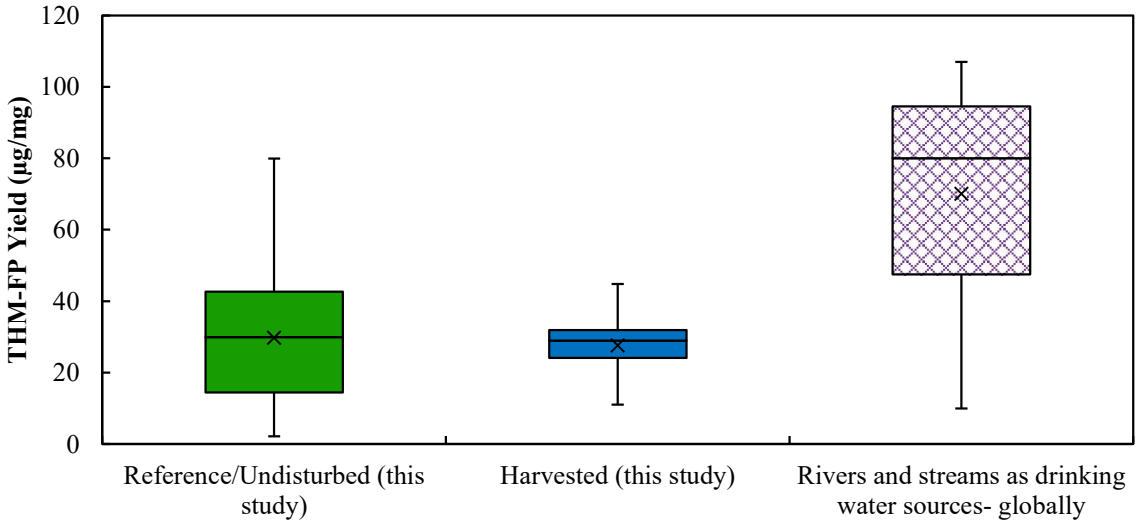

Figure 4-17 Comparison of stream THM-FP yields in this study and those in streams and rivers serving as drinking water sources globally (Adapted from Table 2-5).

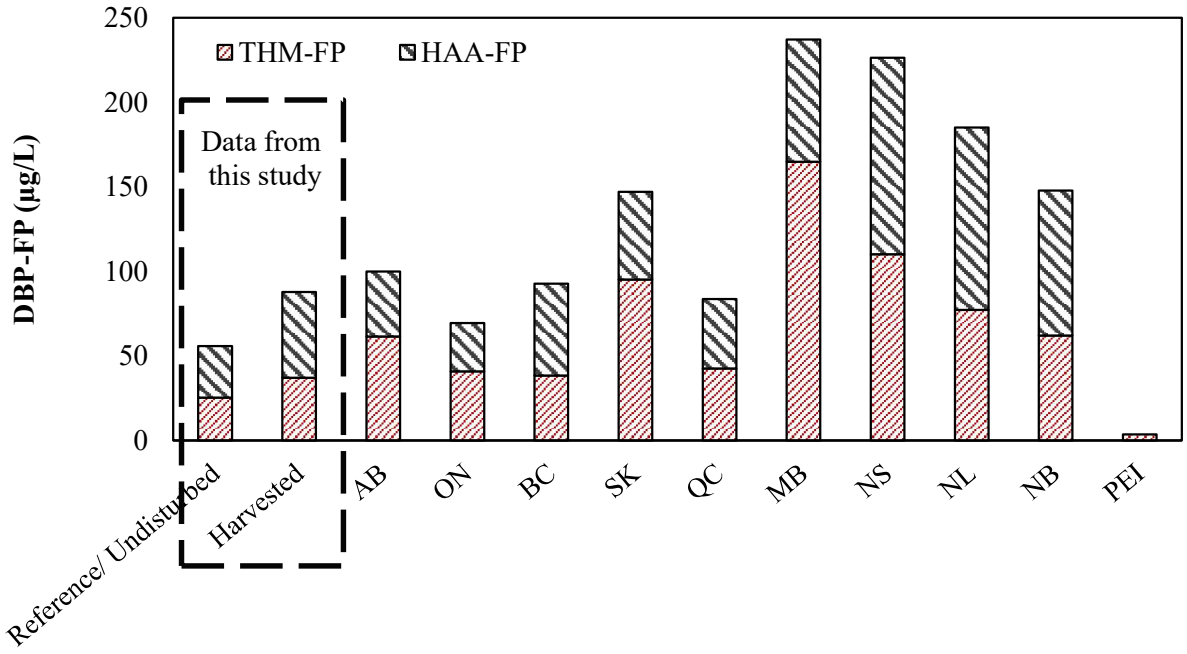

Figure 4-18 Comparison of stream DBP-FPs in this study and Canadian drinking water sources on average (Adapted from Ahmad & Husain, 2015).

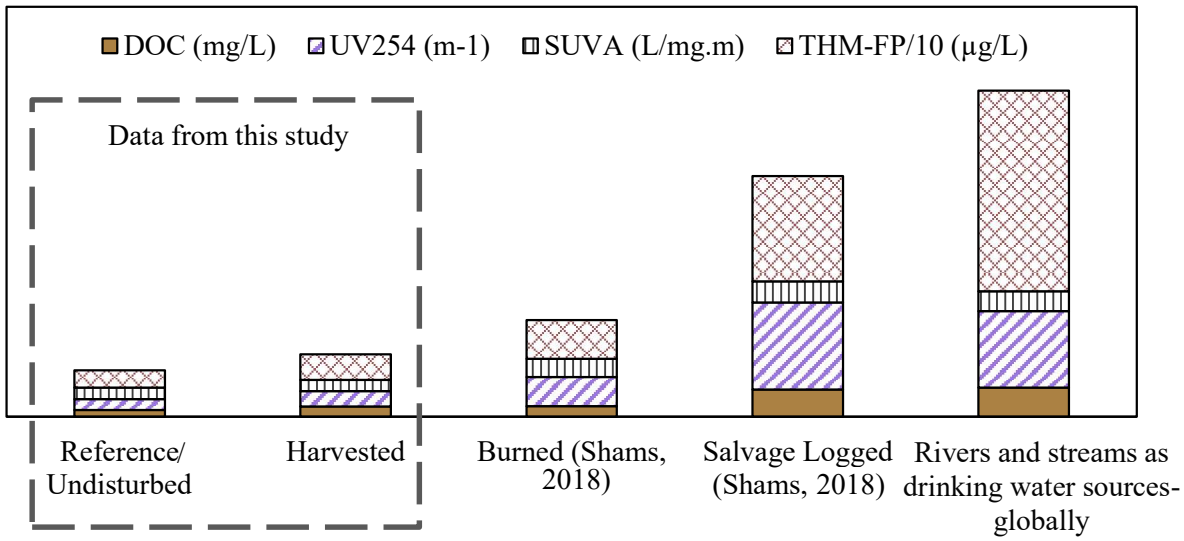

Figure 4-19 Comparison among drinking water quality and treatability parameters measured during this study, burned and salvage logged SRWP research watersheds (Adapted from Shams, 2018), and rivers and streams serving as drinking water sources globally (Adapted from Table 2-5).

Table 4-15 THM FP constituents by percentage mass.

	Average Percentage of THMs			
	Chloroform		Bromodichromethane	
	mean	SD	mean	SD
Ref 1	97	3	6	1
Clear cut	98	3	6	3
Strip cut	98	2	4	0
Partial cut	98	2	3	0
Overall	98	3	5	2

Table 4-16 HAA FP constituents by percentage of mass.

	Average Percentage of THMs			
	Trichloroacetic Acid		Dichloroacetic Acid	
	mean	SD	mean	SD
Ref 1	65	3	35	3
Clear cut	59	8	36	2
Strip cut	62	8	34	2
Partial cut	66	1	34	1
Overall	63	7	35	2

Table 4-17 Correlation between various metrics of aqueous NOM and DBP-FPs.

		DOC	UV_{254}	SUVA	Humics	THM-FP
UV_{254}	R^2	0.84				
	p-value	1.84E-30				
SUVA	R^2	0.44	0.72			
	p-value	9.42E-11	2.70E-21			
Humic Substances	R^2	0.96	0.86	0.66		
	p-value	4.72E-24	7.58E-15	5.52E-09		
THM-FP	R^2	0.76	0.92	0.60	0.81	
	p-value	1.83E-22	2.79E-36	5.35E-15	4.27E-11	
HAA-FP	R^2	0.81	0.92	0.52	0.88	0.97
	p-value	4.31E-20	1.24E-27	1.57E-09	1.45E-09	2.50E-40

5 Conclusions and Implications

The focus of this research was to investigate the impacts of contemporary forest harvesting on drinking water source quality and treatability. To achieve this goal, three types of contemporary forest harvesting (clear-cut with patch retention, strip-shelterwood cut, and partial cut) were investigated at the watershed-scale in the eastern slopes of the Rocky Mountains of southwestern Alberta, Canada. Reference and harvested stream water turbidity and various water quality metrics related to NOM (and associated DBP formation potentials) were evaluated during and over the first three years after forest harvesting. Reported pre-disturbance data from the study watersheds were included in this analysis. Finally, the correlations between THM- and HAA-FPs and several proxy indicators (particularly, aromaticity) were investigated. The following conclusions were drawn from this research:

1. **Contemporary forest harvesting (clear cut with patch retention, strip-shelterwood cut, and partial cut) with careful implementation of BMPs for erosion control did not yield any substantial impacts on drinking water treatability as measured by turbidity or aqueous NOM (i.e., DOC, UV_{254}, humic substances, biopolymers, and building blocks).**

   The present investigation is the first to demonstrate that forest harvesting can be conducted without compromising drinking water source quality and treatability. These reported results—the lack of substantial impacts of forest harvesting on DOC concentration/character and turbidity—provide an important contrast to many historical investigations of forest harvesting impacts on water, which generally suggest that some extent of water quality deterioration (i.e., relatively elevated NOM and/or turbidity) in receiving streams, which can last for years or longer, can be expected after forest harvesting. While the results reported herein comprise a synoptic evaluation of water quality and treatability, they are critically consistent and connected with the findings of Shams (2018), who reported that DOC and turbidity were generally elevated at higher stream flows in the study region; and Corrigan (2017), who showed that the combined impact of the rapid harvesting and road decommissioning on suspended solids and turbidity was largely negligible in the study watersheds, and turbidity was often higher at locations just upstream of forest harvesting. Collectively, these

three investigations compellingly demonstrate that forest harvesting can be conducted without compromising drinking water source quality and treatability.

2. **Contemporary forest harvesting approaches coupled with state-of-the-art BMPs for erosion control show promise as "green" SWP technologies for mitigating severe climate change-associated disturbance risks to drinking water treatability, even in source water regions rich with glacially-derived fine sediments, such as those found in many parts of western North America.**

   One of the greatest potential threats of forest harvesting to drinking water treatability is the potential release of bioavailable phosphorus-enriched fine sediments that can promote the proliferation of microorganisms, and especially potentially toxin-forming cyanobacteria and other algae that may produce taste and odor compounds. As shown by Emelko *et al*. (2016), these fine sediments may remain in source waters and untreated water reservoirs for many years, in some cases—most drinking water reservoirs are not designed to manage these fine sediments to mitigate these threats, which can lead to treatment challenges, service disruptions, and in the most severe cases, water outages. The present investigation provides the required linkage to Corrigan (2017) and Shams (2018) to demonstrate that not only can forest harvesting operations can be conducted without compromising drinking water source quality and treatability (Conclusion #1 above), but they can be conducted in a manner (detailed by Corrigan, 2017) that minimizes and essentially prevents any meaningfully disturbance-associated threats to drinking water treatability. Thus, this work highlights a starting point for the consideration of forest management approaches and important associated BMPs for implementation as "green" SWP technologies and climate change adaptation strategies. Further, it has been suggested that "on average, every $1 spent on source-water protection saved an average of $27 in water treatment costs" (Winiecki, 2012). By demonstrating that forest harvesting can be conducted without adverse impacts on water quality and treatability, this work demonstrates that the use of harvesting as a forest management tool *for promoting forest health* can be applied to produce water that is less expensive to treat, transport, and store. Given this impact and the associated societal co-benefits (*e.g.*, recreational use, habitat, etc. as described in Gartner *et al.*, 2014) suggests that such active approaches for protecting/managing forested watersheds can yield significant economic benefits for drinking water utilities.

3. **The development of forest management-based SWP approaches/technologies requires significant investment and a paradigm shift within the drinking water industry. Specifically, to maximize the impacts of forest management-based approaches to SWP and to develop climate change adaptation strategies—in lieu of traditional landscape-level, time series trend monitoring—focused paired catchment investigations that are designed as before-after-control-impact (BACI) studies are urgently needed.**

Traditionally, the drinking water industry has advocated landscape-level, time series trend monitoring investigations that provide a "snapshot" of hydrological conditions in a watershed. As described by Neary (2016), these types of investigations are well-suited for determining the efficacy of protection or restoration activities; however, they are frequently inadequate for discerning hydrological processes and their causes because factors such as hydrograph time resolution, sampling frequency, climate variability, stream gauge accuracy, and mixed land uses (among others) preclude accurate trend detection amidst the "noise" or natural variability associated with watershed-scale investigations. Thus, traditional landscape-level time series trend monitoring approaches are inherently inadequate for demonstrating forest management impacts on source water quality and treatability. This point is punctuated by the present investigation.

Here, a landscape-scale time series trend monitoring experiment was conducted. The headwaters sampling locations were remote and challenging to access; nonetheless, an effort was made to collect samples across a representative range of stream flow conditions (i.e., baseflows, stormflows, freshet) to capture not only shifts in the baseline values of the water quality and treatability metrics investigated, but also changes in their variability. It should be emphasized that in the absence of pre-disturbance data, water quality and treatability (as described by aqueous NOM concentration and character) in streams draining harvesting-impacted watersheds appeared deteriorated (i.e., statistically) relative to the reference stream in at least some aspects of water quality. Critically, however, the range of both NOM-associated and turbidity values observed during the investigation suggested that none of the source water matrices investigated herein would pose any meaningful challenges to conventional drinking water treatment. Moreover, the NOM-associated and turbidity values reported herein were collected over a representative range of flow conditions—the low range of observed values (regardless of metric evaluated) that were typical of the of high quality forested headwaters regions of the eastern slopes of the Rocky Mountains suggest that water quality and

variability evaluated during the present investigation fluctuated within the range of natural variability.

The range of natural variability in the study region was further supported by comparing the data collected during the present investigation to available, published pre-disturbance data from nearby undisturbed (at the time, in the case of Star Creek) watersheds in which samples were collected at downstream locations relative to the sampling locations utilized herein. Finally, the conclusion that the data observed during the present investigation were within the range of natural variability of water quality in the region was further supported by Corringan's (2017) intensive, high frequency monitoring of receiving stream turbidity and suspended solids concentrations upstream and downstream of the harvested watersheds, which was conducted concurrently with the present investigation). Thus, the present investigation demonstrates some of the potential pitfalls associated with landscape-scale time series trend monitoring investigations: while in some cases they are inadequate for detecting an impact attributable to landscape disturbance, in others, they may suggest an impact where one does not exist; likely by chance or due to some other factor—this occurred in the present investigation because baseline NOM-associated water quality concentrations and character in streams draining the partial cut watershed especially were likely higher than those in the reference watershed. Thus, this investigation emphasizes the importance and utility of properly BACI designed paired catchment studies for informing for evaluating (1) landscape disturbance impacts on water quality and treatability and (2) forest management-based approaches as SWP technologies and climate change adaptation strategies.

4. DOC concentration and aromaticity (measured as UV_{254}) remain the most informative proxy indicators of NOM/DOC reactivity in describing disturbance-associated threats to drinking water treatability.

Although none of the observed differences in aqueous NOM characteristics posed significant challenges to drinking water treatability, DOC concentration and aromaticity generally correlated well with THM- and HAA-FP at the watershed-scale and over multiple flow regimes in the study watersheds. This observation is consistent with other reports of drinking water treatability proxy indicators evaluated during investigations of landscape disturbances (Shams, 2018). Critically, while these proxy indicators inform DBP formation potential threats, they do not inform potential

challenges to pre-treatment processes (i.e. coagulation, flocculation, clarification) that—although plant specific—can lead to potentially catastrophic service disruptions. More research is needed to better anticipate such risks.

Overall, this work has demonstrated that carefully implemented contemporary forest harvesting with implementation of best management practices that minimize erosion management, such as minimal density and/or duration of linear disturbances (i.e., roads) can have minimal or no appreciable impact on drinking water source quality and treatability. Nonetheless, further investigations are needed to elucidate the long-term impacts of harvesting approaches and associated BMPs on water quality and treatability. These evaluations should include other water quality and treatability parameters, such as UFC, that are comparable among different watersheds. The considerations of the inclusion of less site specific metrics, can enable better-informed decisions on the impact of climate change and source water protection.

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Appendix A

Dataset for each water quality and treatability metric

DOC concentrations (mg/L)

Time	Location	Ref 1	Undist 1	Undist 2	Clear Cut	Strip Cut	Partial Cut
2009	Jan-09		0.5	0.5			
	Mar-09		0.4	0.5			
	Apr-09	0.70	1.3	0.8			
	May-09	1.40	1.67	1.27			
	Jun-09	0.60	0.80	1.18			
	Jul-09	0.65	0.70	1.13			
	Aug-09	0.90	0.70	0.90			
	Sep-09	0.60	0.70	1.30			
	Oct-09	0.30	1.10	0.50			
	Nov-09		0.80	0.90			
	Dec-09	0.60	0.30	0.80			
2010	Jan-10	0.30	0.60	0.50			
	Mar-10	0.60	0.50	0.70			
	Apr-10	0.55	0.90	0.93			
	May-10	0.88	1.28	1.13			
	Jun-10	1.23	1.45	1.98			
	Jul-10	0.45	0.60	1.30			
	Aug-10	0.80	0.80	1.10			
	Sep-10	0.90	0.95	2.45			
	Oct-10	0.70	0.90	1.00			
	Dec-10	1.00	0.90	1.00			
2013	Apr-13		1.051	1.180			
	May-13		1.330	1.193			
	Jul-13			1.029			
	Sep-13		0.780	0.971			

Time	Location	Ref 1	Undist 1	Undist 2	Clear Cut	Strip Cut	Partial Cut
2014	Apr-14		0.63	0.95			
	May-14		1.60	1.58			
	Jul-14		0.70	0.81			
	Aug-14		0.56	0.90			
	Sep-14		0.58	0.75			
	Oct-14		0.78	0.98			
2015	Apr-15	0.82			1.09	1.53	1.54
	May-15				1.48	1.52	1.47
	Jun-15	1.12			1.29	1.32	1.73
	Sep-15	0.99			1.42	1.09	
2016	Mar-16	0.75			0.94	1.01	
	May-16	1.04			1.19	0.95	1.64
	Jun-16	0.85			0.90	1.01	1.52
	Aug-16	0.68			0.81	0.85	
2017	Jun-17	0.65			0.79	0.98	1.64
	Aug-17	0.49			0.76	0.98	
	Oct-17	0.36			0.58	0.67	
2018	Apr-19	1.03			1.37	2.17	
	May-19	3.45			4.15	4.79	7.38
	Jun-19	1.28			0.85	0.94	1.70
	Aug-19	0.43			0.72	0.89	
	Oct-19	0.65			0.67		

UV_{254} (m^{-1})

Time \ Location		Ref 1	Ref 2	Ref 3	Clear Cut	Strip Cut	Partial Cut
2013	Apr-13		1.29	1.44			
	May-13		2.51	1.47			
	Jul-13			2.03			
	Sep-13		1.35	1.72			
2014	Apr-14		1.35	1.85			
	May-14		4.7	4.6			
	Jul-14		0.3	1.3			
	Aug-14		1.0	1.6			
	Sep-14		1.4	1.6			
	Oct-14		1.8	2.6			
2015	Apr-15	0.87			1.61	2.37	2.57
	May-15				2.19	2.35	2.15
	Jun-15	1.09			1.58	1.73	2.9
	Sep-15	1.09			2.02	1.75	
2016	Mar-16	0.86			1.72	2	
	May-16	2.61			2.53	1.92	4.25
	Jun-16	1.34			1.63	2.23	3.6
	Aug-16	0.73			1.61	1.34	
2017	Jun-17	1.11			1.94	1.78	3.28
	Aug-17	1.09			1.6	1.34	
	Oct-17	0.87			1.46	1.97	
2018	Apr-19	0.83			1.13	1.67	
	May-19	2.77			2.87	4.10	8.33
	Jun-19	1.00			1.37	1.63	3.60
	Aug-19	0.70			1.80	1.37	
	Oct-19	1.30			1.30	2.20	

SUVA (L/mg.m)

Time	Location	Ref 1	Undist 1	Undist 2	Clear Cut	Strip Cut	Partial Cut
2013	Apr-13		1.23	1.22			
	May-13		1.88	1.23			
	Jul-13			1.97			
	Sep-13		1.74	1.77			
2014	Apr-14		2.15	1.94			
	May-14		3.0	2.9			
	Jul-14		0.5	1.6			
	Aug-14		1.8	1.8			
	Sep-14		2.4	2.2			
	Oct-14		2.3	2.6			
2015	Apr-15	1.06			1.48	1.55	1.67
	May-15	1.54			1.48	1.55	1.46
	Jun-15	0.97			1.22	1.31	1.68
	Sep-15	1.10			1.42	1.61	
2016	Mar-16	1.15			1.83	1.98	
	May-16	2.51			2.13	2.02	2.59
	Jun-16	1.58			1.81	2.21	2.37
	Aug-16	1.07			1.99	1.58	
2017	Jun-17	1.71			2.46	1.82	2.00
	Aug-17	2.22			2.11	1.37	
	Oct-17	2.42			2.52	2.94	
2018	Apr-19	1.21			1.30	1.36	
	May-19	2.49			2.37	3.28	4.99
	Jun-19	0.78			1.61	1.75	1.21
	Aug-19	0.62			0.40	0.65	
	Oct-19	2.00			1.94		

Humic substances (mg/L)

Time	Location	Ref 1	Ref 2	Ref 3	Clear Cut	Strip Cut	Partial Cut
2014	Apr-14		0.35	0.68			
	May-14		1.28	0.85			
	Jul-14		0.41	0.51			
	Aug-14		0.30	0.55			
	Sep-14		0.33	0.50			
	Oct-14		0.37	0.36			
2015	Apr-15	0.62			0.74	0.75	1.13
	May-15				0.99	0.85	1.02
	Jun-15	0.77			0.91	0.95	1.28
2017	Jun-17	0.10			0.61	0.64	1.14
	Aug-17	0.31			0.42	0.62	
	Oct-17	0.13			0.31	0.08	

Biopolymers (mg/L)

Time / Location		Ref 1	Undist 1	Undist 2	Clear Cut	Strip Cut	Partial Cut
2014	Apr-14		0.008	0.012			
	May-14		0.025	0.011			
	Jul-14		0.013	0.010			
	Aug-14		0.010	0.007			
	Sep-14		0.007	0.010			
	Oct-14		0.009	0.009			
2015	Apr-15	0.005			0.008	0.003	0.004
	May-15				0.022	0.008	0.010
	Jun-15	0.002			0.006	0.005	0.003
2017	Jun-17				0.010	0.005	0.007
	Aug-17	0.010			0.005	0.004	
	Oct-17	0.006			0.005		

Building Blocks (mg/L)

Time	Location	Ref 1	Undist 1	Undist 2	Clear Cut	Strip Cut	Partial Cut
2014	Apr-14		0.06	0.12			
	May-14		0.19	0.20			
	Jul-14		0.09	0.09			
	Aug-14		0.07	0.11			
	Sep-14		0.06	0.09			
	Oct-14		0.08	0.08			
2015	Apr-15	0.16			0.21	0.28	0.29
	May-15				0.21	0.18	0.24
	Jun-15	0.20			0.22	0.22	0.36
2017	Jun-17	0.02			0.09	0.13	0.19
	Aug-17	0.05			0.09	0.14	
	Oct-17	0.03			0.07	0.01	

LMW Acids (mg/L)

Time	Location	Ref 1	Undist 1	Undist 2	Clear Cut	Strip Cut	Partial Cut
2014	Apr-14						
	May-14						
	Jul-14		0.01	0.00			
	Aug-14						
	Sep-14		0.01				
	Oct-14		0.01	0.00			
2015	Apr-15					0.17	
	May-15						
	Jun-15						
2017	Jun-17	0.003			0.03	0.01	0.03
	Aug-17	0.01			0.004	0.02	
	Oct-17	0.02			0.02	0.02	

LMW Neutrals (mg/L)

Time	Location	Ref 1	Undist 1	Undist 2	Clear Cut	Strip Cut	Partial Cut
2014	Apr-14		0.18	0.11			
	May-14		0.13	0.36			
	Jul-14		0.12	0.09			
	Aug-14		0.14	0.15			
	Sep-14		0.14	0.12			
	Oct-14		0.31	0.54			
2015	Apr-15	0.08			0.10	0.28	0.16
	May-15	0.00			0.24	0.50	0.25
	Jun-15	0.14			0.16	0.14	0.20
2017	Jun-17	0.50			0.07	0.20	
	Aug-17	0.01			0.03	0.02	
	Oct-17	0.14			0.12	0.52	

THM-FP (µg/L)

Time / Location		Ref 1	Undist 1	Undist 2	Clear Cut	Strip Cut	Partial Cut
2013	Apr-13		10.00	8.20			
	May-13		17.00	13.00			
	Jul-13			8.00			
	Sep-13		11.00	10.00			
2014	Apr-14		38.00	42.00			
	May-14		128.0				
	Jul-14		32.0	39.0			
	Aug-14		26.0	33.0			
	Sep-14		32.0	27.0			
	Oct-14		33.0	32.0			
2015	Apr-15	25			31	37	45
	May-15				43	39	35
	Jun-15	23			27	25	50
	Sep-15	17			23	27	
2016	Mar-16	15			28	31	
	May-16	36			39	29	50
	Jun-16	25			26	34	48
	Aug-16	14			20	26	
2017	Jun-17						
	Aug-17	13			21	24	
	Oct-17	16			25	30	
2018	Apr-19	15			19	24	
	May-19	7.4			48	65	168
	Jun-19	24			31	32	65
	Aug-19	15			22	31	
	Oct-19	25			22	36	

HAA-FP (µg/L)

Time / Location		Ref 1	Undist 1	Undist 2	Clear Cut	Strip Cut	Partial Cut
2013	Apr-13						
	May-13						
	Jul-13						
	Sep-13						
2014	Apr-14		42.00	39.00			
	May-14						
	Jul-14						
	Aug-14						
	Sep-14						
	Oct-14						
2015	Apr-15	37			36	42	57
	May-15				91	68	64
	Jun-15	50			38	32	74
	Sep-15	24			29	34	
2016	Mar-16	16			35	32	
	May-16	50			53	35	59
	Jun-16	30			33	34	61
	Aug-16	20			23	32	
2017	Jun-17						
	Aug-17	16			21.5	24.6	
	Oct-17	22.6			37.4	42.7	
2018	Apr-19	19.2			24.1	29.6	
	May-19				64.1	78.5	258
	Jun-19	28.9			34.2	33.7	77.2
	Aug-19	24.4			39.3	59.3	
	Oct-19	38.4			35.2	54.1	

Appendix B

ANOVA results for 9 points DOC measurements

p-value	Ref 1	Undist 1	Undist 2	Clear cut	Strip cut	Partial cut
May-13		0.9943	0.9990			
Sep-13		0.9922	0.9409			
Apr-14		0.8298	0.9621			
May-14		0.7849	0.9169			
Jul-14		0.6210	0.6702			
Aug-14		0.2623	0.1623			
Sep-14		0.9902	0.4304			
Oct-14		0.6447	0.1087			
Apr-15	0.7612			0.8976	0.5057	0.8030
Mar-16	0.7860			0.0815	0.1016	
May-16	0.9470			0.9805	0.2899	0.6013
Jun-16	0.9987			0.9079	0.9434	0.9888
Aug-16	0.7169			0.9971	0.9991	
Jun-17	0.8961			0.9064	0.5908	0.7042
Aug-17	0.8294			0.5920	0.4136	
Oct-17	0.9428			0.0911	0.1536	
Apr-19	0.9898			0.9600	0.7931	
May-19	1			1	0.9977	0.9910
Jun-19	0.9037			0.9221	0.9066	0.8397
Aug-19	0.9590			0.8860	0.9119	
Oct-19	0.9458			0.4732		

Appendix C

Non-parametric Mann-Whitney U test results

In the following tables the right hand side represents the results of non-parametric test with all available data and left hand side represents results from a data set without the inclusion of elevated dataset from May 2018. The purpose was to investigate the possible impact of May 2018 data on statistical analysis.

For DOC (mg/L) dataset

All Data

	U	*p*-value
Ref 1 (2015-18) vs. Undist 1 (2013-15)	66	0.95
Ref 1 (2015-18) vs. Undist 2 (2013-15)	50	0.18
Undist 1 (2013-15) vs. Undist 2 (2013-15)	28	0.18
Ref 1 (2009-15) vs. Undist 1 (2009-15)	210	0.21
Ref 1 (2009-15) vs. Undist 2 (2009-15)	140	0.003
Ref 1 vs. Undist 1 (2009-18)	436	0.42
Ref 1 vs. Undist 2 (2009-18)	306	0.005
Undist 1 vs. Undist 2 (2009-18)	327	0.047

Excluding May 2018 Data

	U	*p*-value
Ref 1 (2015-18) vs. Undist 1 (2013-15)	57	0.73
Ref 1 (2015-18) vs. Undist 2 (2013-15)	40	0.08
Undist 1 (2013-15) vs. Undist 2 (2013-15)	28	0.18
Ref 1 (2009-15) vs. Undist 1 (2009-15)	210	0.21
Ref 1 (2009-15) vs. Undist 2 (2009-15)	140	0.003
Ref 1 vs. Undist 1 (2009-18)	57	0.30
Ref 1 vs. Undist 2 (2009-18)	40	0.002
Undist 1 vs. Undist 2 (2009-18)	327	0.047

All Data

	U	*p*-value
Ref 1 (2015-18) vs. Clear cut (2015-18)	82.5	0.14
Ref 1 (2015-18) vs. Partial cut (2015-18)	7	0.000
Ref 1 (2015-18) vs. Strip cut (2015-18)	63.5	0.04
Ref 1 (all) vs. Clear cut (2015-18)	151	0.02
Ref 1 (all) vs. Partial cut (2015-18)	7	0.00
Ref 1 (all) vs. Strip cut (2015-18)	102.5	0.001

Excluding May 2018 Data

	U	*p*-value
Ref 1 (2015-18) vs. Clear cut (2015-18)	67.5	0.10
Ref 1 (2015-18) vs. Partial cut (2015-18)	0	0.000
Ref 1 (2015-18) vs. Strip cut (2015-18)	49.5	0.02
Ref 1 (all) vs. Clear cut (2015-18)	136	0.02
Ref 1 (all) vs. Partial cut (2015-18)	0	0.000
Ref 1 (all) vs. Strip cut (2015-18)	88.5	0.001

	U	*p*-value
Undist 1 (2013-14) vs. Clear cut (2015-18)	47	0.17
Undist 1 (2013-14) vs. Partial cut (2015-18)	3	0.001
Undist 1 (2013-14) vs. Strip cut (2015-18)	36	0.06
Undist 1 (2009-15) vs. Clear cut (2015-18)	163.5	0.08
Undist 1 (2009-15) vs. Partial cut (2015-18)	8	0.000
Undist 1 (2009-15) vs. Strip cut (2015-18)	109.5	0.004

	U	*p*-value
Undist 1 (2013-14) vs. Clear cut (2015-18)	47	0.24
Undist 1 (2013-14) vs. Partial cut (2015-18)	3	0.001
Undist 1 (2013-14) vs. Strip cut (2015-18)	36	0.10
Undist 1 (2009-15) vs. Clear cut (2015-18)	163.5	0.14
Undist 1 (2009-15) vs. Partial cut (2015-18)	8	0.000
Undist 1 (2009-15) vs. Strip cut (2015-18)	109.5	0.010

All Data

	U	*p*-value
Undist 2 (2013-14) vs. Clear cut (15-18)	73	0.74
Undist 2 (2013-14) vs. Partial cut (2015-18)	3	0.000
Undist 2 (2013-14) vs. Strip cut (2015-18)	62	0.50
Undist 2 (09-15) vs. Clear cut (2015-18)	245	0.96
Undist 2 (2009-15) vs. Partial cut (2015-18)	17	0.000
Undist 2 (2009-15) vs. Strip cut (2015-18)	185	0.27

Excluding May 2018 Data

	U	*p*-value
Undist 2 (2013-14) vs. Clear cut (2015-18)	63	0.53
Undist 2 (2013-14) vs. Partial cut (2015-18)	3	0.001
Undist 2 (2013-14) vs. Strip cut (2015-18)	62	0.67
Undist 2 (2009-15) vs. Clear cut (2015-18)	220	0.78
Undist 2 (2009-15) vs. Partial cut (2015-18)	17	0.000
Undist 2 (2009-15) vs. Strip cut (2015-18)	185	0.44

All Data

	U	*p*-value
Clear cut vs. Strip cut (2015-18)	83.5	0.23
Clear cut vs. Partial cut (2015-18)	8	0.000
Strip cut vs. Partial cut (2015-18)	17.5	0.004

Excluding May 2018 Data

	U	*p*-value
Clear cut vs. Strip cut (2015-18)	75.5	0.20
Clear cut vs. Partial cut (2015-18)	1	0.000
Strip cut vs. Partial cut (2015-18)	10.5	0.003

For UV_{254} (m^{-1})

All Data

	U	*p*-value
Ref 1 (2015-18) vs. Undist 1 (2013-15)	37	0.07
Ref 1 (2015-18) vs. Undist 2 (2013-15)	20	0.001
Undist 1 vs. Undist 2 (2013-15)	27	0.16
Strip cut vs. Clear cut (2015-18)	94.5	0.21
Strip cut vs. Partial cut (2015-18)	10	0.000
Clear cut vs. Partial cut (2015-18)	4	0.000
Ref 1 vs. Clear cut (2015-18)	33.5	0.000
Ref 1 vs. Partial cut (2015-18)	4	0.000
Ref 1 vs. Strip cut (2015-18)	31	0.000
Undist 1 (2013-14) vs. Clear cut (2015-18)	49	0.21
Undist 1 (2013-14) vs. Partial cut (2015-18)	8	0.01
Undist 1 (2013-14) vs. Strip cut (2015-18)	46	0.15
Undist 2 (2013-14) vs. Clear cut (2015-18)	69	0.59
Undist 2 (2013-14) vs. Partial cut (2015-18)	8	0.003
Undist 2 (2013-14) vs. Strip cut (2015-18)	68	0.55

Excluding May 2018 Data

	U	*p*-value
Ref 1 (2015-18) vs. Undist 1 (2013-15)	29	0.03
Ref 1 (2015-18) vs. Undist 2 (2013-15)	11	0.000
Undist 1 vs. Undist 2 (2013-15)	27	0.16
Strip cut vs. Clear cut (2015-18)	74.5	0.17
Strip cut vs. Partial cut (2015-18)	4	0.000
Clear cut vs. Partial cut (2015-18)	2	0.000
Ref 1 vs. Clear cut (2015-18)	18.5	0.000
Ref 1 vs. Partial cut (2015-18)	2	0.000
Ref 1 vs. Strip cut (2015-18)	16	0.000
Undist 1 (2013-14) vs. Clear cut (2015-18)	48	0.26
Undist 1 (2013-14) vs. Partial cut (2015-18)	8	0.01
Undist 1 (2013-14) vs. Strip cut (2015-18)	45	0.19
Undist 2 (2013-14) vs. Clear cut (2015-18)	60	0.43
Undist 2 (2013-14) vs. Partial cut (2015-18)	8	0.007
Undist 2 (2013-14) vs. Strip cut (2015-18)	67	0.68

For SUVA (L/mg.m)

All Data

	U	*p*-value
Ref 1 (2015-18) vs. Undist 1 (2013-15)	49	0.21
Ref 1 (2015-18) vs. Undist 2 (2013-15)	46	0.08
Undist 1 vs. Undist 2 (2013-15)	44	0.97
Strip cut vs. Clear cut (2015-18)	115	0.86
Strip cut vs. Partial cut (2015-18)	47	0.43
Clear cut vs. Partial cut (2015-18)	53	0.53
Ref 1 vs. Clear cut (2015-18)	95	0.22
Ref 1 vs. Partial cut (2015-18)	35.5	0.08
Ref 1 vs. Strip cut (2015-18)	88.5	0.22
Undist 1 (2013-14) vs. Clear cut (2015-18)	60	0.52
Undist 1 (2013-14) vs. Partial cut (2015-18)	36	0.96
Undist 1 (2013-14) vs. Strip cut (2015-18)	54	0.45
Undist 2 (2013-14) vs. Clear cut (2015-18)	70	0.62
Undist 2 (2013-14) vs. Partial cut (2015-18)	39	0.97
Undist 2 (2013-14) vs. Strip cut (2015-18)	64	0.57

Excluding May 2018 Data

	U	*p*-value
Ref 1 (2015-18) vs. Undist 1 (2013-15)	41	0.12
Ref 1 (2015-18) vs. Undist 2 (2013-15)	38	0.04
Undist 1 vs. Undist 2 (2013-15)	44	0.97
Strip cut vs. Clear cut (2015-18)	98	0.78
Strip cut vs. Partial cut (2015-18)	40	0.54
Clear cut vs. Partial cut (2015-18)	47	0.73
Ref 1 vs. Clear cut (2015-18)	78	0.16
Ref 1 vs. Partial cut (2015-18)	29.5	0.11
Ref 1 vs. Strip cut (2015-18)	75.5	0.20
Undist 1 (2013-14) vs. Clear cut (2015-18)	53	0.41
Undist 1 (2013-14) vs. Partial cut (2015-18)	27	0.68
Undist 1 (2013-14) vs. Strip cut (2015-18)	45	0.28
Undist 2 (2013-14) vs. Clear cut (2015-18)	62	0.50
Undist 2 (2013-14) vs. Partial cut (2015-18)	31	0.74
Undist 2 (2013-14) vs. Strip cut (2015-18)	54	0.37

For THM-FP (µg/L)

All Data

	U	*p*-value
Ref 1 (2015-18) vs. Undist 1 (2013-15)	36.5	0.10
Ref 1 (2015-18) vs. Undist 2 (2013-15)	54.5	0.60
Undist 1 vs. Undist 2 (2013-15)	34	0.60
Strip cut vs. Clear cut (2015-18)	73	0.11
Strip cut vs. Partial cut (2015-18)	8.5	0.001
Clear cut vs. Partial cut (2015-18)	4.5	0.000
Ref 1 vs. Clear cut (2015-18)	43	0.01
Ref 1 vs. Partial cut (2015-18)	1	0.000
Ref 1 vs. Strip cut (2015-18)	20	0.000
Undist 1 (2013-14) vs. Clear cut (2015-18)	64.5	0.86
Undist 1 (2013-14) vs. Partial cut (2015-17)	7	0.01
Undist 1 (2013-14) vs. Strip cut (2015-18)	60.5	0.68
Undist 2 (2013-14) vs. Clear cut (2015-18)	58	0.60
Undist 2 (2013-14) vs. Partial cut (2015-18)	2	0.001
Undist 2 (2013-14) vs. Strip cut (2015-18)	51.5	0.35

Excluding May 2018 Data

	U	*p*-value
Ref 1 (2015-18) vs. Undist 1 (2013-15)	36.5	0.14
Ref 1 (15-18) vs. Undist 2 (2013-15)	54.5	0.79
Undist 1 vs. Undist 2 (2013-15)	34	0.60
Strip cut vs. Clear cut (2015-18)	59	0.08
Strip cut vs. Partial cut (2015-18)	3	0.000
Clear cut vs. Partial cut (2015-18)	2	0.000
Ref 1 vs. Clear cut (2015-18)	43	0.02
Ref 1 vs. Partial cut (2015-18)	1	0.000
Ref 1 vs. Strip cut (2015-18)	20	0.000
Undist 1 (2013-14) vs. Clear cut (2015-18)	56.5	0.69
Undist 1 (2013-14) vs. Partial cut (2015-17)	7	0.02
Undist 1 (2013-14) vs. Strip cut (2015-18)	59.5	0.83
Undist 2 (2013-14) vs. Clear cut (2015-18)	58	0.78
Undist 2 (2013-14) vs. Partial cut (2015-18)	2	0.002
Undist 2 (2013-14) vs. Strip cut (2015-18)	51.5	0.48

For HAA-FP (µg/L)

All Data

	U	*p*-value
Strip cut vs. Clear cut (2015-18)	106.5	0.81
Strip cut vs. Partial cut (2015-18)	12	0.003
Clear cut vs. Partial cut (2015-18)	10	0.002
Ref 1 vs. Clear cut (2015-18)	57	0.06
Ref 1 vs. Partial cut (2015-17)	0	0.000
Ref 1 vs. Strip cut (2015-17)	43	0.01

Excluding May 2018 Data

	U	*p*-value
Strip cut vs. Clear cut (2015-18)	93	0.80
Strip cut vs. Partial cut (2015-18)	6	0.002
Clear cut vs. Partial cut (2015-18)	6	0.002
Ref 1 vs. Clear cut (2015-18)	57	0.10
Ref 1 vs. Partial cut (2015-18)	0	0.00
Ref 1 vs. Strip cut (2015-18)	43	0.02