DO IMPACTS FROM FOREST HARVESTING SPATIALLY ACCUMULATE IN STREAM NETWORKS?

by

Kelli Charbonneau

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Supervisors:	Karen Kidd, PhD., Biology Department & Canadian Rivers Institute
	Michelle Gray, PhD., Faculty of Forestry and Environmental Management & Canadian Rivers Institute
Examining Board:	Donald Baird, PhD., Biology Department & Canadian Rivers Institute, Chair
	Paul Sibley, PhD., School of Environmental Sciences, University of Guelph
	Dave Kreutzweiser, PhD., Canadian Forest Service, Natural Resources Canada

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ABSTRACT

Despite a suite of provincial guidelines working in concert with federal policy to promote sustainable forest management in Canada, legacies of ecosystem degradation from forestry persist, particularly within nearby streams. Harvesting-induced impacts to small headwaters tend to be well-studied and reasonably predictable, but it is lesser known how impacts manifest in larger downstream areas and if best management practices (BMPs) designed to protect against such impacts are effective at broader spatial scales. To address this uncertainty, I examined ecosystem indicators affected by selection-based harvesting under BMPs within mixed hardwood forest catchments along a spatial gradient (1st to 4th order streams, drainage areas 49 to 1856 ha). Indicators included water chemistry, dissolved organic matter (DOM) quality, sediment deposition and organic matter content, leaf litter decomposition and associated macroinvertebrate community structures, analysis of the algal contribution to stream consumer diet, and mercury (Hg) in water and biota. Indicator responses along the spatial gradient for streams within paired harvested and reference catchments were compared using two-factor ANOVAs. A significant interaction between treatment (reference vs. harvested) and sample site location (upstream, middle-reach, and downstream) was used as evidence for spatially cumulative trends within the harvested catchment (i.e., indicator response over spatial range differed in harvested catchment compared to reference catchment). A treatment effect with potential to have adverse implications at downstream sites in harvested catchments indicated that BMPs were not effective at preventing impacts at broader spatial scales. Overall, I found strong evidence for forest harvesting-induced spatially cumulative trends for % organic content of coarse sediment

and various endpoints for Hg in water and biota. BMPs were effective at preventing an adverse impact from harvesting at downstream sites for all indicators except sediment deposition, % dietary algae, Hg(II) in biota, and bioaccumulation factor for MeHg in biota, but they were not effective at upstream sites (i.e., local spatial scale) for conductivity, total nitrogen, % organic content of sediment, % shredders, % dietary algae, and MeHg and Hg(II) in water and biota. My study contributes to a comprehensive and predictive understanding of the potential for spatially cumulative trends within harvested catchments, which is critical to help forest managers maintain healthy forest streams and their provisioning of aquatic ecosystem services for future generations of humans and wildlife alike.

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List of Abbreviations

Abbreviation	Full Term
AES	Aquatic Ecosystem Services
AIC_c	Akaike Information Criteria (adjusted for small sample sizes)
ANOSIM	Analysis of Similarities
ANOVA	Analysis of Variance
BAF	Bioaccumulation Factor
BAT	Batchawana Catchment
BMF	Biomagnification Factor
BMP	Best Management Practice
CH_3Hg^+	Methylmercury (Organic)
CRM	Certified Reference Material
CV-AFS	Cold Vapour-Atomic Fluorescence Spectroscopy
Dec	Deciduous
DD	Degree Days
Dist to DS	Distance to Downstream
DOC	Dissolved Organic Carbon
DOM	Dissolved Organic Matter
DOM_PC1 & DOM_PC2	Dissolved Organic Matter Principal Components 1 & 2
dw	Dry Weight
E2E3	E2-to-E3 Ratio
EPT	Ephemeroptera, Plecoptera, Trichoptera
EV	Explanatory Variable
EVSA	Effective Variable Source Area
FI	Fluorescence Index
FMR	Forest Management Regulation
FMT	Fall Mean Temperature
GLFC	Great Lakes Forestry Centre
HARV	Harvested
HCl	Hydrochloric Acid
Hg	Mercury
Hg(II)	Divalent Mercury (Inorganic)
Hg^0	Elemental Mercury
HIX	Humification Index
KER A	Kerwin A Catchment
KER B	Kerwin B Catchment
LOI	Loss on Ignition
LRD	Logging Road Density
MeHg	Methylmercury
NMDS	Non-Metric Multi-Dimensional Scaling
PAN	Pancake Catchment
PC	Principal Component
PCA	Principal Component Analysis
PF	Polyestersulfone Filter

ppb	Parts Per Billion
ppm	Parts Per Million
QFF	Quartz Fiber Filter
REF	Reference
RV	Response Variable
SAC_{340}	Specific Absorbance Coefficient at 340 nm
SD	Standard Deviation
SMT	Summer Maximum Temperature
SRP	Soluble Reactive Phosphorus
SUVA	Specific UV Absorbance at 254 nm
TAH	Total Area Harvested
THg	Total Mercury
TIC	Total Inorganic Carbon
TN	Total Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorus
VIF	Variance Inflation Factor
WC_PC1 & WC_PC2	Water Chemistry Principal Components 1 & 2
ww	Wet Weight
β:α	Freshness Index
β:α	Freshness Index
$\delta^{13}C$	Stable Isotopes of Carbon
$\delta^{2}H$	Stable Isotopes of Hydrogen

1. Introduction

1.1. Forests and Forest Streams

In order to understand and address the impacts of forest harvesting on streams over broad spatial scales, it is necessary to first examine forests in Canada, their linkages to adjacent aquatic ecosystems, and current forest management practices. Thirty-five percent of Canada's land mass is forested, collectively accounting for nine percent of the world's total forested lands (Natural Resources Canada 2016). The majority of forests in Canada are dominated by coniferous species, and the leading forest age class is between 81 and 120 years old (Natural Resources Canada 2017a). Canadian forests are important carbon sinks, provide habitat for a wide variety of species of plants, animals, and microbes, and hold extreme economic, recreational, aesthetic, and spiritual value for Canadians (Natural Resources Canada 2017b). Within Canada, distinct topography, soils, climate, and vegetation allow for forest classification by ecozone, forest region, forest composition, and plant hardiness zone (Natural Resources Canada 2017c). Forest lands are almost entirely publicly owned in Canada (94%; Natural Resources Canada 2016); in concert with federal policy, each province and territory is responsible for sustainable management of its forests, with advisory boards and councils representing private, non-profit, and academic stakeholder interests. This management includes the forest terrestrial landscape, but also adjacent aquatic systems (e.g., lakes, streams, and wetlands) within the forest.

There are strong linkages between land and water in a forest, and subsidies of matter, energy, and nutrients routinely cross terrestrial and aquatic ecosystem boundaries. For instance, terrestrial organic matter inputs (e.g., leaf litter and dissolved

organic matter) constitute over 90% of total organic matter in small forested streams (Richardson et al. 2005). Conversely, aquatic insects are a source of food for riparian terrestrial consumers (e.g., arthropods, birds, bats, herptiles) when they emerge from the water as adults (Baxter et al. 2005). In particular, there is direct ecological feedback between headwater streams and the terrestrial catchment (Hynes 1970). Positive feedback is amplified in headwater streams due to their high edge-to-volume ratio, which maximizes the interface between land and water (Vannote et al. 1980). Forest headwater streams are also critical components of larger river networks because they provide water and organic matter to support downstream food webs (Wipfli et al. 2007). However, this land-water linkage potentially puts stream ecosystems at risk as disturbances in the terrestrial catchment may translate into disturbances in the aquatic environment. For example, disturbances to forest soils (e.g., by logging or forest fire) can impact the quantity and quality of nutrients, sediments, and organic matter delivered to a stream, or perhaps mobilize contaminants from the soil to the stream (e.g., Kreutzweiser et al. 2004; Bishop et al. 2009). Terrestrial catchment disturbance has also been noted to impact the delivery of downstream aquatic ecosystem services (AES) that include mitigating large hydrologic fluxes, filtration of pollutants, production and maintenance of clean water supplies, and habitat provisions for invertebrates, fish, and other wildlife (Furniss et al. 2010; PEW Environment Group 2011; Brogna et al. 2017). Accordingly, the value of healthy forest catchments for providing AES is being increasingly recognized. For instance, New York City has avoided building a multi-billion-dollar water treatment facility by purchasing and preserving undeveloped forest lands in the Catskill/Delaware Watershed which are deemed to be hydrologically

sensitive for water quality (Pires 2004). Moreover, forests have been shown to positively influence raw water quality and even decrease water prices for consumers in France (Fiquepron et al. 2013).

1.2. Forest Management in Canada

Forest management involves any activity that extracts and replaces forest resources, and as a result can alter the composition or structure of a forest. These activities may include harvesting, site preparation, road construction, pesticide application, vegetation management, and stand tending. Forests in Canada are managed for timber, fiber, biomass, non-timber forest products, and conservation values. The majority of managed forests are in Southern Canada because forest quality tends to decrease as you move northward in terms of tree quantity, size, and suitability for market goods. Despite a decrease in demand for Canadian wood products within the last ten years, forest products remain a staple of Canada's economy, contributing over $20 billion to the country's GDP and employing over 200 000 people in 2016 (Natural Resources Canada 2017c). However, accessing timber stocks comes with an environmental cost. In 2013, forestry activities accounted for 4% of total water use and 5% of total industrial and household greenhouse gas emissions (Statistics Canada 2017). Forest managers are thus tasked with balancing the economic and environmental sustainability of one of the country's most important resources while meeting consumer demand for forest products.

1.3. Impacts of Forest Management Activities on Streams

1.3.1. Temperature & Hydrology

The impacts of forest management on small headwater streams have been well-studied and are, as a result, reasonably predictable. For instance, removal of over-stream canopy vegetation can increase the reception of incident radiation and decrease the evapotranspiration capacity of a forest. Increased solar radiation may elevate stream temperatures, either through direct exposure to the stream or via transport of warmed soil or soil-water in overland run-off (Moore et al. 2005). Higher stream temperatures may increase the rate of metabolism and other biochemical reactions in stream fauna, as well as potentially alter life-cycle timing and species composition due to thermal stress (Holtby 1988; Clapcott and Barmuta 2010). Primary productivity will increase as more sunlight reaches the stream, which can lead to a cascade of increasing biomass at all trophic levels as food availability and foraging conditions improve (Murphy and Hall 1981). Decreased evapotranspiration and loss of canopy interception can also result in heavier snow cover, increased run-off, prolonged waterlogged suboxic pockets in the soil, increased groundwater levels, and greater run-off flashiness and discharge (Bosch and Hewlett 1982; Eklöf et al. 2016). Additionally, construction and use of logging roads, landings, and skid trails may compact soil and prevent infiltration (i.e., reducing groundwater recharge), capture or re-direct surface run-off to streams, and leave streams more susceptible to changes in peak flows during storm events (Harr et al. 1975).

1.3.2. Transport of Particulate & Dissolved Materials

Ground disturbance associated with tree removal, road construction/use, and stand tending can cause soil erosion, destabilization of debris slopes, and disrupt biogeochemical processes in the soil. These activities, especially in combination with increased overland run-off, can increase the delivery of particulate and dissolved matter

to forest streams. One study observed that mean bedload estimates of fine, inorganic sediment in low-order forested streams increased to approximately 4000 g/m^2 in catchments that were highly disturbed by logging road activities compared to 400 g/m^2 pre-manipulation (Kreutzweiser and Capell 2001). Fine particles are of high concern because they tend to travel the farthest downstream, and may reduce the porosity and hydraulic connectivity of gravel stream beds, interfere with movement, respiration, and feeding of invertebrates, and shift dominant substrate cover for biota (Lenat et al. 1979; Lemly 1982; Brunke and Gonser 1997; Hand 1997). In addition, increased overland run-off and groundwater rising through new soil horizons may elevate the export of dissolved organic material (DOM) to nearby streams, including dissolved organic carbon (DOC) and various nutrients. Numerous studies have reported DOC increased 1.3 to 2.6 times in streams of logged catchments relative to undisturbed catchments, even with the application of riparian buffers, as a result of increased run-off (Lamontagne et al. 2000; Laudon et al. 2009; Schelker et al. 2012; Eckley et al. 2018). An increase in DOC can affect food web structure, increase transport of trace metals complexed to DOC, act as a microbial substrate, and decrease pH (Tipping et al. 1991; Köhler et al. 2002; Berggren et al. 2007; Jansson et al. 2007). Nitrogen, phosphorus, and base cations (Ca, Mg, and K) may also leach from forest soils or become bound to soil particles and increase in concentration in aquatic ecosystems (Lamontagne et al. 2000; Eklöf et al. 2012; De Wit et al. 2014). Their mobility may be enhanced by increased microbial activity in the upper soil layers of the forest floor as the water table rises, soil erosion following vegetation removal, or increased input of nutrients from logging residues and decreased uptake of nutrients as vegetation is removed (Rosen and

Lundmark-Thelin 1987; Hornberger et al. 1994; Creed et al. 1996). Higher nutrient concentrations can increase primary productivity and microbial activity and disrupt biogeochemical processes (Planas et al. 2000; Kreutzweiser et al. 2008b)

1.3.3. Organic Matter Cycling

Autumn leaves and, to a lesser degree, harvesting residues provide fresh organic carbon to forest streams as they decompose and are critically important sources of energy for lotic food webs. When wetted by the stream, leaves begin to leach inorganic and organic components, primarily soluble carbohydrates and polyphenols (Suberkropp et al. 1976). Because the dissolved organic fraction is 45 to 50% organic carbon by mass, this leached material is often referred to as DOC (Allan 1995). Wetted leaves are colonized by microbial communities of bacteria and fungi, and subsequently fragmented by leaf-shredding macroinvertebrates (e.g., families of Trichoptera, Plecoptera, Crustacea, and Diptera). Colonisation by microbes increases the nutritional quality and palatability of stream leaf-litter to higher level consumers, possibly because the addition of microbial tissue and exoenzymes increases nutrient content, and/or due to partial exoenzymatic digestion of polysaccharides from plant cell walls (Bärlocher 1982, 1985). Shredders feeding on leaf litter can increase the availability of fine, organic particulate matter as a food source for suspension-feeders and the release of DOC (Short and Maslin 1977; Meyer and O'Hop 1983). As a result, lotic ecosystems in temperate regions derive most of their energy from terrestrial organic matter (Allan 1995; Richardson et al. 2009; Tank et al. 2010). However, forest harvesting can influence the amount of leaf material delivered to streams as well as the rates of decomposition of leaf litter in-stream, which can affect the provision of energy and particulate matter

downstream. Leaf litter decomposition rate and associated cycling of energy and nutrients have been found to decrease in recently logged headwater streams, a likely result of burial by sediments, reduced microbial function, and/or fostering of conditions that are favourable to macroinvertebrates that are less efficient at decomposing leaf litter (Pozo et al. 1998; Kreutzweiser et al. 2008a; Lecerf and Richardson 2010; Emilson et al. 2016). Alternatively, leaf litter breakdown in clear-cut streams has also been found to accelerate post-harvest, perhaps due to increased nitrate concentrations stimulating microbial decomposition and attracting shredders, or due to physical abrasion from high sediment loads and increased variability of hydrologic regimes (Benfield et al. 2001; McKie and Malmqvist 2009). Other impacts from harvesting operations, such as changes in flow, reduced water quality, elevated conductivity, and lower organic matter content of sediment, may also influence leaf litter decomposition (Kreutzweiser et al. 2008a; Emilson et al. 2017).

In addition, DOM leached from leaves, forest residues, and soils controls geochemical processes, transport of bound contaminants, dissolution and precipitation reactions, photic zone depth, and others (Weishaar et al. 2003). DOM molecules can vary widely in structure, weight, protein content, aromaticity, and lability (i.e., the quality of DOM), often reflecting source, processing time, biological activity, and hydrologic connectivity (Ravichandarn 2004). Generally, low-molecular weight and aliphatic compounds tend to be more easily degraded and labile than high-molecular weight and aromatic compounds, which are more reactive (Kiikkilä et al. 2014). The degradability or quality of DOM influences the ability of microorganisms to uptake carbon and cycle it through the food web. Watershed disturbances like forest harvesting

have been noted to influence DOM quality, particularly by altering water table level and associated aerobic/anaerobic conditions (Kiikkilä et al. 2014).

1.3.4. Stream Biota

Impacts of forest harvesting on physical and chemical components of stream ecosystems are more commonly reported than those on the abundance and diversity of stream biota, though the two are inherently linked. For instance, harvesting to stream sides may increase periphyton biomass, particularly filamentous algae, due to higher availability of nutrients and light to support algal growth (Graynoth 1979; Shortreed and Stockner 1983; Noel et al. 1986). Increased algal growth in streams with harvested catchments has been associated with an increase in scraper taxa who feed on periphyton on substrate surfaces (Newbold et al. 1980). Alternatively, transport of fine sediment from logging may reduce species richness and biomass of filter-feeding invertebrates (e.g., Trichoptera and Diptera) due to clogging of nets and filtering apparatuses, and may also reduce photosynthetic production as higher turbidity precludes light penetration (Lemly 1982; Ryan 1991). Abundance of shredder taxa and taxonomic richness in leaf litter-dwelling invertebrate communities have also been noted to be lower in logged catchments; lower shredder abundance may be associated with lower leaf litter decomposition, which may also decrease the amount of fine, particulate organic matter produced by shredders as a food source to support downstream taxa (Kreutzweiser et al. 2008a; Yeung et al. 2017). Furthermore, decreased access to detritus due to vegetation removal may cause an increase in the abundance of specialist herbivores, a decrease in the consumption of detritus by generalist and detritivore

species, and an increase in the consumption of algae across most functional feeding groups (Richardson 1991; Göthe et al. 2009).

Changes in abundance and/or species richness of macroinvertebrates may have cascading impacts within aquatic food webs, including reduced efficiency in utilization of energy inputs, or increased biomass at higher trophic levels owing to greater inputs of nutrients and sunlight (Wallace et al. 1977; Gregory et al. 1987). Additionally, higher trophic level consumers may be directly influenced by harvesting activities; for example, incidence and density of juvenile Atlantic Salmon have been noted to decline as areal coverage of logging increased, a potential result of deteriorating habitat quality from fine sediment deposition (Deschênes et al. 2007). Density and biomass of salmonid fishes have alternatively been observed to increase in response to recent clear-cut logging, likely due to the creation of refuge habitat by large woody debris or due to the cascade of increasing biomass providing greater food availability (Gregory et al. 1987; Mellina and Hinch 2009).

Another important biotic consideration is the determination of food sources for primary stream consumers (i.e., the basal energy resources in stream food webs) as terrestrial (allochthonous) or aquatic (autochthonous). Terrestrial organic matter is generally considered the quantitatively dominant basal energy source for headwater stream food webs through contribution of large amounts of detritus and because shading by riparian vegetation limits in-stream autotrophic production (Hynes 1975; Vannote et al. 1980). Webster and Meyer (1997) reported a median ratio of leaf litter to algal production in 28 streams of 2.8. As stream size increases, so does the importance of autochthonous primary production and residual fine organic matter transported from

upstream, with a decrease in reliance on direct allochthonous inputs (Vannote et al. 1980). Accordingly, the morphology and feeding behaviour of stream invertebrates reflects these shifts in organic matter source; shredders typically dominate headwater assemblages, and collectors typically dominate assemblages in larger, downstream areas (Vannote et al. 1980). However, it has recently been suggested that the importance of autochthonous material (algae) in the diet of stream consumers is greater than previously thought due to its higher nutritional quality (i.e., the accessibility of energy and suitability for synthesizing new biomass) relative to allochthonous material (Brett et al. 2009, 2017; Guo et al. 2016b). The lignocellulose comprising terrestrial organic material is highly resistant to enzymatic breakdown by most consumers, and detritus is a poor source of the biochemical compounds essential for growth and reproduction in stream biota (Martin et al. 1991; Brett et al. 2009). Furthermore, amino acid stable isotope analyses and profiles of essential fatty acids in stream consumers increasingly indicate a much stronger reliance on algae than previously believed (Guo et al. 2016a; Thorp and Bowes 2017). Therefore, the biochemical composition of algae may outweigh the higher quantity of terrestrial organic matter, and algae consumption may have a significant influence on the growth and development of consumers in a given stream system. However, the relative contribution of either source to the diet of stream consumers may be altered by harvesting activities. Many studies have reported an increase in autochthonous contributions to consumer diet, likely due to increased delivery of nutrients and light to streams stimulating aquatic primary production, or due to stream-side vegetation removal limiting the subsidization of terrestrial detritus (Rounick et al. 1982; England and Rosemond 2004; Göthe et al. 2009).

To determine the relative contribution of allochthonous and autochthonous food sources to the diet of consumers, ecologists often use stable isotope analysis. Stable isotopes provide a time-integrated measure of trophic energy flow because stable isotope ratios in consumers tend to reflect their diet (Peterson and Fry 1987). Stable isotopes of carbon (δ^{13}C) are commonly used as tracers of diet, but isotopic separation between potential terrestrial and aquatic sources of organic matter may be limited (Finlay et al. 2010). Environmental variation or physiological differences in organic matter may result in overlap in δ^{13}C and prevent isotopic separation of allochthonous and autochthonous dietary sources (Finlay and Kendall 2007). Hydrogen stable isotopes (δ^{2}H) have been proposed as an alternative to δ^{13}C due to the distinct isotopic separation between terrestrial and aquatic sources of organic matter, even across broad environmental gradients and where δ^{13}C values have overlapped (Doucett et al. 2007; Jardine et al. 2009). For example, Doucett et al. (2007) found autochthonous organic matter to be depleted in δ^{2}H by approximately 100‰ relative to allochthonous organic matter due to preferential loss of hydrogen to the atmosphere during evapotranspiration in terrestrial autotrophs (i.e., kinetic fractionation), and overall concluded that hydrogen stable isotopes are useful tools in tracing energy sources and flows in aquatic food webs.

1.3.5. Mercury

Various functional groups comprising the organic matter of forest soils, particularly thiols and carboxylic acids, may bind and retain atmospherically deposited or weathered contaminants, such as mercury (Hg) and other metals (Haitzer et al. 2002; Wood et al. 2011). Atmospheric Hg may be of natural (e.g., from geologic weathering, volcanic eruptions) or anthropogenic (e.g., from fossil fuel combustion, artisanal gold

mining) origin, where it exists in two forms: elemental Hg (Hg^0) and divalent inorganic Hg (Hg(II)). Following atmospheric oxidation of Hg^0 to Hg(II), Hg(II) may reach the earth's surface by wet deposition in precipitation or dry deposition via scavenging and settling of airborne particles. Soil erosion and changes to hydrologic regimes from forest harvesting can enhance Hg mobilization from the catchment and increase Hg concentrations in stream water. Many studies have found that logging in forested watersheds increases loading of organic matter-bound Hg to aquatic systems and in biota (e.g., Garcia and Carignan 2000; Povari et al. 2003; Ward et al. 2010; De Wit et al. 2014; Wu et al. 2018). Inorganic Hg may be methylated, primarily during the oxidation of DOM by sulphate- and iron-reducing bacteria in anoxic environments, to its organic form, methylmercury (CH_3Hg^+, MeHg). MeHg is a potent neurotoxicant that binds preferentially to cysteine amino acids in the protein of muscle tissue due to its strong affinity for thiol or sulphhydryl functional groups, resulting in a tendency to bioaccumulate in stream consumers (i.e., accumulation that exceeds catabolism or excretion) and biomagnify in stream food webs (i.e., magnification of concentration with trophic level) (Harris et al. 2003). In high-level stream consumers like fish, MeHg has been noted to exceed Canadian wildlife consumption guidelines of 33 µg/kg ww (Canadian Council of Ministers of the Environment 2000), and has been shown to negatively impact the nervous system, physiology, immune function, and reproduction of stream biota and their human consumers (Clarkson 1990; Wiener et al. 2003; Burgess and Meyer 2008).

In addition to mobilization of Hg, harvesting activities may stimulate methylating microbes, via the creation of labile sources of DOC and mobilization of

nutrients, as well as create waterlogged suboxic environments (e.g., standing water in skidder tracks) that increase Hg methylation (Eklöf et al. 2016). However, DOC plays a dual role in influencing the incorporation of MeHg into stream food webs: terrestrial DOC may facilitate the solubility and transport of MeHg to surface waters, but when complexed to Hg(II) may limit its bioavailability to methylating bacteria as its molecular weight is too large to cross their cell membranes (Winfrey and Rudd 1990; Ravichandran 2004). The relationship between DOC and Hg methylation, including its implications for MeHg bioaccumulation, is complex and under ongoing investigation.

1.4. Regulations & Guidelines for Forest Management in Canada

To mitigate the impacts of forestry operations on terrestrial catchments and their adjacent water bodies, provincial/territorial governments provide forest management regulations (FMRs), which are mandated operational requirements, and a set of guidelines known as best management practices (BMPs), which are voluntary measures to reduce environmental impacts. FMRs and BMPs are proactive and practical considerations that aim to minimize disturbances to forests and associated habitats caused by forestry activities and promote their sustainable management. Contemporary FMRs and BMPs guide forest harvesting method and intensity (including licensing and restrictions for forestry companies), road construction, soil rutting, soil compaction, trail orientation, stream crossings, managing designated protected areas, and others. Among the most commonly used BMPs is the application of a no-harvest buffer, typically about 30 m wide or more, maintained along the length of the watercourse to minimize bank erosion, stream-side vegetation removal, and sediment deposition (Jeglum et al. 2003). However, studies have noted their variable effectiveness. For example, Davies and

Nelson (1994) found that logging did not significantly decrease riffle macroinvertebrate abundance when riparian buffers of 30 m or more were maintained, and Kreutzweiser et al. (2010) found that even partially-harvested riparian buffers posed little risk of harm to aquatic invertebrate communities. Broadmeadow and Nisbet (2004) also deemed 10 to 30 m buffers adequate to replicate the canopy cover necessary to maintain the ecological integrity (e.g., shading, provision of woody debris) of a woodland stream. Conversely, Kreutzweiser et al. (2008a) noted declines in leaf litter decomposition in logged catchments even with application of over 30 m no-harvest buffer zones. Application of riparian buffers also does not emulate patterns of natural disturbance, and often results in an unnatural strip of older-growth forest stands around streams and lakes (Buttle 2002; Macdonald et al. 2004). While the application of several BMPs to protect water resources has been frequently studied and usually found to be effective, less is known about their effectiveness over large spatial and temporal scales.

1.5. Cumulative Impacts

Environmental processes in lotic ecosystems and disturbances thereof naturally operate at a variety of spatial and temporal scales; for instance, disturbances to headwater streams can disrupt material and energy provisions for downstream consumers (e.g., Wipfli et al. 2007), and can be exacerbated by fluctuating hydrologic regimes as seasons change (e.g., Brown et al. 2005). Forestry operations are also undertaken at various times and locations within the catchment, and can integrate to impact downstream ecosystems (Richardson 2008). Paradoxically, studies that observe impacts from harvesting and BMP effectiveness are typically bound by discrete spatial and temporal scales. The potential for these impacts to accumulate over space and time

is uncertain and under-investigated, particularly when compounded with expansion of harvesting into previously unmanaged forests, intensification of harvesting in currently managed forests, overlap with other resource developments, and interactions with climate change (Kreutzweiser et al. 2013). Cumulative impacts occur when individual effects integrate in an additive, synergistic, or antagonistic manner over space and/or time, and can be non-linear and difficult to predict. The ecological thresholds for cumulative impacts from harvesting activities from which aquatic ecosystem biodiversity cannot reasonably recover are largely unknown (Kreutzweiser et al. 2013). While cumulative effects assessment is a requirement for industrial development or installations under many provincial environmental assessment laws, little is known about the potential for cumulative impacts resulting from forest management practices as they exist in Canada. Consideration of cumulative impacts may be an impetus for forest managers to adjust scales of thinking from site-specific to the broader landscape, as well as from immediate concerns to periods of several years or decades.

To understand whether impacts from harvesting are accumulating *spatially* in forest drainage networks (i.e., the focus of this thesis), it is necessary to consider how rivers naturally change and accumulate changes at the watershed scale. Many conceptual models for how physical, chemical, and biological processes change along the length of a river have been proposed, including the river continuum concept (Vannote et al. 1980), the riverine productivity model (Thorp and Delong 2002), the process domain concept (Montgomery 1999), and the patch dynamics theory (Pringle et al. 1988). Each model attempts to describe the gradient in environmental processes, physical characteristics, and biota from headwaters to downstream reaches (i.e., along the “river continuum”).

For example, in a study measuring the variance in carbon-to-nitrogen ratio of fine particulate organic matter in streams of increasing width (a proxy for size), variance was found to be lower in wider streams (indicating a lower sensitivity to extrinsic environmental factors and a spatially homogenous biogeochemical condition) and higher in narrower streams (indicating a higher sensitivity to extrinsic environmental factors and a less stable biogeochemical condition) (Sakamaki and Richardson 2013). Deviations from the environmental baseline along the river's length can help to quantify impacts from harvesting relative to the natural "river continuum" (Vannote et al. 1980). Indicators used to quantify impacts must therefore not only well-represent baseline ecosystem health, but should also be sensitive to changes over broad spatial ranges.

While much of the published literature has focused on the reach-scale when examining impacts from forestry operations (e.g., Kreutzweiser et al. 2005; Klimesh et al. 2015; Clapott and Barmuta 2010), a few studies have noted impacts at large spatial scales. For example, Zhang et al. (2009) observed the impact of past forestry practices on current stream habitats and benthic invertebrate assemblages at large spatial scales. Zhang et al. (2009) found a significant decrease in taxon richness, relative abundance, and overall biomass at sites with a legacy of past forest management (i.e., up to 40 years prior) and at sites with drainage areas ranging from 0.54 to 6.74 km^2. A study by Deschênes et al. (2007) examined the incidence and density of juvenile Atlantic Salmon at the sub-basin and 8, 2, and 0.5 km upstream of study sites in response to forestry activities. Fish have been known to relate to their local environment differently than to catchment-level characteristics, and trends noted at smaller spatial scales do not always extrapolate to larger spatial scales (Wiens 1989; Folt et al. 1998). Deschênes et al.

(2007) found that incidence and density of the salmon declined with areal coverage of harvesting, but only at larger spatial scales (sub-basin and 8 km). Therefore, there is a need for studies quantifying the impact of forest management on stream ecosystems to address impacts at reach-level and broader spatial scales, including whether these impacts accumulate throughout the stream network.

1.6. Research Question & Objectives

In this Master's thesis, I assess how impacts to forest streams from harvesting of northern hardwood stands in Central Ontario, Canada changed with spatial scale to answer the question: do impacts from forest harvesting spatially accumulate in stream networks? The objectives of this research were to (1) assess and quantify the effect of (a) selection-based forest harvesting under Ontario BMPs and (b) spatial location of sample site on indicator responses; (2) determine if indicators exhibit spatially cumulative trends between headwaters and larger downstream areas in managed catchments relative to minimally-managed catchments (where spatially cumulative trends are interpreted as a *difference* in spatial trends between treatments); and (3) appraise if BMPs designed to minimize local or stand-level impacts were effective at protecting against adverse treatment impacts at broader spatial scales (where an adverse treatment impact is a significant difference between treatments for an indicator value, with the direction of change in the harvested catchment linked to a negative outcome for the ecosystem).

1.7. Hypotheses

The first null hypothesis for my research question is that there was no difference (i.e., in magnitude and/or direction) in the spatial trend of an indicator between the

harvested and reference catchments, suggesting there was not a spatially cumulative trend in indicator response in the harvested catchment. The first alternative hypothesis is that there was a difference in the spatial trend of an indicator between the harvested and reference catchments, suggesting that there was a spatially cumulative trend in indicator response in the harvested catchment. The second null hypothesis is that there was no adverse impact of harvesting at downstream sites, suggesting that BMPs applied at the operational and local scale are effective at mitigating impacts at broader spatial scales. The second alternative hypothesis is that there was an adverse treatment impact at downstream sites, suggesting that BMPs applied at the operational and local scales are not effective at mitigating impacts at broader spatial scales. Due to the lack of studies that have addressed spatially cumulative impacts related to forest management, it was difficult to anticipate whether indicators will behave similarly over space in harvested and reference catchments, and at what scale BMPs will be effective at preventing adverse treatment effects.

1.8. Approach

1.8.1. Study Area

My research was conducted in the Lower Batchawana River Watershed and the adjacent Pancake River Watershed, located in the Great Lakes-Saint Lawrence forest region on the northeast shore of Lake Superior in Ontario, Canada. Both watersheds consist of mixed conifer and hardwood forest but are dominated by hardwood stands in the lower reaches of the watersheds where my study occurred. The predominant hardwood tree species are sugar maple (*Acer saccharum* Marsh) and yellow birch (*Betula alleghaniensis* Britton). Within the watersheds, four smaller catchments were

selected for my study: three in the Lower Batchawana River Watershed, and one in the Pancake River Watershed. Two of the catchments have been subject to substantial forest management within 5 years prior to sampling (i.e., the harvested catchments), and two that have been minimally managed within 20 years prior to sampling (i.e., the reference catchments). Harvesting in the managed catchments was mostly selection-based, usually at a rate of 30 to 50% basal area removal, with very small clear-cut sections. This method of harvesting creates forest openings that emulate the patchy and small-scale natural disturbances (e.g., windthrow) typical of hardwood forests (Lorimer 1989). All harvesting was conducted according to Ontario's FMRs and BMPs, including the application of 30 m or wider no-harvest buffers along all mapped or observed streams.

1.8.2. Experimental Design

This study involved two paired-catchment comparisons; each of the two harvested catchments are paired with a reference catchment similar in size, forest composition, and geomorphology. The paired comparison design has been widely used to isolate the effects of vegetation removal and natural resource development from baseline levels (Brown et al. 2005). Each study catchment contained three sample sites positioned along a spatial gradient, from headwaters (1^{st} to 2^{nd} order stream), to middle-reaches (2^{nd} to 3^{rd} order stream), to downstream reaches (3^{rd} to 4^{th} order stream). Each site within a harvested catchment was assessed using a suite of indicators and compared with its sister site (i.e., the site with the same spatial orientation) within its reference catchment pair. This design allowed for the spatial trend in indicators in harvested catchments to be compared to that of reference catchments to assess change from the natural river continuum, and if these impacts accumulated over the spatial range (i.e.,

magnified or accumulated treatment impact at downstream sites). In addition to the paired-catchment comparisons, all sites were considered collectively to examine the impact of various explanatory variables (harvesting, catchment, and reach characteristics) on each indicator using linear models.

1.8.3. Indicators

Abiotic, biotic, and contaminant indicators were used to quantify impacts to stream condition from harvesting. The indicators used to quantify harvesting-induced impacts to forest streams were chosen on the basis of the fundamental characteristics of lotic environments as described by Hynes (1970), as well as to reflect both abiotic and biotic ecosystem components. Abiotic indicators assessed in this study include water chemistry, DOM quality, and sediment deposition. Water temperature and water flow were also measured for descriptive purposes, but were not used as indicators because they were not adequately characterized throughout the sampling period. Biological indicators assessed in this study include leaf litter decomposition, leaf litter macroinvertebrate community structure, and aquatic versus terrestrial organic matter contribution to the diet of stream consumers. An additional contaminant indicator used was the concentration of Hg in various environmental compartments.

1.9. Project Significance

Aquatic ecosystems are dynamic and multi-faceted, with components responding differently to disturbance over varying spatial and temporal scales. For example, abiotic characteristics like water chemistry and sediment deposition tend to respond rapidly to catchment disturbance, whereas biotic characteristics such as invertebrate community structure tend to change over longer timeframes. Therefore, to understand how forest

harvesting will impact the structure and function of stream ecosystems, it is necessary to use indicators that reflect multiple components of the system and are adequately sensitive to respond to such perturbations. Surprisingly, many harvesting-impact studies observe only one or a few indicators (e.g., Carignan et al. 2000; Macdonald et al. 2003; Kibler et al. 2013), and often focus on abiotic parameters (e.g., hydrology, temperature, and water quality) with a lack of emphasis on food web structure. There is a current paucity of research integrating information from multiple indicators, particularly in the context of cumulative impacts over space and time. In this study I used abiotic, biotic, and contaminant indicators to obtain a holistic and comprehensive understanding of stream condition in response to forest harvesting at multiple spatial scales. The use of multiple indicators can help to identify changes to the entire ecosystem from harvesting, as well as shortcomings in current BMPs at preventing impacts locally and downstream. Moreover, indicators used herein were chosen to reflect concerns relevant to the current ecological and social context. For example, the use of Hg in biota as an indicator is particularly timely given recent concerns surrounding Hg biomagnification in the Great Lakes region (e.g., Omara et al. 2015; Rolfhus et al. 2015) and Hg poisoning in indigenous communities, such as legacy contamination from Hg-contaminated pulp mill effluent in the Grassy Narrows First Nation (Porter 2017).

Forestry has been predicted to intensify in the near future in response to a growing world population, demand for lignocellulosic feedstocks to fuel biorefineries, and interest in forest products for innovative applications (e.g., nanocrystalline cellulose as an alternative to stainless steel) (Ferguson 2012; Creed et al. 2016). In response to this intensification, it is ever more critical that a framework for cumulative impacts

assessment for forest management be developed and integrated into Canadian environmental assessment legislation. My research will address knowledge gaps surrounding the potential for spatially cumulative trends resulting from forestry operations and provide insight on their potential magnitude and/or direction in similar northern hardwood forests. Ultimately, the significance of my research lies in its challenge of the status-quo for environmental protection guidelines surrounding forest management activities in Canada by determining if BMPs designed to minimize local or stand-level impacts are effective at larger spatial scales. My study will also contribute to establishing a comprehensive and predictive understanding of the individual and cumulative effects associated with forest management in northern hardwood catchments, a keystone to maintaining healthy future forests and their provision of aquatic ecosystem services.

2. Methods

2.1. Study Area & Site Selection

My research was conducted approximately 10 to 30 km inland of the northeastern shore of Lake Superior and 60 km north of the city of Sault Ste. Marie in Ontario, Canada (Figure 1). The watersheds delineating the study area are the Lower Batchawana River and Pancake River Watersheds, which ultimately drain in to Batchawana Bay and Pancake Bay of Lake Superior, respectively. The watersheds are captured by the Boreal Shield ecozone and the Great Lakes-Saint Lawrence forest region, and accordingly their forests include mixed hardwood stands predominated (>90%) by sugar maple (*Acer saccharum* Marsh), with some yellow birch (*Betula alleghaniensis* Britton), white birch (*Betula papyrifera*), red maple (*Acer rubrum*), trembling aspen (*Populus tremuloides*), white pine (*Pinus strobus*), and white spruce (*Picea glauca*). From 1980 to 2007, the mean annual air temperature was 4.63 ± 1.15°C (increasing at a rate of 0.09°C per year), and the mean annual precipitation was 1198 ± 143 mm (2/3 rain, 1/3 snow) (Environment Canada 2014).

Within the watersheds are four smaller catchments: two with either little or no recent harvest (PAN and KER A, <6% catchment area harvested within five years of sampling) and two with substantial selection-based harvest under BMPs (BAT and KER B, >32% catchment area harvested within five years of sampling) (Figure 1, Table 1). Catchment characteristics were determined using the Arc-GIS flow accumulation grid based on the digital elevation model (DEM) obtained from the Land Information Ontario (LIO) database (20 m x 20 m resolution) and WhiteBox (Lindsay 2016). Harvesting on crown land in the managed catchments was conducted in accordance with

Ontario FMRs (OMNR 2010), and harvesting on private lands was managed by VanderMolen Forestry Inc. following similar BMPs (Ron VanderMolen, personal communication). In both the crown and private forests, no-harvest buffers of at least 30 m wide were retained along permanent or obvious stream channels. Other BMPs included minimizing sensitive soil disturbances, avoidance of stream-side wet areas, approved stream crossings and culverts, and sediment abatement procedures associated with logging roads and skid trails.

Both harvested catchments were paired with a minimally-harvested catchment to isolate the effects of harvesting; accordingly, PAN (reference) and BAT (harvested) are one pair, and KER A (reference) and KER B (harvested) are a second pair (Figure 1). The PAN and BAT catchments were comprised of 75 to 95% deciduous tree species (primarily sugar maple and yellow birch) and had a road density of 12.9 to 13.8 m/ha; the KER A and KER B catchments were comprised of 67 to 87% deciduous tree species and had a road density of 6.8 to 10.4 m/ha (Ontario Forest Resource Industry Database) (Table 1). All catchments were dominated (>70%) by bedrock knobs (dominant landform) and ground moraines composed of glacial till (subordinate landform) with high relief and a generally dry ground surface (Ministry of Norther Development and Mines Geology Terrain Database).

Within each catchment are three sample sites (i.e., ~30 m long reaches) positioned along a spatial gradient (Figure 2). Our ability to achieve a spatial gradient adequate for detecting both the natural river continuum and potential cumulative trends from forest harvesting was difficult to due limited access to sites within the forested landscape. When possible, sites were chosen on the basis of (1) increasing stream order,

(2) increasing drainage area (generally positively related to stream size, Downing et al. 2012), and (3) were ideally hydrologically connected along the same channel. Because the total area of the catchment and the total area harvested increased with stream size (i.e., greater in lower reaches than in headwaters), this design allowed for detection of an accumulation of effects throughout the catchment. Selected sites ranged from headwaters (1^{st} to 2^{nd} order streams, average stream width = 1.48 m), to middle-reach (2^{nd} to 3^{rd} order, average stream width = 2.56 m), to downstream (3^{rd} to 4^{th} order, average stream width = 5.23 m) (Figure 2). In 2017, the middle-reach site in the BAT catchment was changed (i.e., from BAT 2 to a new site, BAT 2new*) as to have a site that was better positioned along the stream network to represent a mid-way point between upstream (BAT 3) and downstream (BAT 1). Therefore, BAT 2 represented the middle-reach site for 2016 data, but BAT 2new* represented the middle-reach site for 2017 data. All sites were assessed for natural changes or impacts from harvesting using indicators of impact (see below).

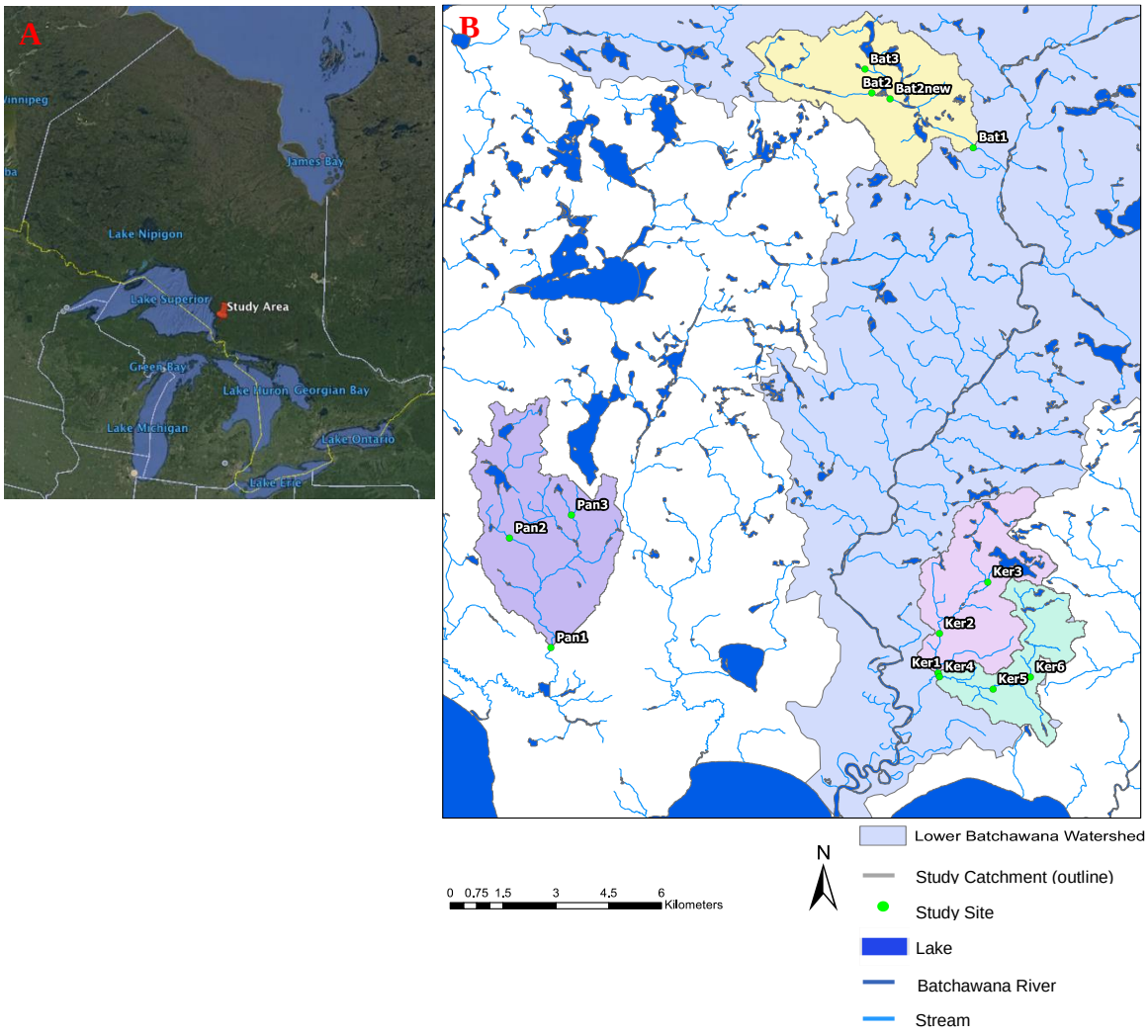

Figure 1: Map of study location on northeast shore of Lake Superior in Ontario, Canada (A, Google Earth 2017) and location of study catchments and sample sites therein (B).

Table 1: Catchment and site/sub-catchment characteristics and harvesting information for 13 study sites. Note: Site "BAT 2new*" added in the second season of sampling; no data for this site in 2016.

Catchment	Site/Sub-Catchment	Latitude	Longitude	Position	Dist to DS (km)	Area (ha)	% EVSA	% Dec	TAH (ha)	% Harv	LRD (m/ha)
Catchment Pair 1											
PAN *(Reference)*	PAN 3	47.01053	-84.6239	HW	5	74	17.02	95.00	0	0	1.77
	PAN 2	47.00508	-84.6473	MR	4.1	544	11.27	89.27	0	0	17.45
	PAN 1	46.97642	-84.633	DS	0	1856	13.37	87.49	0	0	13.79
BAT *(Harvested)*	BAT 3	47.12349	-84.5088	HW	4.36	49	13.78	86.67	47.79	97.53	27.52
	BAT 2	47.11725	-84.5065	MR	3.56	497	11.57	75.32	301.91	60.75	11.99
	BAT 2new*	47.11535	-84.49937	MR	2.96	921	8.75	76.21	301.29	32.71	13.73
	BAT 1	47.10231	-84.4692	DS	0	1688	9.55	75.15	736.79	43.65	12.87
Catchment Pair 2											
KER A *(Reference)*	KER 3	46.98988	-84.4691	HW	3.66	189	16.90	81.43	0	0	3.66
	KER 2	46.97781	-84.4875	MR	1.2	959	11.37	67.74	0	0	9.04
	KER 1	46.96698	-84.4891	DS	0	1278	10.92	71.04	66.72	5.22	10.37
KER B *(Harvested)*	KER 6	46.96558	-84.4547	HW	3.02	392	11.25	86.50	0.95	0.241	11.15
	KER 5	46.96192	-84.4684	MR	1.82	594	11.68	86.18	130.03	21.89	8.18
	KER 4	46.96579	-84.4893	DS	0	785	11.35	86.16	34.94	31.94	6.77

[1]Position = Categorical description of site location (HW = Headwater, MR = Middle-Reach, DS = Downstream); Dist to DS = Stream distance (km) from current site to farthest downstream site within the catchment; Area = sub-catchment area (Ha); % EVSA = % Effective Variable Source Area (indicates % soil wetness as determined using the topographic index within catchment); % Dec = % of deciduous tree species in sub-catchment; TAH = Total area of sub-catchment harvested (ha) within the last 5 years; % Harv = % Area of sub-catchment harvested within the last 5 years; LRD = Logging road density of sub-catchment (m road per ha catchment).

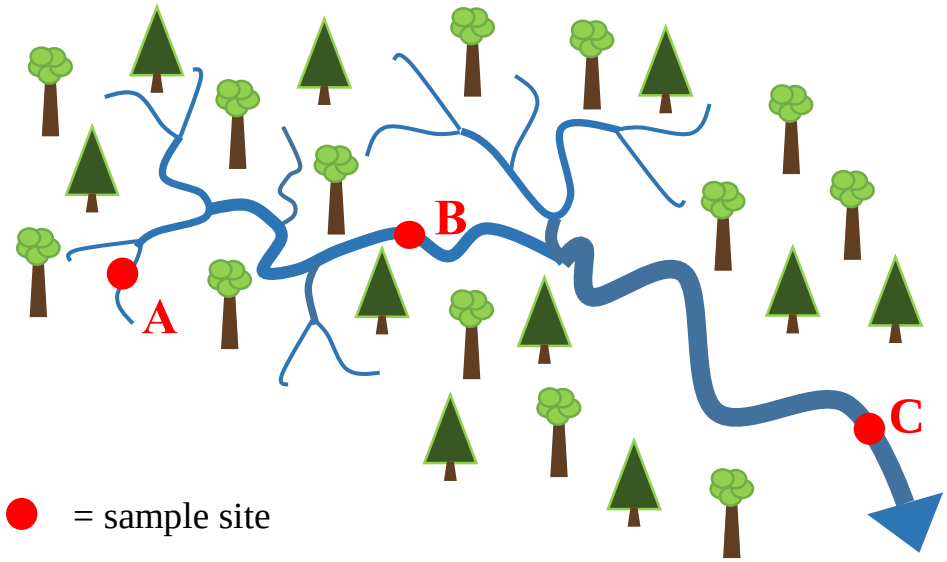

Figure 2: Example of sample site positioning within a catchment: headwater/upstream (A), middle-reach (B), and downstream (C).

2.2. Indicators of Spatially Cumulative Impacts

2.2.1. Sampling Details

Sampling was carried out at 12 sites (i.e., all except BAT 2new*) in September and October in 2016 and at all 13 sites in August, September, and October in 2017 (Table 1). When sites were near road crossings, sampling occurred immediately upstream. Collections/measurements were taken by the same individual(s) of the field team during each sampling event to ensure consistency over time.

2.2.2. Stream Temperature

Stream water temperature was recorded bi-hourly from July to October in 2016 and from June to October in 2017 using submersible temperature loggers (HOBO TidbiT v2 water temperature data loggers, Onset Computer Application). Mean fall temperature (Sept 1st until temperature logger removal in October), summer maximum temperature (from deployment in June (2017) or July (2016) until August 31st), and monthly degree days (sum of daily average temperatures for a given month) were

calculated in °C. No temperature data was collected for sites PAN 3 and 1 and BAT 2 in 2016 as temperature loggers were lost or dead upon retrieval. See Appendix Table 1 & Appendix Table 2 for site-specific logger deployment and removal dates.

2.2.3. Stream Size & Flow

Stream water flow characteristics, including mean water depth, stream width, flow velocity, and discharge volume, were determined in August, September, and October of 2017 (with the exception of PAN 2, which was not measured in August due to adverse weather conditions). Mean water depth and stream width were measured using a meter stick and a tape measurer, respectively, with depth averaged for measurements recorded at 0.1 to 0.5 m intervals across the width of the stream. Water flow velocity and discharge were also recorded at 0.1 to 0.5 m intervals across the width of the stream using a current meter (Global Flow Probe model FP201; Global Flow Instrumentation, Gold River, California) at mid-depth at the same locations as depths and width, and averaged for each study reach.

2.2.4. Water Chemistry & Dissolved Organic Matter (DOM) Quality

Stream water samples for water chemistry analyses (N=1/site) were collected in September and October in 2016 and August, September, and October in 2017. Sub-surface grab samples of water were stored in clean, 500 mL polyethylene bottles upstream of stream activity (i.e., road crossings, other sampling, etc.) and stored at 4°C until analysis. Prior to analysis, samples were passed through Whatman No. 41 filters to remove particulate matter. The parameters measured were pH, conductivity, alkalinity, cations and anions (Ca, K, Mg, Na, SO_4, Cl, SiO_2), nutrients (NO_2+NO_3, NH_4, total N, total organic carbon (TOC), total inorganic carbon (TIC), soluble reactive P (SRP), and

total P (TP)), and metals (Al, Fe, Mn, Zn, Cd, Cu, Ni, Pb). Water chemistry analyses were conducted by the Water Chemistry Laboratory at the Great Lakes Forestry Centre (GLFC) in Sault Ste. Marie, ON, using standardized procedures and quality-control methods (Nicholson 1988). Stream water pH was measured using an Orion Ross Ultra Glass Electrode (Man-Tech PC Titrate, Orion Thermo combination pH electrode, Version 3 Software). Conductivity (µmho/cm at 25°C) was measured using a PCE-96-CT1003 conductivity cell (Man-Tech PC Titrate with 4510 conductivity meter, Version 3 Software). Alkalinity (meq/L) was measured using electrometric titration (Man-Tech PC-Titrate, Titra-Sip titrator, Orion Thermo combination pH electrode, Version 3 Software). Ca, K, Mg, Al, Fe, Mn, Zn, Cd, Ni, and Pb (ppm) were measured using inductively coupled plasma mass spectrometry (ICP MS; Agilent 7700x Inductively Coupled Plasma Instrument, Masshunter Software). Sulfate and chlorine (ppm) were measured using conductance-ion exchange suppression (Dionex ICS 1100 Ion Chromatograph, Chromeleon 7.0 Software). SiO_2 (ppm) was measured by automated ascorbic acid (Technicon Autoanalyzer II, N.A.P. Software Version 4.4). NO_2+NO_3 and NH_4 (ppm) were measured by automated cadmium reduction and automated sodium nitroprusside, respectively (Seal Analytical AA3 Autonanalyzer, AACE 6.07 Software). TOC and TIC (ppm) were measured by autoclave digestion – cadmium reduction, acid – and sulfuric acid – carbon dioxide methods, respectively (Seal Analytical AA3 Autoanalyzer, AACE 6.07 Software). Soluble reactive P and total P were measured using automated molybdophosphoric blue (Technicon Autoanalyzer II, N.A.P. Software Version 4.4 – Autoclave Digestion). Total N was measured using automated cadmium

reduction (Technicon Autoanalyzer II, NO_2 and NO_3 channel – autoclave digestion, N.A.P. Software Version 4.4).

Stream water samples for DOM quality analyses (N=1/site) were collected in September and October in 2016 and August, September, and October in 2017. The quality of DOM refers to the ease of its microbial decomposition (i.e., higher ease = higher quality), which is dependent on molecular weight and structure (Thurman 1985). Sub-surface grab samples of water were collected in clean, 50 mL polypropylene centrifuge tubes upstream of stream activity and stored at 4°C until analysis. Prior to analysis, samples were passed through Whatman No. 41 filters to remove particulate matter. Using Cary eclipse and Cary 60 UV-Vis spectrophotometers (three-dimensional fluorescence scans at 5 nm excitation steps from 250 to 450 nm and 2 nm emission steps from 300 to 600 nm), optically-derived properties of DOM were obtained from corrected and adjusted excitation-emission matrices for each water sample. These properties include humification index (HIX, an indicator of the degree of DOM humification; Ohno 2002), fluorescence index (FI, an indicator of terrestrially- (1.8) vs. microbially- (1.2) derived DOM; Johnson et al. 2011), freshness index (β:α, an indicator of recently derived (β) or older (α) DOM; Huguet et al. 2009), specific UV absorbance at 254 nm (SUVA, an indicator of DOM aromaticity; Weishaar et al. 2003), E2-to-E3 ratio (E2E3, an indicator of DOM molecular weight and aromaticity; Peuravuori and Pihlaja 1997), and specific absorbance coefficient at 340 nm (SAC_{340}, an indicator of the amount of aromatic DOM; Wood et al. 2011).

2.2.5. Sediment Deposition

Sediment deposition was measured over a six-week period between September and October in 2016 and again between September and October in 2017 (see Appendix Table 14 & Appendix Table 15 for exact deployment and removal dates). Sediment deposition was measured by deploying six (2017) to seven (2016) collectors (50 mL polypropylene centrifuge tubes with caps removed) on the stream floor in depositional areas of each 30 to 50 m reach. To keep their position in the stream, the collectors were wedged into the holes of a brick (protruding about 2 cm above) and tied to a branch or root on the bank. Previous studies have suggested that this collector design is sufficiently deep to retain fine particles deposited from the water column (Kreutzweiser and Capell 2001). Following the six-week deployment period, the collectors were capped, removed, and then preserved with formaldehyde until analysis.

Sediment retained in the collectors was size-fractionated to obtain the fine (1.5 µm to 250 µm) and coarse (1 mm to 250 µm) fractions. To separate the fractions, collector contents were rinsed through a bank of 4 mm, 1 mm, and 250 µm sieves. The material retained in the 250 µm sieve was the coarse fraction. The filtrate washed through the sieves was vacuum-filtered onto pre-ashed, 1.5 µm glass microfiber filters, with the mass on the filter representing the fine fraction. The sediment fractions were oven dried in pre-weighed aluminum dishes at 60°C for two days and weighed, and then combusted at 500°C for two hours to remove their organic components and re-weighed. The sediment remaining after combustion was the inorganic fraction. The organic fraction was calculated by subtracting the inorganic weight from the initial dry weight (i.e., prior to combustion). Inorganic sediment deposition per day was calculated by dividing coarse or fine deposition by the number of days the collectors were deployed.

The percentage of organic material in the sediment fractions was calculated by dividing the organic mass by the total dry mass.

2.2.6. Leaf Litter Decomposition & Associated Macroinvertebrates

Leaf litter decomposition and associated macroinvertebrate community structure were measured in standardized leaf packs deployed over a six-week period between September and October at the time of natural leaf fall in 2016 and 2017 (see Appendix Table 19 & Appendix Table 20 for exact deployment and removal dates). Leaf packs are plastic coarse-mesh bags (5 mm x 10 mm mesh size) containing approximately 10 g of leached, air-dried, and pre-weighed senescent speckled alder leaves (*Alnus rugosa*, a ubiquitous deciduous species in the riparian zone of all study reaches) and a circular wire frame (20 cm x 15 cm, to help leaf pack keep its shape and facilitate movement of water/oxygen and macroinvertebrates) (Kreutzweiser et al. 2008a). Leaf packs have been suggested to be ecologically relevant for assessing the effects of land use on stream health due to the critical role of allochthonous litter in stream ecosystem functioning, the sensitivity of leaf-litter breakdown to anthropogenic disturbance, and the relative ease of leaf pack implementation and processing (Gessner and Chauvet 2002). The mesh size used in this study was chosen such that stream macroinvertebrates were able to penetrate the bag and access the leaf litter; mass loss of leaf material in these coarse mesh leaf packs was therefore the cumulative result of macroinvertebrate feeding, microbial decomposition, and physical abrasion. Previous studies have suggested a deployment period of at least five weeks to be adequate time for measurable decomposition; this time period was also employed to avoid rain- and snow-induced flooding common to the region after late October (Kreutzweiser et al. 2008a).

Three leaf packs were deployed in depositional areas of the stream floor along a 30 to 50 m reach. Leaf packs were attached to the same brick as the sediment collectors, being careful to avoid pools with heavy silt, areas of high turbulence, and plunge pools (Kreutzweiser and Capell 2001). Upon their retrieval in October, a 0.5 mm mesh net was placed under and downstream of the brick to prevent loss of material or invertebrates. Leaf packs (and material retained in the net) were placed in a collection container and preserved with formaldehyde until analysis.

At the GLFC in Sault Ste. Marie, ON, the contents of the collection containers were rinsed through a 250 µm sieve to remove the preservative. The rinsed material was transferred to an elutriation trough to facilitate separation of macroinvertebrates from leaf material. The contents of the trough were then rinsed through a stack of 4 mm, 2 mm, and 500 µm (2016) or 250 µm (2017) sieves and transferred to sorting trays. The use of sieves eased identification of macroinvertebrates in leaf litter and debris, and all material in sieves was sorted (i.e., no sub-sampling). Inter-year differences in sieve sizes was unintentional (human error). Leaf material was retained in a pre-weighed aluminum dish, while macroinvertebrates, made more visible by addition of a protein-binding dye (phloxine B), were retained separately in vials with ethanol. The leaf fragments were dried to constant mass in a drying oven at 60°C and weighed.

Percent leaf mass lost per degree day was calculated as the difference between initial and post-deployment dry leaf mass divided by the initial leaf mass, then divided again by the number of degree days (sum of daily mean temperatures during deployment period). Leaf breakdown rates were determined using the following equation (Petersen and Cummins 1974):

$$k = - \ln \left(\frac{M_t}{M_0} \right) / t$$

where k is breakdown rate, M_t is the post-deployment dry leaf mass, M_0 is the initial dry leaf mass, and t is the total degree days over the deployment period. Both percent leaf mass lost and breakdown rate were corrected for temperature and duration of deployment because mass lost and decomposition are known to slow with time and temperature (Irons III et al. 1994; Lecerf and Richardson 2010). Note that because no temperature data was recorded for sites PAN 3 and 1 and BAT 2 in 2016, degree days at sites within the other catchment with the same treatment (i.e., KER 3, 1, and 5, respectively) were used to correct percent leaf mass lost and breakdown rate for temperature.

The macroinvertebrates separated from the leaf material were identified to genus by conventional taxonomic methods, with the exception of Chironomidae (3 subgroups identified: Tanypodinae, Tanytarsini, and "other"), Ceratopogonidae, Tabanidae, Gomphidae, Elmidae, and Dytiscidae, which were identified to family, and Oligochaeta, Gastropoda, and Hirudinea which were identified to order. Metrics used to compare macroinvertebrate community structure include total abundance (average of total number of individuals in leaf packs), taxonomic richness (average number of taxa present in leaf packs), % EPT (average % of individuals in orders Ephemeroptera, Plecoptera, and Trichoptera in leaf packs), % shredders (average % of detritivores consuming coarse particulate organic matter in leaf packs, as identified by Merritt and Cummins (1996)), % Chironomidae (average % of individuals belonging to the Family Chironomidae in leaf packs), Margalef's Richness (average measurement of richness independent of sample size in leaf packs), and Shannon's Diversity Index (average

diversity index accounting for abundance and richness in leaf packs; higher index = more diverse community). These metrics were chosen due to their common use in many biomonitoring programs (e.g., Davis et al. 2001) and due to their relevance to leaf-litter decomposition (Kreutzweiser et al. 2008a).

2.2.7. Hydrogen Stable Isotopes

Hydrogen stable isotope values (i.e., the ratio of deuterium (^{2}H) to hydrogen (H)) were measured in late instars of larval caddisflies (Family: Hydropsychidae), leaves, biofilm, sediment, and water in October 2016 and August, September, and October 2017 to determine diet autochthony (i.e., % algae in diet) for Hydropsychids. Hydropsychids are filter-collectors who spin silken nets to capture food particles (e.g., phytoplankton or smaller items) from the water column, and account for approximately 80% of all Trichopterans in North American streams (Robach 1962). Hydropsychids were chosen for hydrogen stable isotope analysis because they were present at all study sites, and because they use a filter-collecting feeding strategy, which renders their diet a good indicator of the source of organic matter within the suspended pool (Jonsson et al. 2018). For each sampling event, single composite samples of 50 to 100 Hydropsychids were collected by electro-shocking and disturbing rocks around cobble-based riffles over a 30 to 50 m reach at each study site and placed in Whirl-pak® bags with stream water for transport to the lab. In the lab, Hydropsychids were transferred to petri dishes with stream water overnight to facilitate egestion of gut contents, and then frozen at -20°C until analysis (Haro et al. 2013). No results are reported for caddisflies at site PAN 3 in October 2016 due to insufficient sample mass. Leaves (N=1 composite sample per site per sampling event) were scavenged from the water surface or debris jams, placed in

Whirl-pak® bags, and frozen at -20°C until analysis. Biofilm (N=1 composite sample per site per sampling event) was scraped from in-stream rock surfaces with a toothbrush, rinsed with stream water into a Whirl-pak® bag, and stored at -20°C until analysis. Sediment (N=1 composite sample per site per sampling event) was collected using a turkey baster in depositional areas of study streams and stored in Whirl-pak® bags at -20°C until analysis. Stream water (N=3 samples per site per sampling event) was collected at each site using a 10 mL syringe with 0.22 µm filter and stored in 1.5 mL amber glass vials in the dark at 4°C until analysis.

Hydropsychid, leaf, biofilm, and sediment samples were examined for impurities (e.g., invertebrates and sediment on leaf material or in biofilm matrix), then freeze-dried, homogenized, and weighed into silver microcapsules (5 x 3.5 mm). Sample mass required for analysis was determined based on % hydrogen of the material in order to produce a return of 0.1 mg hydrogen (i.e., 1.4 mg for Hydropsychids, 1.6 mg for leaves, and 8 mg for biofilm). Composite samples were analyzed in triplicate for bulk hydrogen stable isotopes by converting sample hydrogen to hydrogen gas and measuring the isotopic composition using a Thermo-Finnigan Delta Plus XP continuous-flow-isotope-ratio mass spectrometer at the G.G. Hatch Stable Isotope Laboratory (Ottawa, ON). Internal standards included hair, kaolinite, and cellulose, and were calibrated to international standards with precision of ± 2‰ (SD) for organic material and ± 3‰ (SD) for inorganic material. Samples and standards were acclimated to local atmospheric hydrogen prior to analysis to correct for exchangeable hydrogen (Wassenaar and Hobson 2003). Hydrogen stable isotope values in water were measured using an LGR

Liquid Water Isotope Analyzer DLT-100 at the Colorado Plateau Stable Isotope Laboratory (Flagstaff, Arizona, USA).

Hydrogen stable isotope values are relative to the Vienna Standard Mean Ocean Water and expressed as δ^2H in parts per thousand (‰), according to this equation:

$$\delta^2H = \left(\frac{R_{sample} - R_{standard}}{R_{standard}} - 1\right) * 1000$$

where R is the corresponding $^2H/H$ ratio (Fry 2006).

Furthermore, because a consumer's isotope ratio is a combination of environmental water and assimilated food, the environmental water contribution (δ^2H_{WC}) was accounted for before calculation of % algae, according to this equation:

$$\delta^2H_{WC} = \delta^2H_{cons} - (\delta^2H_{cons} - \omega_{tot} * \delta^2H_{water})/(1 - \omega_{tot}),$$

where δ^2H_{water} is the isotopic value (‰) of stream water, and ω_{tot} is the total contribution of dietary water to the isotopic value (‰) of the consumer (δ^2H_{cons}), calculated according to:

$$\omega_{tot} = 1-(1- \omega)^{\tau},$$

where ω is the per-trophic-level contribution of dietary water to consumers and τ is the trophic level ($\tau = 1$ for primary consumers, including Hydropsychids). A ω of 0.20 ± 0.1 was assumed based on previous studies by Solomon et al. (2009), Wang et al. (2009), and Wilkinson et al. (2015).

The relative contribution of aquatic organic matter to the diet of Hydropsychids (% algae of diet) was estimated using a two-source mixing model, according to this equation:

$$\% \text{ Algae of Diet} = (\delta^2H_{cons} - \delta^2H_{ter}) - (\delta^2H_{aq} - \delta^2H_{ter}) * 100$$

where δ^2H_{cons} is the isotope value (‰) of the consumer (Hydropsychids, corrected for environmental water contribution), δ^2H_{ter} is the isotopic value (‰) of the terrestrial food source (leaves), and δ^2H_{aq} is the isotope value (‰) of the aquatic food source (biofilm).

Sediment was excluded from the analysis of % algae of diet as plots of δ^2H indicated it was well outside the diet range of caddisflies (Appendix Figure 4). Additionally, Hydropsychid δ^2H values were often found outside the boundaries of terrestrial and aquatic food sources, or there was overlap between the two food sources, rendering the mixing model variably informative. It is proposed that this was due to biofilms not reflecting the isotopic value of aquatic primary producers, but instead being more representative of the mix of components (i.e., algae, bacteria, fungi, sediment, etc.) included in the biofilm matrix. As a solution to this issue, an attempt was made to determine % algae of diet by calculating algal isotope values by subtracting 170 ± 15‰ from the δ^2H value for environmental water for that site, as per studies that suggest primary producers have 160 to 170‰ lower δ^2H than environmental water due to fractionation against deuterium during photosynthesis (Solomon et al. 2011; Hondula et al. 2014). Therefore, the two sources used in mixing models for 2016 and 2017 were leaves (measured, terrestrial/allochthonous food source) and algae (calculated, aquatic/autochthonous food source).

2.2.8. Mercury (Hg)

Different species of Hg were measured in multiple environmental compartments over the two sampling years. In October 2016, MeHg was measured in stream water and MeHg and Hg(II) were measured in Hydropsychids. In August, September, and October

2017, MeHg was measured in stream water, MeHg and Hg(II) were measured in seston and Hydropsychids, and THg was measured in sediment.

Stream water samples (N=4 to 6 samples per site per sampling event) were collected in 250 mL amber glass bottles with Teflon caps by grab-sampling with powder-free gloves upstream of steam activity. Within 12 hours, half of the samples collected at each site were filtered via vacuum filtration using 0.45 µm polyestersulfone (2016) or pre-ashed quartz fibre (2017) filters. Polyestersulfone filters (PFs) were swapped for quartz fibre filters (QFFs) in 2017 as QFFs are known to have low background Hg levels but produce results comparable to PFs (Lewis and Brigham 2004). All water samples (filtered and unfiltered) were preserved with 0.5% HCl (ACS reagent grade, ~34%) and stored in the dark at 4°C until analysis.

Hydropsychids were collected as described above (i.e., same composite sample used as for hydrogen isotope analyses). In October 2017, no Hydropsychids were available for collection for Hg analysis at sites BAT 2 and BAT 3, presumably because they had been washed away by strong flows caused by an extended period of heavy rainfall.

In 2016, an attempt was made to determine Hg in seston by subtracting Hg in filtered stream water from Hg in unfiltered stream water. However, since dissolved Hg often makes up the majority of the waterborne pool, the difference between unfiltered and filtered water, in combination with instrument error, was too small to determine sestonic Hg concentration (Morrison and Watras 1999). In 2017, an alternative method was developed to collect seston in the field (J. Ogorek, USGS, personal communication). Stream water (100 to 300 L) was poured through a clean, 4”-diameter

PVC pipe containing a 63 µm Nitex screen. The retained material was back-washed from the screen into a clean, 500 mL Teflon bottle using deionized water, producing one composite sample per site per sampling event in 2017. Within 12 hours, the retained material was size fractioned to 63 to 500 µm and frozen at -20°C until analysis. This size fraction was chosen to represent the size of particulate matter filtered by the nets of the Hydropsychid caddisflies (Fuller and Mackay 1980). Due to the low concentrations of suspended material in study streams, insufficient seston mass for Hg analysis was collected for 67% of sites over the three sampling events (Appendix Table 36), so monthly data were combined to produce a single 2017 data set for Hg in seston.

In 2017, sediment was collected for total Hg (THg) analysis as described above (i.e., same composite sample used as for hydrogen isotope analyses). THg in sediment was size-fractioned from 63 to 500 µm (i.e., same fraction as seston) and frozen at -20°C until analysis. Data and analyses for THg in sediment are shown in Appendix Table 37 and Appendix Table 38.

MeHg was extracted from water samples via direct aqueous ethylation and analyzed with a Brooks-Rand automated MERX system. Direct ethylation required pH adjustment to 4.5 to 5.0 using 25% KOH to ensure effectiveness of the 2 M acetate buffer and ethylating agent (1.33 M tetraethylborate in 2% KOH). MeHg was measured using derivatization, purge and trap, and cold vapour atomic fluorescence spectrometry with the Model II Brooks-Rand detector (US EPA Method 1630 modified for direct ethylation (Mansfield and Black 2015; Brooks Rand Analytical notes)). MeHg concentration was calculated based on external calibration curves with minimum five data points for known standards ($R^2 = 0.99$) and reported as ng MeHg per L water.

Samples with 50 pg matrix spike recoveries below 90% and above 110% were recovery-corrected (2016 and 2017). The method blank concentration was subtracted from all samples in 2017 (no method blank in 2016). MeHg in all samples was above the method detection limit (0.59 pg for 2016 and 0.50 pg for 2017), calculated as three times the standard deviation of the calibration (2016) or the method (2017) blanks. Relative standard deviation for sampling triplicates was 17.6% (n=48) for 2016 and 7.9% (n=12) for 2017. Relative standard deviation for analytical triplicates was 15.8% (n=8) for 2016 and 8.3% (n=48) for 2017.

MeHg and Hg(II) in freeze-dried and homogenized Hydropsychids (average moisture = 85%) were extracted and measured following the same analytical procedure and instrumentation as for water, with the addition of prior sample digestion in 25% KOH in MeOH, shaking for one hour, and baking for one hour at 90 to 95°C. Triplicate composite samples for each site were analyzed. MeHg and Hg(II) concentrations were calculated based on external calibration curves with minimum five data points for known standards ($R^2 = 0.99$) and reported as µg MeHg or Hg(II) per kg dry weight (dw) of Hydropsychid. Certified reference material (CRM; DOLT-5, National Research Council Canada) was additionally used to ensure instrument precision and accuracy, and recovery was on average 108.2 ± 30.6% for MeHg (n = 4) and 69.0 ± 17.6% for Hg(II) (n= 4) in 2016 and 111.7 ± 11.6% for MeHg (n = 6) and 75.5 ± 7.5% for Hg(II) (n = 6) in 2017. Hg(II) was recovery-corrected based on CRMs in 2016, and based on a rolling average of ongoing recovery check standards in 2017. MeHg and Hg(II) in all samples were above the method detection limit (0.82 pg for MeHg and 8.69 pg for Hg(II) in 2016, and 0.52 pg for MeHg and 1.56 pg for Hg(II) in 2017). The method blank

concentration was subtracted from all samples in 2016 and 2017. Relative standard deviation for analytical triplicates was 16.2% (n=12) for MeHg and 20.2% (n=12) for Hg(II) in 2016, and 11.1% (n=6) for MeHg and 17.6% (n=6) for Hg(II) in 2017. The % MeHg was calculated as the concentration of MeHg divided by the sum of concentrations of MeHg and Hg(II) (assumed to represent total mercury species in tissue) and multiplied by 100.

MeHg and Hg(II) in freeze-dried and homogenized seston was determined following the same digestion, ethylation, and analytical procedure as described for invertebrate tissue. Composite samples for each site were analyzed individually due to mass restrictions (i.e., not run in triplicate as for Hydropsychids), with the exception of two samples run in duplicate to determine analytical precision. MeHg and Hg(II) concentrations were calculated based on external calibration curves with minimum five data points for known standards ($R^2 = 0.99$) and reported as µg MeHg or Hg(II) per kg dw of seston. MeHg and Hg(II) in all samples were above the method detection limit (0.37 pg for MeHg and 2.24 pg for Hg(II)). Relative standard deviation for analytical duplicates was 3.4% (n=2) for MeHg and 1.1% (n=2) for Hg(II). % MeHg was calculated as for Hydropsychids.

THg in freeze-dried sediment was measured using a Nippon Instruments Corporation Mercury Analyzer (NIC-MA 3000). THg was not measured at the upstream site in KER A (site KER 3) for August 2017 due to insufficient sample mass. CRM (MESS-3, National Research Council Canada) recovery was on average 102.3 ± 2.2% for THg (n = 3). THg in all samples were above method detection limit (0.002 ng). Relative standard deviation for analytical triplicates was 14.9% (n=3). THg in sediment

was reported as ppb (dw). Organic matter content of sediment was also determined by combusting pre-weighed samples at 550°C for two hours. The mass of the sediment following combustion was subtracted from the initial mass to determine the proportion of organic matter lost on ignition (LOI). LOI was reported as a percentage of mass lost relative to the initial sample mass (data shown in Appendix Figure 8).

Bioaccumulation factor (BAF) for MeHg in Hydropsychids was calculated for October 2016 and August, September, and October 2017 using the following equation:

$$BAF = \frac{[MeHg]_H}{[MeHg]_{FW}}$$

where $[MeHg]_H$ is the monthly average concentration of MeHg in Hydropsychids (dw) and $[MeHg]_{FW}$ is the monthly average concentration of MeHg in filtered water (e.g., Rolfhus et al. 2011).

Biomagnification factor (BMF) for MeHg in Hydropsychids was calculated for August, September, and October 2017 using the following equation:

$$BMF = \frac{[MeHg]_H}{[MeHg]_S}$$

where $[MeHg]_H$ is the monthly average concentration of MeHg in Hydropsychids (dw) and $[MeHg]_S$ is the 2017 average concentration of MeHg in seston (dw) (e.g., Rolfhus et al. 2011).

2.3. Statistical Analysis

Data for stream temperature, size, and flow are reported in the results for descriptive purposes but were not analyzed statistically or used to make inferences about harvesting impacts because they were not adequately characterized throughout the sampling period and/or were unlikely to be influenced by a selection-based harvesting

approach. The remaining indicators underwent statistical analysis using a combination of multi-variate techniques, analysis of variance, and multiple linear regression.

2.3.1. Principal Component Analysis

Because many parameters were measured for water chemistry (N=25) and DOM quality (N=7), Principal Component Analyses (PCAs) were used to reduce the number of variables for each indicator and identify the parameters contributing most significantly to variation among sites. Prior to conducting PCAs, all values for water chemistry and DOM quality were averaged among the five sampling events (i.e., September and October of 2016 and August, September, and October of 2017) to obtain a more robust measurement, and all averaged values were scaled and centered (i.e., to minimize the influence of large values due to differences in units or ranges among variables). Both PCAs were done in R (R Core Team 2016). Variables with the highest correlations with the first two PC axes for water chemistry (WC_PC1 and WC_PC2) and DOM quality (DOM_PC1 and DOM_PC2) were selected for further analysis of spatially cumulative trends and spatial scale of harvesting impact using analysis of variance. Scores for each site on WC_PC1, WC_PC2, DOM_PC1, and DOM_PC2 were used as explanatory variables in mixed effects linear models.

2.3.2. Non-Metric Multi-Dimensional Scaling

Non-metric multi-dimensional scaling (NMDS) analyses were used to observe leaf pack invertebrate community structures in multi-variate space among sites (i.e., all catchments considered, not just within pairs) in 2016 and in 2017. Prior to NMDS analyses, taxa that were present at less than 10% of sites were removed from the data set to minimize potential bias from highly uncommon taxa (Appendix Table 31). Average

abundance of taxa within sites for each year were auto-transformed (Wisconsin's double standardization and square-root transformation) to reduce the influence of dominant taxa on the ordination. NMDS analyses were carried out in R (R Core Team 2016) using the following packages: *vegan* (Oksanen et al. 2017) and *MASS* (Venables and Ripley 2002). An analysis of similarities (ANOSIM) was conducted for both years to test for significant differences in invertebrate communities with varying treatment (i.e., reference vs. harvested) and/or site (i.e., upstream, middle-reach, and downstream locations).

2.3.3. Analysis of Variance (ANOVA)

Two-factor analyses of variance (ANOVA) were conducted within paired-catchment comparisons and sampling events to test the null hypothesis of no interaction between treatment (i.e., reference vs. harvested, a categorical variable) and site (i.e., upstream, middle-reach, and downstream positions, a categorical variable), as well as no individual effect of treatment and/or site, on average indicator response within catchment comparisons (GraphPad Prism Version 7.0 d Software, GraphPad Software Inc., San Diego, CA, USA). Rejection of the null hypothesis ($\alpha = 0.5$) for the interaction between the treatment and site was used as evidence of spatially cumulative trends, as it indicated that the spatial trend in indicator values in harvested catchments was different than in reference catchments. "Limited evidence", "some evidence", and "strong evidence" of spatially cumulative trends was provided if >50%, 50 to 75%, and >75%, respectively, of paired catchment comparisons had a significant interaction between treatment and site. The presence of an interaction was supplemented by testing for an effect of treatment within sites and an effect of site within treatments using the same

two-factor ANOVA. The same criteria for strength of evidence (i.e., “limited”, “some”, or “strong” evidence among paired comparisons) were used for individual effects of treatment and site. When no interaction was found, a multiple comparisons test was used to test for effects of treatment (Sidak’s Multiple Comparisons) or site (Tukey’s Multiple Comparisons). When an interaction was found, a t-test or a one-factor ANOVA was used to test for the individual effects of treatment or site, respectively. The presence of an adverse treatment effect at the downstream site within a paired comparison was used as evidence for lack of effectiveness of BMPs at broader spatial scales. Prior to running ANOVAs, the normality of the distribution for an indicator response across all sites and years was assessed using the Shapiro-Wilks test and values were transformed as necessary to improve normality and homogeneity of variance.

2.3.4. Linear Mixed Effects Models

Linear mixed effects models were built to identify the combination of explanatory variables (EVs; i.e., catchment, reach, harvesting, and sampling characteristics) that best predicted variance in response variables (RVs; i.e., abiotic, biotic, and contaminant indicators) (Table 2 & Table 3). This approach supplemented ANOVA results by determining if treatment (% area harvested within the last five years, a continuous variable) and site (distance to the farthest downstream site multiplied by -1, a continuous variable), and/or an interaction between the two, were significant predictors of variability in the RVs. This approach also allowed for determination of the influence of other EVs (e.g., year, % EVSA) on variation in the RVs. Site positions (PAN 3, 2, 1; BAT 3, 2 or 2new*, 1; KER A 3, 2, 1; KER B 6, 5, 4) and catchment treatment (PAN_{REF}, BAT_{HARV}, KER A_{REF}, KER B_{HARV}) were included in models as

random effects, unless their standard deviation was 0, in which case they were changed to fixed effects or excluded from the model. All other EVs were included as fixed effects. The mixed effects design is desirable for a nested study with limited replication as it models the error structure to overcome omitted variable bias and facilitates more general conclusions (Zuur et al. 2007). All models were built in R (R Core Team 2016) using the following packages: *effects* (Fox 2003), *car* (Fox and Weisberg 2011), *lme4* (Bates et al. 2015), *influence.ME* (Nieuwenhuis et al. 2012), *lmerTest* (Kuznetsova et al. 2017), and *MuMIn* (Bartoń 2018).

For each RV, values for all reference (PAN and KER A) and harvested (BAT and KER B) catchments were analyzed collectively in one model as opposed to upholding the paired-catchment comparison design as was done for the ANOVAs (i.e., PAN vs. BAT, KER A vs. KER B). While catchments were originally chosen to be more similar within pairs than among pairs, all were ultimately dominated by the same geology and forest composition, so it was deemed acceptable to combine catchments within treatments for modelling purposes. The rationale behind doing this was to increase the sample size and thus statistical power of the models. Values were not averaged or separated by year as was done for ANOVAs, with the exception of macroinvertebrate community metrics which were separated by year due to different sieve sizes among years (see above).

Prior to building models, the normality of the distribution for RVs across all sites and years was assessed using the Shapiro-Wilks test and values were transformed as necessary to improve normality and homogeneity of variance. All proportions (in EVs or RVs) were logit-transformed and all EVs were scaled to minimize the influence of

large values due to differences in units or ranges among variables. Proportions below 0 or above 100 were changed to 0 and 100, respectively, to meet the assumptions of the logit transformation (see % algae in Hydropsychid diets). Note that RVs were sometimes used as EVs in models for other indicators; for example, sediment deposition was included as an EV in models where macroinvertebrate abundance was the RV. Individual replicates were used in models as opposed to averages when applicable.

For EV selection, a group of potential EVs and interactions among EVs was compiled based on an a priori determination of those that are likely to be ecologically significant in modelling the RV. Next, a Pearson's correlation analysis was used to identify correlated EVs. When two or more variables (PCA scores for water chemistry and DOM quality) were more than 70% correlated, one of the highly correlated EVs was chosen as the representative variable based on the strength of correlation with other variables within the group (Montgomery and Peck 1992; Allison 2001). The ecologically-relevant EVs filtered by those chosen as representative from the correlation analysis were then used to build a mixed effects model. Variance inflation factors (VIFs) were further screened for co-linearity among EVs included in the model; EVs with VIFs greater than 5 indicated strong multi-collinearity, and one of the co-linear variables was used as the representative variable for that group, and the others were removed from the model (O'Brien 2007). The model containing all ecologically-relevant but not correlated or co-linear EVs was the "full" or "global" model used for model selection. Variables that were identified as correlated or co-linear by Pearson's correlation analysis or VIFs and accordingly not included in the full model are reported in the text as they may be alternative predictors of RVs.

Model selection was conducted using an automated model selection function in R. Using the full and null models, the *dredge* function (MuMIn package; Bartoń 2018) identified combinations of EVs that maximized goodness of fit and model complexity, with a lower Akaike Information Criteria adjusted for small sample sizes (AIC_c) value indicating a better model fit for the data (Zuur et al. 2009). The *model.avg* function was used to find the average model from all models identified by *dredge* that had ΔAIC_c of 7 or less. Models with a difference in AIC_c of less than 7 are considered plausible and containing variables with relevant information (Anderson 2008). The average model for each RV was determined to be the best-fitting model.

Model validation was carried out by observing the distribution of model residuals to ensure normality, observing a plot of residuals versus model-fitted values to ensure homogeneity, and applying the Cook's Distance test to identify outliers (points with Cook's Distance > 1 were deemed influential and removed from data set; Fox 2002). The average best-fitting model (meeting the above assumptions) for each RV is reported in the text. For each EV in the average model, the estimate for the slope coefficient, the 95% confidence interval, and the relative variable importance are reported. EVs with confidence intervals that do not overlap zero were considered to be statistically significant predictors of the RV. The conditional R^2 (R^2 associated with both fixed and random effects) for the average model is also reported.

Table 2: Summary of indicators and their corresponding response variables (RVs) that were assessed using the statistical analyses described above. RVs in bold are shown in the "Results" section; all others are included in the Appendix.

	Indicator	Response Variable
Abiotic	Water Chemistry	• **Conductivity (µmho/cm)** • **TN (ppm)**
	DOM Quality	• **SUVA** • FI • β:α
	Sediment Deposition	• **Fine, inorganic deposition (day^{-1})** • Coarse, inorganic deposition (day^{-1}) • **Fine % organic content** • Coarse % organic content
Biotic	Leaf Decomposition	• **% Leaf mass lost (degree day^{-1})** • Leaf breakdown rate
	Macroinvertebrate Communities	• **Abundance** • **Taxonomic Richness** • % EPT • **% Shredders** • **% Chironomidae** • Margalef's Richness • Shannon's Diversity Index
	Aquatic vs. Terrestrial Contribution to Diet	• **% Algae**
Contaminant	Mercury Concentration in Water & Biota	• **MeHg in water, Hydropsychids, seston** • **Hg(II) in Hydropsychids**, seston • % MeHg in Hydropsychids, seston • THg in Sediment • **BAF and BMF in Hydropsychids**

Table 3: Summary of environmental and sampling characteristics considered for inclusion as explanatory variables (EVs) in linear mixed effects models, prior to the EV selection process described above. Select response variables (RVs) act as EVs for other indicators.

Environmental / Sampling Characteristics	Explanatory Variables
Catchment Characteristics	• Catchment size (ha) • Effective Variable Source Area (% EVSA, m^2) • Extent of deciduous vegetation (% Dec)
Reach Characteristics	• Site (Continuous): Distance from site to farthest downstream site within catchment (multiplied by -1) • Stream flow (velocity, m/s) • Water Chemistry: WC_PC1, WC_PC2 • DOM: DOM_PC1, DOM_PC2 • Sediment Deposition: Fine, inorganic deposition per day, % Organic content of fine sediment • Leaf Litter Decomposition: % Leaf mass lost per degree day • Hydrogen Stable Isotopes: % Algae in Hydropsychid diets • Mercury: MeHg in filtered water, MeHg and Hg(II) in seston, BAF
Harvesting Characteristics	• Treatment: % Area harvested within last 5 years • Logging road density (m/ha)
Sampling Characteristics	• Year of sampling (ordered 2016, 2017) • Month of sampling (ordered August, September, October)

3. Results

3.1. Site Characteristics

3.1.1. Stream Temperature

In 2016, summer maximum temperature (SMT) and fall mean temperature (FMT) across all sites ranged from 15.5 to 24.1°C and 11.6 to 12.5°C, respectively (Figure 3). In 2017, SMT and FMT were generally lower than in 2016 (inter-annual differences within sites ranged from 0.6 to 5.1°C and 0.5 to 1.9°C, respectively) at 16.4 to 22.5°C and 10.3 to 13.7°C across all sites, respectively.

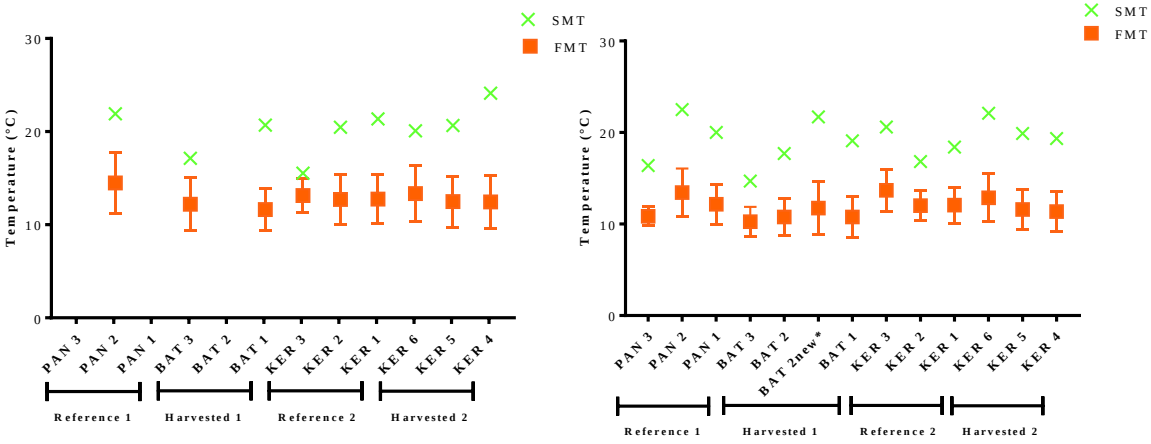

Figure 3. Fall mean (± SD) temperature (FMT; orange squares, average of bi-hourly recordings from September 1st until temperature logger removal in late October) and summer maximum temperature (SMT; green Xs, maximum of bi-hourly recordings 12.5 from July 1st until August 31st) for all study sites in A) 2016 and B) 2017. Note that no temperature data was recorded for sites PAN 3 and PAN 1 in 2016 as temperature loggers were lost/dead upon retrieval. See Appendix Table 1 to Appendix Table 4 for exact logger deployment and removal dates and monthly temperature data.

3.1.2. Stream Size & Velocity

Measurements for stream size and velocity were taken in August, September, and October 2017, but not in 2016.

3.1.2.1. Stream Width & Depth

In 2017, stream width and mean depth across all sites and sampling months ranged from 0.4 to 8.0 m and 0.08 to 0.42 m, respectively (Table 4). There was substantial variation in stream width and mean depth across months and, for this reason, monthly measurements were not averaged. Stream width was greatest in October at all sites than in other months (except PAN 1 and 3, higher in August). Stream mean depth was also consistently greatest in October at all sites (except PAN 1, BAT 2new*, and KER 4, higher in August). A strong spatial gradient in stream width (i.e., increasing from upstream to downstream) was observed in the PAN_{REF} and BAT_{HARV} catchments, but less so in the KER A_{REF} and KER B_{HARV} catchments. The spatial gradient in mean stream depth was not consistent among sites.

Table 4: Stream width and mean depth in August, September, and October 2017. No width or depth measurements were taken for site PAN 2 in August due to adverse weather conditions.

Catchment Name	Site Name	Aug Width (m)	Sept Width (m)	Oct Width (m)	Aug Depth (m)	Sept Depth (m)	Oct Depth (m)
PAN_{REF}	PAN 3	1.5	0.4	1.6	0.17	0.09	0.16
	PAN 2	No data	1.7	2.4	No data	0.16	0.19
	PAN 1	8.0	5.4	4.5	0.22	0.15	0.20
BAT_{HARV}	BAT 3	0.9	0.5	1.3	0.08	0.13	0.15
	BAT 2	3.4	3.0	4	0.32	0.31	0.37
	BAT 2new*	1.7	1.6	1.5	0.33	0.31	0.42
	BAT 1	8.0	7.5	11	0.20	0.18	0.22
KER A_{REF}	KER 3	1.4	1.1	2.3	0.09	0.10	0.21
	KER 2	3.4	3.5	6.5	0.20	0.22	0.26
	KER 1	2.3	3.6	4.9	0.08	0.10	0.24
KER B_{HARV}	KER 6	2.0	1.9	2.8	0.11	0.12	0.19
	KER 5	2.0	1.4	2.7	0.27	0.28	0.34
	KER 4	3.1	2	2.6	0.16	0.14	0.21

3.1.2.2. Stream Velocity

In 2017, monthly mean and maximum stream velocity across all sites ranged from 0.11 to 0.45 m/s and from 0.15 to 0.72 m/s, respectively (Figure 4). Mean velocity tended to increase downstream for all catchments except KER B_{HARV}, which is one of

the catchments that also did not display a strong spatial gradient in stream width or mean depth (Table 4). Similar trends for all sites were also noted for mean and maximum stream discharge (Appendix Figure 5).

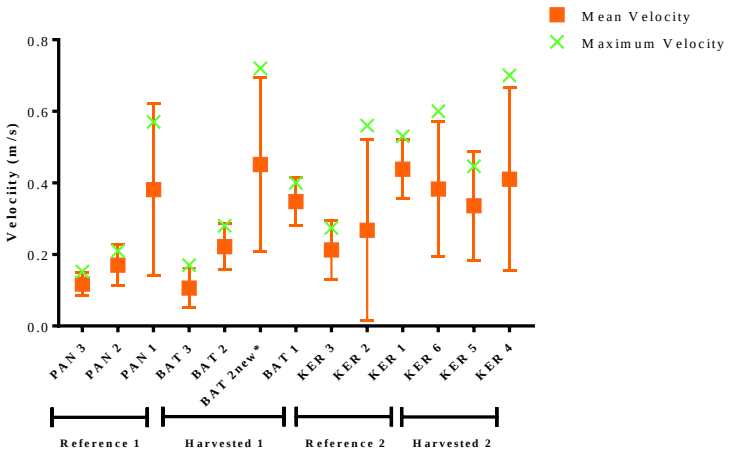

Figure 4: Mean (± SD; orange, mean of measurements in August, September, and October) and maximum (green, maximum of measurements in August, September and October) stream velocity (m/s) for all study sites in 2017. No velocity measurements were taken for site PAN 2 in August due to adverse weather conditions.

3.2. Indicators of Spatially Cumulative Impacts

3.2.1. Water Chemistry

3.2.1.1. Multivariate Water Chemistry Analyses

Water chemistry data are shown in Appendix Table 8, and generally indicate similar concentrations of NH_4, SRP, TP, Zn, Cd, Cu, Ni, and Pb among sites and sampling months in both 2016 and 2017. A PCA of all 25 water chemistry parameters (values averaged for 2016 and 2017 samples) produced two PC axes; the first explained 47.6% of variability within the data, and the second explained 19.3% of variability within the data (Figure 5). The first PC axis (WC_PC1) was strongly and negatively correlated with Mg, conductivity, Fe, alkalinity, Al, and TIC (r<-0.85) and negatively correlated with pH, K, Ca, Na, TOC, TP, and Mn (r<-0.60). The second PC axis (WC_PC2) was positively correlated with TN, NO_2+NO_3, SiO_2, and K (r>0.60). The

middle-reach and downstream sites for the KER B_{HARV} catchment (sites KER 5 and 4) were positioned on the far left of WC_PC1, and these sites had among the highest values for conductivity, Mg, Fe, Na, and TP. The upstream site for the BAT_{HARV} catchment (BAT 3) and the middle-reach site for KER A_{REF} (KER 2) exhibited large variation in the PCA and were positioned on opposite ends of WC_PC2; BAT 3 had the highest values for TN, NO_2+NO_3, SiO_2, and K, whereas KER 2 had relatively low values for those parameters. Three out of four sites in BAT_{HARV} (sites BAT 2, BAT 2new*, and BAT 1) were clustered closely together in the PCA, suggesting similar water quality at these sites, but no other obvious clustering was noted among sites on the basis of treatment or site.

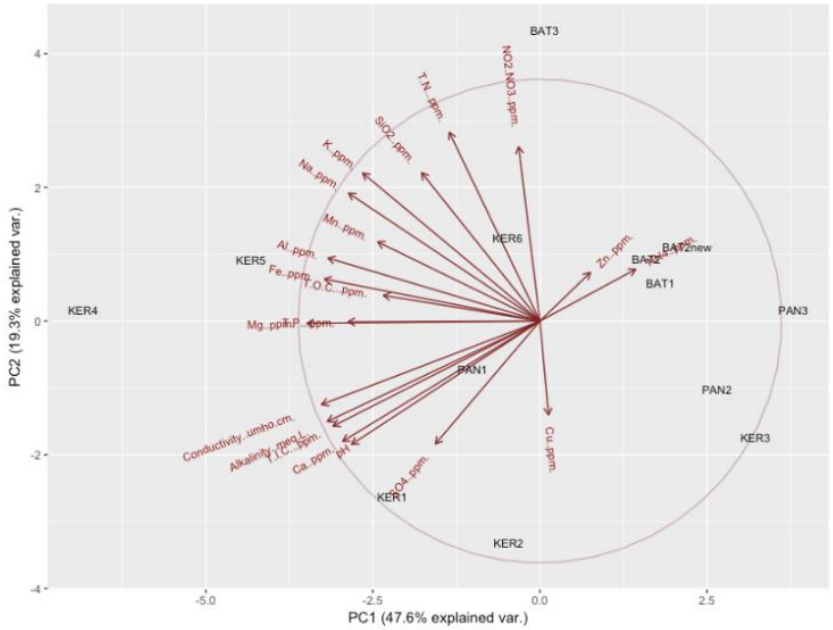

Figure 5: Principal Component Analysis (PCA) for water chemistry parameters (red arrows) for all sites (black text), with values averaged for all collections in 2016 (N=2 for all sites except BAT 2new*, where N=0) and 2017 (N=3).

The two water quality variables chosen for further analyses (see below) were conductivity (i.e., to represent variation in WC_PC1) and TN (i.e., to represent variation in WC_PC2). Conductivity was chosen because it was strongly correlated with

WC_PC1, pH, alkalinity, Ca, Mg, and TIC (r>0.85), and is a holistic measure of cations in the environment. TN had the strongest correlation with WC_PC2 (r=0.78) and was also correlated with K, Na, SiO_2, and NO_2+NO_3 (r>0.6). See Appendix Figure 2 and Appendix Table 9 for correlation coefficients for all water chemistry parameters among themselves and with first two PC axes.

3.2.1.2. Conductivity

Average conductivity in 2016 and 2017 ranged from 17.5 to 35.9 µmho/cm across all sites (Appendix Table 8). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}), there was an interaction between treatment and site (two-factor ANOVA, p<0.0001) (Figure 6). Within sites, there was an effect of treatment; average conductivity was higher in BAT_{HARV} than PAN_{REF} at the upstream site and lower in BAT_{HARV} than PAN_{REF} at the downstream site (multiple t-tests, p=0.0031, 0.0031, respectively). Within treatments, there was an effect of site in PAN_{REF} only (one-factor ANOVA, p<0.0001); average conductivity was higher at the middle-reach site than the upstream site (Tukey's Multiple Comparisons, p=0.0064), and higher at the downstream site than either the upstream or middle-reach sites (p=<0.0001, 0.0004, respectively). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}), there was no interaction between treatment and site (two-factor ANOVA, p=0.46) and no effect of treatment within sites (p=0.27). Within treatments, there was an effect of site (p<0.0001); average conductivity at the middle-reach and downstream sites was higher than at the upstream site for KER A_{REF} (Tukey's Multiple Comparisons, p<0.0001) and for KER B_{HARV} (p=0.0001, 0.0001, respectively).

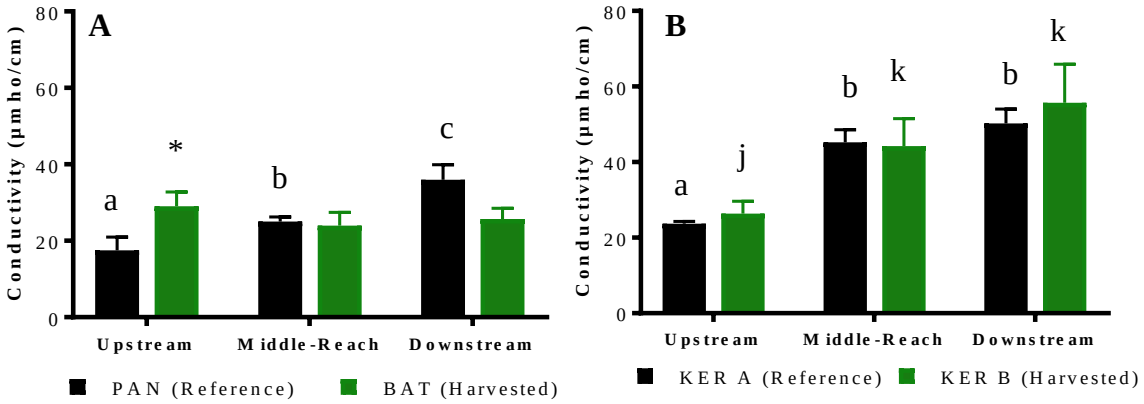

Figure 6: Average (± SD) conductivity (µmho/cm) in 2016 and 2017 (N=5 except for BAT middle-reach, where N=3) at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT, and B) the reference catchment KER A compared to the harvested catchment KER B. The asterisk (*) indicates significant difference among average conductivity between the two paired catchments at that site. The letters indicate significant difference among average conductivity between site locations within a treatment.

The average best-fitting model for conductivity indicated positive effects of treatment (% area harvested in the last five years), site (distance to farthest downstream site multiplied by -1), and road density, and negative effects of % EVSA and flow velocity (Table 5). The effects of site and treatment were significant, and these variables also had the greatest explanatory power (greatest relative variable importance and slope coefficients). The conditional R^2 was 0.94, indicating a high proportion of total variance was explained by the selected EVs.

Table 5: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for conductivity (µmho/cm), as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables				
		Site	**Treatment**	% EVSA	Road	Flow
Conductivity (µmho/cm)	Slope Coefficient	**0.25**	**0.19**	-0.04	0.07	-0.03
	95% Confidence Interval	**0.11 to 0.39**	**0.02 to 0.36**	-0.24 to 0.17	-0.08 to 0.02	-0.07 to 0.01
	Relative Variable Importance	**0.74**	**0.09**	0.05	0.04	0.03

[1]Site = distance (km) to farthest downstream site within catchment multiplied by -1; Treatment = % sub-catchment area harvested within the last 5 years; % EVSA = % effective variable source area compared to sub-catchment area; Road = road density (m per ha sub-catchment); Flow = stream flow velocity (m/s).
[2]Site highly correlated (r=0.86) with catchment size (not included in full model); % Dec (% deciduous tree species) and % EVSA had high VIFs (>4), so % Dec removed from full model.

Overall, there was some evidence of spatially cumulative trends for conductivity in harvested catchments as indicated by an interaction between treatment and site for only one of two (50%) paired comparisons. There was some evidence (50%) of an effect of treatment within sites and strong evidence (100%) for an effect of site within treatments in both paired catchment comparisons, which was also noted in linear models (positive).

3.2.1.3. Total Nitrogen (TN)

Average TN in 2016 and 2017 ranged from 0.22 to 0.61 ppm across all sites (Appendix Table 8). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}), there was an interaction between treatment and site (two-factor ANOVA, p=0.011) (Figure 7). Within sites, there was an effect of treatment; average TN was higher in BAT_{HARV} than PAN_{REF} at the upstream and middle-reach sites (multiple t-tests; p=0.010, 0.016, respectively). Within treatments, there was an effect of site in

BAT_{HARV} only (one-factor ANOVA, p=0.0005); average TN was higher at the upstream site than at the middle-reach and downstream sites (Tukey's Multiple Comparisons, p=0.0005, 0.0058, respectively). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}), there was no interaction between treatment and site (two-factor ANOVA, p=0.96), no effect of treatment (p=0.06), and no effect of site (p=0.99).

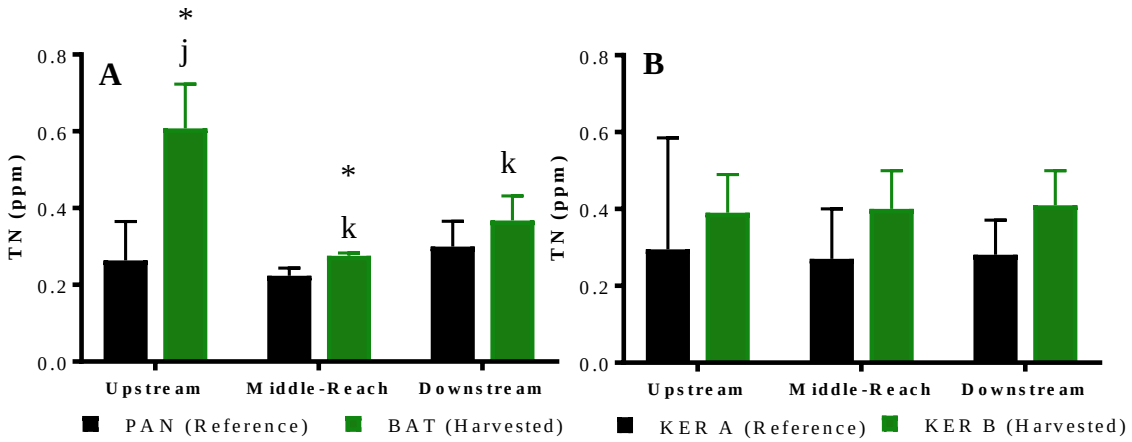

Figure 7: Mean (± SD) total nitrogen (TN, ppm) averaged for 2016 and 2017 measurements (N=5 except for BAT middle-reach, where N=3) in at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT, and B) the reference catchment KER A compared to the harvested catchment KER B. The asterisk (*) indicates a significant difference among average TN (log-transformed) between the two paired catchments at a site. The letters indicate significant difference among average TN between site locations within treatments.

The average best-fitting model for TN indicated positive effects of treatment, road density, % EVSA, and fine sediment deposition, and a negative effect of flow velocity (Table 6). The effect of treatment was significant, and this variable also had the greatest explanatory power (greatest relative variable importance and slope coefficients). The conditional R^2 was 0.53, indicating a moderate proportion of total variance was explained by the selected EVs.

Table 6: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for total nitrogen (TN, ppm), as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables				
		Treatment	Road	Flow	% EVSA	Dep
TN (ppm)	Slope Coefficient	**0.23**	0.05	-0.05	0.03	0.03
	95% Confidence Interval	**0.12 to 0.34**	-0.05 to 0.14	-0.12 to 0.03	-0.05 to 0.11	-0.06 to 0.12
	Relative Variable Importance	**0.82**	0.04	0.03	0.03	0.03

[1]Treatment = % sub-catchment area harvested within the last 5 years; Road = road density (m per ha sub-catchment); Flow = stream flow velocity (m/s); % EVSA = % effective variable source area compared to sub-catchment area; Dep = Fine, inorganic sediment deposition (g/day).
[2]Site highly correlated (r=0.86) with catchment size (not included in full model).

Overall, there was some evidence of spatially cumulative trends for TN in harvested catchments as indicated by an interaction between treatment and site for one of two (50%) paired comparisons. There was some evidence (50%) for an effect of treatment in paired comparisons, and it was a positive predictor in linear models. There was also some evidence (50%) for an effect of site in paired comparisons, but this was not observed in linear models.

3.2.2. DOM Quality

3.2.2.1. Multivariate DOM Quality Analyses

DOM quality data are shown in Appendix Table 10, and generally indicate similar values for HIX and E2E3 among sites and sampling periods, but more variable concentrations for other DOM measures. A PCA of all seven DOM optical properties (values averaged for 2016 and 2017 samples) produced two PC axes; the first explained 74.6% of variance within the data, and the second explained 15.7% of variance within the data (Figure 8). The first PC axis (DOM_PC1) was strongly and positively

correlated with HIX, mHIX, SUVA, E2E3, and SAC_{340} (r>0.85), and negatively correlated with FI (r=-0.82). The second PC axis (DOM_PC2) was strongly and negatively correlated with β:α (r=-0.97). Sites in the KER B_{HARV} catchment grouped relatively close together on the far right of DOM_PC1, and had among the highest values for HIX, SUVA, E2E3, and SAC_{340}. The upstream site for the KER A_{REF} catchment (KER 3) has positioned on the bottom end of DOM_PC2, but did not exhibit higher or lower β:α when compared to other sites. No other obvious clustering of sites was noted on the basis of treatment or site position.

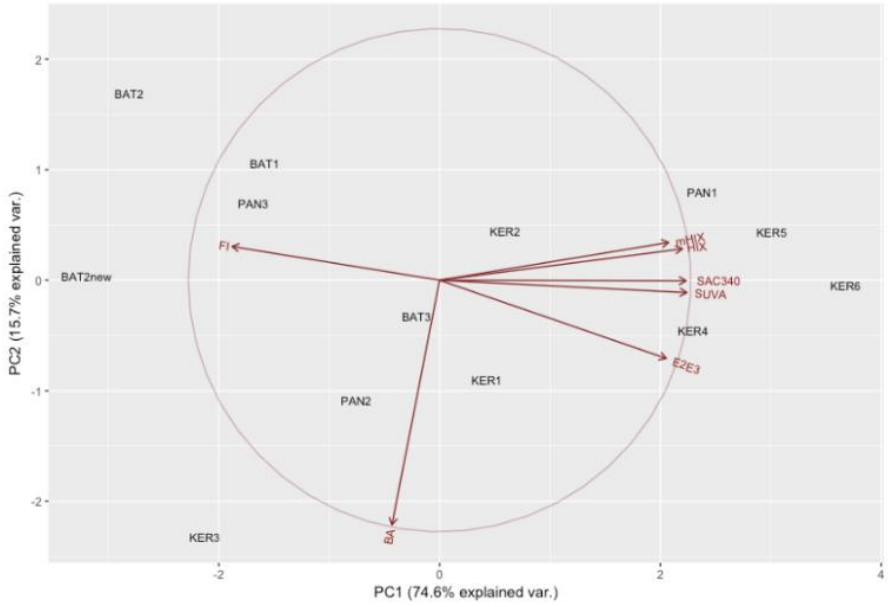

Figure 8: Principal Component Analysis (PCA) for DOM optical properties (red arrows) for all sites (black text), with values averaged for all collections in 2016 (N=1 for all sites except for BAT 2new*, where N=0) and 2017 (N=3).

The three DOM quality variables chosen for further analyses (see below) were SUVA, FI, and β:α. SUVA was chosen because it was strongly and positively correlated with DOM_PC2, HIX, E2E3, and SAC_{340} (r>0.90). Similarly, FI was negatively correlated with DOM_PC1, HIX, SUVA, E2E3, and SAC_{340} (r>0.60). β:α was chosen because it was strongly and negatively correlated with DOM_PC2 (r=-0.97). While β:α

was weakly and negatively correlated with all parameters besides E2E3 (r>-0.30), it is included to represent variance in a second dimension (see Appendix Figure 3 and Appendix Table 11 for correlation coefficients of all variables among themselves and with PC axes).

3.2.2.2. Specific UV Absorbance at 254 nm (SUVA), Fluorescence Index (FI), & Freshness Index (β:α)

Average SUVA in 2016 and 2017 ranged from 13.8 to 34.6 across all sites (Appendix Table 10). Within both paired-catchment comparisons (PAN_{REF} vs. BAT_{HARV} and KER A_{REF} vs. KER B_{HARV}), there was no interaction between treatment and site (two-factor ANOVA, p=0.10 and p=0.72, respectively) and no effect of treatment (p=0.08, Sidak's Multiple Comparisons p>0.05, respectively) or site (p=0.17 and p=0.99, respectively) (Figure 9). Similar results were noted for FI and β:α (see Appendix Table 12).

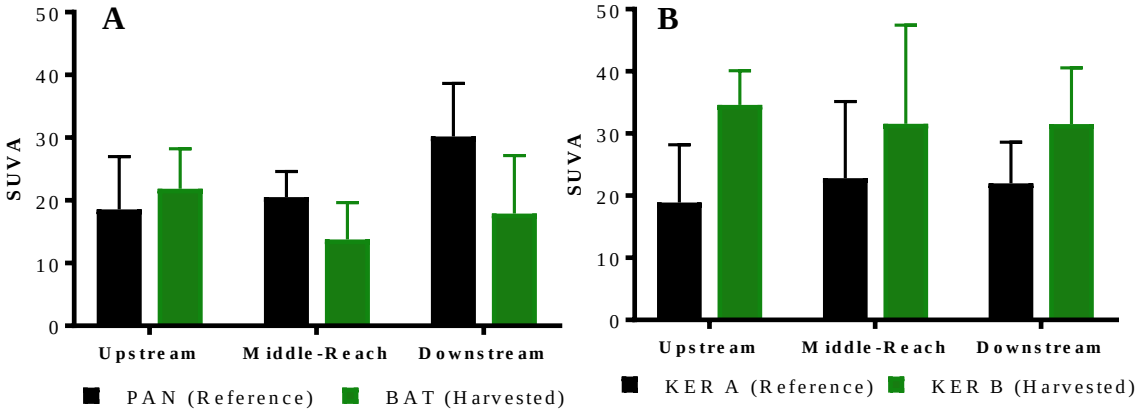

Figure 9: Mean (± SD) specific UV absorbance at 254 nm (SUVA), averaged for 2016 and 2017 measurements (N=5 for all sites except for BAT middle-reach, where N=3) at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT, and B) the reference catchment KER A compared to the harvested catchment KER B.

The average best-fitting model for SUVA indicated positive effects of road density, site, fine sediment deposition, treatment, and flow velocity, and negative effects

of year, % EVSA, month, and interaction between treatment and site (Table 7). The effects of year, road density, and the interaction between treatment and site were significant, and these variables also had the greatest explanatory power (greatest relative variable importance for year and road density and greatest slope coefficients for all). The conditional R^2 was 0.65, indicating a moderate proportion of total variance was explained by the selected EVs. See Appendix Table 13 for average model for β:α (null model was average best-fitting model for FI).

Table 7: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for specific UV absorbance at 254 nm (SUVA), as determined via AIC_c model selection ($\Delta AIC_c <$ 7). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables								
		Year	**Road**	Site	Dep	Treatment	Flow	% EVSA	Month	**Treat:Site**
SUVA	Slope Coefficient	**-0.20**	**0.13**	0.08	0.08	0.02	0.03	-0.03	-0.02	**-0.14**
	95% Confidence Interval	**-0.28 to -0.12**	**0.03 to 0.22**	-0.004 to 0.16	-0.04 to 0.21	-0.16 to 0.20	-0.06 to 0.13	-0.12 to 0.06	-0.11 to 0.06	**-0.23 to -0.05**
	Relative Variable Importance	**0.91**	**0.39**	0.13	0.06	0.06	0.02	0.01	0.01	**0.01**

[1]Year = year of sampling; Road = road density (m per ha sub-catchment); Site = distance (km) to farthest downstream site within catchment multiplied by -1; Dep = fine, inorganic sediment deposition (g/day); Treatment = % sub-catchment area harvested within the last 5 years; Flow = stream flow velocity (m/s); % EVSA = % effective variable source area compared to sub-catchment area; Month = month of sampling (1=August, 2=September, 3=October); Treat:Site = interaction between treatment and site.
[2]Site highly correlated (r=0.86) with catchment size (not included in full model); Treatment highly correlated (r=0.78) with WC_PC2 (not included in full model); Dep highly correlated (r=-0.70) with WC_PC1 (not included in full model).

Overall, there was no evidence of spatially cumulative trends for SUVA in harvested catchments as indicated by the absence of an interaction between treatment and site in both paired comparisons. There was also a lack of evidence (0%) for an effect of treatment or site in both paired comparisons and in linear models.

3.2.3. Sediment Deposition

3.2.3.1. Inorganic Sediment Deposition

Average fine, inorganic sediment deposition in 2016 ranged from 0.02 to 0.28 g/day across all sites (Appendix Table 14). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2016, there was an interaction between treatment and site (two-factor ANOVA, p=0.036) (Figure 10). Within sites, there was an effect of treatment; average deposition was higher in PAN_{REF} than BAT_{HARV} at all sites (multiple t-tests, p=0.079, 0.0070, 0.019, respectively). Within treatments, there was an effect of site in PAN_{REF} only (one-factor ANOVA, p=0.037); average deposition was higher at the downstream site than the middle-reach site (Tukey's Multiple Comparisons, p=0.050). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for 2016, sediment deposition overall was approximately 2 to 4 times higher than in PAN_{REF} or $BAT_{HARV,}$ there was no interaction between treatment and site (two-factor ANOVA, p=0.34) and no effect of treatment (Sidak's Multiple Comparisons, p>0.05). Within treatments, there was an effect of site (two-factor ANOVA, p<0.0001): average deposition was higher downstream than middle-reach in KER A_{REF} (Tukey's Multiple Comparisons, p=0.0059), and higher upstream and downstream than middle-reach in KER B_{HARV} (p=0.0025, 0.0002, respectively). Average coarse, inorganic sediment deposition per day in 2016 followed similar treatment and spatial trends (data are shown in Appendix Table 16).

Average fine, inorganic sediment deposition in 2017 ranged from 0.004 to 0.180 g/day across all sites (Appendix Table 15). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2017, there was no interaction between

treatment and site (two-factor ANOVA, p=0.78) and no effect of treatment within sites (p=0.11) (Figure 10). Within treatments, however, there was an effect of site (p<0.0001): average deposition was higher downstream than upstream and at the middle-reach site in both PAN_{REF} (Tukey's Multiple Comparisons, p=0.0005, <0.0001, respectively) and BAT_{HARV} (p=0.0068, 0.0007, respectively). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for 2017, there was no interaction between treatment and site (two-factor ANOVA, p=0.49). Within sites, there was an effect of treatment (p=0.003): average deposition was higher in KER B_{HARV} than KER A_{REF} at the downstream site only (Sidak's Multiple Comparisons, p=0.024). Within treatments, there was an effect of site (two-factor ANOVA, p=0.007): average deposition was higher downstream than upstream for KER B_{HARV} (Tukey's Multiple Comparisons, p=0.012). Average coarse, inorganic sediment deposition per day in 2017 followed similar treatment and spatial trends (data are shown in Appendix Table 16).

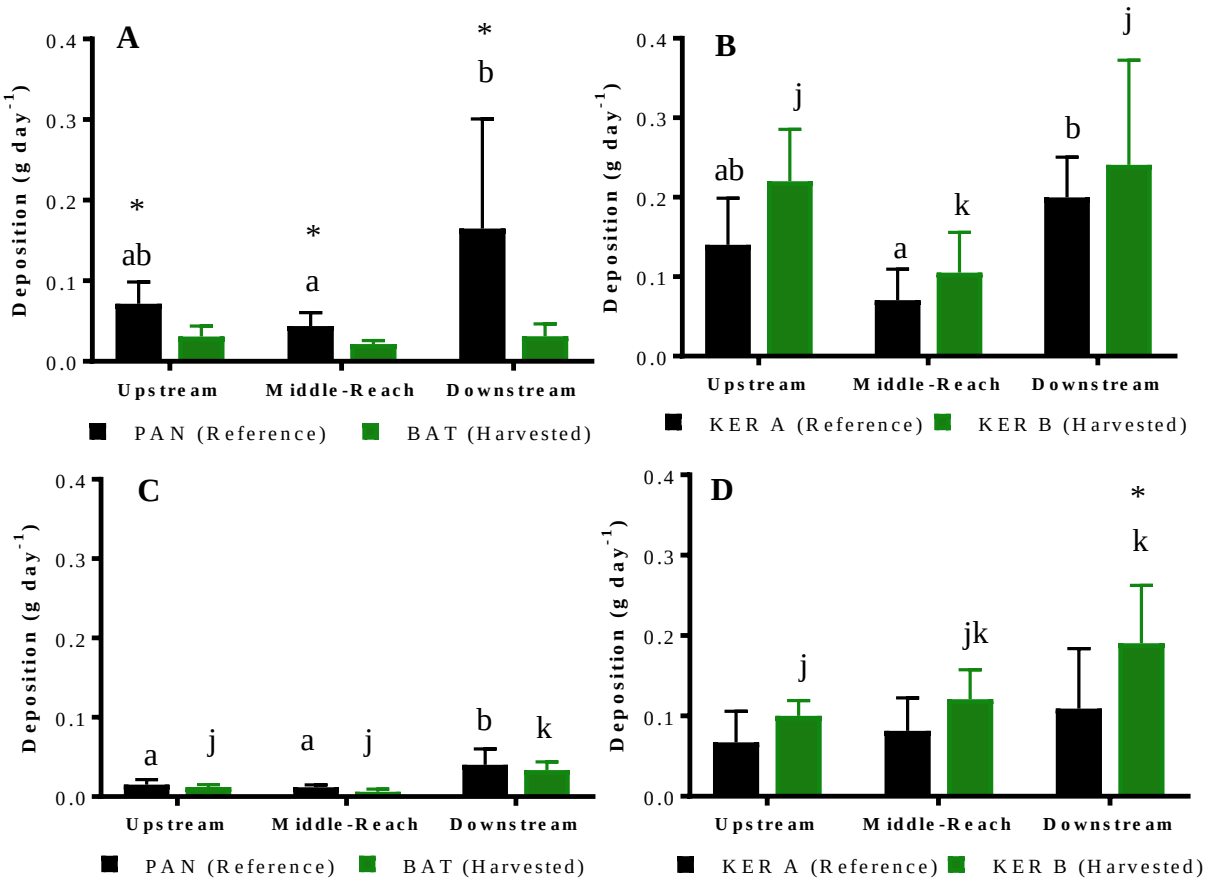

Figure 10: Mean (± SD, N=3-7) fine (1.5-250 µm), inorganic sediment deposition (g/day) at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT in 2016, B) the reference catchment KER A compared to the harvested catchment KER B in 2016, C) the reference catchment PAN compared to the harvested catchment BAT in 2017, and D) the reference catchment KER A compared to the harvested catchment KER B in 2017. The asterisk (*) indicates a significant difference in average deposition between the two paired catchments within a site. The letters indicate significant difference among average deposition between site locations within treatments.

The average best-fitting model for fine, inorganic sediment deposition indicated positive effects of % EVSA, site, and road density, and negative effects of year and treatment (Table 8). The effects of year, site, and % EVSA were statistically significant, and these variables also had the greatest explanatory power (greatest relative variable importance slope coefficients). The conditional R^2 was 0.74, indicating a moderate

proportion of total variance was explained by the selected EVs. See Appendix Table 17 for the average model for coarse, inorganic sediment deposition.

Table 8: Slope coefficient estimates, 95% confidence interval, and relative variable importance for explanatory variables (EVs) included in the average model for fine, inorganic sediment deposition per day determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables				
		Year	**Site**	**% EVSA**	Treatment	Road
Fine, Inorganic Sediment Deposition (g/day)	Slope Coefficient	**-0.41**	**0.47**	**0.32**	-0.09	0.05
	95% Confidence Interval	**-0.52 to -0.30**	**0.29 to 0.66**	**0.15 to 0.49**	-0.34 to 0.17	-0.11 to 0.21
	Relative Variable Importance	**0.92**	**0.89**	**0.81**	0.09	0.05

[1]Year = year of sampling; Site = distance (km) to farthest downstream site within catchment multiplied by -1; % EVSA = % effective variable source area compared to sub-catchment area; Treatment = % sub-catchment area harvested within the last 5 years; Road = road density (m per ha sub-catchment).
[2]Site highly correlated (r=0.89) with catchment size (not included in full model); Site highly correlated (r=0.79) with Flow (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4), so % Dec removed from full model.

Overall, there was limited evidence (25%) of spatially cumulative trends for fine, inorganic sediment deposition in harvested catchments in paired comparisons. There was limited evidence (25%) of an effect of treatment and some evidence (75%) of an effect of site in paired comparisons. Site was also a positive predictor in linear models.

3.2.3.2. Organic Content of Sediment

Average organic content of fine sediment ranged in 2016 ranged from 2.4 to 22.6% across all sites (Appendix Table 14). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2016, there was an interaction between treatment and site (two-factor ANOVA, p=0.011) (Figure 11). Within sites, there was an effect of treatment; average % organic content was higher in BAT_{HARV} at all sites (multiple t-tests, p=0.0074, <0.0001, 0.0074, respectively). Within both treatments,

there was an effect of site (one-factor ANOVA, p<0.0001, <0.0001, respectively); average % organic content was higher at the upstream and middle-reach sites than the downstream site in PAN_{REF} (Tukey's Multiple Comparisons, p=<0.0001, <0.0001, respectively), and higher at the middle-reach site compared to the upstream and downstream sites and at the upstream site compared to the downstream site for BAT_{HARV} (p=0.0005, <0.0001, <0.0001, respectively). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for 2016, there was an interaction between treatment and site (two-factor ANOVA, p=0.0047). Within sites, there was an effect of treatment; average % organic was higher in KER A_{REF} at the upstream and middle-reach sites (multiple t-tests, p=0.0003, 0.0170, respectively). Within treatments, there was an effect of site for KER A_{REF} only (one-factor ANOVA, p=0.0006); average % organic was higher at the upstream and middle-reach sites than the downstream site (Tukey's Multiple Comparisons, p=0.0004, 0.0430, respectively). Average organic content of coarse sediment in 2016 followed similar treatment and spatial trends (data are shown in Appendix Table 16).

Average organic content of fine sediment in 2017 ranged from 3.4 to 37.0% across all sites (Appendix Table 15). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2017, there was no interaction between treatment and site (two-factor ANOVA, p=0.91) and no effect of treatment (p=0.13) (Figure 11). Within treatments, there was an effect of site (p<0.0001); average % organic was higher at the upstream and middle-reach sites than the downstream site in PAN_{REF} (Tukey's Multiple Comparisons, p=0.0028, 0.0002, respectively) and in BAT_{HARV} (p=0.0020, <0.0001, respectively). Within the second paired-catchment comparison (KER A_{REF} vs. KER

B_{HARV}) for 2017, there was no interaction between treatment and site (two-factor ANOVA, p=0.39). Within sites, there was an effect of treatment (p=0.0008); average % organic was higher for KER A_{REF} than KER B_{HARV} at the upstream site only (Sidak's Multiple Comparisons, p=0.0077). Within treatments, there was an effect of site in KER A_{REF} only (p=0.0013); average % organic was higher upstream than at the middle-reach and downstream sites (Tukey's Multiple Comparisons, p=0.0390, 0.0025, respectively). Average organic content of coarse sediment in 2017 followed similar treatment and spatial trends (data are shown in Appendix Table 16).

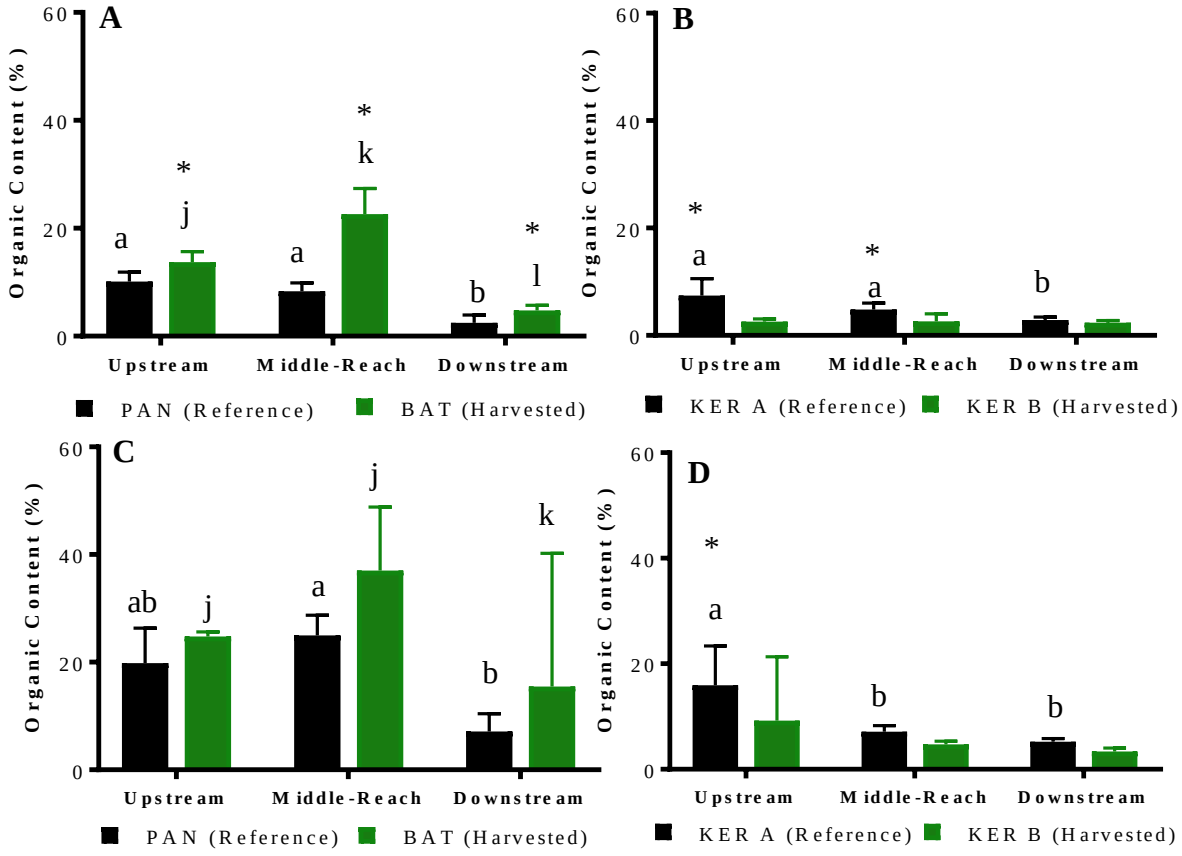

Figure 11: Mean (± SD, N=3-7) % organic content of fine (1.5-250 µm) sediment at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT in 2016, B) the reference catchment KER A compared to the harvested catchment KER B in 2016, C) the reference catchment PAN compared to the harvested catchment BAT in 2017, and D) the reference catchment KER A compared to the harvested catchment KER B in 2017. The asterisk (*) indicates

a significant difference among average % organic between the two paired catchments within a site. The letters indicate significant difference among average % organic between site locations within a treatment.

The average best-fitting model for organic content of fine sediment indicated positive effects of year and treatment, and negative effects of site, % EVSA, road density, and leaf mass lost (Table 9). The effects of site and year were statistically significant, and these variables also had the greatest explanatory power (greatest relative variable importance slope coefficients). The conditional R^2 was 0.79, indicating a moderate proportion of total variance was explained by the selected EVs. See Appendix Table 18 for the average model for % organic content of coarse sediment.

Table 9: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for % organic content of fine sediment, as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables					
		Site	**Year**	% EVSA	Treatment	Road	% Leaf Lost
% Organic of Fine Sediment	Slope Coefficient	**-0.58**	**0.39**	-0.16	0.03	-0.06	-0.02
	95% Confidence Interval	**-0.77 to -0.39**	**0.31 to 0.47**	-0.38 to 0.05	-0.29 to 0.35	-0.27 to 0.15	-0.15 to 0.11
	Relative Variable Importance	**0.94**	**0.94**	0.19	0.10	0.07	0.03

[1]Site = distance (km) to farthest downstream site within catchment multiplied by -1; Year = year of sampling; % EVSA = % effective variable source area compared to sub-catchment area; Treatment = % sub-catchment area harvested within the last 5 years; Road = road density (m per ha sub-catchment); % Leaf Lost = % leaf mass lost per degree day.
[2]Site highly correlated (r=0.89) with catchment size (not included in full model); Site highly correlated (r=0.79) with Flow (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4), so % Dec removed from full model.

Overall, there was some evidence (50%) of spatially cumulative trends for organic content of fine sediment in harvested catchments in paired comparisons. There was strong evidence (100%) of an effect of treatment eand site in paired comparisons. Site was also a negative predictor in linear models.

3.2.4. Leaf Litter Decomposition

3.2.4.1. Leaf Mass Lost & Breakdown Rate

Average leaf mass lost per degree day in 2016 ranged from 0.05 to 0.10% across all sites (Appendix Table 19). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2016, there was an interaction between treatment and site (two-factor ANOVA, 0.024) (Figure 12). There was no effect of treatment (multiple t-tests, $p>0.05$), but there was an effect of site in BAT_{HARV} only (one-factor ANOVA, $p=0.026$); average % leaf mass lost was higher at the upstream site than the middle-reach site (Tukey's Multiple Comparisons, $p=0.022$). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for 2016, there was no interaction between treatment and site (two-factor ANOVA, $p=0.23$), no effect of treatment ($p=0.33$), and no effect of site ($p=0.87$). Similar treatment and spatial trends were noted for leaf litter breakdown rate for 2016 (data are shown in Appendix Table 21).

Average leaf mass lost per degree day in 2017 ranged from 0.06 to 0.12% across all sites (Appendix Table 20). Within both paired-catchment comparisons (PAN_{REF} vs. BAT_{HARV} and KER A_{REF} vs. KER B_{HARV}) in 2017, there was no interaction between treatment and site (two-factor ANOVA, $p=0.50$ and $p=0.30$, respectively), no effect of treatment ($p=0.05$ and $p=0.09$, respectively), and no effect of site ($p=0.77$ and $p=0.19$, respectively) (Figure 12). Similar treatment and spatial trends were noted for leaf litter breakdown rate for 2017 (Appendix Table 21).

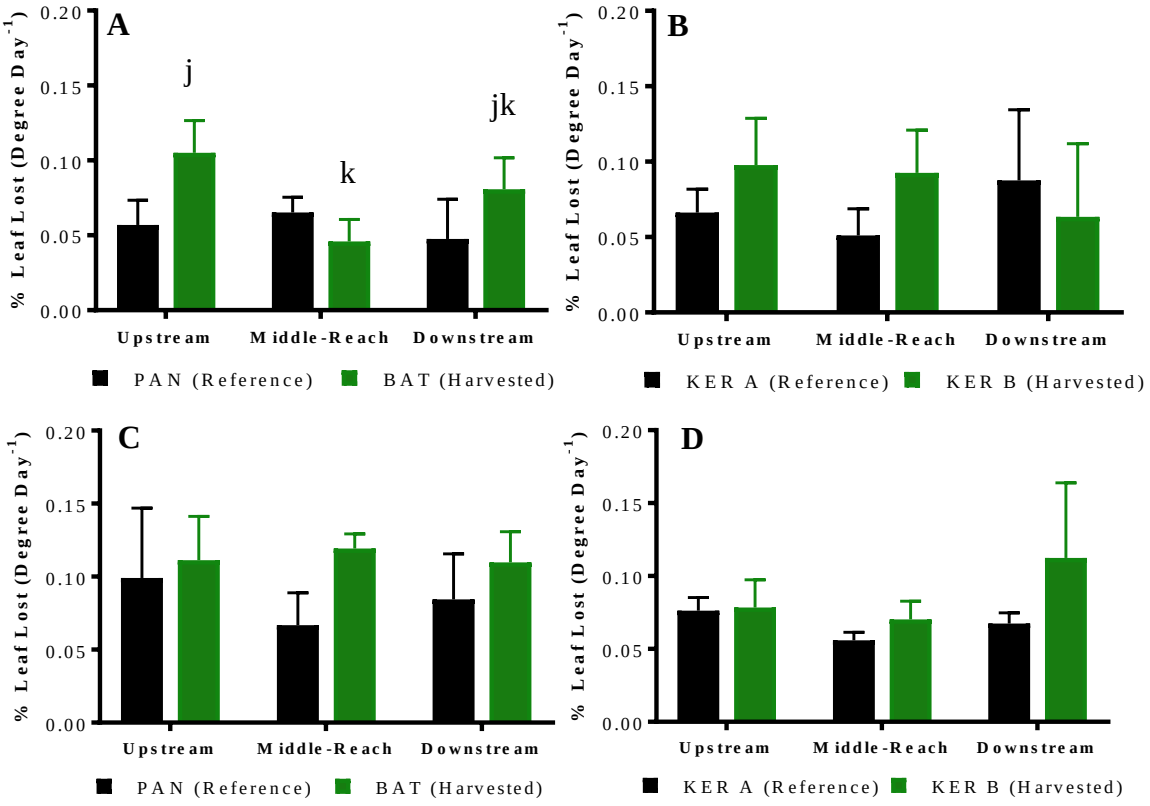

Figure 12: Mean (± SD, N=3) % leaf mass lost per degree day at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT in 2016, B) the reference catchment KER A compared to the harvested catchment KER B in 2016, C) the reference catchment PAN compared to the harvested catchment BAT in 2017, and D) the reference catchment KER A compared to the harvested catchment KER B in 2017. The letters indicate significant difference among average % leaf mass lost between site locations within a treatment.

The average best-fitting model for leaf mass lost indicated positive effects of year, treatment, % shredders, and % EVSA, and negative effects of DOM_PC2, fine sediment deposition, DOM_PC1, and site (Table 10). The effects of year, treatment, and % shredders were statistically significant, and these variables also had the greatest explanatory power (greatest relative variable importance and slope coefficients). The conditional R^2 was 0.37, indicating a limited proportion of total variance was explained by the selected EVs. See Appendix Table 22 for the average best-fitting model for leaf litter breakdown rate.

Table 10: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in average model for % leaf mass lost per degree day, as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables							
		Year	**Treatment**	**Shredders**	DOM_PC2	Dep	DOM_PC1	Site	% EVSA
% Leaf Mass Lost (Degree Day^{-1})	Slope Coefficient	**0.14**	**0.13**	**0.11**	-0.09	-0.04	-0.02	-0.02	0.002
	95% Confidence Interval	**0.04 to 0.24**	**0.03 to 0.22**	**0.002 to 0.22**	-0.20 to 0.01	-0.19 to 0.10	-0.15 to 0.11	-0.12 to 0.07	-0.10 to 0.10
	Relative Variable Importance	**0.60**	**0.13**	**0.11**	0.10	0.03	0.03	0.01	0.01

[1]Year = year of sampling; Treatment = % sub-catchment area harvested within the last 5 years; Shredders = % of invertebrates with shredding feeding strategy; DOM_PC2 = scores for DOM PCA axis 2; Dep = fine, inorganic sediment deposition per day; DOM_PC1 = scores for DOM PCA axis 1; Site = distance (km) to farthest downstream site within catchment multiplied by -1; % EVSA = % effective variable source area compared to sub-catchment area.
[2]Site highly correlated (r=0.86) with catchment size (not included in full model); % EVSA highly correlated (r=-0.70) with flow (not included in full model); Treatment highly correlated (r=0.78) with WC_PC2 (not included in full model); DOM_PC1 and DOM_PC2 highly correlated (r=-0.68, -0.75) with WC_PC1 (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4), so % Dec removed from full model; Road and Dep had high VIFs (>4), so Road removed from full model.

Overall, there was limited evidence (25%) of spatially cumulative impacts for % leaf mass lost per degree day in paired comparisons. There was no evidence (0%) of an effect of treatment and limited evidence (25%) of an effect of site in paired comparisons. Treatment was a positive predictor in linear models.

3.2.5. Leaf Litter Macroinvertebrates

See Appendix Table 23 and Appendix Table 24 for complete list and counts of invertebrates identified in leaf packs. Leaf litter macroinvertebrate communities could not be compared between 2016 and 2017 as different sieve sizes were used to separate them from leaf litter. Therefore, separate models were generated for each year. ANOVA and model results for % EPT, Margalef's Richness, and Shannon's Diversity Index metrics are shown in the Appendix Table 28 to Appendix Table 30.

3.2.5.1. Multivariate Invertebrate Community Analysis

In 2016, sites in KER A_{REF} (KER 3, 2, 1) and sites in KER B_{HARV} (KER 6, 5, 4) catchments clustered separately in ordination space, suggesting distinct leaf pack macroinvertebrate assemblages among treatments for these catchments (Figure 13). In contrast, sites in PAN_{REF} (PAN 3, 2, 1) and BAT_{HARV} (BAT 3, 2, 1) were more scattered in ordination space, suggesting that assemblages did not segregate by treatment for these catchments. There was also no obvious clustering of sites by spatial orientation (i.e., headwater, middle-reach, and downstream). Overall, the ANOSIM did not detect statistically significant differences among communities on the basis of treatment or site (R=0.04, -0.07, respectively; p=0.24, 0.74, respectively).

In 2017, the grouping of leaf pack macroinvertebrate communities by treatment was similar to 2016, including sites in KER A_{REF} and KER B_{HARV} which also clustered in ordination space (Figure 13). Again, there was no obvious grouping of sites by spatial orientation. Overall, there were no significant differences in communities at sites on the basis of treatments or location in 2017, as indicated by the ANOSIM analysis (R=0.02, -0.01, respectively; p=0.35, 0.50, respectively).

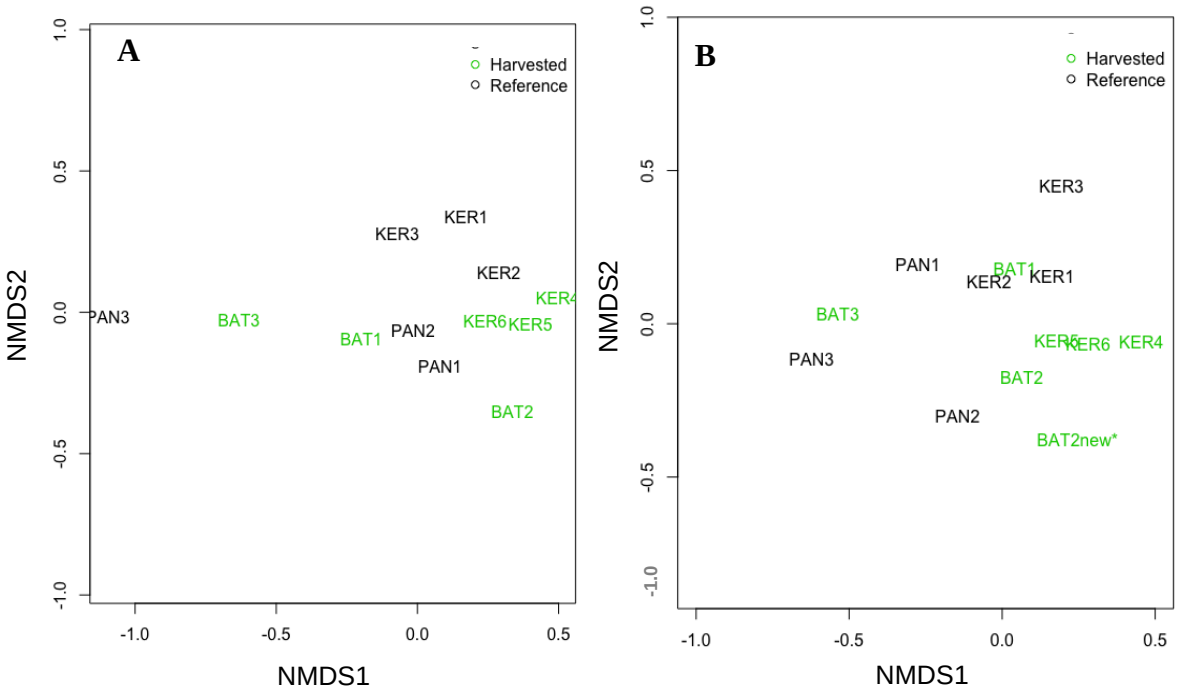

Figure 13: NMDS ordination plots of leaf-litter invertebrate community composition (based on average abundance per site) at A) 6 reference and 6 harvested sites in 2016. (Stress = 0.073) and at B) 6 reference and 7 harvested sites in 2017 (Stress = 0.12). Sites in harvested catchments are shown in green and sites in reference catchments are shown in black.

3.2.5.2. Abundance

Average abundance of leaf pack macroinvertebrates in 2016 ranged from 42 to 254 individuals across all sites (Appendix Table 23). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2016, there was no interaction between treatment and site (two-factor ANOVA, p=0.55) and no effect of treatment (Sidak's Multiple Comparisons, p>0.05) (Figure 14). Within treatments, there was an effect of site (p=0.0002): average abundance was higher at the middle-reach and downstream sites than the upstream site for both PAN_{REF} (Tukey Multiple Comparisons, p=0.015, 0.001, respectively) and BAT_{HARV} (p=0.033, 0.014, respectively). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for 2016, there was no

interaction between treatment and site (two-factor ANOVA, p=0.56), and no effect of treatment (p=0.84) or site (p=0.64) on average abundance.

Average abundance of leaf pack macroinvertebrates in 2017 ranged from 243 to 2039 individuals across all sites (Appendix Table 24). Within both paired-catchment comparisons (PAN_{REF} vs. BAT_{HARV} and KER A_{REF} vs. KER B_{HARV}) for 2017, there was no interaction between treatment and site (two-factor ANOVA, p=0.39 and p=0.98, respectively) and no effect of treatment (p=0.17 and p=0.40, respectively) or site (p=0.90 and p=0.28, respectively) on average invertebrate abundance (Figure 14).

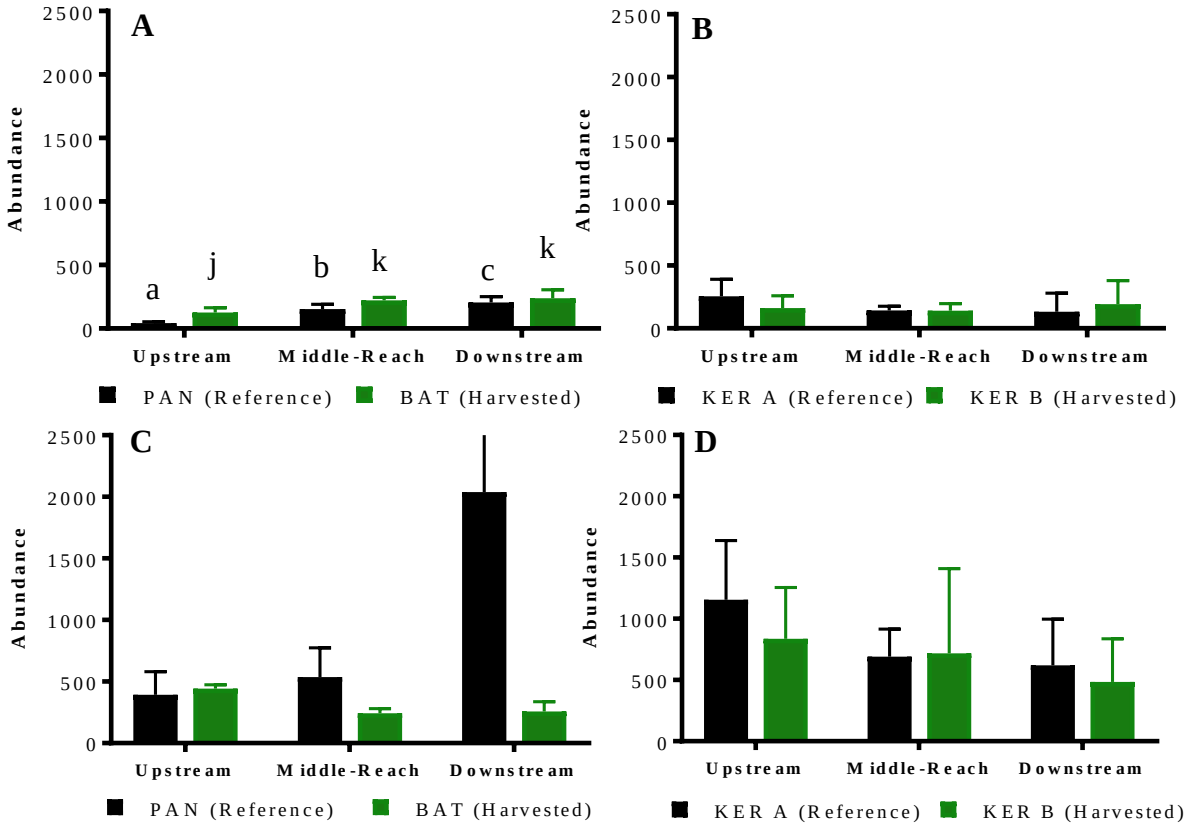

Figure 14: Average (± SD, N=3) leaf pack macroinvertebrate abundance (total number of individuals) at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT in 2016, B) the reference catchment KER A compared to the harvested catchment KER B in 2016, C) the reference catchment PAN compared to the harvested catchment BAT in 2017, and D) the reference catchment KER A compared to the harvested catchment KER B in 2017.

The letters indicate a significant difference in average invertebrate abundance between site locations within treatments.

The average best-fitting model for leaf pack macroinvertebrate abundance in 2016 indicated positive effects of site, road density, treatment, and DOM_PC2, and negative effects of % EVSA and fine sediment deposition (Table 11). The conditional R^2 was 0.51, indicating a moderate proportion of total variance was explained by the selected EVs. The average best-fitting model in 2017 indicated positive effects of % EVSA, DOM_PC1, and road density, and negative effects of treatment, DOM_PC2, and site. The conditional R^2 was 0.24, indicating a limited proportion of total variance was explained by the selected EVs. In both years, no effects were statistically significant or had great explanatory power.

Table 11: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for leaf pack macroinvertebrate abundance in 2016 and 2017, as determined via AIC_c model selection ($\Delta AIC_c < 7$).

Response Variable		**Explanatory Variables**						
		% EVSA	Site	Road	Treatment	Dep	DOM_PC2	DOM_PC1
Abundance 2016	Slope Coefficient	-0.17	0.14	0.12	0.08	-0.03	0.006	
	95% Confidence Interval	-0.46 to 0.11	-0.15 to 0.43	-0.17 to 0.41	-0.22 to 0.38	-0.34 to 0.27	-0.30 to 0.31	
	Relative Variable Importance	0.09	0.07	0.06	0.05	0.05	0.05	
Abundance 2017	Slope Coefficient	0.21	-0.06	0.07	-0.23		-0.20	0.19
	95% Confidence Interval	-0.05 to 0.48	-0.32 to 0.20	-0.24 to 0.39	-0.50 to 0.04		-0.47 to 0.07	-0.11 to 0.49
	Relative Variable Importance	0.16	0.03	0.05	0.17		0.14	0.13

[1]% EVSA = % effective variable source area compared to sub-catchment area; Site = distance (km) to farthest downstream site within catchment multiplied by -1; Road = road density (m per ha sub-catchment); Treatment = % sub-catchment area harvested within the last 5 years; Dep = fine, inorganic

sediment deposition per day; DOM_PC2 = scores for DOM PCA axis 2; DOM_PC1 = scores for DOM PCA axis 1.
[2]Site in 2016 and 2017 highly correlated (r=0.87, 0.85) with catchment size (not included in full models); Treatment in 2016 and 2017 highly correlated (r=0.77, 0.78) with WC_PC2 (not included in full models); DOM_PC2 in 2016 was highly correlated (r=0.75) with WC_PC1 (not included in full models); % EVSA in 2017 highly correlated (r=-0.70) with Flow (not included in full models); Dep and DOM_PC1 in 2017 highly correlated (r=0.87, 0.75) with WC_PC1 (not included in full models); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4) in 2016 and 2017, so % Dec removed from full models; % Organic and Site had high VIFs (>4) in 2016 and 2017, so % Organic removed from full models; Road and DOM_PC1 had high VIFs (>4) in 2016, so DOM_PC1 removed from full model; Road and Dep had high VIFs (>4) in 2017, so Road removed from full models.

Overall, there was no evidence (0%) of spatially cumulative trends for invertebrate abundance in paired comparisons. There was no evidence (0%) of an effect of treatment and limited evidence (25%) of an effect of site in paired comparisons. Neither treatment nor site were significant predictors in linear models.

3.2.5.3. Taxonomic Richness

Average leaf pack macroinvertebrate taxonomic richness in 2016 ranged from 7.0 to 22.3 taxa across all sites (Appendix Table 25). Within both paired-catchment comparisons (PAN_{REF} vs. BAT_{HARV} and KER A_{REF} vs. KER B_{HARV}) for 2016, there was no interaction between treatment and site (two-factor ANOVA, p=0.18 and p=0.39, respectively) and no effect of treatment (p=0.98 and p=0.89, respectively) or site (p=0.35 and p=0.33, respectively) on average taxonomic richness (Figure 15).

Average leaf pack macroinvertebrate taxonomic richness in 2017 ranged from 13.7 to 28.3 taxa across all sites (Appendix Table 26). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2017, there was no interaction between treatment and site (two-factor ANOVA, p=0.37) and no effect of treatment (p=0.32) on average taxonomic richness (Figure 15). Within treatments, there was an effect of site in BAT_{HARV} only (p=0.0024); average taxonomic richness was higher at the downstream site than the upstream site (Tukey's Multiple Comparisons, p=0.0046). Within the

second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for 2017, there was no interaction between treatment and site (two-factor ANOVA, p=0.76), and no effect of treatment (p=0.72) or site (p=0.35).

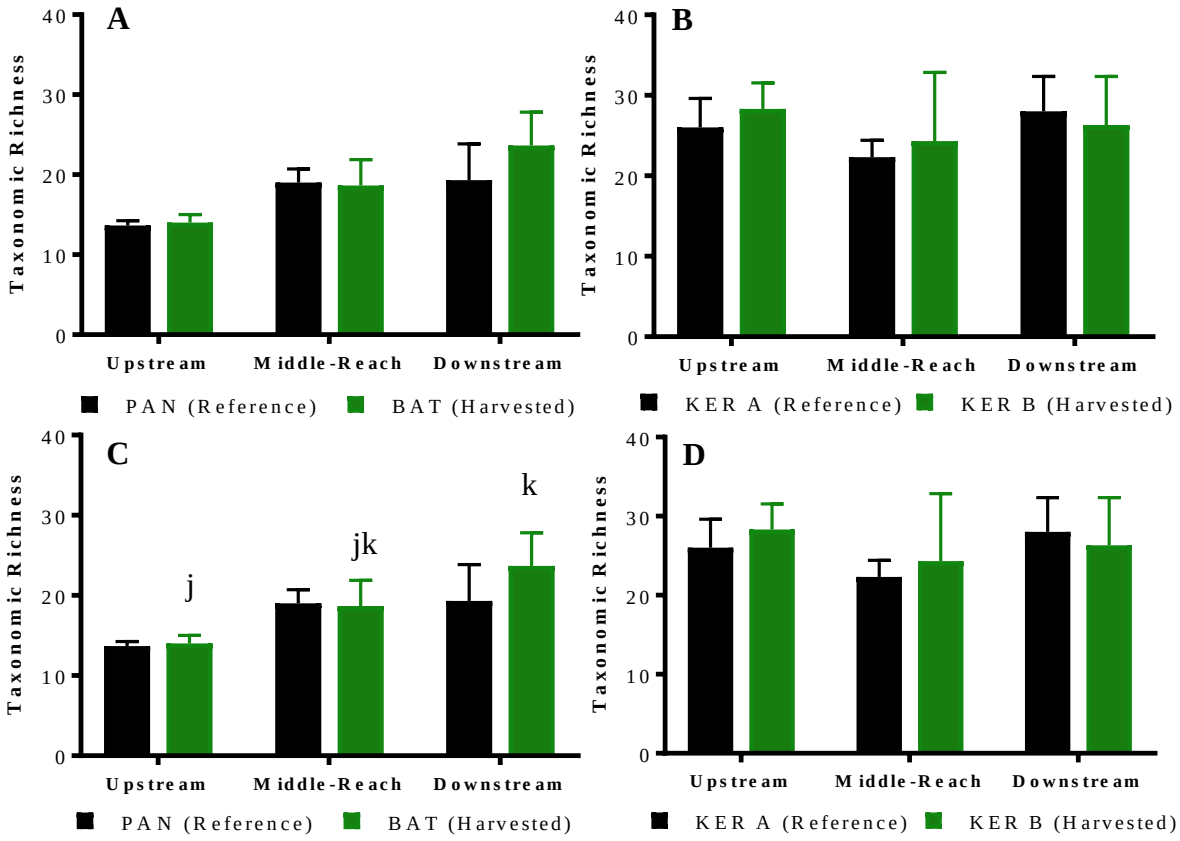

Figure 15: Average (± SD, N=3) leaf pack macroinvertebrate taxonomic richness (number of taxa) at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT in 2016, B) the reference catchment KER A compared to the harvested catchment KER B in 2016, C) the reference catchment PAN compared to the harvested catchment BAT in 2017, and D) the reference catchment KER A compared to the harvested catchment KER B in 2017. The letters indicate a significant difference in average taxonomic richness between site locations within treatments.

The average best-fitting model for leaf pack macroinvertebrate taxonomic richness in 2016 indicated positive effects of site, and negative effects of % EVSA, treatment, DOM_PC2, road density, DOM_PC1, and an interaction between treatment and site (Table 12). No effects of were statistically significant or had great explanatory

power. The conditional R^2 was 0.59, indicating a moderate proportion of total variance was explained by the selected EVs. The average best-fitting model in 2017 indicated positive effects of site, road density, and an interaction between treatment and site, and negative effects of treatment, DOM_PC2, and % EVSA. The effect of site was statistically significant, and this variable also had the greatest explanatory power (highest relative variable importance and slope coefficient). The conditional R^2 was 0.64, indicating a moderate proportion of total variance was explained by the selected EVs.

Table 12: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for leaf pack macroinvertebrate taxonomic richness in 2016 and 2017, as determined via AIC_c model selection ($\Delta AIC_c < 7$).

Response Variable		**Explanatory Variables**						
		% EVSA	**Site**	Treatment	DOM_PC2	Road	DOM_PC1	Treat:Site
Taxonomic Richness 2016	Slope Coefficient	-2.68	1.98	-1.49	-1.28	-0.34	-0.72	-0.29
	95% Confidence Interval	-6.51 to 1.16	-1.90 to 5.85	-5.09 to 2.10	-4.41 to 1.86	-4.72 to 4.04	-4.11 to 2.65	-4.93 to 4.34
	Relative Variable Importance	0.71	0.70	0.64	0.50	0.48	0.45	0.22
Taxonomic Richness 2017	Slope Coefficient	-0.71	**2.39**	-2.14	-1.92	0.14		1.07
	95% Confidence Interval	-3.43 to 2.00	**0.39 to 4.39**	-4.99 to 0.70	-3.91 to 0.07	-3.36 to 3.64		-1.36 to 3.49
	Relative Variable Importance	0.40	**0.90**	0.78	0.73	0.41		0.29

[1]% EVSA = % effective variable source area compared to sub-catchment area; Site = distance (km) to farthest downstream site within catchment multiplied by -1; Treatment = % sub-catchment area harvested within the last 5 years; DOM_PC2 = scores for DOM PCA axis 2; Road = road density (m per ha sub-catchment); DOM_PC1 = scores for DOM PCA axis 1; Treat:Site = interaction between Treatment and Site.

[2]Site in 2016 and 2017 highly correlated (r=0.87, 0.85) with catchment size (not included in full model); Treatment in 2016 and 2017 highly correlated (r=0.77, 0.78) with WC_PC2 (not included in full model); DOM_PC2 in 2016 was highly correlated (r=0.75) with WC_PC1 (not included in full model); % EVSA in 2017 highly correlated (r=-0.70) with flow (not included in full model); Dep and DOM_PC1 in 2017

highly correlated (r=0.87, 0.75) with WC_PC1 (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4) in 2016 and 2017, so % Dec removed from full model; % Organic and Site had high VIFs (>4) in 2016 and 2017, so % Organic removed from full model; Road and DOM_PC1 had high VIFs (>4) in 2016, so DOM_PC1 removed from full model; Road and Dep had high VIFs (>4) in 2017, so Road removed from full model.

Overall, there was no evidence (0%) of spatially cumulative trends for invertebrate taxonomic richness in paired comparisons. There was no evidence (0%) of an effect of treatment and limited evidence (25%) of an effect of site in paired comparisons. Site was a positive predictor in linear models in 2017 only.

3.2.5.4. % Shredders

Average leaf pack % shredders in 2016 ranged from 1.6 to 26.2% across all sites (Appendix Table 25). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2016, there was no interaction between treatment and site (two-factor ANOVA, p=0.15) (Figure 16). Within sites, there was an effect of treatment (p=0.02); average % shredders was higher for BAT_{HARV} than PAN_{REF} at the upstream site only (Sidak's Multiple Comparisons, p=0.04). Within treatments, there was an effect of site in the BAT_{HARV} catchment only (two-factor ANOVA, p=0.0029); average % shredders was higher at the upstream site than the middle-reach and downstream sites (Tukey's Multiple Comparisons, p=0.0019, 0.0390, respectively). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for 2016, there was no interaction between treatment and site (two-factor ANOVA, p=0.29) and no effect of site (p=0.13) on average % shredders. Within sites, there was an effect of treatment (p=0.016); average % shredders was higher in KER B_{HARV} at the upstream site only (Sidak's Multiple Comparisons, p=0.049).

Average leaf pack % shredders in 2017 ranged from 2.8 to 21.3% across all sites (Appendix Table 26). Within both paired-catchment comparisons (PAN_{REF} vs. BAT_{HARV}

and KER A_{REF} vs. KER B_{HARV}) for 2017, there was no interaction between treatment and site (two-factor ANOVA, p=0.45 and p=0.20, respectively), and no effect of treatment (p=0.51 and p=0.49, respectively) or site (p=0.75 and p=0.18, respectively) for average % shredders (Figure 16).

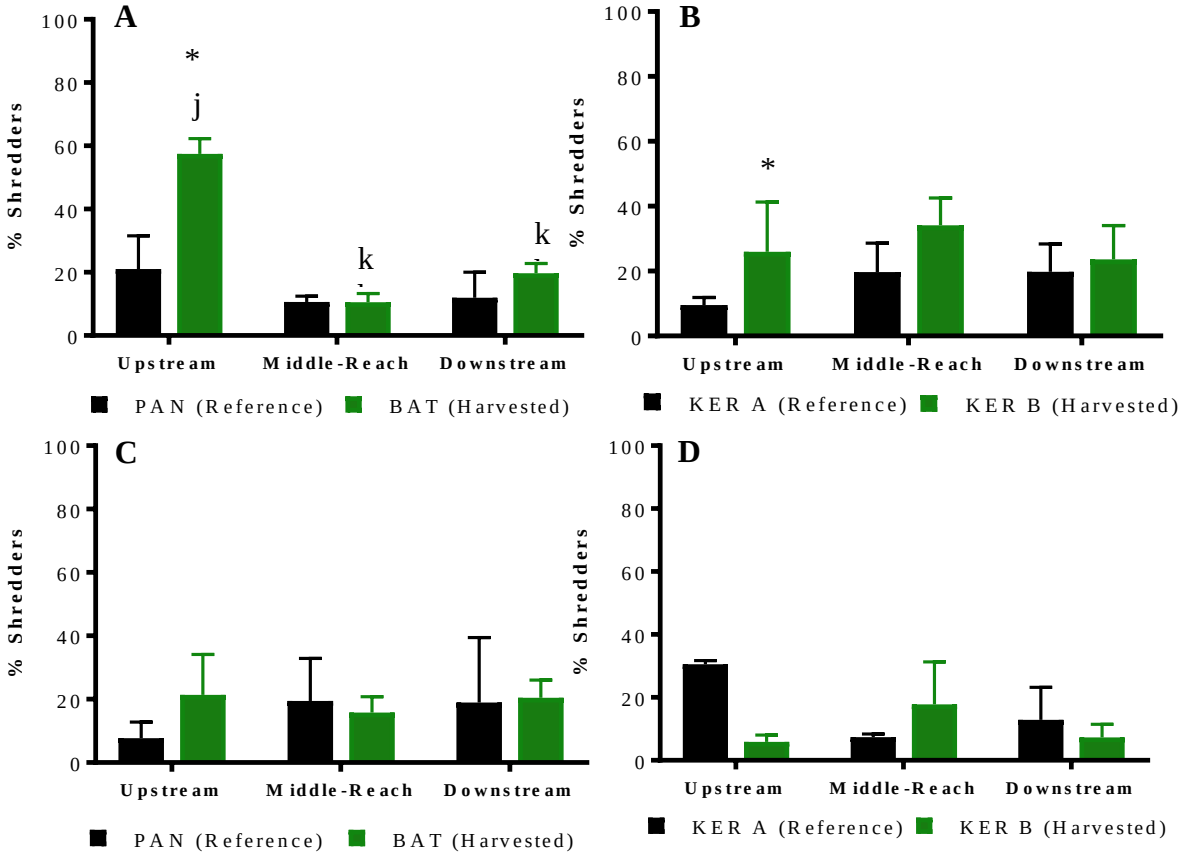

Figure 16: Average (± SD, N=3) leaf pack % shredders (% of invertebrates with shredding feeding strategy) at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT in 2016, B) the reference catchment KER A compared to the harvested catchment KER B in 2016, C) the reference catchment PAN compared to the harvested catchment BAT in 2017, and D) the reference catchment KER A compared to the harvested catchment KER B in 2017. The asterisk (*) indicates a significant difference in average % shredders between the two paired catchments within a site. The letters indicate a significant difference in average % shredders between site locations within treatments.

The average best-fitting model for leaf pack % shredders in 2016 indicated positive effects of treatment, road density, and DOM_PC2, and negative effects of % EVSA and site (Table 13). The effect of treatment was statistically significant, and this

variable also had the greatest explanatory power (highest relative variable importance and slope coefficient). The conditional R^2 was 0.70, indicating a moderate proportion of total variance was explained by the selected EVs. The average best-fitting model in 2017 indicated positive effects of road density, treatment, DOM_PC2, site, % organic of fine sediment, and DOM_PC1, and a negative effect of % EVSA. The effects of road density and DOM_PC2 were statistically significant, and these variables also had the greatest explanatory power (greatest relative variable importance and slope coefficients). The conditional R^2 was 0.54, indicating a moderate proportion of total variance was explained by the selected EVs.

Table 13: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for leaf pack % shredders in 2016 and 2017, as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables						
		Treatment	**Road**	**DOM_PC2**	% EVSA	Site	DOM_PC1	% Organic
% Shredders 2016	Slope Coefficient	**0.47**	0.28	0.004	-0.002	-0.05		
	95% Confidence Interval	**0.07 to 0.85**	-0.21 to 0.77	-0.45 to 0.45	-0.45 to 0.45	-0.48 to 0.37		
	Relative Variable Importance	**0.53**	0.18	0.08	0.08	0.08		
% Shredders 2017	Slope Coefficient	0.29	**0.43**	**0.28**	-0.27	0.14	0.05	0.12
	95% Confidence Interval	-0.13 to 0.71	**0.13 to 0.73**	**0.0005 to 0.55**	-0.58 to 0.05	-0.16 to 0.45	-0.39 to 0.50	-0.21 to 0.45
	Relative Variable Importance	0.17	**0.56**	**0.28**	0.18	0.06	0.06	0.06

[1]Treatment = % sub-catchment area harvested within the last 5 years; Road = road density (m per ha sub-catchment); DOM_PC2 = scores for DOM PCA axis 2; % EVSA = % effective variable source area compared to sub-catchment area; Site = distance (km) to farthest downstream site within catchment multiplied by -1; DOM_PC1 = scores for DOM PCA axis 1; % Organic = % organic of fine sediment.
[2]Site in 2016 and 2017 highly correlated (r=0.87, 0.85) with catchment size (not included in full models); Treatment in 2016 and 2017 highly correlated (r=0.77, 0.78) with WC_PC2 (not included in full models); DOM_PC2 in 2016 was highly correlated (r=0.75) with WC_PC1 (not included in full model); % EVSA in 2017 highly correlated (r=-0.70) with flow (not included in full model); Dep and DOM_PC1 in 2017

highly correlated (r=0.87, 0.75) with WC_PC1 (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4) in 2016 and 2017, so % Dec removed from full models; Road and DOM_PC1 had high VIFs (>4) in 2016, so DOM_PC1 removed from full model; Interaction between Treatment and Site, Road, and Dep had high VIFs (>4) in 2017, so Interaction and Dep removed from full model.

Overall, there was no evidence (0%) of spatially cumulative trends for invertebrate % shredder in paired comparisons. There was some evidence (50%) of an effect of treatment and limited evidence (25%) of an effect of site in paired comparisons. Treatment was a positive predictor in linear models in 2016 only.

3.2.5.5. % Chironomidae

Average leaf pack % Chironomidae in 2016 ranged from 30.0 to 63.9% across all sites (Appendix Table 25). Within both paired-catchment comparisons (PAN_{REF} vs. BAT_{HARV} and KER A_{REF} vs. KER B_{HARV}) for 2016, there was no interaction between treatment and site (two-factor ANOVA, p=0.09 and p=0.64, respectively) and no effect of treatment (p=0.42 and p=0.08, respectively) or site (p=0.70 and p=0.07, respectively) for average % Chironomidae (Figure 17).

Average % Chironomidae in 2017 ranged from 14.4 to 82.2% across all sites (Appendix Table 26). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2017, there was no interaction between treatment and site (two-factor ANOVA, p=0.71) and no effect of treatment (Sidak's Multiple Comparisons, p>0.05) or site (two-factor ANOVA, p=0.21) for average % Chironomidae (Figure 17). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) in 2017, there was no interaction between treatment and site (p=0.11) and no effect of treatment (p=0.35). There was an effect of site (two-factor ANOVA, p<0.0001); average % Chironomidae was higher upstream than downstream in KER A_{REF} (Tukey's Multiple Comparisons,

p=0.0140), and higher upstream and middle-reach than downstream for KER B_{HARV} (p<0.0001, 0.0042, respectively).

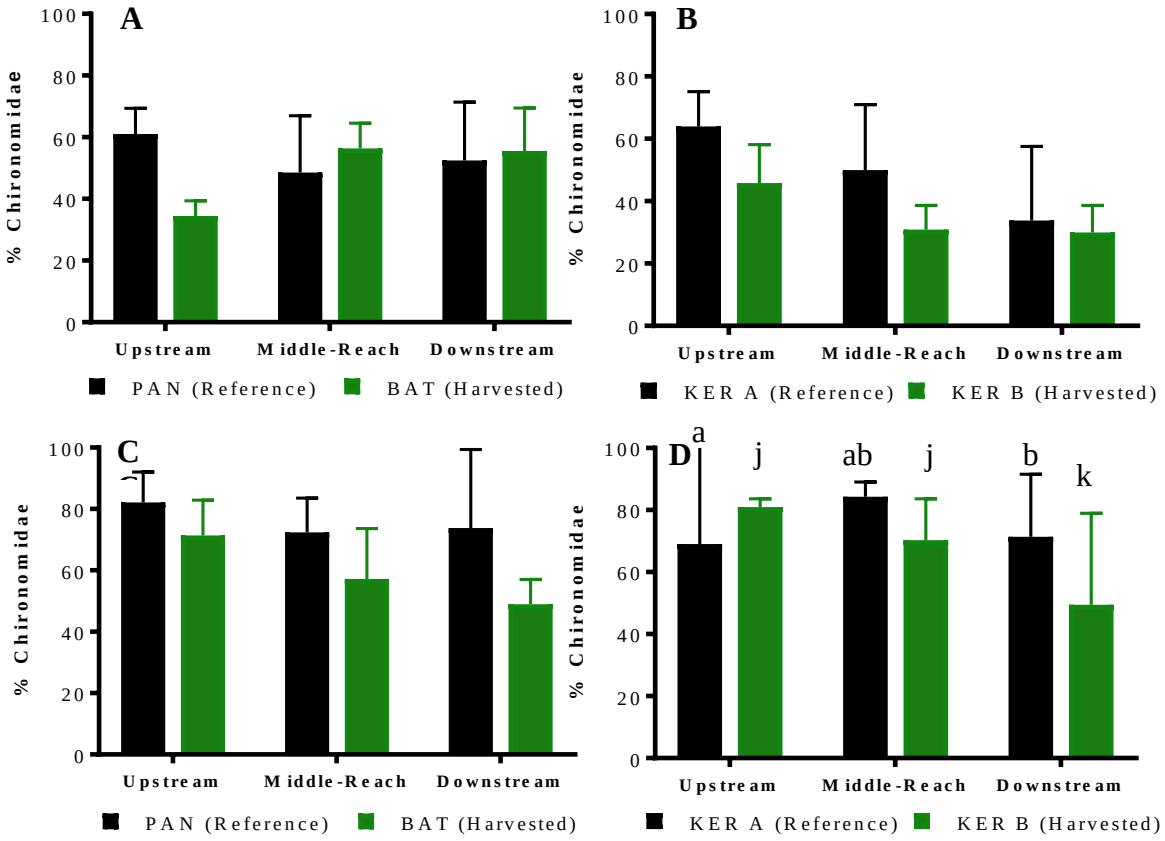

Figure 17: Average (± SD, N=3) leaf pack % Chironomidae (% of invertebrates in the family Chironomidae) at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT in 2016, B) the reference catchment KER A compared to the harvested catchment KER B in 2016, C) the reference catchment PAN compared to the harvested catchment BAT in 2017, and D) the reference catchment KER A compared to the harvested catchment KER B in 2017. The letters indicate a significant difference in average % Chironomidae between site locations within treatments.

The average best-fitting model for % Chironomidae in 2016 indicated positive effects of % EVSA and DOM_PC2 and negative effects of treatment, road density, and site (Table 14). The effect of treatment was statistically significant, and this variable also had the greatest explanatory power (greatest relative variable importance and slope coefficient). The conditional R^2 was 0.60, indicating a moderate proportion of total

variance was explained by the selected EVs. The average best-fitting model in 2017 indicated positive effects of % EVSA, DOM_PC1, and road density, and negative effects of treatment, site, % organic of fine sediment, and DOM_PC2. The effects of % EVSA and treatment were statistically significant, and these variables also had the greatest explanatory power (greatest relative variable importance and slope coefficients). The conditional R^2 was 0.52, indicating a moderate proportion of total variance was explained by the selected EVs.

Table 14: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for leaf pack % Chironomidae in 2016 and 2017, as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables						
		Treatment	**% EVSA**	Road	Site	DOM_PC2	% Organic	DOM_PC1
% Chironomidae 2016	Slope Coefficient	**-0.50**	0.22	-0.24	-0.17	0.08		
	95% Confidence Interval	**-0.91 to -0.06**	-0.08 to 0.50	-0.54 to 0.06	-0.47 to 0.13	-0.27 to 0.39		
	Relative Variable Importance	**0.34**	0.19	0.15	0.12	0.06		
% Chironomidae 2017	Slope Coefficient	**-0.38**	**0.36**	0.07	-0.30	-0.03	-0.26	0.21
	95% Confidence Interval	**-0.70 to -0.05**	**0.07 to 0.66**	-0.37 to 0.51	-0.64 to 0.04	-0.38 to 0.33	-0.57 to 0.06	-0.28 to 0.71
	Relative Variable Importance	**0.43**	**0.45**	0.07	0.22	0.05	0.17	0.10

[1]Treatment = % sub-catchment area harvested within the last 5 years; % EVSA = % effective variable source area compared to sub-catchment area; Road = road density (m per ha sub-catchment); Site = distance (km) to farthest downstream site within catchment multiplied by -1; DOM_PC2 = scores for DOM PCA axis 2; % Organic = % organic of fine sediment; DOM_PC1 = scores for DOM PCA axis 1.
[2]Site in 2016 and 2017 highly correlated (r=0.87, 0.85) with catchment size (not included in full models); Treatment in 2016 and 2017 highly correlated (r=0.77, 0.78) with WC_PC2 (not included in full models); DOM_PC2 in 2016 was highly correlated (r=0.75) with WC_PC1 (not included in full model); % EVSA in 2017 highly correlated (r=-0.70) with Flow (not included in full model); Dep and DOM_PC1 in 2017 highly correlated (r=0.87, 0.75) with WC_PC1 (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4) in 2016 and 2017, so % Dec removed from full model; Road and DOM_PC1 had high VIFs (>4) in 2016, so DOM_PC1 removed from full model; Interaction between

Treatment and Site, Road, and Dep had high VIFs (>4) in 2016 and 2017, so Interaction and Dep removed from full model.

Overall, there was no evidence (0%) of spatially cumulative trends for leaf pack % Chironomidae in paired comparisons. There was no evidence of an effect of treatment (0%) and limited evidence of an effect of site (25%) in paired comparisons. Treatment was a negative predictor in linear models in both 2016 and 2017.

3.2.6. Hydrogen Stable Isotopes

3.2.6.1. % Algae in Hydropsychid Diets

Average δ^2H in Hydropsychids ranged from -170.2 to -95.9‰ across all sites in 2016 and 2017 (Appendix Figure 4 to Appendix Figure 7, Appendix Table 32 and Appendix Table 33). An example plot for δ^2H values of Hydropsychids and all measured food sources is shown in Figure 18. See Appendix Figure 4 to Appendix Figure 7, Appendix Table 32 and Appendix Table 33 for raw data and plots of δ^2H for Hydropsychids and all measured food sources for catchments and all sampling events.

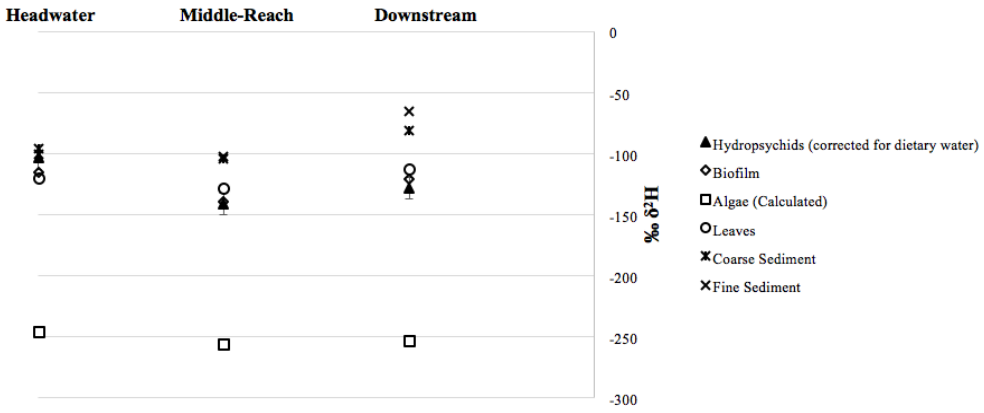

Figure 18: Average (± SD, N=1-3) hydrogen isotope values (δ^2H, ‰) for Hydropsychids (consumer) and all possible food sources in the BAT_{HARV} catchment in October 2016.

δ^2H was not measured for the upstream site in PAN_{REF} in October 2016 due to insufficient sample mass, so average % algae in Hydropsychid diets for the PAN_{REF} vs.

BAT_{HARV} paired-catchment comparisons was not assessed for this year. Average % algae in Hydropsychid diets in October 2016 ranged from 4.1 to 28.2% across sites within the KER A_{REF} vs. KER B_{HARV} paired comparison (Appendix Table 32). Within this comparison, there was no interaction between treatment and site (two-factor ANOVA, p=0.53) and no effect of treatment (p=0.21) for average % algae (Figure 19). There was an effect of site in KER A_{REF} only (p=0.04); average % algae was higher at the downstream than the middle-reach site (Tukey's Multiple Comparisons, p=0.04).

Average % algae in Hydropsychid diets in September 2017 ranged from -7.4 to 23.9% across all sites (Appendix Table 33). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) in September 2017, there was no interaction between treatment and site (two-factor ANOVA, p=0.24) (Figure 19); an effect of treatment and site individually could only be calculated for BAT_{HARV} due to limited replication in PAN_{REF}. For BAT_{HARV}, there was no effect of treatment (p=0.93) but there was an effect of site (p=0.0011); average % algae was higher at the middle-reach and downstream sites than the upstream site (Tukey's Multiple Comparisons, p=0.0006). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) in September 2017, there was an interaction between treatment and site (two-factor ANOVA, p=0.0002) for average % algae. Within sites, there was an effect of treatment; average % algae was higher at in KER B_{HARV} than KER A_{REF} at the middle-reach site only (multiple t-tests, p=0.027). Within treatments, there was an effect of site for KER B_{HARV} only (one-factor ANOVA, p=0.023); average % algae was higher at the middle-reach site than the upstream and downstream sites (Tukey's Multiple Comparisons, p=0.037, 0.001, respectively), and higher at the upstream site than the downstream site

(p=0.024). Similar treatment and spatial trends were noted for average % algae in Hydropsychid diets in August and October 2017 (Appendix Table 34).

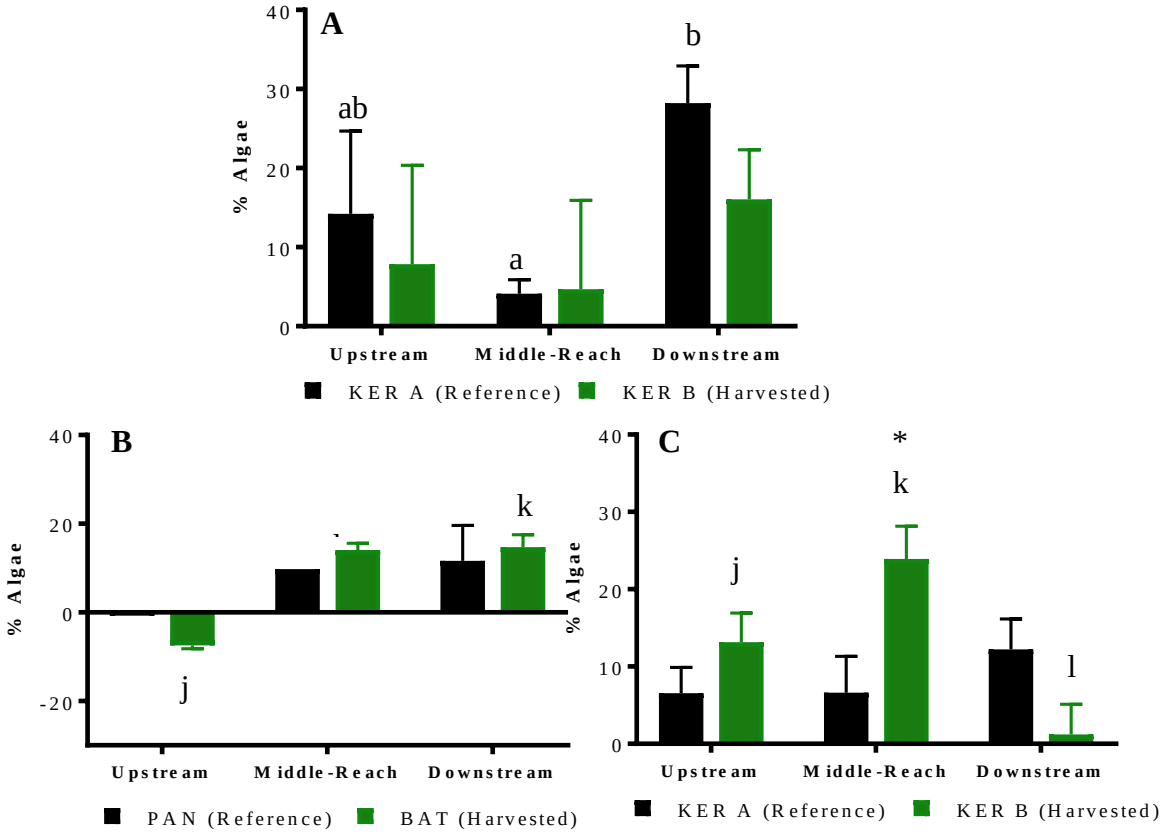

Figure 19: Average (± SD, N=1-3) % algae in Hydropsychid diets (% organic matter in Hydropsychid diet with algal δ^2H signature) at upstream, middle-reach, and downstream sites in A) the reference catchment KER A compared to the harvested catchment KER B in 2016, B) the reference catchment PAN compared to the harvested catchment BAT in September 2017, and C) the reference catchment KER A compared to the harvested catchment KER B in September 2017. The asterisk (*) indicates a significant difference among average % algae between the two paired catchments within a site. The letters indicate a significant difference in average % algae between site locations within treatments.

The average best-fitting model for % algae in Hydropsychid diets indicated positive effects of DOM_PC2, fine sediment deposition, and year, and negative effects of % EVSA, DOM_PC1, % deciduous tree species, treatment, site, flow, % organic of fine sediment, and month (Table 15). The effect of % EVSA only was statistically significant, and this variable also had the greatest explanatory power (greatest relative

variable importance and slope coefficient). The conditional R^2 was 0.48, indicating a moderate proportion of total variance was explained by the selected EVs.

Table 15: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for % algae in Hydropsychid diets, as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables										
		% EVSA	DOM_ PC1	% Dec	Treatment	Site	DOM_ PC2	Dep	Flow	% Organic	Month	Year
%Algae in Hydropsychid Diets	Slope Coefficient	**-0.37**	-0.21	-0.22	-0.20	-0.002	0.10	0.03	-0.04	-0.02	-0.04	0.02
	95% Confidence Interval	**-0.66 to -0.08**	-0.48 to 0.05	-0.57 to 0.13	-0.50 to 0.09	-0.40 to 0.40	-0.15 to 0.34	-0.20 to 0.27	-0.20 to 0.13	-0.19 to 0.15	-0.17 to 0.08	-0.11 to 0.15
	Relative Variable Importance	**0.47**	0.17	0.14	0.13	0.07	0.05	0.03	0.02	0.02	0.01	0.01

[1]% EVSA = % effective variable source area compared to sub-catchment area; DOM_PC1 = scores for DOM PCA axis 1; % Dec= % deciduous tree species; Treatment = % sub-catchment area harvested within the last 5 years; Site = distance (km) to farthest downstream site within catchment multiplied by -1; DOM_PC2 = scores for DOM PCA axis 2; Dep = fine, inorganic sediment deposition (g/day); Flow = stream velocity (m/s); % Organic = % organic of fine sediment; Month = month of sampling (1=August, 2=September, 3=October); Year = year of sampling.
[2]Site highly correlated (r=0.70) with catchment size (not included in full model); DOM_PC1 and Dep highly correlated (r=-0.74, -0.74) with WC_PC1 (not included in full model); Treatment highly correlated (r=0.75) with WC_PC2 (not included in full model); Road and Treat:Site Interaction had high VIFs (>4), so Road removed from full model.

Overall, there was some (57%) evidence of spatially cumulative trends for % algae in paired comparisons (including data shown in Appendix Table 34). There was also some evidence (57%) of an effect of treatment and site in paired comparisons. Neither treatment nor site were significant predictors in linear models.

3.2.7. Mercury (Hg)

Only MeHg in filtered and unfiltered water and MeHg, Hg(II), and % MeHg in Hydropsychids were modelled; other Hg endpoints did not have sufficient replication or had too few or missing data among years and sampling events to be modeled.

3.2.7.1. Water

Average MeHg in filtered water in October 2016 ranged from 0.09 to 0.35 ng/L across all sites (Appendix Table 36). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for October 2016, there was an interaction between treatment and site (two-factor ANOVA, p=0.0002) for average MeHg in filtered water (Figure 20). Within sites, there was an effect of treatment; average MeHg in filtered water was higher in BAT_{HARV} than PAN_{REF} at the upstream site only (multiple t-tests, p=0.009). Within treatments, there was an effect of site for KER A_{REF} and KER B_{HARV} (one-factor ANOVA, p=0.027, 0.008, respectively); average MeHg in filtered water was higher downstream than upstream for PAN_{REF} (Tukey's Multiple Comparisons, p=0.027), and was higher upstream than at the middle-reach and downstream sites for BAT_{HARV} (p=0.008). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for October 2016, there was also an interaction between treatment and site (two-factor ANOVA, p=0.005) for average MeHg in filtered water. There was an effect of treatment; average MeHg in filtered water was higher in KER B_{HARV} than KER A_{REF} at the upstream site only (multiple t-tests, p=0.021). Within treatments, there was an effect of site in KER A_{REF} only (one-factor ANOVA, p=0.017); average MeHg in filtered water was higher at the middle-reach site than the upstream site (Tukey's Multiple Comparisons, p=0.014). Average MeHg in unfiltered water for October 2016 followed similar treatment and spatial trends (data are shown in Appendix Table 38).

Treatment and spatial trends for average MeHg in filtered water were similar in August, September, and October in 2017 and sometimes only had one collection per site, so the average of monthly values were used for the ANOVAs (see Appendix Table 37 for monthly concentrations). Average MeHg in filtered water in 2017 ranged from

0.06 to 0.28 ng/L across all sites (Appendix Table 37). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2017, there was an interaction between treatment and site (two-factor ANOVA, p=0.0024) for average MeHg in filtered water (Figure 20). Within sites, there was an effect of treatment; average MeHg in filtered water was higher in PAN_{REF} than BAT_{HARV} at the downstream site only (multiple t-tests, p=0.0016). Within treatments, there was an effect of site in PAN_{REF} only (one-factor ANOVA, p=0.0061); average MeHg in filtered water was higher at the downstream site than the upstream site (p=0.0050). Similarly, within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for 2017, there was an interaction between treatment and site (two-factor ANOVA, p=0.018) for average MeHg in filtered water. Within sites, there was an effect of treatment; average MeHg in filtered water was higher in KER B_{HARV} than KER A_{REF} at the upstream site only (multiple t-tests, p=0.030). Within treatments, there was an effect of site in KER B_{HARV} only (one-way ANOVA, p=0.025); average MeHg in filtered water was higher at the upstream site than the downstream site (Tukey's Multiple Comparisons, p=0.023). Average MeHg in unfiltered water for 2017 followed similar treatment and spatial trends (data are shown in Appendix Table 38).

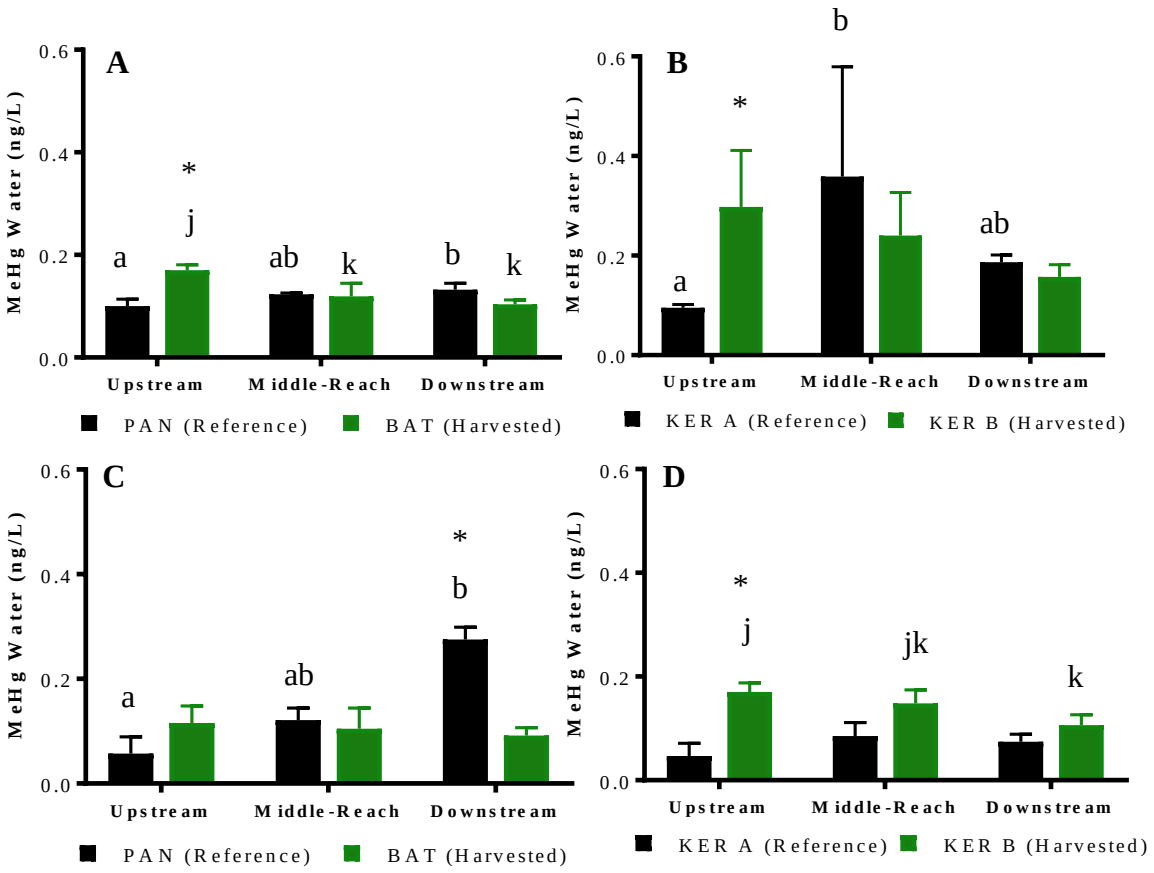

Figure 20: Average (± SD, N = 3) concentration of methylmercury (MeHg, ng/L) in filtered (0.45 µm) water at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT in 2016, B) the reference catchment KER A compared to the harvested catchment KER B in 2016, C) the reference catchment PAN compared to the harvested catchment BAT in 2017, and D) the reference catchment KER A compared to the harvested catchment KER B in 2017. The asterisk (*) indicates a significant difference in average MeHg in filtered water between the two paired catchments. The letters indicate significant difference among average MeHg in filtered water between site locations.

The average best-fitting model for MeHg in filtered water indicated positive effects of DOM_PC1, DOM_PC2, treatment, % organic content of fine sediment, site, and month of sampling, and negative effects of year, % EVSA, % deciduous tree species, and stream flow (Table 16). The effect of year, DOM_PC1, and % EVSA were statistically significant, and these variables also had the greatest explanatory power (greatest relative variable importance and slope coefficients). The conditional R^2 was

0.64, indicating a moderate proportion of total variance was explained by the selected EVs. See Appendix Table 39 for the average model for MeHg in unfiltered water.

Table 16: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for MeHg in filtered stream water, as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables									
		Year	**DOM_ PC1**	**% EVSA**	DOM_ PC2	% Dec	Treat ment	Flow	% Organic	Site	Month
MeHg in Filtered Water	Slope Coefficient	**-0.24**	**0.30**	**-0.16**	0.088	-0.048	0.068	-0.059	0.024	0.012	0.026
	95% Confidence Interval	**-0.32 to -0.17**	**0.16 to 0.43**	**-0.29 to -0.030**	-0.051 to 0.23	-0.23 to 0.14	-0.084 to 0.22	-0.17 to 0.052	-0.091 to 0.14	-0.099 to 0.12	-0.045 to 0.097
	Relative Variable Importance	**0.89**	**0.87**	**0.38**	0.07	0.05	0.05	0.05	0.03	0.03	0.02

[1]Year = year of sampling; DOM_PC1 = scores for DOM PCA axis 1; % EVSA = % effective variable source area compared to sub-catchment area; DOM_PC2 = scores for DOM PCA axis 2; % Dec= % deciduous tree species; Treatment = % sub-catchment area harvested within the last 5 years; Flow = stream velocity (m/s); % Organic = % organic content of fine sediment; Site = distance (km) to farthest downstream site within catchment multiplied by -1; Month = month of sampling (1=August, 2=September, 3=October).
[2]Site highly correlated (r=0.70) with catchment size (not included in full model); Treatment highly correlated (r=0.75) with Road (not included in full model); Treatment highly correlated (r=0.78) with WC_PC2 (not included in full model); DOM_PC1 and Dep highly correlated (r=-0.73, -0.73) with WC_PC1 (not included in full model); DOM_PC1 and Dep had high VIFs (>4), so Dep removed from full model; % Dec and Catchment (as a fixed effect) had high VIFs (>4), so Catchment removed from full model.

Overall, there was strong evidence (100%) of spatially cumulative trends for MeHg in filtered water in paired comparisons. There was also strong evidence (100%) of an effect of treatment and site in paired comparisons. However, neither treatment nor site were significant predictors in linear models.

3.2.7.2. Hydropsychids

3.2.7.2.1. MeHg & % MeHg

Average MeHg in Hydropsychids in October 2016 ranged from 29.7 to 136.1 µg/kg dw across all sites (Appendix Table 35). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for October 2016, there was an interaction between

treatment and site (two-factor ANOVA, p<0.0001) for average MeHg in Hydropsychids (Figure 21). Within sites, there was an effect of treatment; average MeHg in Hydropsychids was higher in BAT_{HARV} than PAN_{REF} at the upstream site only (multiple t-tests, p<0.0001). Within treatments, there was an effect of site (one-factor ANOVA, p=0.0290, 0.0004, respectively); average MeHg in Hydropsychids was higher at the middle-reach site than the upstream site for PAN_{REF}, and higher at the upstream and downstream sites than the middle-reach site for BAT_{HARV} (p=0.0240, 0.0004, 0.0019, respectively). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for October 2016, there was an interaction between treatment and site (two-factor ANOVA, p<0.0001) for average MeHg in Hydropsychids. Within sites, there was an effect of treatment; average MeHg in Hydropsychids was higher in KER B_{HARV} than KER A_{REF} at the upstream and middle-reach sites (multiple t-tests, p=0.0003, 0.0003, respectively). Within treatments, there was an effect of site (one-way ANOVA, p=<0.0001, 0.0016, respectively); average MeHg in Hydropsychids was higher at the downstream and middle-reach sites than the upstream site and higher at the downstream site than the middle-reach site in KER A_{REF} (Tukey's Multiple Comparisons, p=0.0001, <0001, 0.0280, respectively), and higher at the middle-reach than the upstream and downstream sites in KER B_{HARV} (p=0.0086, 0.0015, respectively).

Treatment and spatial trends in average MeHg in Hydropsychids in 2017 were similar among sampling trips, so only results for September 2017 are shown for the ANOVAs below (see Appendix Table 37 for results for other dates). Average MeHg in Hydropsychids in September 2017 ranged from 58.2 to 192.2 µg/kg dw across all sites (Appendix Table 37). Within the first paired-catchment comparison (PAN_{REF} vs.

BAT_{HARV}) for September 2017, there was an interaction between treatment and site (two-factor ANOVA, p=0.0004) in average MeHg in Hydropsychids (Figure 21). Within sites, there was an effect of treatment; average MeHg in Hydropsychids was higher in BAT_{HARV} than PAN_{REF} at the upstream site only (multiple t-tests, p=0.0002). Within treatments, there was an effect of site in PAN_{REF} only (one-factor ANOVA, p=0.0007); average MeHg in Hydropsychids was higher at the middle-reach and downstream sites than the upstream site (Tukey's Multiple Comparisons, p=0.0008, 0.0017, respectively). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for September 2017, there was an interaction between treatment and site (two-factor ANOVA, p=0.0020) in average MeHg in Hydropsychids. Within sites, there was an effect of treatment; average MeHg in Hydropsychids was higher in KER B_{HARV} than KER A_{REF} at the middle-reach site and higher in KER A_{REF} than KER B_{HARV} at the downstream site (multiple t-tests, p=0.016, 0.032, respectively). Within treatments, there was an effect of site in KER B_{HARV} only (one-factor ANOVA, p<0.0001); average MeHg in Hydropsychids was higher at the upstream and middle-reach sites than the downstream site, and higher at the middle-reach site than the downstream site (Tukey's Multiple Comparisons, p=0.0001, 0.0200, <0.0001, respectively). Trends in % MeHg were similar to MeHg concentration (data are shown in Appendix Table 38).

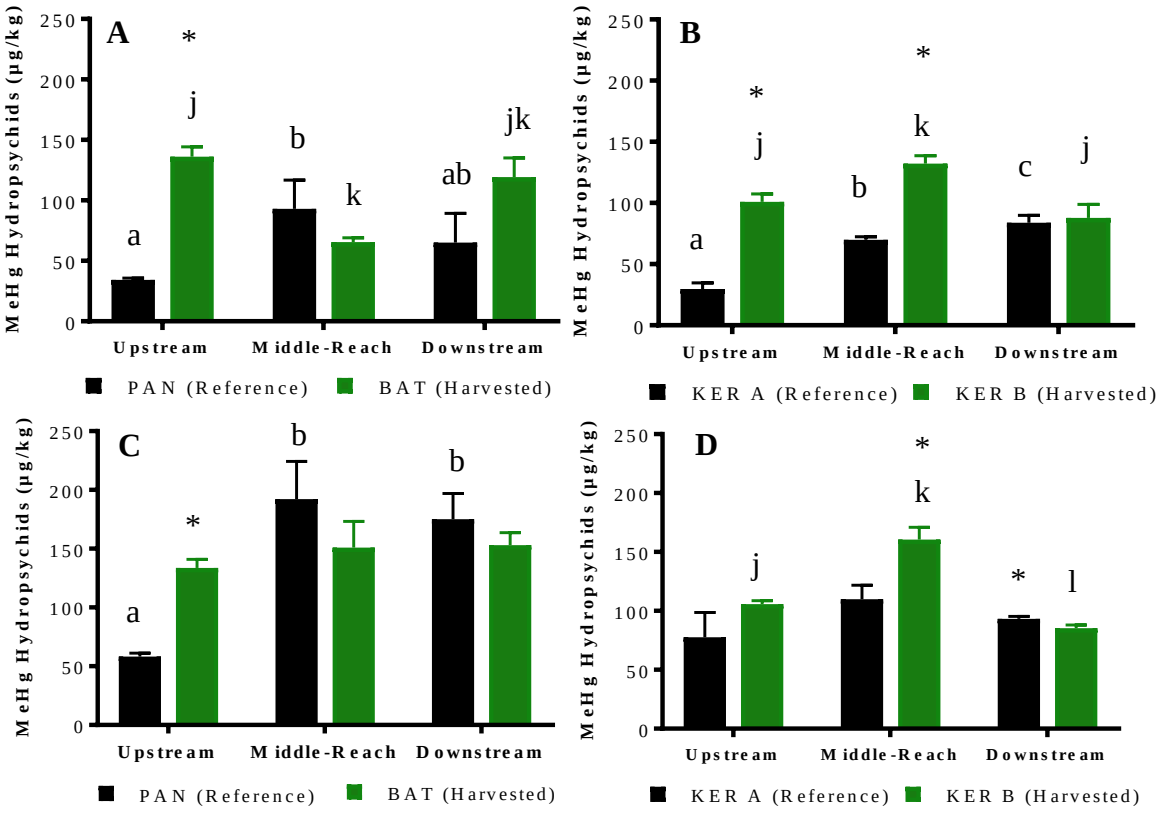

Figure 21: Average (± SD, N=3) concentration of methylmercury (MeHg, µg/kg dw) in Hydropsychids for October 2016 at upstream, middle-reach, and downstream sites A) the reference catchment PAN compared to the harvested catchment BAT in 2016, B) the reference catchment KER A compared to the harvested catchment KER B in 2016, C) the reference catchment PAN compared to the harvested catchment BAT in 2017, and D) the reference catchment KER A compared to the harvested catchment KER B in 2017. The asterisk (*) indicates a significant difference in average MeHg in Hydropsychids between the two paired catchments within a site. The letters indicate a significant difference in average MeHg in Hydropsychids between site locations within treatments.

The average best-fitting model for MeHg in Hydropsychids indicated positive effects of treatment, MeHg in filtered water, month, year, DOM_PC1, MeHg in seston, and fine sediment deposition, and negative effects of % EVSA, DOM_PC2, site, % organic of fine sediment, interaction between treatment and year, and interaction between treatment and site (Table 17). The effects of MeHg in filtered water, year, % EVSA, treatment, and month were statistically significant, and these variables also had

the greatest explanatory power (greatest relative variable importance and slope coefficients). The conditional R^2 was 0.80, indicating a high proportion of total variance was explained by the selected EVs. See Appendix Table 40 for the average model for % MeHg in Hydropsychids.

Table 17: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for MeHg in Hydropsychids, as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables												
		MeHg F Water	**Year**	**% EVSA**	**Treatment**	**Month**	Site	DOM_PC1	MeHg Seston	DOM_PC2	Dep	% Organic	Treat:Year	Treat:Site
MeHg in Hydropsychids (µg/kg dw)	Slope Coefficient	**0.55**	**0.87**	**-0.98**	**0.80**	**0.31**	-0.05	0.25	0.10	-0.21	0.081	-0.14	-0.13	-0.17
	95% Confidence Interval	**0.23 to 0.87**	**0.59 to 1.16**	**-1.74 to -0.22**	**0.09 to 1.52**	**0.08 to 0.54**	-0.99 to 0.89	-0.56 to 1.05	-0.75 to 0.95	-0.98 to 0.56	-0.48 to 0.64	-0.54 to 0.26	-0.36 to 0.10	-1.00 to 0.67
	Relative Variable Importance	**0.63**	**0.63**	**0.58**	**0.51**	**0.50**	0.15	0.12	0.12	0.12	0.06	0.05	0.04	0.02

[1]MeHg F water = concentration of MeHg (ng/L) in filtered water; Year = year of sampling; % EVSA = % effective variable source area compared to sub-catchment area; Treatment = % sub-catchment area harvested within the last 5 years; Month = month of sampling (1=August, 2=September, 3=October); Site = distance (km) to farthest downstream site within catchment multiplied by -1; DOM_PC1 = scores for DOM PCA axis 1; MeHg Seston = concentration of MeHg (µg/kg dw) in seston; DOM_PC2 = scores for DOM PCA axis 2; Dep = deposition of fine, inorganic sediment (g/day); % Organic = % organic of fine sediment; Treat:Year = interaction between Treatment and Year; Treat:Site = interaction between Treatment and Site.
[2]Site highly correlated (r=0.85) with catchment size (not included in full model); Treatment highly correlated (r=0.73) with WC_PC2 (not included in full model); DOM_PC1 and Dep highly correlated (r=-0.75, -0.74) with WC_PC1 (not included in full model); MeHg in filtered water highly correlated (r=0.70) with MeHg in unfiltered water; % Dec and % EVSA had high VIFs (>4), so % Dec removed from full models; Road and Treat:Site had high VIFs (>4), so Road removed from full models.

Overall, there was strong evidence (100%) of spatially cumulative trends for MeHg in Hydropsychids in paired comparisons (including data in Appendix Table 37). There was also strong evidence (100%) of an effect of treatment and site in paired comparisons. Treatment was a positive predictor in linear models.

3.2.7.2.2. Hg(II)

Average Hg(II) in Hydropsychids in October 2016 ranged from 31.4 to 157.2 µg/kg dw across all sites (Appendix Table 35). Within the first paired-catchment

comparison (PAN_{REF} vs. BAT_{HARV}) for October 2016, there was an interaction between treatment and site (two-factor ANOVA, p=0.0007) for average Hg(II) in Hydropsychids (Figure 22). Within sites, there was an effect of treatment; average Hg(II) in Hydropsychids was higher in BAT_{HARV} than PAN_{REF} at the middle-reach and downstream sites (multiple t-tests, p=0.024, 0.024, respectively). Within treatments, there was an effect of site in BAT_{HARV} only (one-factor ANOVA, p=0.0002); average Hg(II) in Hydropsychids was higher in the middle-reach and downstream sites than the upstream site (Tukey's Multiple Comparisons, p=0.0003, 0.0004, respectively). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for October 2016, there was an interaction between treatment and site (two-factor ANOVA, p<0.0001) for average Hg(II) in Hydropsychids. Within sites, there was an effect of treatment; average Hg(II) in Hydropsychids was higher in KER B_{HARV} than KER A_{REF} at the upstream site (multiple t-tests, p=0.0045). Within treatments, there was an effect of site in both KER A_{REF} and KER B_{HARV} (one-way ANOVA, p=0.0002, 0.0180, respectively); average Hg(II) in Hydropsychids was higher at the middle-reach and downstream sites than the upstream site in KER A_{REF} (Tukey's Multiple Comparisons, p=0.0002, 0.0008, respectively), and higher at the middle-reach site than the downstream site in KER B_{HARV} (p=0.0170).

Treatment and spatial trends in average Hg(II) in Hydropsychids in 2017 were similar among sampling trips, so only results for September 2017 are shown for the ANOVAs below (see Appendix Table 37 for results for other dates). Average Hg(II) in Hydropsychids in September 2017 ranged from 38.7 to 213.6 µg/kg across all sites (Appendix Table 37). Within the first paired-catchment comparison (PAN_{REF} vs.

BAT_{HARV}), there was no interaction between treatment and site (two-factor ANOVA, p=0.46) and no effect of treatment (Sidak's Multiple Comparisons, p>0.05) for average Hg(II) in Hydropsychids (Figure 22). Within treatments, there was an effect of site (two-factor ANOVA, p<0.0001); average Hg(II) in Hydropsychids was higher at the middle-reach and downstream sites than the upstream site in both PAN_{REF} (Tukey's Multiple Comparisons, p=0.0016, 0.0075, respectively) and BAT_{HARV}-(p=0.0001, 0.0006, respectively). Within the second-paired comparison (KER_{REF} vs. KER_{BAT}), there was no interaction between treatment and site (two-factor ANOVA, p=0.32) and no effect of treatment (p=0.96) or site (p=0.09) for average Hg(II) in Hydropsychids.

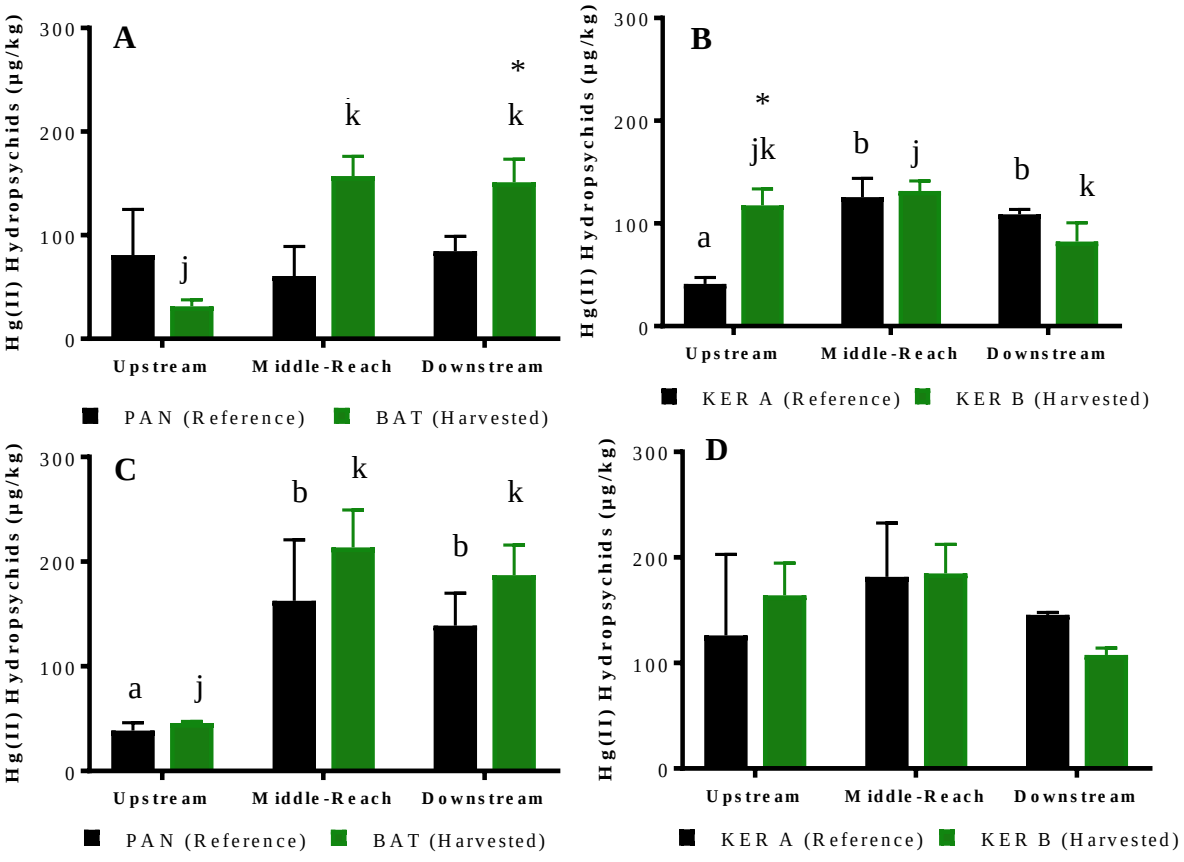

Figure 22: Average (± SD, N=3) concentration of divalent mercury (Hg(II), µg/kg dw) in Hydropsychids at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT in 2016, B) the reference catchment KER A compared to the harvested catchment KER B in 2016, C) the reference catchment PAN compared to the harvested catchment BAT in September

2017, and D) the reference catchment KER A compared to the harvested catchment KER B in September 2017. The asterisk (*) indicates a significant difference in average Hg(II) in Hydropsychids between the two paired catchments within a site. The letters indicate significant difference in average Hg(II) in Hydropsychids between site locations within treatments.

The average best-fitting model for Hg(II) in Hydropsychids indicated positive effects of year, MeHg in filtered water, DOM_PC2, Hg(II) in seston, MeHg in seston, and site, and negative effects of % EVSA, treatment, DOM_PC1, month, fine sediment deposition, and % organic of fine sediment (Table 18). The effects of year, % EVSA, treatment, and MeHg in filtered water were statistically significant, and these variables also had the greatest explanatory power (greatest relative variable importance and slope coefficients). The conditional R^2 was 0.74, indicating a moderate proportion of total variance was explained by the selected EVs.

Table 18: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in average model for Hg(II) in Hydropsychids, as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables											
		Year	**% EVSA**	**Treatment**	**MeHg F Water**	DOM_PC2	Site	DOM_PC1	Hg(II) Seston	MeHg Seston	Dep	Month	% Organic
Hg(II) in Hydropsychids (µg/kg dw)	Slope Coefficient	**0.15**	**-0.31**	**-0.15**	**0.08**	0.10	0.08	-0.04	0.06	0.06	-0.04	-0.04	-0.02
	95% Confidence Interval	**0.09 to 0.21**	**-0.47 to -0.15**	**-0.29 to -0.01**	**0.004 to 0.15**	-0.074 to 0.28	-0.16 to 0.31	-0.20 to 0.12	-0.11 to 0.24	-0.12 to 0.24	-0.16 to 0.08	-0.10 to 0.01	-0.12 to 0.07
	Relative Variable Importance	**0.73**	**0.66**	**0.16**	**0.10**	0.06	0.03	0.03	0.02	0.02	0.02	0.02	0.01

[1]Year = year of sampling; % EVSA = % effective variable source area compared to sub-catchment area; Treatment = % sub-catchment area harvested within the last 5 years; MeHg F water = concentration of MeHg (ng/L) in filtered water; DOM_PC2 = scores for DOM PCA axis 2; Site = distance (km) to farthest downstream site within catchment multiplied by -1); DOM_PC1 = scores for DOM PCA axis 1; Hg(II) Seston = concentration of Hg(II) (µg/kg dw) in seston; MeHg Seston = concentration of MeHg (µg/kg dw) in seston; Dep = Deposition of fine, inorganic sediment (g/day); Month = month of sampling (1=August, 2=September, 3=October); % Organic = % organic of fine sediment.
[2]Site highly correlated (*r*=0.85) with catchment size (not included in full model); Treatment highly correlated (*r*=0.73) with WC_PC2 (not included in full model); DOM_PC1 and Dep highly correlated (*r*=-0.75, -0.74) with WC_PC1 (not included in full model); MeHg in filtered water highly correlated (*r*=0.70) with MeHg in unfiltered water; % Dec and % EVSA had high VIFs (>4), so % Dec removed from full model; Road and Treat:Site had high VIFs (>4), so Road removed from full model.

Overall, there was limited evidence (29%) of spatially cumulative trends for Hg(II) in Hydropsychids in harvested catchments, as indicated by an interaction between treatment and site for two of seven paired comparisons (including data in Appendix Table 37). There was also limited evidence of an effect of treatment (29%) and some evidence of an effect of site (71%) in paired comparisons. Treatment was a negative predictor in linear models.

3.2.7.3. Seston

Hg in seston was measured only in 2017. Due to missing data for many sites throughout the three sampling trips in 2017, values for Hg in seston were averaged/compiled for all trips in 2017 in the ANOVAs below (see Appendix Table 37 for results for individual sampling events).

3.2.7.3.1. MeHg, % MeHg, & Hg(II)

Average MeHg concentration in seston in 2017 ranged from 2.3 to 13.9 µg/kg across all sites (Appendix Table 37). Within both paired-catchment comparisons (PAN_{REF} vs. BAT_{HARV} and KER A_{REF} vs. KER B_{HARV}) for 2017, there was no interaction between treatment and site (two-factor ANOVA, p=0.29 and p=0.82, respectively) and no effect of treatment (p=0.51 and p=0.34, respectively) or site (p=0.24 and p=0.80, respectively) for average MeHg in seston (Figure 23). Trends in % MeHg and Hg(II) in seston were similar to those for MeHg concentration (data are shown in Appendix Table 38). Overall, there was no evidence (0%) of spatially cumulative trends for MeHg in seston. There was also no evidence (0%) of an effect of treatment or site in paired comparisons.

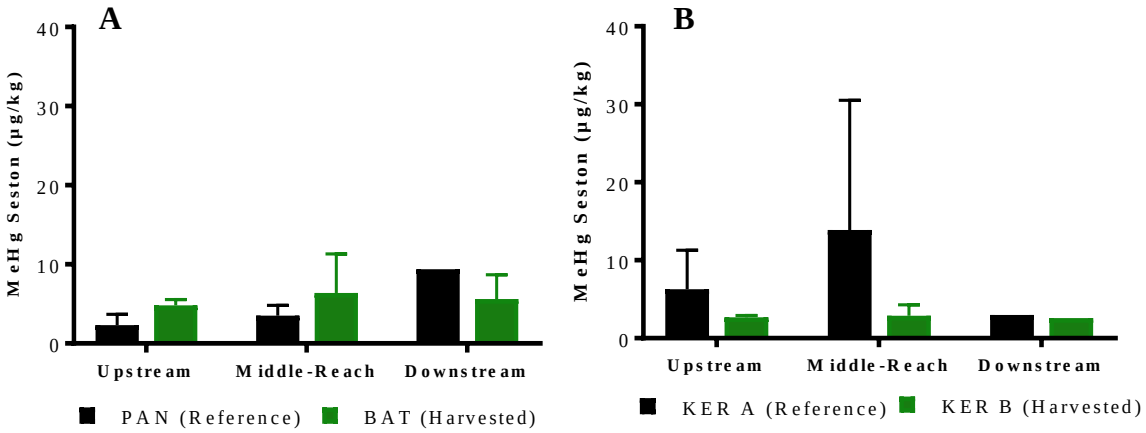

Figure 23: Average (±SD, N=1-3) concentration of methylmercury (MeHg, µg/kg dw) in seston in 2017 at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT, and B) the reference catchment KER A compared to the harvested catchment KER B.

3.2.7.4. Bioaccumulation Factor (BAF) & Biomagnification Factor (BMF) for Hydropsychids

Due to sample compositing, only single BAF values were calculated for MeHg in Hydropsychids for each site in 2016 (i.e., not an average) and this precluded conducting ANOVAs. Hence, ANOVAs were only conducted on BAFs in 2017 (see Appendix Table 37 for 2016 BAFs). Treatment and spatial trends in BAF in 2017 were similar among sampling trips, so monthly values were averaged for the ANOVAs described below (see Appendix Table 37 for results for individual months).

Average MeHg BAFs in 2017 ranged from 538 324 to 1 727 663 across all sites (Appendix Table 37). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2017, there was no interaction between treatment and site (two-factor ANOVA, $p>0.99$) and no effect of site ($p=0.99$) for average BAF (Figure 24). Within sites, there was an effect of treatment ($p=0.047$); average BAF was higher for BAT_{HARV} than PAN_{REF} at the downstream site (Sidak's Multiple Comparisons, $p=0.047$). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for 2017, there

was no interaction between treatment and site (p=0.14) and no effect of site within treatment (p=0.96) for average BAF. Within sites, there was an effect of treatment (p=0.035): average BAF was higher in KER A_{REF} than KER B_{HARV} at the upstream site only (Sidak's Multiple Comparisons, p=0.026). Overall, there was no evidence (0%) of spatially cumulative trends for MeHg BAFs in paired comparisons. There was strong evidence (100%) of an effect of treatment and no evidence (0%) of an effect of site in paired comparisons.

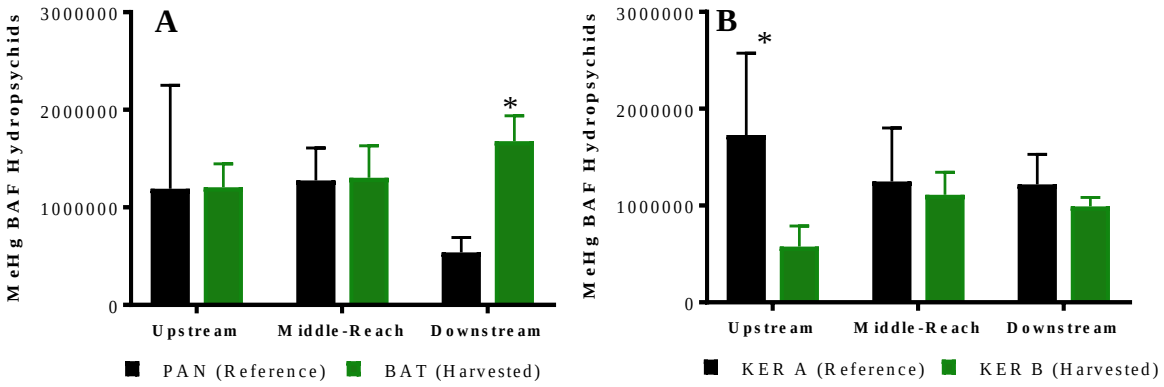

Figure 24: Average (±SD, N=3) bioaccumulation factor (BAF) for MeHg in Hydropsychids for 2017 at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT, and B) the reference catchment KER A compared to the harvested catchment KER B). The asterisk (*) indicates a significant difference in average BAF for MeHg in Hydropsychids between the two paired catchments within a site.

BMFs for MeHg in Hydropsychids were not calculated in 2016 as there was no data for MeHg in seston. Treatment and spatial trends in BMFs in 2017 were similar among sampling trips, so monthly values were averaged for the ANOVAs described below (see Appendix Table 37 for results for other dates). BMFs for MeHg in 2017 ranged from 7.3 to 58.5 across all sites (Appendix Table 37). Within the first paired-catchment comparison (PAN_{REF} vs. BAT_{HARV}) for 2017, there was an interaction between treatment and site (two-factor ANOVA, p=0.0098) for average BMF (Figure 25). There was no effect of treatment (multiple t-tests, p>0.05), but there was an effect

of site for PAN_{REF} (one-way ANOVA, p=0.032); average BMF was higher at the middle-reach site than the downstream site (Tukey's Multiple Comparisons, p=0.037). Within the second paired-catchment comparison (KER A_{REF} vs. KER B_{HARV}) for 2017, there was an interaction between treatment and site (two-factor ANOVA, p=0.033) for average BMF. Within sites, there was an effect of treatment; average BMF was higher in KER B_{HARV} than KER A_{REF} at the middle-reach site only (multiple t-tests, p=0.037). Within treatments, there was an effect of site for KER A_{REF} only (one-way ANOVA, p=0.0002); average BMF was higher at the downstream site than the upstream and middle-reach sites (Tukey's Multiple Comparisons, p=0.0007, 0.0003, respectively). Overall, there was strong evidence (100%) of spatially cumulative trends for MeHg BMFs in paired comparisons. There was some evidence (50%) of an effect of treatment and strong evidence (100%) of an effect of site in paired comparisons.

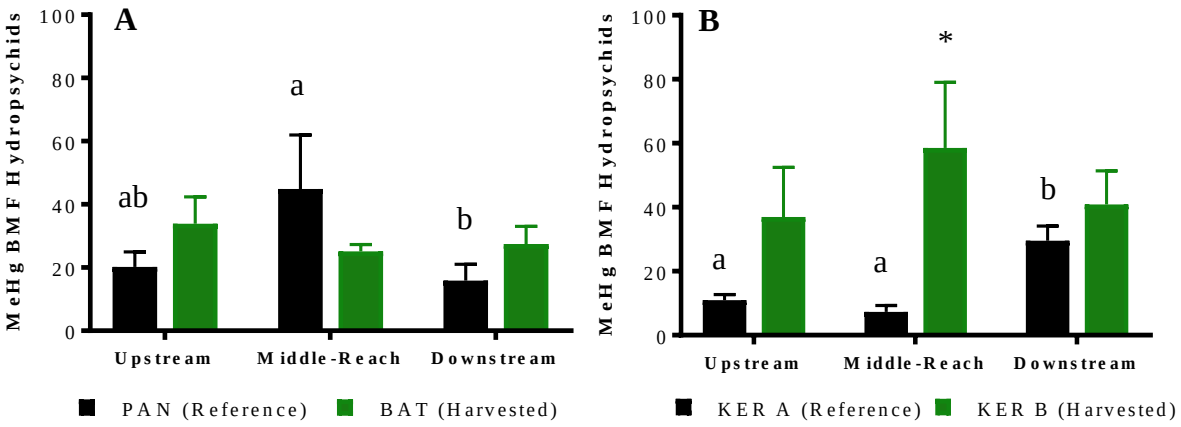

Figure 25: Average (± SD, N=3) biomagnification factor (BMF) for MeHg in Hydropsychids in 2017 at upstream, middle-reach, and downstream sites in A) the reference catchment PAN compared to the harvested catchment BAT, and B) the reference catchment KER A compared to the harvested catchment KER B). The asterisk (*) indicates a significant difference in average BMF for MeHg in Hydropsychids between the two paired catchments within a site. The letters indicate a significant difference in average BMF for MeHg in Hydropsychids between site locations within treatments.

3.2.8. Summary

The table below summarizes the strength of evidence for an interaction between treatment and site, an effect of treatment, and an effect of site, as well as the statistically significant EVs in average best-fitting linear mixed effects models, for all RVs presented above and those included in the Appendix.

Table 19: Summary table of the presence of an interaction between treatment and site, effect of treatment, and effect of site identified in ANOVA analyses, and statistically significant explanatory variables (EVs) identified in the average best-fitting model, for all response variables (RVs) examined in this section, as well as those included in the Appendix (indicated by an *). Strength of evidence (i.e., based on proportion of paired comparisons with interaction or effect present compared to absent) and/or direction of effect are indicated in brackets, where R = reference catchment, H = harvested catchment, U = upstream, M = middle-reach, and D = downstream.

Indicator	RV	Interaction?	Effect of Treatment?	Effect of Site?	Significant EVs
Water Chemistry	Conductivity	**Yes** (Some)	**Yes** (Strong; Variable)	**Yes** (Strong; D>M+U)	Treatment (+) Site (+)
	Total Nitrogen	**Yes** (Some)	**Yes** (Some; H>R)	**Yes** (Some; U>M+D)	Treatment (+)
DOM Quality	Specific UV Absorbance at 254 nm	No	No	No	Year (-) Road density (+) Interaction Treatment & Site (-)
	*Fluorescence Index	No	No	No	N/A
	*Freshness Index	No	**Yes** (Some; R>H)	**Yes** (Some; U>M)	Year (-) Fine sediment (+)

Sediment Deposition	Fine, Inorganic Deposition Per Day	**Yes** (Limited)	**Yes** (Some; R>H)	**Yes** (Strong; D> M+U)	Year (-) Site (+) % EVSA (+)
	*Coarse, Inorganic Deposition Per Day	**Yes** (Limited)	**Yes** (Limited; R>H)	**Yes** (Strong D>M+U)	Year (-) Site (+)
	% Organic Content of Fine Sediment	**Yes** (Some)	**Yes** (Strong; Variable)	**Yes** (Strong; U>M+D)	Site (-) Year (+)
	*% Organic Content of Coarse Sediment	**Yes** (Strong)	**Yes** (Some; Variable)	**Yes** (Strong; U>M+D)	Site (-) Year (+)
Leaf Litter Decomposition	% Leaf Mass Lost Per Degree Day	**Yes** (Limited)	No	**Yes** (Limited; U>M)	Year (+) % Shredders (+) Treatment (+)
	*Leaf Litter Breakdown Rate	**Yes** (Limited)	No	**Yes** (Limited; U>M)	Year (+) Treatment (+)
Leaf Litter Macroinvertebrates	Abundance	No	No	**Yes** (Limited; M+D>U)	2016: None 2017: None
	Taxonomic Richness	No	No	**Yes** (Limited; M+D>U)	2016: None 2017: Site (+)
	% Shredders	No	**Yes** (Some; H>R)	**Yes** (Limited; U>M+D)	2016: Treatment (+) 2017: Road density (+), DOM_PC2 (+)
	% Chironomidae	No	No	**Yes** (Limited; U>D)	2016: Treatment (-) 2017: Treatment (-), % EVSA (+)
	*% EPT	No	No	No	2016: None 2017: Treatment (+)
	*Margalef's Richness	No	**Yes** (Limited; H>R)	**Yes** (Limited; M+D>U)	2016: % EVSA (+) 2017: % Organic (+)
	*Shannon's Diversity Index	**Yes** (Limited)	No	**Yes** (Some; Variable)	2016: % EVSA (-), Site (+) 2017: % EVSA (-), Site (+)

Hydrogen Stable Isotopes	% Algae in Hydropsychid Diets	**Yes** (Some)	**Yes** (Some; H>R)	**Yes** (Some; Variable)	% EVSA (-)
Mercury	MeHg in Filtered Water	**Yes** (Strong)	**Yes** (Strong; H>R)	**Yes** (Strong; Variable)	Year (-) DOM_PC1 (+) % EVSA (-)
	*MeHg in Unfiltered Water	**Yes** (Strong)	**Yes** (Strong; Variable)	**Yes** (Strong; Variable)	Year (-) DOM_PC1 (+)
	MeHg in Hydropsychids	**Yes** (Strong)	**Yes** (Strong; H>R)	**Yes** (Strong; Variable)	MeHg in filtered water (+) Year (+) % EVSA (-) Treatment (+) Month (+)
	MeHg in Seston	No	No	No	N/A
	Hg(II) in Hydropsychids	**Yes** (Limited)	**Yes** (Limited; H>R)	**Yes** (Some; M+D>U)	Year (+) % EVSA (-) Treatment (-) MeHg in filtered water (+)
	*Hg(II) in Seston	No	No	No	N/A
	*% MeHg in Hydropsychids	N/A	No	No	Year (-) Interaction Treatment & Year (-) % EVSA (+) % Organic (-)
	*% MeHg in Seston	N/A	No	No	N/A
	*THg in Sediment	**Yes** (Some)	**Yes** (Strong; Variable)	**Yes** (Some; M>U>D)	N/A
	MeHg BAF for Hydropsychids	No	**Yes** (Strong; Variable)	No	N/A
	MeHg BMF for Hydropsychids	**Yes** (Strong)	**Yes** (Some; H>R)	**Yes** (Strong; Variable)	N/A

4. Discussion

4.1. Indicators of Spatially Cumulative Impacts

4.1.1. Water Chemistry

Conductivity is a surrogate for total ions in stream water. There was some evidence of spatially cumulative trends for conductivity in the paired comparisons, indicating that spatial trends in conductivity may not have followed the natural river continuum in harvested catchments (Figure 6). Generally, one would expect conductivity to increase downstream as it typically increases with stream volume (Robson et al. 1992). This was observed in my study in both reference catchments and one harvested catchment, and supported by the fact that site location was a positive predictor of conductivity in linear models (Table 5). Harvesting increased conductivity (considered an adverse treatment response due to its impacts on organism physiology and osmolarity) at an upstream site in one paired comparison (some evidence), but there was no evidence of an adverse treatment effect at downstream sites, indicating that BMPs were potentially effective at broader spatial scales. Other researchers have also noted increases in conductivity post-harvest. For example, stream water conductivity increased for two to eight years following clear-cutting and clearing of riparian forest in studies by Feller and Kimmins (1984) and Richardson and Béraud (2014), respectively. Higher conductivity may be caused by run-off from logging roads and skid trails carrying dissolved materials and ions to streams, or increased nutrient and ion supply from leaching of logging residues (Trombulak and Frissell 2000; Kreutzweiser et al. 2008b; Emilson et al. 2017). Higher conductivity after harvesting may also be related to the regeneration of forests after tree removal as younger trees and sparse or non-

vegetated patches are less efficient at intercepting precipitation; the result is greater infiltration to deeper soils where conductivity is higher, with this highly conductive water being subsequently delivered to the stream (Chang 2003; Gordon et al. 2004; Musetta-Lambert et al. 2017).

Nitrogen is a critical forest nutrient as it is a major component of protein, urea, and chlorophyll, and is essential for all organisms to grow, reproduce, and survive. There was some evidence of spatially cumulative trends for TN in the paired comparisons, indicating that spatial trends in TN may not have been following the natural river continuum in harvested catchments (Figure 7). Nitrogen cycles while it is carried downstream; its denitrification rate (i.e., the rate of anaerobic microbially-mediated reduction of nitrate to nitrogen gas, which is lost to the atmosphere) increases downstream due to higher ambient nitrate concentrations, suggesting that TN concentrations are not expected to increase downstream (Kellman and Hillaire-Marcel 1998). This is consistent within most catchments in my study, as well as the fact that site was not a significant predictor of TN in linear models (Table 6). Harvesting increased TN (considered an adverse treatment response due to stimulation of primary productivity that can cause cascading increases in biomass at higher trophic levels) at an upstream and a middle-reach site in one paired comparison (some evidence), but there was no evidence of an adverse treatment effect at the furthest downstream sites, indicating that BMPs were likely effective at broader spatial scales. Harvesting-induced impacts on TN in headwaters is in agreement with other studies that noted higher TN and nitrate (up to an order of magnitude) in clear-cut forests (Dahlgren and Driscoll 1994; Danehy et al. 2007). Greater export of nitrogen to streams with harvested

catchments can be caused by associated alterations in soil temperature, moisture, organic matter content, and uptake of nutrients, because the resulting warm, aerated soil conditions can increase microbial metabolism and therefore enhance mineralization and nitrification (Kreutzweiser et al. 2008b; Webster et al. 2015).

Overall, the observed RVs for stream water chemistry were impacted by (a) selection-based forest harvesting under Ontario BMPs (some evidence) and (b) spatial position of the sample site (some or strong evidence) (Objective 1), and there was some evidence of spatially cumulative trends in harvested catchments for both conductivity and TN (Objective 2; H_0 #1 rejected). While the lack of an adverse treatment effect at downstream sites in harvested catchments indicates that BMPs were effective at preventing impacts from forest management at broader spatial scales, the presence of an effect at the upstream and middle-reach sites indicates that these guidelines did not prevent water chemistry changes at the local or reach-level (Objective 3; H_0 #2 not rejected).

4.1.2. DOM Quality

Highly aromatic DOM (i.e., indicated by higher SUVA values) may be of lower quality/less available for microbial decomposition, which can decrease overall stream metabolism and influence in-stream nutrient cycling (Weishaar et al. 2003). There was no evidence of spatially cumulative trends for SUVA in paired catchment comparisons, indicating that spatial trends in SUVA were following the natural river continuum in harvested catchments (Figure 9). However, a negative interaction between treatment and site was a significant predictor in linear models, which may alternatively suggest evidence of spatially cumulative trends for this indicator (Table 19). There was no

difference in SUVA among site locations in either treatment, but this is contrary to what may be expected in stream systems, as high molecular weight, recalcitrant DOM is typically exported downstream and lower molecular weight, labile DOM is preferentially taken up at the headwaters (Aiken 2014). Moreover, higher SUVA values in downstream lotic waters compared to lakes have been reported, a result of surrounding peatlands and/or increased photodegradation in open waters which lowers SUVA values (Cory et al. 2007; Lescord et al. 2018). The lack of impact of harvesting at all sites is consistent with the literature (e.g., Skyllberg et al. 2009; De Wit et al. 2014; Glaz et al. 2015; Eckley et al. 2018). In addition, road density was a positive predictor of SUVA in linear models, which may be the result of altered hydrologic connectivity of the sub-catchment enhancing the delivery of terrestrially-derived DOM (i.e., DOM that is more humic, aromatic, and complex) to my study streams (McKnight et al. 2001; Roiha et al. 2012; Erdozain et al. 2018). Year was a negative predictor of SUVA, which may be related to substantially greater precipitation and run-off in September of 2016 than in 2017 (Appendix Table 5).

Overall, SUVA (an RV of DOM quality) did not appear to be affected by (a) selection-based forest harvesting under Ontario BMPs or (b) spatial position of sample site (Objective 1), and there was evidence from linear models only of spatially cumulative trends for SUVA in harvested catchments (Objective 2; H_0 #1 possibly rejected). However, there was no evidence of an adverse treatment effect on DOM quality at any spatial location, indicating that BMPs were likely effective at preventing impacts from forest management at broader spatial scales (Objective 3; H_0 #2 not rejected).

4.1.3. Sediment Deposition

Increased deposition of fine, inorganic sediments in streams of harvested catchments is commonly associated with heavy logging road construction and use (e.g., Kreutzweiser et al. 2005; Wang et al. 2013). There was limited evidence of spatially cumulative trends for fine, inorganic sediment deposition, indicating that this response variable was likely following the natural river continuum in harvested catchments (Figure 10). Generally, downstream areas are expected have higher deposition as they receive sediment inputs from both adjacent overland run-off and their headwaters (Vannote et al. 1980). Higher deposition downstream was noted in most reference catchments and some harvested catchments in the paired comparisons, and site was a positive predictor in linear models (Table 8). In my study, harvesting increased fine, inorganic sediment deposition (an adverse treatment effect due to the tendency of fine, inorganic sediments to cause scouring, clogging, and increase water turbidity) at the downstream site in one paired comparison, providing limited evidence that BMPs were not effective at broader spatial scales. However, road density was not a significant predictor of fine, inorganic sediment deposition in linear models, possibly because there was no actual construction of logging roads during the time of sampling (personal observation), as well as due to the presence of riparian buffers, which reduce soil movement and run-off velocity (Cooper et al. 1987; Lakel III et al. 2006), and other sediment abatement practices. There was a positive association with % EVSA in linear models, which is likely due to greater hydrologic connectivity, and therefore sediment transport, in areas with a higher proportion of EVSAs. This may have been exacerbated by tertiary roads and/or skid trails crossing EVSAs, although I did not measure the

degree of equipment intrusions into EVSAs. A negative effect of year was also noted, and may be related to higher degree of run-off transporting inorganic particles in September 2016 than in 2017 (Appendix Table 5).

The organic content of fine sediment transported to forest streams is a function of the structural characteristics of the stream channel, the physical properties of terrestrial organic matter, and the hydrology of the catchment and stream (Richardson et al. 2009). There was some evidence of spatially cumulative trends for organic content of fine sediment in the paired comparisons, indicating that this indicator was not following the natural river continuum in harvested catchments (Figure 11). Generally, organic content of sediment is higher in headwater streams due to higher terrestrial inputs from surrounding vegetation that are lost by breakdown or consumption downstream, and more storage by woody debris (i.e., in pools, backwaters, and debris jams) in the headwaters as wood loading decreases with channel size (Bilby and Likens 1980). This trend was generally consistent with reference catchments in my paired comparisons, as well as in linear models, where site was a negative predictor (Table 9). In the current study, harvesting increased and decreased organic content of fine sediment (both of which can be considered an adverse treatment impact due to changing the provision of organic carbon at the base of the food web) at headwater (some evidence), middle-reach (some evidence), and downstream (limited evidence) sites, indicating that BMPs were not completely effective at preventing adverse treatment impacts at multiple spatial scales. Other studies have similarly noted logging activities increasing and decreasing the transport of organic particulate matter in forest streams. Organic content of sediment may increase due to the fresh sources of terrestrial detritus from logging residues and

higher delivery in run-off, or it may decrease as vegetation removal limits the contribution of woody debris to trap and store organic matter in streams (Palviainen et al. 2004; Hassan et al. 2005). However, in my study area, neither of these mechanisms were likely to occur because the riparian buffers prevented harvesting within at least 30 m of stream edges. As discussed previously for DOM quality and inorganic sediment deposition, the positive effect of year in linear models may be related to lower precipitation or run-off in 2017, resulting in more retention of organic matter and higher % organic content at headwater sites (Appendix Table 5).

Among the RVs observed for sediment deposition, both daily inorganic deposition and organic content of fine sediment were affected by (a) selection-based forest harvesting under Ontario BMPs (limited or some evidence) and (b) spatial position of the sample site (strong evidence) (Objective 1). There was limited or some evidence of spatially cumulative trends for fine, inorganic deposition and % organic content of sediment in harvested catchments, respectively (Objective 2; H_0 #1 possibly rejected). There was limited evidence for an adverse treatment effect downstream for both sediment deposition RVs, indicating limited evidence that BMPs were not effective at broader spatial scales (Objective 3; H_0 #2 possibly rejected). There was also some evidence for an adverse treatment impact at upstream and middle-reach sites for % organic content of fine sediment, further indicating that BMPs may not have been effective at local spatial scales as well.

4.1.4. Leaf Litter Decomposition

Leaf litter is a critical source of particulate and dissolved organic matter to forest streams, but its importance generally decreases downstream where land and water are

less tightly coupled and less shading facilitates larger contributions of autotrophic production to the food web (Hynes 1970; Vannote et al. 1980). There was limited evidence of spatially cumulative trends for % leaf mass lost, indicating this indicator was likely following the natural river continuum in harvested catchments (Figure 12). One might expect lower leaf decomposition at downstream sites as there is a general shift from shredder-dominated macroinvertebrate communities to collector- and scraper-dominated communities (Vannote et al. 1980). However, this trend was not noted in reference or harvested paired comparisons in my study, perhaps because the gradient of stream size was not sufficiently large to observe changes in the functional feeding groups of macroinvertebrates. There was also no evidence of an effect of harvesting at any site, indicating that BMPs were effective at all spatial scales. Both decreases (e.g., Pozo et al. 1998; Lecerf and Richardson 2010) and increases (e.g., Benfield et al. 2001; McKie and Malmqvist 2009) in leaf decomposition have been observed in other studies of streams with managed catchments. The lack of response at my sites may be due to the relatively low intensity of selection-based harvesting compared to clear-cutting and/or the maintenance of riparian buffers. Treatment was, however, a positive predictor in the average best-fitting models for both RVs (Table 10), which could be a result of increased microbial decomposition stimulated by higher TN at some harvested sites (see above) and/or the increased activity of shredding macroinvertebrates (supported by % shredders as a positive predictor in the linear model and by higher % shredders at some harvested sites, see below). There was also a positive effect of year in linear models; yearly variation in leaf decomposition in boreal forest streams was also noted by Kreutzweiser et al. (2010).

Overall, there was (a) evidence from linear models only that % leaf mass lost (indicator of leaf decomposition) was affected by selection-based forest harvesting under Ontario BMPs, and (b) limited evidence from paired comparisons of an effect of spatial position of the sample site (Objective 1). There was also limited evidence for spatially cumulative trends in harvested catchments (Objective 2; H_0 #1 not rejected). The lack of an adverse treatment impact at the downstream sites indicates that BMPs were likely effective at preventing negative impacts of forest management at all spatial scales (Objective 3; H_0 #2 not rejected).

4.1.5. Leaf Litter Macroinvertebrates

Leaf litter macroinvertebrate communities were similar among years but did not ordinate based on treatment or site (Figure 13), suggesting that other catchment or reach factors were likely driving macroinvertebrate community assemblages in my study area. For example, in comparing macroinvertebrate NMDS plots to PCAs for water chemistry (Figure 5) and DOM quality (Figure 8), some similarities can be noted. The BAT 3 and PAN 3 sites are highly isolated in both the NMDS plots and in the PCA for water chemistry, suggesting that water chemistry may be influencing the unique community structures at these sites. In addition, the KER B_{HARV} sites clustered together on the NMDS plots and the PCA for DOM quality, suggesting that DOM quality attributes may be influencing their similar community structures. Catchment topography, surficial geology, channel substrate, riparian vegetation, stream morphology, and hydrology are also important determinants of stream macroinvertebrate community structure (e.g., Richards et al. 1996, 1997; Sandin and Johnson 2004; Petkovska and Urbanič 2015). However, the sites in my study had similar catchment characteristics; variability in

invertebrate communities may therefore have been more related to reach and patch level differences, such as local water chemistry, DOM quality, proportion of organic matter availability, and others.

The abundance and taxonomic richness of leaf litter macroinvertebrates are commonly used to observe the impacts of land use on biotic assemblages (e.g., Davis et al. 2001). There was limited evidence of spatially cumulative trends for abundance and taxonomic richness, indicating that these indicators were following the natural river continuum in harvested catchments (Figure 14 & Figure 15). Abundance and taxonomic richness of invertebrates in stream networks is dependent on many factors, including the export of invertebrates and organic matter from headwater streams (Richards et al. 1996; Wipfli et al. 2007). There was limited evidence for an effect of site in my study for both metrics, but site was a positive predictor of richness in linear models for 2017 (Table 12). There was no evidence of an effect of treatment on either metric, as in Erdozain (2018), indicating that BMPs were effective at preventing adverse treatment effects at broader spatial scale. The lack of treatment impact contrasts with, for example, Erman et al. (1977) and Newbold et al. (1980), who observed increases in abundance and diversity, and Lemly (1982) and Lenat (1979) who observed decreases in abundance and richness. The reported impacts of harvesting on invertebrate abundance and impact in these earlier studies almost certainly reflect the fact that forestry BMPs were less developed and applied in those studies than under current forestry operations.

Shredding invertebrates (e.g., *Leuctra*, *Lepidostoma*) are particularly abundant in many forested headwater streams, and are responsible for breaking down coarse, particulate organic matter (namely leaves) into finer particles that are transported

downstream, where they act as an important food source for filter-collector and detritivorous invertebrates (e.g., *Baetis*) (Vannote et al. 1980). There was limited evidence of spatially cumulative trends for leaf pack % shredders, indicating that this indicator followed the natural river continuum in harvested catchments (Figure 16). According to the River Continuum Concept, shredder abundance is expected to be highest in the headwaters (where leaves from overhanging canopy are more abundant) and decrease at downstream reaches (Vannote et al. 1980). This trend was true for only one catchment in my study, so therefore there was limited evidence for an effect of site on % shredders (Figure 16). As noted previously for leaf decomposition, the gradient in stream size used in my study may have been too small observe measurable changes in functional feeding groups, particularly within the KER A and B catchments where average difference in stream width between upstream and downstream sites was 1.2 m (Table 4). Distribution of shredders may also be influenced by other factors, such as hydrology and topography. There was, however, some evidence for an effect of treatment in my study; higher % shredders (considered an adverse treatment impact due to changing the provision of coarse organic matter downstream) at two upstream sites indicated that BMPs were not protective at the headwater scale. Treatment was also a positive predictor of % shredders in linear models (Table 13). This is consistent with the findings of Jackson et al. (2007), Medhurst et al. (2010), and Kreutzweiser et al. (2010), and is a likely result of increased terrestrial detritus delivered to the stream post-harvest. Although harvesting was excluded from within 30 m of streams, organic particles from upland site disturbance and tree removal may have been delivered to streams at or near stream crossings and/or from overland flow under higher run-off events such as

snowmelt. Shredders have also been noted to decline in harvested catchments (e.g., Kreutzweiser et al. 2008a; Klimesh et al. 2015), possibly because nutrient fluxes alter the microbial communities colonizing the leaf litter and thereby reduce leaf palatability to the shredders, or due to decreased input of particulate organic matter following vegetation removal (Pozo et al. 1998; Klimesh et al. 2015). In my study, there was also evidence for a positive association with road density (which is positively correlated with DOM_PC1) and fine sediment deposition in linear models, which supports the notion that increased shredder presence is related to higher inputs of terrestrial organic matter.

Chironomidae are known to be a disturbance-tolerant taxon due to their short generation times, rapid colonization rates, generalist feeding strategy, and ability to use fine, particulate organic matter as a habitat and food source (Hynes 1970; Vannote et al. 1980; Reid et al. 2010). There was no evidence of spatially cumulative trends for leaf pack % Chironomidae, indicating that this indicator followed the natural river continuum in harvested catchments (Figure 17). Chironomids are generalist feeders that tend to be distributed along entire stream corridors (Vannote et al. 1980). This was also true in my study, as there was limited evidence of an effect of site on % Chironomidae. There was also no evidence of an effect of treatment on % Chironomidae at any site, indicating that BMPs were likely effective at local and broader spatial scales. However, treatment was a negative predictor of % Chironomidae in linear models (Table 14). This result is contrary to what would be expected, as Chironomids are often more abundant in disturbed streams due to their ability to tolerate environmental stressors such as low oxygen levels and increased fine sediment deposition (Walshe 1947; Noel et al. 1986; Reid et al. 2010). There was a positive effect of % EVSA in linear models for 2017,

which may have influenced % Chironomidae through the delivery of fine sediment for use as a habitat or food source (Campbell and Doeg 1989). The % of deciduous tree species (correlated with % EVSA) may also have had a positive effect, but this contrasts with Noel et al. (1986) who suggest that reduced canopy cover (i.e., unlike that of a highly deciduous forest) is related to greater density of Chironomids due to higher light intensity and water temperature.

Overall, there was evidence indicating an effect of selection-based forest harvesting under Ontario BMPs on macroinvertebrate % shredders (some evidence) and % Chironomidae (linear models only), but no evidence for effects on abundance and taxonomic richness (Objective 1). There was evidence of an effect of spatial position of the sample sites on abundance and richness (limited evidence plus linear models), % shredders (some evidence), and % Chironomidae (linear models only) (Objective 1). However, there was no evidence of spatially cumulative trends for leaf pack macroinvertebrate communities in harvested catchments for any of the discussed metrics (Objective 2; H_0 #1 not rejected). Overall, there was some evidence of an adverse impact of harvesting on % shredders at the upstream site (i.e., indicating BMPs were perhaps not effective at this spatial scale), but no effect of treatment at downstream sites for all discussed metrics (Objective 3; H_0 #2 not rejected).

4.1.6. Hydrogen Stable Isotopes

The % algae in Hydropsychid diets (i.e., as determined by hydrogen stable isotope values) has important implications for growth and survival because algae is a higher quality food source and results in more efficient energy transfer to higher trophic levels than does terrestrial organic matter (Brett et al. 2009, 2017; Guo et al. 2016a). In

my study, there was up to 30% contribution of algae to the diet of Hydropsychids, adding to recent evidence that consumers use algae even in forested streams where there are high levels of allochthonous inputs and autochthony is generally considered to be low (Guo et al. 2016a; Thorp and Bowes 2017). There was some evidence from paired comparisons for spatially cumulative trends in % dietary algae, suggesting that this indicator did not follow the natural river continuum in harvested catchments (Figure 20, Appendix Table 34). A general increase in % algae used by stream consumers from upstream to downstream is expected as a more open canopy allows greater light penetration for in-stream photosynthesis (Vannote et al. 1980). In my study, there was some evidence for an effect of site, but % algae did not always increase downstream. A recent study by Jonsson et al. (2018) on the autochthony of blackfly and caddisfly larvae noted no effect of drainage area (surrogate for stream size); autochthony was instead promoted by higher lake and wetland cover in the catchment and decreased by forest cover and the presence of forestry. None of my streams originated from lakes, but there were beaver ponds interspersed throughout my catchments and those pond outlets may have contributed to the observed increases in % algae of Hydropsychid diets at some sites, regardless of treatment. There was also some evidence for an effect of treatment on % algae in the diet of Hydropsychids in my study, but the direction of the effect was variable: higher % algae at harvested sites in some comparisons, but lower in others. Lower % dietary algae (an adverse treatment impact because of algae's higher nutritional quality compared to terrestrial organic matter) was noted at upstream and downstream sites in harvested catchments (limited evidence), suggesting that BMPs were potentially not effective at either spatial scale. This is in agreement with other

studies, which have noted that % algae can decrease with a higher degree of forestry due to greater input of terrestrial organic matter from harvesting residues and run-off, or increase in clear-cut streams due to the increased presence of specialist herbivores in communities and the availability of nitrogen for algal growth (Göthe et al. 2009; Jonsson et al. 2018). There was also a negative effect of % EVSA in linear models, which may be because EVSAs increase the delivery of terrestrial organic matter and reduce relative availability of algae to consumers (Neary et al. 2009).

Overall, there was some evidence to suggest that % algae in the diet of Hydropsychids was affected by (a) selection-based forest harvesting under Ontario BMPs and (b) by spatial position (Objective 1). There was also some of evidence indicating that their % dietary algae exhibited spatially cumulative trends in harvested catchments (Objective 2; H_0 #1 rejected). The observation that there was an adverse impact on % algae consumed by Hydropsychids at an upstream and a downstream site provides limited evidence that BMPs were possibly not effective at preventing impacts from forest management at these spatial scales (Objective 3; H_0 possibly rejected).

4.1.7. Mercury (Hg)

The concentration of Hg in walleye, northern pike, and lake trout in Ontario is projected to increase within the next 30 years (Gandhi et al. 2015), and it is therefore critical to quantify the impact of various anthropogenic land uses on Hg transport to and fate in aquatic ecosystems. There was strong evidence for spatially cumulative trends in MeHg in water, indicating that MeHg in water in harvested catchments did not follow the natural river continuum (Figure 20, Appendix Table 38). Spatial variation in MeHg within a stream network is expected; Tsui et al. (2009) found that larger streams had the

highest aqueous MeHg due to the presence of methylation hotspots. MeHg may also increase downstream as a more open canopy may lead to greater rates of Hg photomethylation (O'Driscoll et al. 2004). In my study, this spatial trend was noted in the reference catchments, but the opposite trend was noted in the harvested catchments. There was also strong evidence of treatment impacts in my study, but impacts were only adverse (i.e., higher MeHg in water, due to its negative implications for health, survival, and reproduction) at upstream sites (some evidence), indicating that BMPs may not have been effective at the upstream scale. Increases in MeHg in run-off and surface water in logged catchments compared to undisturbed catchments have been well-documented (e.g., Povari et al. 2003; Eklöf et al. 2014; Olsson et al. 2017) and are commonly attributed to increased exports of DOM from logged catchments, a known carrier ligand for dissolved MeHg (Tsui and Finlay 2011). In agreement, the linear model analyses herein indicated a positive effect of DOM_PC1 (positively correlated with SUVA, Table 13) on aqueous MeHg, a relationship which has also been widely observed in the literature (e.g., Tsui and Finlay 2011; Lescord et al. 2018). Surprisingly, there was a negative effect of % EVSA in linear models; this was unexpected as EVSAs promote transport of DOM and may also represent ideal areas for methylation to occur (Bishop et al. 2009). However, % EVSA was positively correlated with % deciduous tree species in the catchment, which may better explain the negative MeHg relationship as conifers tend to accumulate Hg from the atmosphere more than deciduous trees due to their higher surface area for deposition and adsorption of atmospheric Hg, enhancing the delivery of Hg to the watershed during litterfall and throughfall (Munthe et al. 1995). However, this is likely not the case in my study, because sites with higher MeHg did not

tend to have higher % conifers (Table 1). While MeHg in surface water is influenced by soil organic matter content and Hg deposition, mobility, and methylation, there are conflicting reports of which factor is the most important determinant of MeHg concentration and subsequent accumulation in biota (Skyllberg et al. 2009; Eklöf et al. 2012). My study suggests that organic matter aromaticity appears to be a strong determinant of MeHg in stream water.

Higher MeHg in stream water often translates to higher MeHg in stream biota (Tsui and Finlay 2011; De Wit et al. 2012) and Bishop et al. (2009) estimate that forest harvesting is responsible for between one-tenth and one-quarter of Hg in fish in high-latitude, managed forest landscapes. There was strong evidence for spatially cumulative trends for MeHg in Hydropsychids, indicating that these RVs are not following the natural river continuum in harvested catchments (Figure 21). MeHg in invertebrates may be expected to increase downstream: Tsui and Finlay (2011) and Lescord et al. (2015) reported a positive relationship between catchment drainage area and MeHg in invertebrates. A strong effect of site was noted in my study, but increasing MeHg in Hydropsychids downstream was more common in reference catchments than in harvested catchments. In this study, harvesting was found to increase MeHg in Hydropsychids (an adverse treatment impact) at upstream and middle-reach sites (some evidence), indicating that BMPs were possibly not protective at these spatial scales. In addition, there was limited evidence suggesting that Hg(II) was higher in harvested catchments at the downstream site, which suggests that BMPs may not have been effective at preventing downstream accumulation in this species of Hg (Figure 22). Previous studies have found increased MeHg in biota after harvesting (Garcia and

Carignan 1999, 2000, 2005). In the linear model (Table 17), treatment (a positive predictor of MeHg in Hydropsychids) was positively correlated with WC_PC2 (positively correlated with TN), and therefore nitrogen may have also influenced MeHg in Hydropsychids; aqueous nitrogen is a strong predictor of MeHg in invertebrates, likely due to its stimulation of MeHg methylation (Lescord et al. 2015). Given that trends for MeHg in Hydropsychids were similar to those for filtered water, it is not surprising that there was a positive effect of the latter on the former. This also suggests that SUVA, which was a positive predictor of MeHg in filtered water, did not appear to be limiting Hg bioavailability to biota in these streams (Wood et al. 2011).

Conversely, I found no evidence of an effect of spatially cumulative trends, or effects of site or treatment on Hg in seston (Figure 23, Appendix Table 38). The reasons for this are unclear, but may be because invertebrates were selectively consuming particles in the bulk seston analyzed herein. There was no evidence of spatially cumulative trends for MeHg BAF in Hydropsychids, but there was for MeHg BMF in Hydropsychids (Figure 24 & Figure 25). There was also an adverse treatment (higher values) effect at the downstream and middle-reach sites for BAF and BMF, respectively, suggesting that BMPs may not be protective at these spatial scales. The consequences of increased Hg contamination caused by harvesting can influence terrestrial ecosystems as well, as larval invertebrates emerge as adults and become prey for predators on land (Speir et al. 2014).

Overall, analyses of MeHg in water and Hydropsychids gave a strong evidence of effects of (a) selection-based forest harvesting under Ontario BMPs and (b) spatial position (Objective 1), and there was strong evidence of spatially cumulative trends in

harvested catchments (Objective 2; H_0 #1 rejected for MeHg RVs). Adverse treatment impacts were not observed at downstream sites, suggesting that BMPs were potentially effective at broader spatial scales (Objective 3; H_0 #2 not rejected). There was, however, some evidence for an adverse treatment impact at upstream and middle-reach sites for MeHg in water and Hydropsychids, indicating that BMPs were perhaps not effective at local spatial scales. In addition, there was limited to some evidence for adverse treatment impacts downstream for Hg(II) in Hydropsychids and MeHg BAF, suggesting that BMPs were again possibly not protective for these metrics at broader spatial scales (Objective 3; H_0 #2 possibly rejected).

4.2. Study Limitations & Future Directions

As with all studies and despite best efforts, there were limitations in my ability to detect, quantify, and comment on spatially cumulative trends and adverse impacts from forest harvesting. Firstly, application of the study findings is limited to selection-based forest harvesting within temperate, hardwood streams of similar size, geology, vegetation, and BMPs. Within this remote forested landscape, difficulties in accessing sample sites and identifying catchment pairs limited the amount of site/treatment replication that was possible; it would have been ideal to have at least a third catchment pair with perhaps four or five sites along the stream network. Moreover, while a variety of indicators representing abiotic, biotic, and contaminant parameters of a stream ecosystem that may be impacted by forestry activities were examined, some potentially important indicators were not measured (e.g., algal biomass, substrate cover), not all indicators were adequately characterized due to a lack of time and resources, and some EVs may have been missed. For example, stream hydrology is widely known to be

influenced by forest harvesting (Bosch and Hewlett 1982; Eklöf et al. 2016), but was measured only in 2017 and only at discrete time points throughout the sampling period, making it difficult to use to detect an effect of harvesting and potentially biasing results on the basis of individual fluxes in stream velocity and/or discharge. RVs like Hg in water and biota may be influenced by riparian vegetation composition, proportion of lakes and wetlands in the catchment, extent of canopy coverage of the stream, presence of other land uses in the catchment, and so on, all of which are potential EVs that were not measured in this study. In addition, only one EV was used to describe harvesting in the linear models: % area harvested in the last five years. While this EV attempted to account for intensity of and time since harvesting events within sub-catchments, it did not provide information about the proximity of the harvesting to the study site, as well as how recent within the five-year window the harvesting occurred (i.e., as RVs respond to and recover from disturbance at varying time scales).

There were also methodological limitations to consider. Missing data for Hydropsychids, seston, and biofilm represent instances where insufficient tissue mass was collected (e.g., due to extreme rainfall events causing biota to be carried downstream). Large rain events may have also influenced the results of grab-sample water collections, such as those for water chemistry, DOM quality, hydrogen isotopes, and Hg. Among the most prominent methodological limitations were my attempts to measure hydrogen isotopes in biofilm to indicate aquatic organic material contribution to stream consumer diet, which were largely unsuccessful and resulted in the use of calculated values for algal hydrogen isotope values instead. This was likely due to the presence of terrestrial, bacterial, and fungal organisms confounding the algal signature

of the biofilm. The lipid content of food sources and consumers were also not quantified, which may have limited my ability to compare hydrogen isotope values among sites as lipids are known to be depleted in deuterium relative to other tissues (Sessions et al. 1999). Finally, the use of a two-source mixing model assumed 100% herbivory and that Hydropsychids were feeding only on two food sources, which may be unlikely given that their feeding strategy is passive and food collected in silk nets may also include small animal tissue (i.e., rendering their diet omnivorous).

Some statistical limitations and challenges also presented themselves during data analysis. As methods for the assessment of spatially cumulative impacts of resource extraction and/or land use are largely missing in the literature, there was minimal direction on statistical approaches. Herein I attempted to quantify spatially cumulative trends using both categorical (i.e., the paired catchment comparisons) and continuous (i.e., the mixed effects models) variables for the effect of treatment, site, and their interaction, but the two approaches often gave inconsistent results (see Table 19), and limited replication may have decreased my ability to find spatially cumulative trends among RVs. In addition, there was a high degree of correlation among some of the EVs used, and while the process for excluding variables was standardized and the relationships between included and excluded EVs addressed in the text, it remained difficult to know which correlated variable was more influential.

In addressing the limitations of this study there is also the opportunity to identify areas for future research. In future studies of spatially cumulative trends from forest management, it is recommended that multiple (i.e., greater than two) pairs of reference and harvested treatments be sampled. It is also recommended to include a greater spatial

gradient of sample sites (i.e., greater distance between sites) such that longitudinal impacts may be more perceptible (e.g., changes in proportion of macroinvertebrate functional feeding group). Similar studies should also be conducted in other landscapes and biomes (e.g., softwood forests, tropical regions, etc.) and in areas with different forest harvesting practices (e.g., clear-cut, stump harvest, etc.). These studies should include a wide variety of indicators, RVs, and EVs and ensure that both the structural and functional aspects of stream ecosystems are considered. Perhaps most importantly for forest management studies, the development of an integrated measure of harvesting intensity/scale, time frame, and proximity to the stream could increase the ability to attribute impacts to harvesting activities in the catchment. To address issues related to quantifying the sources of organic matter in the diet of stream consumers, lipid content or carbon-to-nitrogen ratios could be measured and accounted for, a wider variety of food sources and potential consumers could be considered (i.e., with different feeding strategies), and innovative techniques such as fatty acid analysis could be used in conjunction with isotope analysis to distinguish between aquatic and terrestrial primary production. Given the interesting results from the Hg analyses, Hg may be a potential focal point for future forest management studies in general, and may be one of the variables most suited to spatially cumulative trends assessment. To expand on the research presented in this thesis, future studies examining Hg accumulation in more complex food webs (i.e., include higher trophic levels such as foraging and piscivorous fish) are recommended along with improved methods for collecting and measuring sestonic Hg in stream water.

5. Conclusions & Management Implications

In summary, the objectives of this study were to: (1) assess and quantify indicator responses in forest streams with varying (a) selection-based forest harvesting under Ontario BMPs and (b) spatial locations of sample sites; (2) determine if indicators exhibited spatially cumulative trends between headwaters and larger downstream areas in managed catchments relative to minimally-managed catchments; and (3) appraise whether BMPs designed to minimize local or stand-level impacts were potentially effective at protecting against adverse effects at broader spatial scales. For Objective 1, a two-factor ANOVA was used to test for a significant difference in indicator responses between treatments within site location (a) and site locations within a treatment (b). For Objective 2, spatially cumulative trends were defined as a significant difference in the spatial trend of an indicator response in harvested compared to reference catchments, regardless of direction. Evidence of spatially cumulative trends was provided if a two-factor ANOVA detected a significant interaction between treatment and site location within paired-catchment comparisons. Note that these trends may or may not manifest as a magnification of the endpoint at downstream sites. For Objective 3, potential effectiveness of BMPs at broader spatial scale was evaluated based on the presence of a significant treatment effect at downstream sites in harvested catchments, where the treatment response was considered "adverse" as it had the ability to have negative implications in the ecosystem.

Among the 28 RVs tested for spatially cumulative impacts (Table 19), five gave strong evidence (i.e., interaction present in >75% of paired comparisons) of spatially cumulative trends from forest harvesting: organic content of coarse sediment, MeHg in

filtered water, MeHg in unfiltered water, MeHg in Hydropsychids, and MeHg BMF for Hydropsychids. In addition, five gave some evidence (50 to 75% of paired comparisons; conductivity, TN, organic content of fine sediment, % algae in Hydropsychid diets, and THg in sediment), six gave limited evidence (>50% of paired comparisons; fine and coarse inorganic deposition per day, % leaf mass lost, leaf litter breakdown rate, Shannon's Diversity Index, and Hg(II) in Hydropsychids), and 12 gave no evidence (DOM quality metrics, leaf litter macroinvertebrate metrics other than Shannon's Diversity Index, MeHg in seston, Hg(II) in seston, and MeHg BAF for Hydropsychids) of spatially cumulative trends. Most RVs did not have an adverse impact of treatment at the downstream site (with the exception of limited to some evidence for inorganic sediment deposition, organic content of sediment, % dietary algae, Hg(II) in Hydropsychids, and MeHg BAF in Hydropsychids), indicating that BMPs were largely effective at protecting against adverse impacts at broader spatial scales, which is promising and important information for forest managers. RVs including conductivity, TN, organic content of sediment, % dietary algae, MeHg in water, and MeHg and Hg(II) in Hydropsychids saw localized (upstream) impacts of harvesting, suggesting that BMPs were possibly not effective at the reach-level for these indicators. In addition to ANOVAs, linear mixed effects models found that % EVSA and year commonly predicted indicator responses. Results for paired-catchment comparisons and average best-fitting linear models were not always consistent, but each provided important information about indicator responses.

Overall, the results of my study suggest that while indicators may follow different spatial trends along the river continuum in harvested catchments, current FMRs

and BMPs for forest management in Canada are mostly effective at preventing adverse impacts downstream as there was generally little evidence of adverse treatment impacts at these sites. However, because the results of this study suggest that forestry activities may influence abiotic, biotic, and contaminant parameters at individual sites (commonly upstream), forest managers should consider monitoring several variables when implementing forestry practices, and perhaps revise some regulations to better protect headwater sites. In improving our understanding of how impacts from forest harvesting are behaving over space (and time, and in combination with other stressors) in stream networks, mandated regulations and voluntary guidelines can be accordingly revised to better protect aquatic ecosystems for both human and wildlife uses. For example, Bishop et al. (2009) and Hsu-Kim et al. (2013) provide recommendations to reduce the impacts of forestry on Hg in streams, primarily focused on reducing Hg mobilization and methylation by minimizing machinery damage of soils, promoting rapid re-vegetation, avoiding stream-side burning of forest residues, and preventing creation of standing water pools where methylation can occur.

The assessment of spatially cumulative trends will also help managers and scientists to predict changes in forest ecosystems as forestry and climate change are expected to intensify in the near future (Price et al. 2013; Creed et al. 2016). Rising temperatures (4 to 5°C predicted in Canada's boreal zone) could cause tipping points in the state of forest ecosystems through permafrost decreasing, release of greenhouse gases increasing, forest fires intensifying, forest pests migrating northwards, hydrologic fluxes become more flashy, and others (Price et al. 2013). Such effects are predicted to change forest ecosystem services, such as decreased regulation of water flow and

quality, decreased resistance to abiotic and biotic hazards, increased or decreased climate regulation, and others (Pohjanmies et al. 2017). This information is highly relevant as forest sector competitiveness, social license to operate, and third-party certification are being increasingly linked to environmental performance. Overall, my study presents a case for the need to consider spatially cumulative trends of forestry operations in stream networks in an effort to maintain healthy forests and their aquatic ecosystem services for generations to come.

6. Bibliography

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Appendix I

A.1. Stream Temperature

Appendix Table 1: Temperature logger deployment and removal dates for every sample site in 2016.

Site Name	Temperature Logger Deployment Date	Temperature Logger Removal Date
PAN 3	01/07/2016	28/10/2016
PAN 2	01/07/2016	28/10/2016
PAN 1	01/07/2016	28/10/2016
BAT 3	01/07/2016	27/10/2016
BAT 2	01/07/2016	27/10/2016
BAT 1	01/07/2016	27/10/2016
KER 3	01/07/2016	26/10/2016
KER 2	01/07/2016	26/10/2016
KER 1	01/07/2016	25/10/2916
KER 6	01/07/2016	26/10/2016
KER 5	01/07/2016	26/10/2016
KER 4	01/07/2016	25/10/2016

Appendix Table 2: Temperature logger deployment and removal dates for every sample site in 2017.

Site Name	Temperature Logger Deployment Date	Temperature Logger Removal Date
PAN 3	06/22/2017	23/10/2017
PAN 2	06/27/2017	23/10/2017
PAN 1	06/22/2017	23/10/2017
BAT 3	06/22/2017	27/10/2017
BAT 2	06/22/2017	27/10/2017
BAT 2new*	07/21/2017	27/10/2017
BAT 1	06/22/2017	27/10/2017
KER 3	06/22/2017	25/10/2017
KER 2	06/22/2017	25/10/2017
KER 1	06/22/2017	26/10/2017
KER 6	06/22/2017	25/10/2017
KER 5	06/22/2017	26/10/2017
KER 4	06/22/2017	26/10/2017

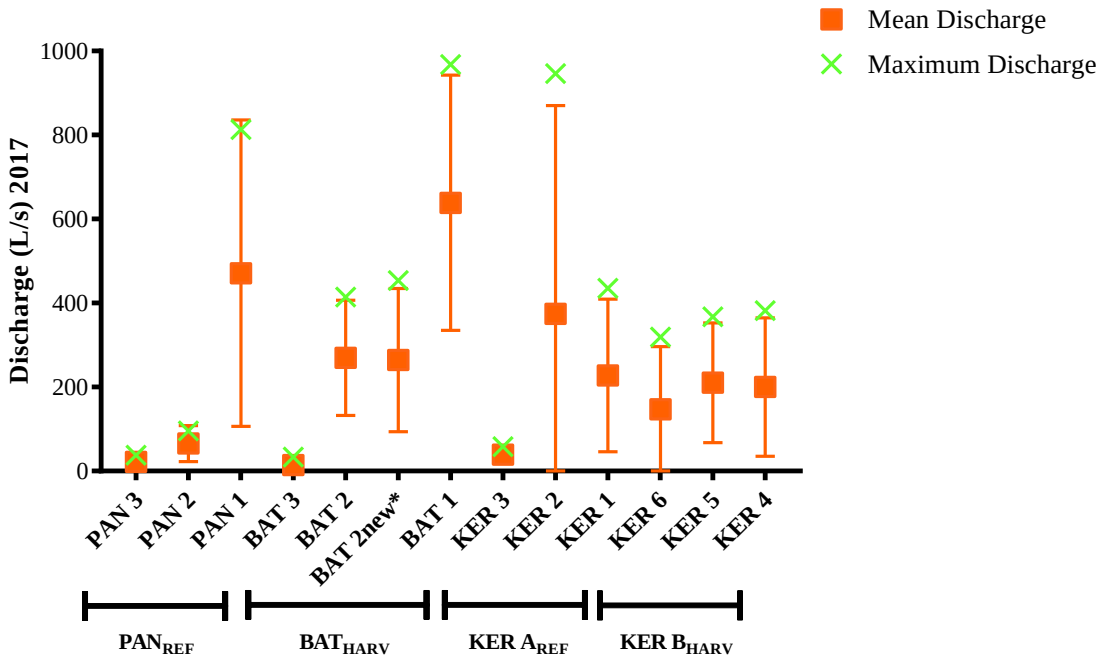
Appendix Table 3: Monthly average (± SD), maximum, and minimum stream temperature, as well as monthly degree days, at all study sites for duration of temperature logger deployment (Appendix Table 1) in 2016. Note that no temperature data was recorded for PAN 3, PAN 1, and BAT 2 because temperature loggers were lost or dead upon retrieval.

Site Name	July Avg (± SD, °C)	July Max (°C)	July Min (°C)	July Degree Days	Aug Avg (± SD, °C)	Aug Max (°C)	Aug Min (°C)	Aug Degree Days	Sept Avg (± SD, °C)	Sept Max (°C)	Sept Min (°C)	Sept Degree Days	Oct Avg (± SD, °C)	Oct Max (°C)	Oct Min (°C)	Oct Degree Days
PAN 3	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data
PAN 2	18.47 (1.57)	21.92	14.31	572.53	18.53 (1.12)	21.51	15.99	574.67	16.75 (1.57)	20.60	13.83	572.53	11.66 (2.50)	15.80	6.41	290.67
PAN 1	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data
BAT 3	13.80 (1.25)	16.46	10.74	427.99	14.73 (0.79)	17.13	12.65	456.61	13.07 (1.34)	16.30	9.78	427.99	9.85 (1.89)	13.69	5.26	245.08
BAT 2	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data	No Data
BAT 1	16.87 (1.69)	20.70	12.82	523.11	16.25 (1.46)	20.29	12.82	503.79	14.10 (1.76)	18.46	9.71	523.11	9.87 (2.08)	13.79	4.71	245.37
KER 3	13.70 (0.91)	15.53	11.66	424.78	12.87 (1.03)	15.22	11.32	398.83	14.22 (1.37)	16.20	11.30	424.78	11.78 (1.42)	13.98	9.29	293.97
KER 2	16.40 (1.67)	20.46	12.61	508.68	16.41 (1.22)	19.10	13.21	508.76	14.43 (1.57)	18.13	10.61	508.68	10.57 (2.23)	15.15	5.28	263.15
KER 1	16.96 (1.70)	21.15	12.87	525.92	17.47 (1.53)	21.37	14.00	541.73	14.50 (1.41)	17.49	11.03	525.92	10.59 (2.20)	14.58	5.02	263.34
KER 6	17.52 (1.17)	20.08	13.76	530.79	17.25 (1.12)	20.08	13.76	534.60	15.32 (1.55)	18.58	12.03	530.79	10.91 (2.47)	15.37	4.45	271.23
KER 5	15.04 (2.30)	20.65	10.10	466.50	13.54 (2.28)	19.13	8.67	419.48	14.09 (1.87)	18.06	9.88	466.50	10.50 (2.31)	15.03	4.71	261.19
KER 4	17.59 (2.36)	23.57	12.63	545.60	17.75 (2.22)	24.12	12.92	550.46	14.24 (1.75)	17.75	9.81	545.60	10.21 (2.27)	14.53	4.71	253.94

Appendix Table 4: Monthly average (± SD), maximum, and minimum stream temperature, as well as monthly degree days, at all study sites for duration of temperature logger deployment (Appendix Table 2) in 2017.

Site Name	June Avg (± SD, °C)	June Max (°C)	June Min (°C)	June Degree Days	July Avg (± SD, °C)	July Max (°C)	July Min (°C)	July Degree Days	Aug Avg (± SD, °C)	Aug Max (°C)	Aug Min (°C)	Aug Degree Days	Sept Avg (± SD, °C)	Sept Max (°C)	Sept Min (°C)	Sept Degree Days	Oct Avg (± SD, °C)	Oct Max (°C)	Oct Min (°C)	Oct Degree Days
PAN 3	11.43 (0.89)	13.40	9.30	102.68	12.87 (1.19)	15.90	10.50	398.00	13.21 (1.15)	16.40	10.90	409.65	11.59 (1.41)	13.70	9.50	347.60	10.12 (1.41)	13.10	10.12	232.53
PAN 2	15.64 (1.11)	17.90	12.90	62.32	18.38 (1.62)	22.50	14.50	567.39	17.11 (1.56)	22.40	13.50	529.19	15.30 (2.48)	21.10	9.80	458.36	11.56 (1.86)	16.10	6.70	265.66
PAN 1	14.08 (0.96)	16.50	11.40	126.45	16.32 (1.48)	19.60	12.80	504.19	15.59 (1.41)	20.00	12.10	482.21	13.73 (2.04)	18.00	9.40	411.84	10.59 (1.78)	14.20	6.90	243.56
BAT 3	11.09 (0.77)	12.50	9.10	99.45	12.51 (0.79)	14.30	10.30	386.56	12.60 (0.96)	14.70	10.00	389.82	11.40 (1.64)	14.90	8.30	341.87	9.10 (1.70)	12.80	5.00	341.87
BAT 2	12.77 (0.92)	14.80	10.30	114.52	14.44 (1.35)	17.70	11.10	445.35	13.63 (1.22)	16.90	10.50	421.07	12.23 (1.30)	15.10	9.00	366.57	9.31 (1.61)	13.20	5.80	250.43
BAT 2new*		No data	No data	No data	16.89 (1.99)	21.70	12.80	184.44	15.58 (2.02)	21.50	10.70	480.03	13.84 (2.28)	19.50	7.80	412.15	9.67 (2.37)	16.10	4.80	258.57
BAT 1	13.40 (1.10)	15.70	10.10	120.13	15.29 (1.53)	19.10	11.70	471.09	14.24 (1.60)	18.90	10.20	439.41	12.36 (1.74)	16.50	8.10	370.21	9.19 (1.90)	13.80	5.50	246.78
KER 3	13.66 (1.02)	16.37	11.42	122.90	16.56 (1.58)	20.60	13.11	513.35	16.97 (1.37)	19.70	11.86	526.17	15.29 (1.91)	19.70	11.86	458.68	12.06 (2.07)	17.56	-2.45	297.64
KER 2	12.86 (0.89)	15.32	10.88	115.76	14.07 (1.07)	16.58	11.69	436.16	14.43 (0.88)	16.82	12.20	447.25	13.18 (1.72)	16.87	9.78	395.38	10.82 (1.44)	14.05	7.87	270.00
KER 1	13.47 (1.04)	16.42	11.05	121.23	15.28 (1.27)	18.41	12.34	473.79	15.00 (1.05)	17.58	12.39	464.97	13.46 (1.76)	17.23	9.88	403.68	10.68 (1.58)	14.03	7.34	276.92
KER 6	15.30 (1.08)	17.63	12.00	137.72	17.55 (1.84)	22.11	13.98	544.19	16.91 (1.74)	21.53	12.27	524.28	14.74 (2.29)	20.03	9.26	442.12	11.00 (2.06)	15.61	6.13	274.52
KER 5	13.79 (1.52)	17.39	10.69	124.13	15.80 (1.51)	19.89	12.80	489.86	13.12 (1.82)	17.03	8.72	477.82	13.12 (1.82)	17.03	8.72	393.75	10.08 (1.84)	14.29	5.39	260.39
KER 4	13.89 (1.26)	17.15	11.13	125.05	15.43 (1.36)	19.34	12.39	478.28	14.71 (1.05)	17.03	11.59	455.99	12.90 (1.59)	16.56	9.81	387.11	9.82 (1.64)	13.04	4.97	253.27

A.2. Stream Hydrology



Appendix Figure 1: Mean (SD, N=3) and maximum discharge (green, maximum of singular measurements in August, September and October) for all study sites in 2017. Note that no discharge measurements were taken for site PAN 2 in August due to adverse weather conditions.

Appendix Table 5: Mean air temperature (°C) and precipitation (mm) for months in which sampling took place, as obtained from Environment & Climate Change Canada's monthly climate summaries database (Sault Ste. Marie station).

Date	Air Temperature (°C)	Precipitation (mm)
September 2016	16.2	166.0
October 2016	8.4	110.3
August 2017	16.5	122.6
September 2017	15.4	37.4
October 2017	9.5	174.0

A.3. Water Chemistry

Appendix Table 6: Water chemistry parameters measured and number of measurements for all sites excluding BAT 2new*. Some parameters have fewer measurements because they were below instrument limit of detection for certain sampling periods. Note that Cl, SRP, Cd, Ni, and Pb are not reported for any sites because they were consistently below instrument quantification limit.

Water Chemistry Parameter	**N** (number of measurements included in average for all sites besides BAT 2new*)	**Details of N**
pH	5	Aug & Sept 2016; Aug, Sept, Oct 2017
Conductivity (umho/cm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
Alkalinity (meq/L)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
Ca (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
K (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
Mg (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
Na (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
SO_4 (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
SiO_2 (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
NO_2+NO_3 (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
NH_4 (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
TOC (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
TIC (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
TP (ppm)	3	Aug, Sept, Oct 2017
TN (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
Al (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
Fe (ppm)	5	Aug & Sept 2016; Aug, Sept, Oct 2017
Mn (ppm)	4	Aug & Sept 2016; Aug, Sept 2017
Zn (ppm)	3	Aug, Sept, Oct 2017
Cu (ppm)	3	Aug, Sept, Oct 2017

Appendix Table 7: Water chemistry parameters measured and number of measurements for BAT 2new*. Some parameters have fewer measurements because they were below instrument limit of detection for certain sampling periods. Note that Cl, SRP, Cd, Ni, and Pb are not reported for any sites because they were consistently below instrument quantification limit.

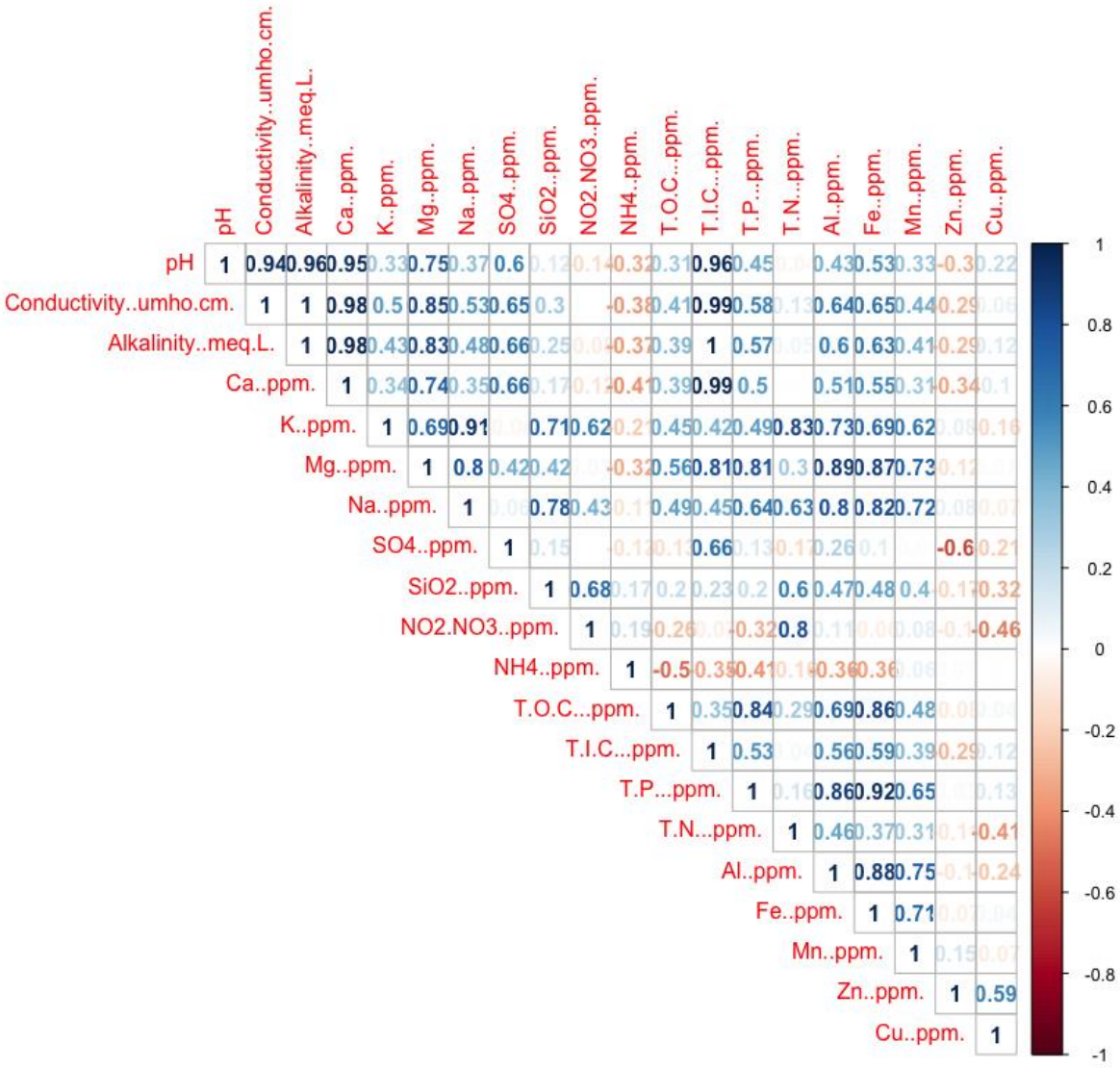
Water Chemistry Parameter	**N** (number of measurements included in average for site BAT 2new*)	**Details of N**
pH	3	Aug, Sept, Oct 2017
Conductivity (umho/cm)	3	Aug, Sept, Oct 2017
Alkalinity (meq/L)	3	Aug, Sept, Oct 2017
Ca (ppm)	3	Aug, Sept, Oct 2017
K (ppm)	3	Aug, Sept, Oct 2017
Mg (ppm)	3	Aug, Sept, Oct 2017
Na (ppm)	3	Aug, Sept, Oct 2017
SO_4 (ppm)	3	Aug, Sept, Oct 2017
SiO_2 (ppm)	3	Aug, Sept, Oct 2017
NO_2+NO_3 (ppm)	3	Aug, Sept, Oct 2017
NH_4 (ppm)	3	Aug, Sept, Oct 2017
TOC (ppm)	3	Aug, Sept, Oct 2017
TIC (ppm)	3	Aug, Sept, Oct 2017
TP (ppm)	3	Aug, Sept, Oct 2017
TN (ppm)	3	Aug, Sept, Oct 2017
Al (ppm)	3	Aug, Sept, Oct 2017
Fe (ppm)	3	Aug, Sept, Oct 2017
Mn (ppm)	2	Aug & Sept 2017
Zn (ppm)	3	Aug, Sept, Oct 2017
Cu (ppm)	3	Aug, Sept, Oct 2017

Appendix Table 8: Raw data for water chemistry parameters for sampling in September and October 2016 and in August, September, and October 2017. Values in bold were below instrument limit of detection.

Site	pH	Conductivity (umho/cm)	Alkalinity (meq/L)	Ca (ppm)	K (ppm)	Mg (ppm)	Na (ppm)	SO4 (ppm)	Cl (ppm)	SiO2 (ppm)	NO2+NO3 (ppm)	NH4 (ppm)	T.O.C. (ppm)	T.I.C. (ppm)	SRP (ppm)	T.P. (ppm)	T.N. (ppm)	Al (ppm)	Fe (ppm)	Mn (ppm)	Zn (ppm)	Cd (ppm)	Cu (ppm)	Ni (ppm)	Pb (ppm)
September 2016																									
PAN 3	6.40	13.36	0.06	1.76	0.24	0.24	0.47	1.66	**0.10**	3.86	0.04	0.02	6.81	0.73	**0.00**	**0.01**	0.39	0.21	0.05	0.00	**0.01**	**0.00**	**0.01**	**0.00**	**0.00**
PAN 2	7.00	24.00	0.16	3.21	0.30	0.49	0.67	1.88	**0.16**	2.97	0.02	0.01	4.71	2.18	**0.00**	**0.01**	0.25	0.05	0.07	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
PAN 1	7.08	32.50	0.22	4.64	0.35	0.73	0.84	2.19	0.21	5.71	0.04	0.01	9.13	2.83	**0.00**	**0.01**	0.40	0.14	0.21	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
BAT 3	6.91	30.00	0.19	3.59	0.75	0.62	1.07	1.81	0.18	7.63	0.46	0.01	6.59	2.27	**0.00**	**0.01**	0.71	0.16	0.14	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
BAT 2	7.04	30.20	0.21	3.67	0.32	0.67	0.92	2.28	0.25	7.37	0.25	0.02	3.80	2.59	**0.00**	**0.00**	0.40	0.07	0.08	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
BAT 1	6.95	25.60	0.15	3.40	0.35	0.60	0.82	2.21	0.23	5.77	0.19	0.00	6.43	1.87	**0.00**	**0.01**	0.46	0.10	0.09	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
KER 3	7.04	24.00	0.16	3.30	0.23	0.50	0.57	2.12	**0.15**	1.04	0.02	0.01	2.83	2.24	**0.00**	**0.00**	0.16	0.01	0.01	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
KER 2	7.29	48.80	0.39	7.85	0.23	0.66	0.60	2.89	**0.16**	3.72	0.08	0.01	5.89	4.90	**0.00**	**0.00**	0.32	0.05	0.07	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
KER 1	7.34	52.00	0.43	8.23	0.34	0.95	0.74	2.91	**0.16**	4.74	0.09	0.01	5.61	5.14	**0.00**	**0.01**	0.32	0.18	0.17	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
KER 6	7.00	30.90	0.19	4.40	0.36	0.84	0.87	2.48	**0.12**	4.60	0.02	0.01	13.62	2.19	**0.00**	**0.01**	0.54	0.20	0.40	0.01	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
KER 5	7.23	45.50	0.33	6.34	0.52	1.23	1.12	2.61	0.43	6.09	0.11	0.01	10.35	4.03	**0.00**	**0.01**	0.54	0.28	0.38	0.01	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
KER 4	7.34	55.70	0.45	7.90	0.58	1.69	1.26	2.68	0.41	6.59	0.13	0.01	9.74	5.04	**0.00**	**0.01**	0.52	0.35	0.45	0.01	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
October 2016																									
PAN 3	6.69	17.02	0.07	2.04	0.18	0.30	0.57	2.19	**0.14**	4.90	0.11	0.01	3.70	1.04	**0.00**	**0.00**	0.16	0.08	0.03	0.00	**0.01**	**0.00**	**0.00**	**0.00**	**0.00**
PAN 2	7.01	23.80	0.16	3.13	0.32	0.49	0.70	2.02	0.19	3.96	0.05	0.00	4.49	2.06	**0.00**	**0.00**	0.20	0.06	0.04	0.00	**0.01**	**0.00**	**0.00**	**0.00**	**0.00**
PAN 1	7.15	32.20	0.24	4.38	0.29	0.69	0.89	2.22	0.22	6.82	0.06	0.00	5.41	2.93	**0.00**	**0.00**	0.22	0.08	0.13	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
BAT 3	6.92	26.50	0.14	2.99	0.58	0.53	0.96	1.93	**0.18**	7.90	0.58	0.01	4.17	1.85	**0.00**	**0.00**	0.67	0.16	0.14	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
BAT 2	7.04	27.80	0.18	3.29	0.31	0.60	0.85	2.26	0.19	7.69	0.36	0.02	1.93	2.33	**0.00**	**0.00**	0.38	0.07	0.08	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
BAT 1	7.03	26.20	0.17	3.24	0.33	0.61	0.86	2.27	0.21	7.01	0.33	0.02	2.45	2.10	**0.00**	**0.00**	0.40	0.10	0.09	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
KER 3	7.05	24.30	0.16	3.47	0.23	0.52	0.56	2.04	**0.15**	1.14	0.01	0.00	3.51	2.22	**0.00**	**0.00**	0.16	0.02	0.02	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
KER 2	7.29	41.90	0.34	6.54	0.23	0.61	0.61	2.37	**0.17**	4.86	0.08	0.00	4.20	4.10	**0.00**	**0.00**	0.21	0.04	0.06	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
KER 1	7.36	46.80	0.39	7.12	0.30	0.87	0.74	2.41	0.18	5.72	0.11	0.00	4.49	4.62	**0.00**	**0.00**	0.22	0.10	0.10	0.00	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
KER 6	6.96	24.20	0.16	3.68	0.44	0.73	0.91	1.95	**0.16**	6.19	0.04	0.01	8.48	1.83	**0.00**	**0.01**	0.30	0.18	0.18	0.00	**0.01**	**0.00**	**0.00**	**0.00**	**0.00**
KER 5	7.24	42.60	0.32	5.77	0.43	1.16	1.16	2.26	0.57	7.00	0.14	0.00	6.47	3.87	**0.00**	**0.00**	0.31	0.16	0.18	0.01	**0.01**	**0.00**	**0.00**	**0.00**	**0.00**
KER 4	7.37	52.00	0.43	6.84	0.47	1.51	1.21	2.31	0.46	7.69	0.15	0.00	6.15	5.04	**0.00**	**0.01**	0.34	0.18	0.20	0.01	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**

August 2017																									
PAN 3	6.90	17.30	0.10	2.35	0.18	0.31	0.67	1.77	**0.09**	4.42	0.14	0.00	4.03	1.26	**0.00**	0.01	0.33	0.10	0.06	0.00	0.01	**0.00**	0.00	0.00	**0.00**
PAN 2	7.10	25.50	0.19	3.95	0.25	0.57	0.83	1.38	**0.16**	3.08	**0.03**	0.02	4.43	2.30	**0.00**	0.00	0.24	0.06	0.15	0.01	0.00	**0.00**	0.01	0.00	**0.00**
PAN 1	7.23	38.10	0.33	5.91	0.31	0.89	1.17	1.53	0.21	6.80	0.07	0.02	6.42	3.76	0.00	0.01	0.31	0.09	0.31	0.00	0.00	**0.00**	0.01	0.00	**0.00**
BAT 3	7.23	34.70	0.23	4.38	0.52	0.74	1.34	1.64	**0.17**	8.54	0.54	0.00	4.20	2.97	**0.00**	0.00	0.63	0.08	0.14	0.00	0.01	**0.00**	0.00	0.00	**0.00**
BAT 2	7.22	27.50	0.19	3.51	0.31	0.62	0.92	1.88	**0.17**	6.96	0.22	0.01	2.36	2.59	**0.00**	0.00	0.31	0.05	0.08	0.00	0.00	**0.00**	0.00	**0.00**	**0.00**
BAT 2new*	7.21	26.90	0.19	3.53	0.31	0.63	0.95	1.83	**0.16**	6.45	0.17	0.01	2.21	2.30	**0.00**	0.00	0.28	0.04	0.08	0.01	0.00	**0.00**	0.00	0.00	**0.00**
BAT 1	7.29	28.10	0.18	3.50	0.30	0.62	0.91	2.00	0.19	6.71	0.26	0.01	2.29	2.54	**0.00**	0.00	0.33	0.03	0.05	0.00	0.00	**0.00**	0.00	0.00	**0.00**
KER 3	7.22	23.30	0.16	3.25	0.21	0.48	0.58	1.73	**0.15**	0.65	0.05	0.01	3.24	2.08	**0.00**	0.00	0.72	0.01	0.03	0.00	0.01	**0.00**	0.00	0.00	**0.00**
KER 2	7.53	46.50	0.38	7.63	0.21	0.67	0.75	1.97	**0.17**	4.39	0.13	0.00	3.36	4.52	**0.00**	0.00	0.25	0.04	0.08	0.00	0.00	**0.00**	0.00	**0.00**	**0.00**
KER 1	7.57	53.30	0.45	8.11	0.29	0.98	0.90	2.02	**0.15**	5.51	0.18	0.01	3.63	5.52	**0.00**	0.00	0.30	0.11	0.13	0.00	0.00	**0.00**	0.00	0.00	**0.00**
KER 6	7.25	26.80	0.20	3.86	0.19	0.71	0.86	1.23	**0.07**	4.96	0.07	0.00	7.85	2.40	0.00	0.01	0.39	0.14	0.25	0.01	0.00	**0.00**	0.00	0.00	**0.00**
KER 5	7.48	52.30	0.42	6.97	0.40	1.45	1.36	1.84	0.69	6.75	0.18	0.01	6.03	4.91	0.00	0.01	0.38	0.30	0.33	0.01	0.00	**0.00**	0.00	0.00	**0.00**
KER 4	7.62	60.10	0.50	8.16	0.46	1.74	1.39	1.87	0.54	7.14	0.19	0.01	6.03	5.94	0.00	0.01	0.40	0.29	0.31	0.01	0.00	**0.00**	0.00	0.00	**0.00**
September 2017																									
PAN 3	6.95	21.90	0.13	2.76	0.16	0.40	0.94	2.40	**0.12**	6.49	0.20	0.01	1.96	1.23	0.00	0.00	0.26	0.05	0.04	0.00	0.00	**0.00**	0.00	0.00	**0.00**
PAN 2	7.13	26.20	0.19	3.74	0.24	0.58	0.86	1.60	**0.17**	2.84	0.04	0.01	3.61	2.37	**0.00**	0.00	0.21	0.03	0.08	0.01	0.00	**0.00**	0.00	0.00	**0.00**
PAN 1	7.36	40.10	0.33	5.62	0.30	0.87	1.23	1.76	0.22	6.75	0.08	0.01	4.78	3.91	0.00	0.01	0.28	0.06	0.27	0.00	0.00	**0.00**	0.00	0.00	**0.00**
BAT 3	7.09	33.80	0.23	4.42	0.47	0.75	1.36	1.68	**0.15**	8.68	0.52	0.02	3.44	2.88	0.00	0.00	0.62	0.08	0.11	0.01	0.00	**0.00**	0.00	0.00	**0.00**
BAT 2	7.15	28.80	0.21	3.87	0.33	0.70	1.07	2.05	**0.18**	7.43	0.24	0.02	1.56	2.59	**0.00**	0.00	0.29	0.03	0.06	0.00	0.00	**0.00**	0.00	**0.00**	**0.00**
BAT 2new*	7.02	24.90	0.17	3.21	0.32	0.59	0.98	1.76	**0.15**	6.25	0.18	0.01	2.60	2.18	**0.00**	0.00	0.27	0.05	0.11	0.01	0.01	**0.00**	0.01	0.00	**0.00**
BAT 1	7.14	27.60	0.22	3.74	0.32	0.66	1.03	2.00	0.19	6.93	0.27	0.01	2.21	2.36	**0.00**	0.00	0.34	0.03	0.08	0.00	0.00	**0.00**	0.00	0.00	**0.00**
KER 3	7.09	22.80	0.16	3.33	0.23	0.52	0.63	1.72	**0.13**	0.80	0.05	0.02	3.47	1.92	**0.00**	0.00	0.20	0.02	0.03	0.00	0.00	**0.00**	0.00	0.00	**0.00**
KER 2	7.39	47.50	0.40	8.16	0.37	0.70	0.86	1.76	**0.14**	4.95	0.11	0.01	4.54	4.78	**0.00**	0.00	0.26	0.04	0.10	0.00	0.01	**0.00**	0.00	0.00	**0.00**
KER 1	7.43	53.60	0.45	9.03	0.31	1.09	0.98	1.92	**0.16**	6.09	0.15	0.03	4.09	5.57	0.00	0.00	0.27	0.12	0.18	0.01	0.00	**0.00**	0.00	0.00	**0.00**
KER 6	7.13	27.50	0.21	4.08	0.24	0.80	1.01	1.17	**0.09**	5.33	0.05	0.02	8.13	2.44	0.00	0.01	0.35	0.17	0.28	0.01	0.00	**0.00**	0.00	0.00	**0.00**
KER 5	7.28	47.90	0.39	6.93	0.42	1.43	1.40	1.65	0.51	6.75	0.14	0.01	6.40	4.64	0.00	0.01	0.36	0.24	0.33	0.01	0.01	**0.00**	0.00	0.00	**0.00**
KER 4	7.54	69.20	0.62	9.40	0.59	2.18	1.75	2.19	0.73	7.82	0.23	0.01	4.42	7.01	0.00	0.01	0.39	0.42	0.43	0.01	0.00	**0.00**	0.00	0.00	**0.00**

October 2017																								
PAN 3	6.87	17.78	0.10	2.04	0.36	0.33	0.71	2.00	**0.17**	4.63	**0.03**	0.01	3.93	1.20	**0.00**	0.00	0.18	0.08	0.05	**0.00**	0.01	**0.00**	0.00	0.00	**0.00**
PAN 2	7.12	25.70	0.18	3.52	0.40	0.55	0.79	1.50	0.28	3.49	0.05	0.01	5.45	2.26	**0.00**	0.01	0.22	0.06	0.09	**0.00**	0.02	**0.00**	0.01	0.00	**0.00**
PAN 1	7.26	36.80	0.28	5.16	0.59	0.85	1.12	1.77	0.38	6.99	0.06	0.00	7.39	3.34	0.00	0.01	0.28	0.09	0.30	**0.00**	0.01	**0.00**	0.01	0.00	0.00
BAT 3	6.79	20.10	0.10	2.33	0.60	0.41	0.79	1.55	0.19	6.95	0.30	0.01	5.73	1.37	**0.00**	0.00	0.41	0.16	0.11	0.00	0.01	**0.00**	0.01	0.00	**0.00**
BAT 2	6.95	23.00	0.15	2.94	0.40	0.54	0.86	1.74	0.19	6.99	0.21	0.01	4.37	1.94	**0.00**	0.00	0.28	0.10	0.11	**0.00**	0.01	**0.00**	0.01	**0.00**	**0.00**
BAT 2new*	6.85	20.10	0.13	2.59	0.41	0.48	0.79	1.58	**0.18**	5.96	0.17	0.01	5.06	1.65	**0.00**	0.00	0.28	0.10	0.08	**0.00**	0.01	**0.00**	0.00	**0.00**	**0.00**
Bat 1	7.00	21.00	0.12	2.58	0.40	0.47	0.76	1.65	0.18	5.96	0.18	0.01	4.90	1.50	0.00	0.00	0.31	0.09	0.08	**0.00**	0.00	**0.00**	0.00	**0.00**	**0.00**
KER 3	7.09	24.00	0.16	3.38	0.31	0.52	0.63	1.77	**0.15**	1.90	0.10	0.01	4.07	2.17	**0.00**	0.00	0.24	0.03	0.05	0.00	0.01	**0.00**	0.01	**0.00**	**0.00**
KER 2	7.31	41.40	0.33	7.19	0.41	0.58	0.60	1.74	0.20	4.64	0.12	0.01	7.57	4.02	**0.00**	0.00	0.31	0.09	0.14	**0.00**	0.01	**0.00**	0.01	0.00	0.00
KER 1	7.39	45.50	0.37	7.41	0.45	0.83	0.76	1.86	0.20	5.56	0.15	0.01	6.34	4.42	**0.00**	0.01	0.30	0.18	0.17	0.00	0.00	**0.00**	0.00	0.00	**0.00**
KER 6	6.84	22.30	0.14	2.92	0.65	0.60	0.84	1.43	0.24	5.62	0.09	0.01	10.16	1.43	0.00	0.01	0.37	0.19	0.24	0.00	0.01	**0.00**	0.01	0.00	0.00
KER 5	7.12	32.90	0.23	4.50	0.66	0.90	0.98	1.64	0.34	6.48	0.20	0.01	8.92	2.69	0.00	0.01	0.41	0.29	0.28	0.00	0.00	**0.00**	0.00	0.00	**0.00**
KER 4	7.30	41.40	0.32	5.72	0.73	1.27	1.09	1.71	0.33	7.17	0.19	0.01	8.42	3.69	0.00	0.01	0.41	0.36	0.36	0.00	0.01	**0.00**	0.01	0.00	0.00



Appendix Figure 2: Correlation matrix for all water chemistry parameters, averaged for all collections in 2016 (N=2 for all sites except BAT 2new*, where N=0) and 2017 (N=3).

Appendix Table 9: Pearson's Correlation Coefficients among all water chemistry parameters averaged for all collections in 2016 (N=2 for all sites except BAT 2new*, where N=0) and 2017 (N=3) and the first two Principal Component Analysis (PCA) axes (WC_PC1 and WC_PC2).

Water Chemistry Parameter	WC_PC1	WC_PC2
pH	-0.78	-0.51
Conductivity (µmho/cm)	-0.90	-0.35
Alkalinity (meq/L)	-0.88	-0.42
Ca (ppm)	-0.82	-0.50
K (ppm)	-0.73	0.61
Mg (ppm)	-0.96	-0.009
Na (ppm)	-0.79	0.53
SO_4 (ppm)	-0.43	-0.51
SiO_2 (ppm)	-0.49	0.61
NO_2+NO_3 (ppm)	-0.09	0.72
NH_4 (ppm)	0.40	0.21
TOC (ppm)	-0.65	0.11
TIC (ppm)	-0.86	-0.43
TP (ppm)	-0.79	-0.003
TN (ppm)	-0.37	0.78
Al (ppm)	-0.88	0.26
Fe (ppm)	-0.89	0.17
Mn (ppm)	-0.67	0.33
Zn (ppm)	0.21	0.20
Cu (ppm)	0.04	-0.39

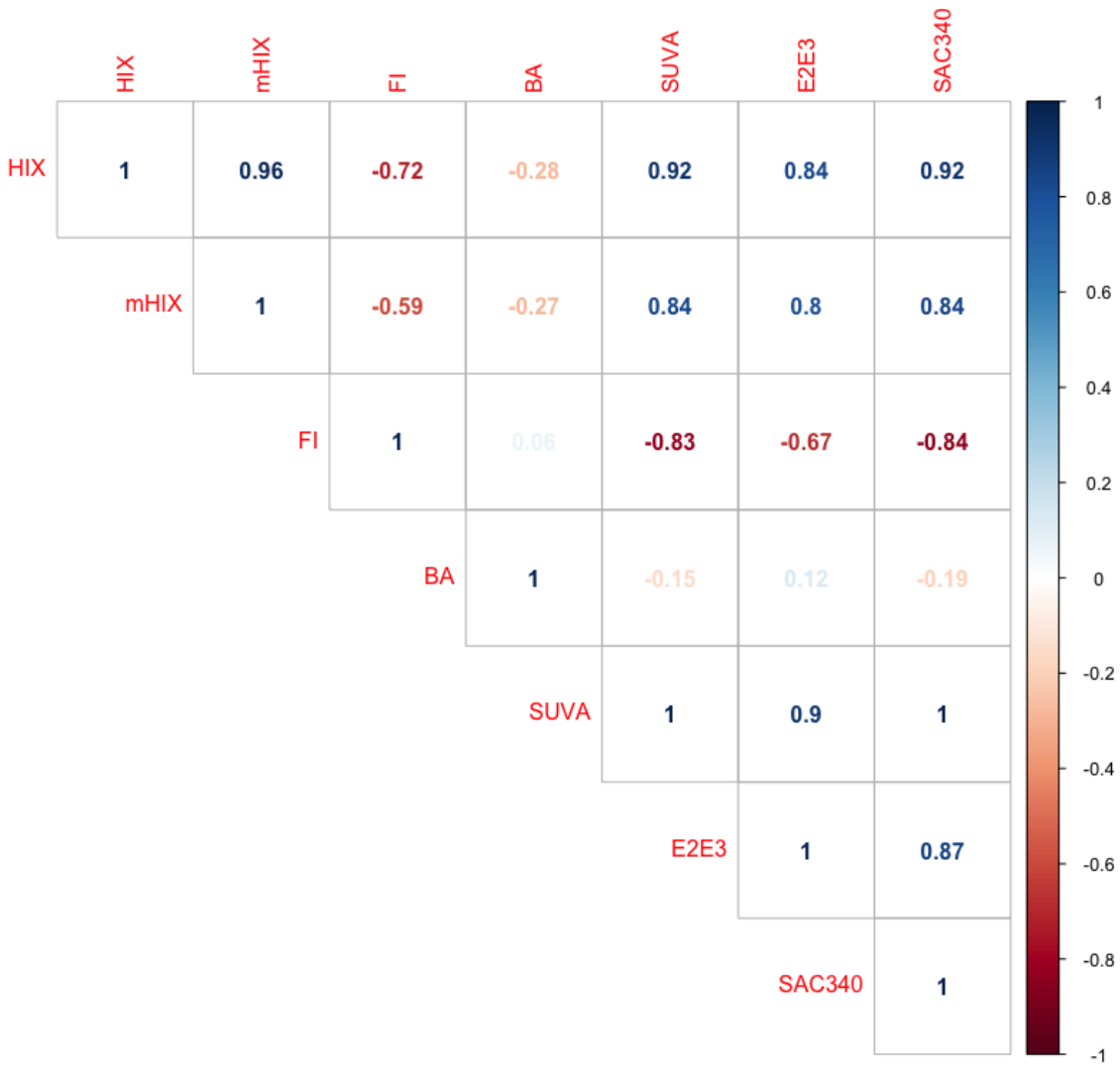
A.4. DOM Quality

Appendix Table 10: Raw data for DOM parameters for sampling in September and October 2016 and in August, September, and October 2017.

Site	HIX	mHIX	FI	β:α	SUVA	E2E3	SAC_{340}
September 2016							
PAN 3	7.19	0.88	1.29	0.46	32.03	3.13	314.14
PAN2	8.57	0.90	1.11	0.40	25.08	2.83	260.58
PAN 1	10.63	0.91	1.25	0.38	43.98	3.39	410.71
BAT 3	10.23	0.91	1.24	0.44	31.27	3.14	305.30
BAT 2	8.02	0.89	1.33	0.39	23.36	2.67	255.28
BAT 1	10.49	0.91	1.36	0.41	32.60	3.06	324.99
KER 3	3.58	0.78	1.32	0.48	15.76	2.33	186.01
KER 2	8.36	0.89	1.32	0.42	28.45	3.02	292.71
KER 1	9.41	0.90	1.28	0.44	28.12	3.06	276.10
KER 6	15.48	0.94	1.19	0.40	60.39	3.67	535.58
KER 5	10.61	0.91	1.23	0.43	46.81	3.49	426.46
KER 4	11.00	0.92	1.29	0.44	44.58	3.43	410.85
October 2016							
PAN 3	8.17	0.89	1.34	0.38	19.77	2.47	228.96
PAN 2	6.75	0.87	1.32	0.45	22.82	2.72	244.58
PAN 1	7.88	0.89	1.26	0.44	28.58	2.95	289.62
BAT 3	6.13	0.86	1.31	0.45	22.39	2.69	243.47
BAT 2	7.21	0.88	1.24	0.37	15.49	2.19	195.50
BAT 1	6.64	0.87	1.19	0.40	17.75	2.35	212.27
KER 3	8.11	0.89	1.31	0.47	40.76	3.25	391.55
KER 2	10.00	0.91	1.30	0.39	23.15	2.72	247.66
KER 1	8.47	0.89	1.32	0.48	24.08	2.76	255.57
KER 6	5.02	0.83	1.27	0.48	18.63	2.51	209.43
KER 5	12.77	0.93	1.32	0.40	32.82	3.06	327.87
KER 4	7.89	0.89	1.28	0.42	32.07	3.04	321.80
August 2017							
PAN 3	8.23	0.89	1.32	0.41	16.77	3.32	155.92
PAN 2	6.05	0.86	1.30	0.45	18.11	3.61	157.00

PAN 1	11.57	0.92	1.19	0.39	25.65	3.90	214.77
BAT 3	10.25	0.91	1.27	0.43	19.05	2.93	193.88
BAT 2	5.33	0.84	1.26	0.40	9.42	2.72	101.13
BAT 2new*	7.55	0.88	1.40	0.45	9.92	2.65	107.32
KER 3	6.28	0.86	1.20	0.45	11.60	3.34	102.56
KER 2	9.52	0.90	1.33	0.41	14.29	3.40	127.67
KER 1	10.85	0.92	1.25	0.43	15.30	3.39	138.28
KER 6	13.59	0.93	1.09	0.40	34.49	4.07	282.69
KER 5	12.62	0.93	1.12	0.42	25.27	3.89	211.29
KER 4	12.44	0.93	1.19	0.40	28.31	3.76	243.41
September 2017							
PAN 3	7.31	0.88	1.33	0.35	9.60	2.80	99.59
PAN 2	6.84	0.87	1.22	0.43	14.78	3.33	133.60
PAN 1	11.52	0.92	1.27	0.39	21.64	3.71	187.57
BAT 3	7.60	0.88	1.38	0.40	13.74	3.41	126.07
BAT 2	4.32	0.81	1.32	0.41	6.32	2.37	72.59
BAT 2new*	4.54	0.82	1.40	0.43	10.86	2.96	107.21
BAT 1	4.46	0.82	1.36	0.38	10.53	2.89	106.34
KER 3	5.70	0.85	1.39	0.42	12.69	3.39	111.25
KER 2	8.87	0.90	1.20	0.39	18.27	3.75	154.83
KER 1	9.12	0.90	1.26	0.40	17.07	3.58	148.96
KER 6	13.62	0.93	1.10	0.39	34.22	3.90	290.44
KER 5	11.97	0.92	1.12	0.40	26.66	3.87	225.53
KER 4	9.41	0.90	1.28	0.41	18.63	3.56	164.95
October 2017							
PAN 3	6.85	0.87	1.45	0.43	14.69	3.45	133.57
PAN 2	7.64	0.88	1.31	0.43	21.76	3.82	184.14
PAN 1	9.93	0.91	1.29	0.36	31.01	3.94	262.80
BAT 3	9.37	0.90	1.26	0.39	22.79	3.94	191.21
BAT 2	6.52	0.87	1.29	0.36	19.14	3.55	170.89
BAT 2new*	6.87	0.87	1.27	0.36	20.55	3.60	181.78
BAT 1	7.28	0.88	1.28	0.39	19.13	3.62	168.66
KER 3	6.29	0.86	1.17	0.44	13.61	3.60	114.65
KER 2	9.60	0.91	1.21	0.40	30.00	4.00	248.33
KER 1	9.61	0.91	1.21	0.41	25.25	3.86	212.06
KER 6	8.92	0.90	1.22	0.35	25.28	3.83	214.56

KER 5	8.44	0.89	1.33	0.35	26.27	3.19	248.43
KER 4	7.43	0.88	1.14	0.40	34.06	4.04	281.39



Appendix Figure 3: Correlation matrix among all DOM optical properties, averaged for all collections in 2016 (N=2 for all sites except BAT 2new*, where N=0) and 2017 (N=3).

Appendix Table 11: Pearson's Correlation Coefficients among all DOM quality optical properties averaged for all collections in 2016 (N=2 for all sites except BAT 2new*, where N=0) and 2017 (N=3) and the first two Principal Component Analysis (PCA) axes (DOM_PC1 and DOM_PC2).

DOM Quality Optical Property	DOM_PC1	DOM_PC2
HIX	0.97	0.12
mHIX	0.91	0.15
FI	-0.83	0.13
β:α	-0.19	-0.97
SUVA	0.98	-0.05
E2E3	0.90	-0.31
SAC_{340}	0.98	-0.002

Appendix Table 12: Test results for significance for interaction between treatment and site, effect of treatment, and effect of site for sediment deposition response variables excluded from results section. All variables were transformed when necessary to meet the assumption of normal distribution.

Paired-Catchment Comparison	Response Variable	Year	Interaction?	Effect of Treatment?	Effect of Site?
PAN_{REF} vs. BAT_{HARV}	Fluorescence Index (FI)	2016 and 2017	**No** (two-factor ANOVA, p=0.068)	**No** (two-factor ANOVA, p=0.35)	**No** (two-factor ANOVA, p=0.27)
KER A_{REF} vs. KER B_{HARV}	Fluorescence Index (FI)	2016 and 2017	**No** (two-factor ANOVA, p=0.55)	**No** (Sidak's Multiple Comparisons, p>0.05):	**No** (two-factor ANOVA, p=0.75)
PAN_{REF} vs. BAT_{HARV}	Freshness Index (β:α)	2016 and 2017	**No** (two-factor ANOVA, p=0.54)	**No** (two-factor ANOVA, p=0.90)	**No** (two-factor ANOVA, p=0.13)
KER A_{REF} vs. KER B_{HARV}	Freshness Index (β:α)	2016 and 2017	**No** (two-factor ANOVA, p=0.18)	**Yes** (two-factor ANOVA, p=0.034: higher in KER A than KER B at upstream site (Sidak's Multiple Comparisons, p=0.030))	**Yes** – KER A only (Tukey's Multiple Comparisons, p<0.05: higher upstream than middle-reach (p=0.02))

Appendix Table 13: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for β:α, as determined via AIC_c model selection ($\Delta AIC_c < 7$). Statistically significant EVs are bolded.

Response Variable		Explanatory Variables	
		Year	**Dep**
β:α	Slope Coefficient	**-0.01**	**0.02**
	95% Confidence Interval	**-0.02 to -0.004**	**0.005 to 0.03**
	Relative Variable Importance	**0.19**	**0.08**

[1]Year = year of sampling; Dep = fine, inorganic sediment deposition (g/day).
[2]ep highly correlated (r=-0.70) with WC_PC1 (not included in full model).

A.5. Sediment Deposition

Appendix Table 14: Average (± SD, N=3-7) daily deposition of coarse (1 mm to 250 µm) inorganic sediment (CIS), fine (1.5 to 250 µm) inorganic sediment (FIS), coarse organic sediment (COS), fine organic sediment (FOS), and average (± SD) %organic for coarse sediment (CS) and fine sediment (FS) in 2016 (including exact deployment and removal dates and number of days deployed for sediment collectors). Note than PAN 3 collectors were found to be stranded after original deployment and re-deployed at end of September.

Site Name	Deployment Date	Removal Date	# Days Deployed	Avg CIS (± SD, g/day)	Avg FIS (± SD, g/day)	Avg COS (± SD, g/day)	Avg FOS (± SD, g/day)	Avg % Organic CS (± SD)	Avg % Organic FS (± SD)
PAN 3	Sept 30, 2016	Oct 28, 2016	28	0.20 (0.073)	0.065 (0.026)	0.0070 (0.0015)	0.0069 (0.0015)	3.38 (137)	9.65 (1.74)
PAN 2	Sept 10, 2016	Oct 28, 2016	49	0.36 (0.15)	0.040 (0.015)	0.0047 (0.0015)	0.0039 (0.0020)	1.29 (1.18)	8.77 (1.57)
PAN 1	Sept 10, 2016	Oct 28, 2016	49	0.44 (0.30)	0.16 (0.13)	0.0039 (0.0026)	0.0035 (0.0025)	0.87 (0.26)	2.15 (1.43)
BAT 3	Sept 8, 2016	Oct 27, 2016	49	0.054 (0.031)	0.027 (0.011)	0.0038 (0.0018)	0.0042 (0.0020)	6.53 (1.60)	13.72 (1.96)
BAT 2	Sept 8, 2016	Oct 27, 2016	49	0.018 (0.026)	0.017 (0.0040)	0.0038 (0.0065)	0.0048 (0.00088)	17.44 (15.46)	22.16 (4.78)
BAT 1	Sept 8, 2016	Oct 27, 2016	49	0.27 (0.15)	0.030 (0.015)	0.0029 (0.0020)	0.0015 (0.00089)	1.07 (0.39)	4.81 (0.95)
KER 3	Sept 9, 2016	Oct 26, 2016	47	0.16 (0.16)	0.13 (0.058)	0.0077 (0.0023)	0.0090 (0.0026)	4.48 (7.27)	6.45 (3.16)
KER 2	Sept 9, 2016	Oct 26, 2016	47	0.24 (0.16)	0.067 (0.038)	0.0046 (0.0038)	0.0033 (0.0016)	1.88 (0.54)	4.62 (1.17)
KER 1	Sept 9, 2016	Oct 25, 2016	46	0.39 (0.050)	0.19 (0.049)	0.0068 (0.0013)	0.0058 (0.0019)	1.69 (2.29)	2.91 (0.54)
KER 6	Sept 9, 2016	Oct 26, 2016	47	0.26 (0.12)	0.21 (0.065)	0.0089 (0.011)	0.0055 (0.0012)	3.26 (8.80)	2.48 (0.49)
KER 5	Sept 9, 2016	Oct 26, 2016	47	0.32 (0.20)	0.087 (0.049)	0.0032 (0.0010)	0.0035 (0.0018)	1.01 (0.65)	3.29 (1.38)
KER 4	Sept 9, 2016	Oct 25, 2016	46	0.41 (0.19)	0.29 (0.13)	0.0061 (0.00088)	0.0074 (0.0041)	1.16 (0.87)	2.33 (0.42)

Appendix Table 15: Average (± SD, N=6) daily deposition of coarse (1 mm to 250 µm) inorganic sediment (CIS), fine (1.5 to 250 µm) inorganic sediment (FIS), coarse organic sediment (COS), fine organic sediment (FOS), and average (± SD) % organic content for coarse sediment (CS) and fine sediment (FS) in 2017 (including exact deployment and removal dates and number of days deployed for sediment collectors).

Site Name	Deployment Date	Removal Date	# Days Deployed	Avg CIS (± SD, g/day)	Avg FIS (± SD, g/day)	Avg COS (± SD, g/day)	Avg FOS (± SD, g/day)	Avg % Organic CS (± SD)	Avg % Organic FS (± SD)
PAN 3	Sept 9, 2017	Oct 23, 2017	44	0.024 (0.029)	0.012 (0.0060)	0.0023 (0.00083)	0.0027 (0.00060)	20.91 (18.34)	1.73 (6.49)
PAN 2	Sept 9, 2017	Oct 23, 2017	44	0.026 (0.032)	0.0088 (0.0028)	0.0024 (0.00047)	0.0029 (0.00062)	22.14 (18.07)	2.46 (3.75)
PAN 1	Sept 9, 2017	Oct 23, 2017	44	0.053 (0.066)	0.038 (0.020)	0.0016 (0.00049)	0.0024 (0.00070)	10.90 (13.34)	1.37 (3.31)
BAT 3	Sept 11, 2017	Oct 27, 2017	46	0.0044 (0.0029)	0.0092 (0.0023)	0.0022 (0.0016)	0.0030 (0.00073)	31.75 (24.62)	0.51 (0.85)
BAT 2	Sept 11, 2017	Oct 27, 2017	46	0.0044 (0.0016)	0.0086 (0.0017)	0.0013 (0.00033)	0.0020 (0.00093)	23.93 (7.06)	0.53 (7.41)
BAT 2new*	Sept 8, 2017	Oct 27, 2017	49	0.0044 (0.0046)	0.0041 (0.0025)	0.00143 (0.0065)	0.0020 (0.0011)	38.04 (3.02)	0.69 (11.81)
BAT 1	Sept 8, 2017	Oct 27, 2017	49	0.23 (0.19)	0.029 (0.014)	0.0064 (0.0065)	0.0047 (0.0072)	2.53 (1.50)	7.22 (24.72)
KER 3	Sept 6, 2017	Oct 25, 2017	49	0.087 (0.089)	0.059 (0.038)	0.0083 (0.0038)	0.0086 (0.0029)	20.23 (18.57)	1.42 (7.46)
KER 2	Sept 6, 2017	Oct 25, 2017	49	0.12 (0.19)	0.076 (0.039)	0.0034 (0.0025)	0.0056 (0.0022)	6.42 (3.74)	1.46 (1.15)
KER 1	Sept 7, 2017	Oct 26, 2017	49	0.23 (0.14)	0.10 (0.71)	0.0060 (0.0042)	0.0054 (0.0034)	2.23 (1.64)	2.14 (0.58)
KER 6	Sept 7, 2017	Oct 25, 2017	48	0.21 (0.15)	0.096 (0.019)	0.0038 (0.0017)	0.0095 (0.012)	2.66 (2.13)	2.18 (12.08)
KER 5	Sept 7, 2017	Oct 26, 2017	49	0.18 (0.16)	0.12 (0.035)	0.0048 (0.0019)	0.0057 (0.0016)	4.70 (3.70)	1.53 (0.60)
KER 4	Sept 12, 2017	Oct 26, 2017	44	0.46 (0.27)	0.18 (0.069)	0.0064 (0.0030)	0.0065 (0.0027)	1.60 (0.67)	2.47 (0.67)

Appendix Table 16: Test results for significance for interaction between treatment and site, effect of treatment, and effect of site for sediment deposition response variables excluded from results section. All variables were transformed when necessary to meet the assumption of normal distribution.

Paired-Catchment Comparison	Response Variable	Year	Interaction?	Effect of Treatment?	Effect of Site?
PAN_{REF} vs. BAT_{HARV}	Coarse Sediment Deposition Per Day	2016	**No** (two-factor ANOVA, p=0.20)	**Yes** (two-factor ANOVA, p<0.0001): higher PAN than BAT at middle-reach (Sidak's Multiple Comparisons, p=0.0006)	**Yes** (two-factor ANOVA, p=0.001: higher downstream than upstream for PAN (Tukey's Multiple Comparisons, p=0.016), higher downstream than upstream and middle-reach for BAT (p=0.033, 0.011))
KER A_{REF} vs. KER B_{HARV}	Coarse Sediment Deposition Per Day	2016	**No** (two-factor ANOVA, p=0.95)	**No** (Sidak's Multiple Comparisons, p>0.05):	**No** (two-factor ANOVA, p=0.18)
PAN_{REF} vs. BAT_{HARV}	Coarse Sediment Deposition Per Day	2017	**Yes** (two-factor ANOVA, p=0.0076)	**No** (multiple t-tests, p>0.05)	**Yes** - BAT only (one-factor ANOVA, p=0.0033: higher downstream than upstream and middle-reach (Tukey's Multiple Comparisons, p=0.0075, 0.0070))
KER A_{REF} vs. KER B_{HARV}	Coarse Sediment Deposition Per Day	2017	**No** (two-factor ANOVA, p=0.50)	**No** (Sidak's Multiple Comparisons, p>0.05)	**Yes** – KER B only (two-factor ANOVA, p=0.014: higher downstream than middle-reach (Tukey's Multiple Comparisons, p=0.025))
PAN_{REF} vs. BAT_{HARV}	Coarse % Organic Content	2016	**Yes** (two-factor ANOVA, p=0.0005)	**Yes** (higher BAT than PAN at the upstream and middle-reach sites (multiple t-tests, p=0.006, 0.012))	**Yes** (one-factor ANOVA, p=0.0002, 0.0017: higher upstream than middle-reach and downstream for PAN (Tukey's Multiple Comparisons,

					p=0.0037, 0.0002), higher middle-reach than upstream and downstream for BAT (p=0.0211, 0.0015)
KER A_{REF} vs. KER B_{HARV}	Coarse % Organic Content	2016	**Yes** (two-factor ANOVA, p=0.0024)	**Yes** (higher KER A at the upstream and middle-reach sites (multiple t-tests, p=0.005, 0.014))	**Yes** (one-factor ANOVA, p=0.0017: higher upstream than downstream (Tukey's Multiple Comparisons, p=0.0012))
PAN_{REF} vs. BAT_{HARV}	Coarse % Organic Content	2017	**No** (two-factor ANOVA, p=0.15)	**No** (two-factor ANOVA, p=0.25)	**Yes** – BAT only (two-factor ANOVA, p=0.0019: higher at upstream and middle-reach than downstream (Tukey's Multiple Comparisons, p=0.0078, 0.0013))
KER A_{REF} vs. KER B_{HARV}	Coarse % Organic Content	2017	**Yes** (two-factor ANOVA, p=0.0090)	**No** (multiple t-tests, p>0.05)	**Yes** – KER A only (one-factor ANOVA, p=0.031: higher upstream than downstream (Tukey's Multiple Comparisons, p=0.031))

Appendix Table 17: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for coarse inorganic sediment deposition per day, as determined via AIC_c model selection ($\Delta AIC_c < 7$). The conditional R^2 was 0.54. Statistically significant EVs are bolded.

Response Variable		Explanatory Variables						
		Year	**Site**	% EVSA	Treatment	Road	Treat: Site	Treat: Year
Coarse Sediment Deposition (g/day) 2017	Slope Coefficient	**-0.76**	**0.84**	0.43	-0.06	0.36	0.75	0.05
	95% Confidence Interval	**-0.96 to -0.56**	**0.27 to 1.40**	-0.33 to 1.19	-0.90 to 0.77	-0.82 to 1.55	-0.007 to 1.52	-0.15 to 0.25
	Relative Variable Importance	**0.96**	**0.89**	0.34	0.45	0.31	0.26	0.02

[1]Year = year of sampling; Site = distance (km) to farthest downstream site within catchment multiplied by -1; % EVSA = % effective variable source area compared to sub-catchment area; Treatment = % sub-catchment area harvested within the last 5 years; Road = road density (m per ha sub-catchment); Treat:Site = interaction between Treatment and Site; Treat:Year = interaction between Treatment and Year.
[2]Site highly correlated (r=0.89) with catchment size (not included in full model); Site highly correlated (r=0.79) with Flow (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4), so %Dec removed from full model.

Appendix Table 18: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for % organic content coarse sediment, as determined via AIC_c model selection ($\Delta AIC_c < 7$). The conditional R^2 was 0.66. Statistically significant EVs are bolded.

Response Variable		Explanatory Variables						
		Site	**Year**	% EVSA	Treatment	Road	% Leaf Lost	Treat: Site
% Organic Content of Coarse Sediment	Slope Coefficient	**-0.82**	**0.56**	-0.06	0.13	-0.04	0.22	-0.29
	95% Confidence Interval	**-1.18 to -0.45**	**0.38 to 0.73**	-0.53 to 0.41	-0.43 to 0.69	-0.54 to 0.46	-0.04 to 0.48	-0.75 to 0.17
	Relative Variable Importance	**0.90**	**0.90**	0.12	0.22	0.13	0.22	0.06

[1]Site = distance (km) to farthest downstream site within catchment multiplied by -1; Year = year of sampling; % EVSA = % effective variable source area compared to sub-catchment area; Treatment = % sub-catchment area harvested within the last 5 years; Road = road density (m per ha sub-catchment); % Leaf Lost = % leaf mass lost per degree day; Treat:Site = interaction between Treatment and Site.
[2]Site highly correlated (r=0.89) with catchment size (not included in full model); Site highly correlated (r=0.79) with Flow (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4), so % Dec removed from full model.

A.6. Leaf Litter Decomposition

Appendix Table 19: Average (± SD, N=3) % leaf mass lost per degree day (DD) and average (± SD, N=3) leaf litter breakdown rate (K) in 2016 (including exact deployment and removal dates and number of days deployed for leaf packs). Note that because no temperature data was collected for sites PAN 3, 1 and BAT 2 in 2016, degree days at these sites was assumed to be the same as sites with the same orientation the KER reference or harvested catchments (i.e., KER 3, 1, and 5).

Site Name	Deployment Date	Removal Date	# Days Deployed	Avg % Lost (± SD, DD^{-1})	Avg K (± SD, DD^{-1})
PAN 3	Sept 30, 2016	Oct 28, 2016	28	0.057 (0.016)	0.00071 (0.00026)
PAN 2	Sept 10, 2016	Oct 28, 2016	49	0.065 (0.012)	0.00085 (0.00019)
PAN 1	Sept 10, 2016	Oct 28, 2016	49	0.048 (0.026)	0.00058 (0.00035)
BAT 3	Sept 8, 2016	Oct 27, 2016	49	0.090 (0.014)	0.0016 (0.00041)
BAT 2	Sept 8, 2016	Oct 27, 2016	49	0.046 (0.014)	0.00054 (0.00020)
BAT 1	Sept 8, 2016	Oct 27, 2016	49	0.081 (0.0034)	0.0011 (0.000062)
KER 3	Sept 9, 2016	Oct 26, 2016	47	0.066 (0.099)	0.00086 (0.00017)
KER 2	Sept 9, 2016	Oct 26, 2016	47	0.051 (0.0085)	0.00061 (0.00012)
KER 1	Sept 9, 2016	Oct 25, 2016	46	0.088 (0.059)	0.0016 (0.0016)
KER 6	Sept 9, 2016	Oct 26, 2016	47	0.098 (0.040)	0.0017 (0.00093)
KER 5	Sept 9, 2016	Oct 26, 2016	47	0.093 (0.017)	0.0013 (0.00039)
KER 4	Sept 9, 2016	Oct 25, 2016	46	0.063 (0.011)	0.00079 (0.00018)

Appendix Table 20: Average (± SD, N=3) % leaf mass lost per degree day (DD) and average (± SD, N=3) leaf litter breakdown rate (K) in 2017 (including exact deployment and removal dates and number of days deployed for leaf packs).

Site Name	Deployment Date	Removal Date	# Days Deployed	Avg % Lost (± SD, DD^{-1})	Avg K (± SD, DD^{-1})
PAN 3	Sept 9, 2017	Oct 23, 2017	44	0.099 (0.048)	0.0015 (0.0011)
PAN 2	Sept 9, 2017	Oct 23, 2017	44	0.067 (0.022)	0.00089 (0.00036)
PAN 1	Sept 9, 2017	Oct 23, 2017	44	0.084 (0.031)	0.0012 (0.00065)
BAT 3	Sept 11, 2017	Oct 27, 2017	46	0.11 (0.030)	0.0017 (0.00065)
BAT 2	Sept 11, 2017	Oct 27, 2017	46	0.071 (0.0076)	0.00087 (0.00012)
BAT 2new*	Sept 8, 2017	Oct 27, 2017	49	0.12 (0.010)	0.0021 (0.00034)
BAT 1	Sept 8, 2017	Oct 27, 2017	49	0.11 (0.021)	0.0017 (0.00049)
KER 3	Sept 6, 2017	Oct 25, 2017	49	0.076 (0.0089)	0.0011 (0.00019)
KER 2	Sept 6, 2017	Oct 25, 2017	49	0.056 (0.0055)	0.00068 (0.000084)
KER 1	Sept 7, 2017	Oct 26, 2017	49	0.067 (0.0073)	0.00087 (0.00012)
KER 6	Sept 7, 2017	Oct 25, 2017	48	0.078 (0.019)	0.0011 (0.00041)
KER 5	Sept 7, 2017	Oct 26, 2017	49	0.070 (0.013)	0.00091 (0.00021)
KER 4	Sept 12, 2017	Oct 26, 2017	44	0.11 (0.052)	0.0020 (0.0015)

Appendix Table 21: Test results for significance for interaction between treatment and site, effect of treatment, and effect of site for leaf litter decomposition response variables excluded from results section. All variables were transformed when necessary to meet the assumption of normal distribution.

Paired-Catchment Comparison	Response Variable	Year	Interaction?	Effect of Treatment?	Effect of Site?
PAN_{REF} vs. BAT_{HARV}	Leaf Litter Breakdown Rate	2016	**Yes** (two-factor ANOVA, p=0.0078)	**No** (multiple t-tests, p>0.05)	**Yes** – BAT only (one-factor ANOVA, p=0.0088: higher upstream than middle-reach (Tukey's Multiple Comparisons, p=0.0071))
KER A_{REF} vs. KER B_{HARV}	Leaf Litter Breakdown Rate	2016	**No** (two-factor ANOVA, p=0.17)	**No** (two-factor ANOVA, p=0.55)	**No** (two-factor ANOVA, p=0.81)
PAN_{REF} vs. BAT_{HARV}	Leaf Litter Breakdown Rate	2017	**No** (two-factor ANOVA, p=0.41)	**No** (two-factor ANOVA, p=0.079)	**No** (two-factor ANOVA, p=0.91)
KER A_{REF} vs. KER B_{HARV}	Leaf Litter Breakdown Rate	2017	**No** (two-factor ANOVA, p=0.32)	**No** (two-factor ANOVA, p=0.0.15)	**No** (two-factor ANOVA, p=0.26)

Appendix Table 22: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for leaf breakdown rate, as determined via AIC_c model selection ($\Delta AIC_c < 7$). The conditional R^2 was 0.32. Statistically significant EVs are bolded.

Response Variable		Explanatory Variables							
		Year	DOM_ PC2	**Treatment**	Shredders	Dep	DOM_ PC1	Site	% EVSA
Leaf Breakdown Rate	Slope Coefficient	**0.16**	-0.13	**0.16**	0.14	-0.06	-0.05	-0.04	-0.005
	95% Confidence Interval	**0.03 to 0.30**	-0.27 to 0.006	**0.04 to 0.29**	-0.01 to 0.30	-0.24 to 0.11	-0.21 to 0.14	-0.17 to 0.09	-0.14 to 0.13
	Relative Variable Importance	**0.46**	0.16	**0.16**	0.14	0.04	0.03	0.02	0.02

[1]Year = year of sampling; Treatment = % sub-catchment area harvested within the last 5 years; DOM_PC2 = scores for DOM PCA axis 2; Shredders = % of invertebrates with shredding feeding strategy; Dep = fine, inorganic sediment deposition per day; Site2: site name as a category (as fixed effect); DOM_PC1 = scores for DOM PCA axis 1; Site = distance (km) to farthest downstream site within catchment multiplied by -1; % EVSA = % effective variable source area compared to sub-catchment area.
[2]Site highly correlated (r=0.86) with catchment size (not included in full model); % EVSA highly correlated (r=-0.70) with Flow (not included in full model); Treatment highly correlated (r=0.78) with WC_PC1 (not included in full model); DOM_PC1 and DOM_PC2 highly correlated (r=-0.68, -0.75) with WC_PC1 (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4), so % Dec removed from full model; Road and Dep had high VIFs (>4), so Road removed from full model.

A.7. Leaf Litter Macroinvertebrates

Appendix Table 23: Complete list and average counts (N=3) of invertebrates identified in leaf packs in 2016.

ORDER	FAMILY	GENERA (etc.)	SPECIES (etc.)	PAN 3	PAN 2	PAN 1	BAT 3	BAT 2	BAT 1
Ephemeroptera	Baetidae	*Baetis*	early instar	0	0	0	0	0	0
			broken	0	4.00	3.67	0	15.00	2.33
			amplus	0	0	0	0	0	2.00
			brunniecolour	0	0	0	0	1.33	1.00
			flavistriga	0	0	0	0	0.33	0
	Caenidae	*Caenis*		0	0	0	0	0	0
	Ephemerellidae	Immature		0	0	0	0	0	0
		Ephemerella		0	1.67	1.33	0	0.67	11.67
		Eurylophella	*funeralis*	1.33	2.67	1.67	1.67	6.67	2.33
			versimilus	0	1	9	0.33	14.33	0.33
	Heptageniidae	Broken		0	0	0.33	0	0.33	0
		Epeorus		0	0	0	0	0	0
		Stenacron		0	0	0.33	0	0	0
		Stenonema		0	0	0.33	0	1.33	0
		Heptagenia		0	0	0	0	0.33	0.33
		Rhithrogena		0	0	0	0	0	0
	Leptophlebidae	Early instar		0	0	0	0	0	0
		Broken		0	2	1	0	0.67	0.67
		Habrophlebia		0	8	3	0	1	0.33
		Leptophlebia		0	1.33	4	0	1.67	0
		Paraleptophlebia		0	4.33	9.67	0	0.67	8.67
	Ephemeridae	*Litobrancha*		0	0	0	0	0	0
		Ephemera		0	0	1.33	0	0	0
Plecoptera	Pteronarcyidae	*Pteronarcys*		0	0	0	0	0	0
	Perlidae	*Acroneuria*		0	0	0	0	0	0
	Taeniopterygidae	*Taeniopteryx*		0	0.33	0	0	1.67	0
	Nemouridae	*Nemoura*		0	0.33	1.67	0.33	0.67	4
	Chloroperlidae	Broken		0	0	0	0	0	0.33
		Alloperla		4.33	0.67	0	1	0.67	3.67
		Suwalia / Sweltsa		1	0. 33	0	0.67	0	2

	Perlodidae	Broken or Immature		0	0	0	0	0	0
		Isoperla		0	2	2.67	0	1.67	5.67
	Capniidae	*Paracapnia*		0	3.67	8.00	0	1.67	4.33
	Leuctridae	*Leuctra*		1	7	2.33	1	5.33	2.67
Trichoptera	Leptoceridae	*Trianeodes*		0	1	0.33	0	0	0
		Oecetis		0	0	0	0	0	0
	Hydroptillidae	*Hydroptila*		0	0	0.67	0	0	0
		Oxyethira		0	0	0.67	0	20	0
	Polycentropodidae	*Nyctiophylax*		0	0	0	0	0	0
		Polycentropus		0.33	1.33	0.67	0.33	0	0
	Philopotamidae	*Dolophilodes*		0	0.33	0	0	0	0
	Glossomatidae	*Glossosoma*		0	0	0	0	0	0
	Lepidostomatidae	*Lepidostoma*		6	8.33	8.33	71.33	12.00	25.67
	Limnephillidae	Broken or Early Instar		0	0	0.67	0.33	0	0.67
		Hydatophylax		0	0	0.33	0	1.33	0
		Hesperophylax		0	0	0	0	0	0
	Dipseudopsidae	*Phylocentropus*		0	0	0	0	0	0
	Uenoidae	*Neophylax*		0	0	0	0	0	0
	Hydropsychidae	*Hydropsyche*		0	0.33	0.33	0	1.00	0
		Parapsche		0	0	0	0	0	0
		Diplectrona		0	0	0	0	0	0
		Cheumatopsyche		0	0	0	0	0.33	0
	Rhyacophilidae	*Rhyacophila*	*fuscula*	0	0	0	0	0.33	0
			carolina	0	0.33	0.33	0	0	1.00
			invaria	0	0.33	0	0.67	0	0.33
	Phryganeidae	*Oligostomis*		0	0	0	0	0	0
		Ptilostomis		0	0.33	0	0	0	0
Diptera	Chironomidae	*Pupae*		0	0	0	0	0	0
		Tanytarsini		4.33	29.00	56.67	24.33	45.67	84.00
		Tanipodinae		0	11.00	18.33	1.33	17.00	10.67
		Other		20.67	32.00	38.00	16.67	64.00	31.33
	Ceratopogonidae			0	4.00	5.33	2.00	0.67	6.33
	Empididae	*Chelifera*		0	0	0	0	0.67	0
		Clinocera		0	0	0	0	0	0

		Oreogeton	0	0	0	0	0	0
		Other	0	0	0	0	0	0
	Simuliidae	Pupae	0	0	0	0	0	0
		Prosimulium	0	1.33	0	0	0	0
		Simulium	0	0	0.67	0	0.33	0
	Tabanidae		0	0	0	0	0	0
	Tipulidae	*Dicranota*	0	0.67	0	0.33	1.00	3.67
		Dolichopodidae	0	0	0	0	0	0
		Antocha	0	0	0	0	0	0
		Hexatoma	0	0	0	0	0	0
		Limnophila	0	0	1.00	0	0	0
		Molophilus	0	0	0	0	0	0
		Holorusia	0	0	0	0	0	0
		Pedicia	0	0	0	0	0	0
		Tipula	1.33	0.67	1.00	0.33	0.33	0.67
		Pilaria	0	0.67	0	0	0	0
Odonata	Gomphidae		0	0	0	0	0	0
	Aeshnidae	*Boyeria*	0	0.33	0.33	0	0	0
	Cordulegastridae	*Cordulegaster*	0	0	0.33	0	0	0
Annelida	Oligochaeta		1.33	21.33	21.33	3.00	1.67	21.33
	Hirudinea		0	0	0	0	0	0
Coleoptera	Elmidae		0	0	0	0	0	0
	Dytiscidae		0	0	0	0	0	0
Megaloptera	Sialidae	*Sialis*	0	0	0	0	0	0
Amphipoda	Gammaridae	*Gammarus*	0	0.33	0	0	0	0

ORDER	**FAMILY**	**GENERA (etc.)**	**SPECIES (etc.)**	**KER 3**	**KER 2**	**KER 1**	**KER 6**	**KER 5**	**KER 4**
Ephemeroptera	Baetidae	*Baetis*	early instar	0					
			broken	2.00	5.50	16.00	9.00	13.00	17.33
			amplus	0	4.00	0	2.00	4.00	2.00
			brunniecolour	0	0	0	0	0	0
			flavistriga	0	0	0	0	0	0
	Caenidae	*Caenis*		0	0	0	0	1.00	10.67
	Ephemerellidae	Immature		1.00	0	1.00	0	0	0
		Ephemerella		1.00	6.00	0	2.50	11.00	10.50

			versimilus	0	0	0	7.67	3.50	0
	Heptageniidae	Broken		0	1	0	0	0	2.00
		Epeorus		0	1	0	0	0	0
		Stenacron		0	0	0	1.00	0	0
		Stenonema		0	2	3.00	3.00	2.00	8.67
		Heptagenia		0	1	0	0	1.00	0
		Rhithrogena		0	0	0	0	0	0
	Leptophlebidae	Early instar		2.00	1	2.00	0	0	0
		Broken		5.67	0	2.00	2.33	1.00	1.00
		Habrophlebia		2.67	0	0	2.00	0	0
		Leptophlebia		0	0	0	200	2.00	0
		Paraleptophlebia		1.00	6.67	2.50	1.67	5.50	33.50
	Ephemeridae	*Litobrancha*		0	0	1.00	0	1.00	0
		Ephemera		0	0	0	0	0	0
Plecoptera	Pteronarcyidae	*Pteronarcys*		0	0	2.00	0	0	0
	Perlidae	*Acroneuria*		0	1.00	1.00	0	0	0
	Taeniopterygidae	*Taeniopteryx*		2.00	5.50	1.00	6.00	2.50	5.00
	Nemouridae	*Nemoura*		0	0	1.00	7.00	5.67	2.67
	Chloroperlidae	Broken		1.00	0	0	0	0	0
		Alloperla		0	0	0	0	1.00	0
		Suwalia / Sweltsa		2.00	1.33	2.67	1.00	0	0
	Perlodidae	Broken or Immature		1.00	1.50	5.00	0	0	0
		Isoperla		1.00	7.50	2.00	3.33	3.00	3.67
	Capniidae	*Paracapnia*		5.00	9.67	3.00	34.33	17.00	20.33
	Leuctridae	*Leuctra*		4.33	7.00	8.50	9.00	15.50	5.00
Trichoptera	Leptoceridae	*Trianeodes*		1.50	0	0	0	0	0
		Oecetis		0	0	0	0	0	0
	Hydroptillidae	*Hydroptila*		0	0	0	0	0	0
		Oxyethira		1.00	1.00	1.00	0	2.00	0
	Polycentropodidae	*Nyctiophylax*		0	0	0	0	0	0
		Polycentropus		0	1.00	0	0	0	0
	Philopotamidae	*Dolophilodes*		0	0	0	0	0	0
	Glossomatidae	*Glossosoma*		1.00	0	0	0	0	1.00
	Lepidostomatidae	*Lepidostoma*		8.67	5.33	5.67	2.00	20.50	3.50

	Limnephillidae	Broken or Early Instar		0	1.00	0	0	0	0
		Hydatophylax		2.00	0	1.00	0	0	0
		Hesperophylax		0	0	0	0	0	0
	Dipseudopsidae	*Phylocentropus*		3.00	0	0	0	0	0
	Uenoidae	*Neophylax*		0	0	0	0	0	1.00
	Hydropsychidae	*Hydropsyche*		0	1.00	0	0	0	0
		Parapsche		0	1.00	1.00	0	0	1.00
		Diplectrona		0	0	0	0	0	1.00
		Cheumatopsyche		0	0	1.00	0	0	7.00
	Rhyacophilidae	*Rhyacophila*	*fuscula*	0	0	0	0	0	1.00
			carolina	0	0	0	0	0	0
			invaria	1.00	1.00	1.00	2.00	0	3.00
	Phryganeidae	*Oligostomis*		0	0	1.00	0	0	1.00
		Ptilostomis		0	0	0	0	0	0
Diptera	Chironomidae	*Pupae*		0	0	0	0	0	0
		Tanytarsini		77.33	32.67	26.00	27.33	17.33	28.00
		Tanipodinae		25.00	5.00	17.00	9.33	6.67	13.00
		Other		70.33	39.33	36.33	30.67	19.00	18.00
	Ceratopogonidae			13.00	2.00	1.00	3.00	2.00	1.67
	Empididae	*Chelifera*		0	0	0	0	0	0
		Clinocera		0	0	0	0	0	0
		Oreogeton		0	0	1.00	0	0	0
		Other		1.00	0	0	0	0	0
	Simuliidae	Pupae		0	0	0	0	0	0
		Prosimulium		0	1.00	0	0	2.00	1.00
		Simulium		1.00	0	0	0	1.00	0
	Tabanidae			0	0	0	0	0	0
	Tipulidae	*Dicranota*		1.00	1.00	4.00	0	1.50	2.50
		Dolichopodidae		0	0	0	0	0	1.00
		Antocha		0	0	0	0	0	0
		Hexatoma		1.00	0	0	0	0	0
		Limnophila		3.00	0	2.50	0	0	0
		Molophilus		0	0	0	0	0	0
		Holorusia		0	0	0	0	0	0
		Pedicia		0	0	0	0	0	0
		Tipula		2.00	1.00	3.67	1.00	0	2.00

		Pilaria	0	0	0	0	0	0
Odonata	Gomphidae		0	0	0	0	0	0
	Aeshnidae	*Boyeria*	0	0	0	0	0	0
	Cordulegastridae	*Cordulegaster*	2.00	2.00	5.00	5.67	0	2.00
Annelida	Oligochaeta		18.00	4.67	17.33	0	10.00	17.33
	Hirudinea		0	0	0	0	0	0
Coleoptera	Elmidae		0	0	1.00	2.00	0	1.00
	Dytiscidae		0	0	0	0	0	0
Megaloptera	Sialidae	*Sialis*	0	0	0	0	0	1.00
Amphipoda	Gammaridae	*Gammarus*	0	0	0	0	1.00	1.00

Appendix Table 24: Complete list and average counts (N=3) of invertebrates identified in leaf packs in 2017.

ORDER	FAMILY	GENERA (etc.)	SPECIES (etc.)	PAN 3	PAN 2	PAN 1	BAT 3	BAT 2	BAT 2*new	BAT 1
Ephemeroptera	Baetidae	*Baetis*	early instar	0	0	0	0	0	0	0
			broken	9.67	7.00	13.67	2.00	14.67	13.00	5.33
			amplus	0	0	1.00	2.00	5.67	2.00	3.33
			brunniecolour	0	0	0	0	0	0	0
			flavistriga	0	0	0	0	0	1.00	0
	Caenidae	*Caenis*		0	0	0	0	0	0	0
	Ephemerellidae	Immature		0	0	0	0	0	0	0
		Ephemerella		0	2.50	6.00	6.00	7.00	3.33	16.00
		Eurylophella	*funeralis*	8.00	4.00	1.50	1.00	4.67	1.00	7.00
			versimilus	0	1.00	2.00	0	8.33	1.00	1.00
	Heptageniidae	Broken		0	0	1.00	0	0	0	0
		Epeorus		0	0	0	0	0	3.00	0
		Stenacron		0	1.00	2.00	0	0	0	0
		Stenonema		0	1.50	1.67	0	4.00	3.67	1.50
		Heptagenia		0	0	2.00	0	0	0	0
		Rhithrogena		0	0	0	0	0	0	1.00
	Leptophlebidae	Early instar		0	0	0	0	0	0	0
		Broken		0	1.50	0	0	0	3.00	1.00
		Habrophlebia		0	4.50	0	0	0	1.00	0
		Leptophlebia		7.00	700	1.00	1.00	12.00	3.00	2.00
		Paraleptophlebia		0	6.33	54.00	0	2.00	5.00	10.00
	Ephemeridae	*Litobrancha*		0	0	0	0	0	0	0
		Ephemera		0	0	0	0	0	0	0
Plecoptera	Pteronarcyidae	*Pteronarcys*		0	0	0	0	0	0	0
	Perlidae	*Acroneuria*		0	0	1.00	0	0	0	0
	Taeniopterygidae	*Taeniopteryx*		0	0	6.00	1.00	4.50	1.00	2.50
	Nemouridae	*Nemoura*		1.67	1.50	10.50	1.00	2.50	2.33	6.50
	Chloroperlidae	Broken		0	0	0	0	0	0	0
		Alloperla		0	0	0	0	0	1.00	1.00
		Suwalia / Sweltsa		8.00	0	0	2.00	1.00	0	5.67
	Perlodidae	Broken or Immature		0	0	0	0	0	0	0

		Isoperla		0	1.00	0	0	4.50	4.50	5.67
	Capniidae	*Paracapnia*		1.00	1.33	33.33	0	3.33	2.00	3.33
	Leuctridae	*Leuctra*		1.00	2.00	2.00	3.00	6.67	1.00	0
Trichoptera	Leptoceridae	*Trianeodes*		0	0	0	0	0	0	0
		Oecetis		0	0	0	0	0	0	0
	Hydroptillidae	*Hydroptila*		0	0	0	0	0	0	0
		Oxyethira		0	0	0	0	1.00	0	0
	Polycentropodidae	*Nyctiophylax*		0	0	0	0	0	0	0
		Polycentropus		2.50	2.00	4.00	3.50	0	0	0
	Philopotamidae	*Dolophilodes*		0	1.00	0	0	0	0	0
	Glossomatidae	*Glossosoma*		0	0	0	0	0	0	0
	Lepidostomatidae	*Lepidostoma*		22.33	109.00	47.33	87.00	36.00	33.33	37.33
	Limnephillidae	Broken or Early Instar		2.00	1.00	13.33	2.00	3.67	0	0
		Hydatophylax		2.00	3.00	7.00	4.00	19.67	1.00	1.33
		Hesperophylax		0	0	0	0	0	0	0
		Pseudostenophylax		0	0	0	3.00	0	0	0
	Dipseudopsidae	*Phylocentropus*		0	0	0	0	0	0	0
	Molandidae	*Molanna*		0	0	0	0	3.00	0	0
	Uenoidae	*Neophylax*		0	0	0	0	0	0	0
	Hydropsychidae	*Hydropsyche*		0	0	0	0	3.00	2.00	0
		Parapsyche		0	0	0	0	0	0	0
		Diplectrona		0	0	0	0	0	0	0
		Cheumatopsyche		0	0	0	0	0	0	0
	Rhyacophilidae	*Rhyacophila*	*fuscula*	0	0	0	0	0	0	0
			carolina	0	0	1.00	0	0	0	1.00
			invaria	2.00	1.00	0	0	3.00	1.00	0
	Phryganeidae	*Oligostomis*		1.00	0	0	0	0	0	0
		Ptilostomis		0	0	0	0	0	0	0
Diptera	Chrionomidae	*Pupae*		0	0	0	0	1.00	0	0
		Tanytarsini		154.67	167.33	899.33	128.33	172.33	35.67	68.33
		Tanipodinae		46.00	46.67	143.33	54.67	40.33	14.00	7.67
		Other		129.67	166.67	798.33	130.67	281.33	91.33	53.67
	Ceratopogonidae			0	1.00	11.50	2.00	1.00	0	3.50
	Empididae	*Chelifera*		0	0	0	0	3.33	1.00	0
		Clinocera		0	0	0	1.00	0	0	0

		Oreogeton	0	0	0	0	0	0	0
		Other	0	0	0	0	0	0	0
	Simulidae	Pupae	0	0	0	0	0	0	0
		Prosimulium	1.00	0	1.00	0	0	19.50	6.67
		Simulium	0	0	0	0	1.00	20.00	0
	Tabanidae		0	0	0	0	0	0	0
	Tipulidae	*Dicranota*	2.00	0	0	0	4.50	0	4.00
		Dolichopodidae	0	0	0	0	0	0	0
		Antocha	0	0	0	0	0	0	0
		Hexatoma	0	0	0	0	0	0	0
		Limnophila	0	1.00	0	0	0	0	0
		Molophilus	0	0	0	0	0	0	0
		Holorusia	0	0	0	0	0	0	0
		Pedcia	1.00	0	0	0	0	0	1.00
		Tipula	1.00	2.00	1.00	1.00	1.00	0	1.50
		Pilaria	0	0	0	0	0	0	0
Odonata	Gomphidae		0	0	0	0	0	0	0
	Aeshnidae	*Boyeria*	0	1.00	0	0	1.00	1.00	0
	Cordulegastridae	*Cordulegaster*	0	0	0	0	0	0	0
	Cordullidae		0	0	0	0	0	0	1.00
Annelida	Oligochaeta		1.00	0	16.33	18.33	2.50	0	5.00
	Hirudinea		0	0	0	0	0	0	0
Coleoptera	Elmidae		0	0	1.00	0	1.00	0	1.00
	Dytiscidae		0	0	0	0	0	0	0
Megaloptera	Sialidae	*Sialis*	0	0	0	0	0	0	0
Amphipoda	Gammaridae	*Gammarus*	0	0	0	0	0	0	0
Zygoptera	Coenagrionidae		0	0	0	0	0	0	0

ORDER	**FAMILY**	**GENERA (etc.)**	**SPECIES (etc.)**	**KER 3**	**KER 2**	**KER 1**	**KER 6**	**KER 5**	**KER 4**
Ephemeroptera	Baetidae	*Baetis*	early instar	0	0	0	0	0	0
			broken	3.00	3.00	10.00	11.33	23.00	9.50
			amplus	2.00	1.00	4.00	4.00	6.00	22.67
			brunniecol our	0	0	1.00	0	0	0

			flavistriga	1.00	0	0	0	2.00	1.00
	Caenidae	*Caenis*		0	0	0	0	0	0
	Ephemerellidae	Immature		3.50	0	0	0	1.00	0
		Ephemerella		8.67	6.00	9.67	10.00	14.00	17.00
		Eurylophella	*funeralis*	10.17	13.33	2.50	23.67	9.00	3.00
			versimilus	0	1.33	1.00	5.67	7.67	2.00
	Heptageniidae	Broken		0	1.50	3.00	2.00	0	2.00
		Epeorus		0	0	0	0	0	0
		Stenacron		0	0	0	0	0	1.00
		Stenonema		0	2.00	1.50	6.67	5.00	3.33
		Heptagenia		0	1.00	1.00	3.00	1.00	0
		Rhithrogena		0	0	0	0	0	1.50
	Leptophlebidae	Early instar		0	0	2.50	0	1.00	0
		Broken		1.00	0	0	1.33	1.00	2.33
		Habrophlebia		2.00	1.00	0	3.00	0	0
		Leptophlebia		51.00	6.33	2.00	3.33	2.50	4.33
		Paraleptophlebia		3.00	2.67	7.33	20.00	3.67	13.33
	Ephemeridae	*Litobrancha*		0	2.00	0	2.00	5.00	2.00
		Ephemera		0	0	0	0	0	0
Plecoptera	Pteronarcyidae	*Pteronarcys*		0	0	0	0	0	0
	Perlidae	*Acroneuria*		0	0	0	0	0	0
	Taeniopterygidae	*Taeniopteryx*		0	0	4.50	3.00	2.00	1.50
	Nemouridae	*Nemoura*		1.67	2.00	5.00	9.00	11.00	3.33
	Chloroperlidae	Broken		0	0	0	0	0	0
		Alloperla		1.00	1.67	2.67	0	0	0
		Suwalia / Sweltsa		3.50	5.00	3.50	1.00	2.00	0
	Perlodidae	Broken or Immature		0	0	0	0	0	0
		Isoperla		4.00	4.00	2.50	5.67	3.67	3.50
	Capniidae	*Paracapnia*		5.00	3.67	17.67	8.33	19.67	14.33
	Leuctridae	*Leuctra*		5.00	7.67	13.67	4.67	20.00	10.50
Trichoptera	Leptoceridae	*Trianeodes*		1.50	0	0	0	0	0
		Oecetis		0	0	0	0	0	0
	Hydroptillidae	*Hydroptila*		0	0	0	0	0	0
		Oxyethira		0	0	0	0	1.00	0
	Polycentropodidae	*Nyctiophylax*		0	0	0	0	0	0
		Polycentropus		1.00	0	1.00	0	1.00	0

	Philopotamidae	*Dolophilodes*		0	0	0	1.00	0	0
	Glossomatidae	*Glossosoma*		0	0	0	1.00	1.00	0
	Lepidostomatidae	*Lepidostoma*		17.67	35.67	15.00	20.67	37.67	6.50
	Limnephillidae	Broken or Early Instar		0	0	1.00	0	0	0
		Hydatophylax		0	3.50	1.00	5.67	3.00	2.00
		Hesperophylax		0	0	0	0	0	0
		Pseudostenophylax		0	0	0	0	0	0
	Dipseudopsidae	*Phylocentropus*		1.00	0	0	0	0	0
	Molandidae	*Molanna*		0	0	0	3.00	2.00	
	Uenoidae	*Neophylax*		0	0	0	2.50	0	0
	Hydropsychidae	*Hydropsyche*		0	0	2.00	2.50	0	15.67
		Parapsyche		0	0	0	0	0	1.00
		Diplectrona		0	0	0	0	0	0
		Cheumatopsyche		1.50	0	0	0	0	2.00
	Rhyacophilidae	*Rhyacophila*	*fuscula*	0	0	0	0	0	1.00
			carolina	1.00	1.00	1.00	0	0	0
			invaria	1.00	0	2.50	2.50	1.50	1.50
	Phryganeidae	*Oligostomis*		0	0	0	0	0	0
		Ptilostomis		0	0	0	0	0	0
Diptera	Chironomidae	*Pupae*		0	0	0	1.00	0	1.00
		Tanytarsini		578.67	255	151.67	228.33	142.67	75.00
		Tanipodinae		108.00	77.67	34.33	56.00	79.33	37.50
		Other		321.67	256.33	304.00	395.00	326.00	204.33
	Ceratopogonidae			6.67	1.67	3.50	0	2.00	1.50
	Empididae	*Chelifera*		2.00	0	1.33	9.00	6.00	1.50
		Clinocera		0	0	0	0	0	0
		Oreogeton		0	0	0	0	0	0
		Other		0	0	0	0	1.00	0
	Simuliidae	Pupae		0	0	0	0	0	0
		Prosimulium		1.00	1.00	2.50	1.00	1.00	63.50
		Simulium		1.67	0	1.00	2.00	0	2.00
	Tabanidae			0	0	0	0	0	0
	Tipulidae	*Dicranota*		1.00	3.00	6.33	4.00	1.50	2.00
		Dolichopodidae		0	1.00	0	0	0	0
		Antocha		0	0	0	0	0	0
		Hexatoma		1.00	0	0	0	0	0

		Limnophila	0	0	1.00	2.00	1.00	2.00
		Molophilus	1.00	0	0	0	0	0
		Holorusia	0	0	0	0	0	0
		Pedicia	0	0	0	0	0	0
		Tipula	3.67	0	1.00	1.00	2.00	0
		Pilaria	0	0	0	0	0	0
Odonata	Gomphidae		0	0	0	0	0	0
	Aeshnidae	*Boyeria*	0	0	0	0	1.00	0
	Cordulegastridae	*Cordulegaster*	0	0	0	0	0	0
	Cordullidae		0	0	0	1.00	0	0
Annelida	Oligochaeta		18.00	10.00	10.33	5.00	9.50	5.50
	Hirudinea		0	0	0	0	1.00	0
Coleoptera	Elmidae		0	0	1.00	3.00	2.00	3.00
	Dytiscidae		0	0	0	0	0	0
Megaloptera	Sialidae	*Sialis*	0	0	1.00	0	0	0
Amphipoda	Gammaridae	*Gammarus*	0	0	0	0	0	0
Zygoptera	Coenagrionidae		0	0	0	1.00	0	0

Appendix Table 25: Average (± SD, N=3) abundance (Ab), taxonomic richness (TR), % Ephemeroptera, Plecoptera, and Trichoptera (% EPT), % shredders (% Sh), % Chironomidae (% Ch), Margalef's Richness (MR), and Shannon's Diversity Index (SDI) for invertebrate separated from leaf packs in 2016. See Appendix Table 18 above for deployment and removal dates of leaf packs.

Site Name	Avg Ab (± SD)	Avg TR (± SD)	Avg % EPT (± SD)	Avg % Sh (± SD)	Avg % Ch (± SD)	Avg MR (± SD)	Avg SDI (± SD)
PAN 3	41.67 (11.06)	7.00 (2.65)	33.30 (3.64)	21.06 (10.49)	61.03 (8.35)	1.59 (0.62)	1.65 (0.46)
PAN 2	153.00 (37.80)	22.33 (4.16)	34.30 (10.64)	10.64 (1.80)	48.53 (18.44)	4.26 (0.74)	2.12 (0.29)
PAN 1	205.67 (45.35)	22.00 (2.00)	32.36 (13.02)	11.95 (8.11)	52.43 (18.95)	3.96 (0.41)	2.62 (0.34)
BAT 3	125.67 (37.86)	10.00 (3.61)	51.16 (19.99)	57.48 (4.81)	34.46 (4.93)	1.86 (0.69)	1.34 (0.24)
BAT 2	222.33 (23.03)	22.33 (3.51)	41.50 (8.45)	10.53 (2.80)	56.41 (8.13)	3.96 (0.73)	2.10 (0.16)
BAT 1	238.00 (66.57)	20.33 (6.03)	29.74 (21.01)	19.71 (3.07)	55.54 (13.94)	3.52 (0.94)	2.42 (0.42)
KER 3	254.33 (135.40)	21.67 (1.15)	19.98 (5.85)	9.58 (2.28)	63.92 (11.19)	3.86 (0.46)	2.10 (0.21)
KER 2	143.33 (31.77)	20.67 (1.53)	45.29 (23.69)	19.69 (8.93)	49.98 (21.00)	3.98 (0.32)	2.33 (0.37)
KER 1	133.33 (147.55)	20.00 (7.55)	27.09 (3.67)	19.83 (8.52)	33.81 (23.77)	4.15 (0.70)	2.32 (0.12)
KER 6	161.33 (97.90)	19.00 (4.58)	48.56 (13.65)	25.92 (15.41)	45.81 (12.31)	3.60 (0.45)	2.28 (0.07)
KER 5	141.33 (56.58)	18.33 (2.89)	62.13 (16.07)	34.14 (8.40)	30.91 (7.73)	3.53 (0.26)	2.43 (0.05)
KER 4	193.67 (185.65)	22.00 (10.58)	56.45 (3.74)	23.65 (10.43)	29.96 (8.70)	4.11 (1.20)	2.54 (0.19)

Appendix Table 26: Average (± SD, N=3) abundance (Ab), taxonomic richness (TR), % Ephemeroptera, Plecoptera, and Trichoptera (% EPT), % shredders (% Sh), % Chironomidae (% Ch), Margalef's Richness (MR), and Shannon's Diversity Index (SDI) for invertebrate separated from leaf packs in 2017. See Appendix Table 19 above for deployment and removal dates of leaf packs.

Site Name	Avg Ab (± SD)	Avg TR (± SD)	Avg % EPT (± SD)	Avg % Sh (± SD)	Avg % Ch (± SD)	Avg MR (± SD)	Avg SDI (± SD)
PAN 3	393.67 (187.36)	13.67 (0.58)	17.22 (10.13)	7.82 (5.01)	82.26 (9.99)	2.15 (0.14)	1.55 (0.27)
PAN 2	536.67 (237.65)	19.00 (1.73)	17.57 (18.17)	19.44 (13.39)	72.35 (11.17)	2.89 (0.12)	1.64 (0.02)
PAN 1	2039.33 (3028.73)	19.33 (4.51)	25.07 (26.04)	18.99 (20.46)	73.77 (25.57)	2.83 (0.59)	1.72 (0.53)
BAT 3	442.33 (32.33)	14.00 (1.00)	24.12 (13.17)	21.34 (12.75)	71.40 (11.44)	2.14 (0.19)	1.63 (0.11)
BAT 2	635.67 (121.59)	23.00 (3.46)	20.83 (1.77)	12.05 (2.40)	77.44 (2.69)	3.43 (0.63)	1.75 (0.06)
BAT 2new*	243.00 (36.37)	18.67 (3.21)	34.90 (11.43)	15.80 (4.98)	57.19 (16.41)	3.23 (0.63)	1.76 (0.07)
BAT 1	256.67 (79.19)	23.67 (4.16)	42.22 (2.67)	20.48 (5.56)	48.95 (8.07)	4.09 (0.52)	1.98 (0.27)
KER 3	1156.00 (483.44)	26.00 (3.61)	10.27 (2.39)	3.05 (1.13)	69.01 (48.31)	3.57 (0.51)	2.30 (0.08)
KER 2	691.67 (222.98)	22.33 (2.08)	13.58 (4.21)	7.38 (1.04)	84.31 (4.71)	3.29 (0.39)	1.49 (0.10)
KER 1	619.00 (377.21)	28.00 (4.36)	23.65 (17.38)	12.88 (10.33)	71.37 (20.21)	4.35 (0.52)	1.62 (0.10)
KER 6	836.00 (418.57)	28.33 (3.21)	17.05 (2.89)	5.91 (2.11)	80.98 (2.64)	4.10 (0.22)	1.88 (0.51)
KER 5	718.00 (690.69)	24.33 (8.50)	28.06 (14.79)	17.78 (13.53)	70.33 (13.31)	3.72 (0.63)	1.73 (0.12)
KER 4	484.67 (351.90)	26.33 (6.03)	33.46 (15.32)	7.27 (4.19)	49.43 (29.50)	4.18 (0.70)	1.88 (0.08)

Appendix Table 27: Test results for significance for interaction between treatment and site, effect of treatment, and effect of site for leaf litter macroinvertebrate response variables excluded from results section. All variables were transformed when necessary to meet the assumption of normal distribution.

Paired-Catchment Comparison	Response Variable	Year	Interaction?	Effect of Treatment?	Effect of Site?
PAN_{REF} vs. BAT_{HARV}	% Ephemeroptera, Plecoptera, Trichoptera	2016	**No** (two-factor ANOVA, p=0.48)	**No** (two-factor ANOVA, p=0.29)	**No** (two-factor ANOVA, p=0.42)
KER A_{REF} vs. KER B_{HARV}	% Ephemeroptera, Plecoptera, Trichoptera	2016	**No** (two-factor ANOVA, p=0.67)	**No** (Sidak's Multiple Comparisons, p>0.05)	**No** (two-factor ANOVA, p=0.075)
PAN_{REF} vs. BAT_{HARV}	% Ephemeroptera, Plecoptera, Trichoptera	2017	**No** (two-factor ANOVA, p=0.80)	**No** (two-factor ANOVA, p=0.081)	**No** (two-factor ANOVA, p=0.37)
KER A_{REF} vs. KER B_{HARV}	% Ephemeroptera, Plecoptera, Trichoptera	2017	**No** (two-factor ANOVA, p-=0.84)	**No** (two-factor ANOVA, p=0.079)	**No** (two-factor ANOVA, p=0.12)
PAN_{REF} vs. BAT_{HARV}	Margalef's Richness	2016	**No** (two-factor ANOVA, p=0.93)	**No** (two-factor ANOVA, p=0.54)	**No** (two-factor ANOVA, p=0.25)
KER A_{REF} vs. KER B_{HARV}	Margalef's Richness	2016	**No** (two-factor ANOVA, p=0.88)	**No** (two-factor ANOVA, p=0.61)	**No** (two-factor ANOVA, p=0.37)
PAN_{REF} vs. BAT_{HARV}	Margalef's Richness	2017	**No** (two-factor ANOVA, p=0.059)	**Yes** (two-factor ANOVA, p=0.022: higher for BAT than PAN at the downstream site (Sidak's Multiple Comparisons, p=0.0071))	**Yes** – BAT only (two-factor ANOVA, p=0.0005: higher middle-reach and downstream than upstream (Tukey's Multiple Comparisons, p=0.022, 0.0003))
KER A_{REF} vs. KER B_{HARV}	Margalef's Richness	2017	**No** (two-factor ANOVA, p=0.48)	**No** (two-factor ANOVA, p=0.31)	**No** (two-factor ANOVA, p=0.075)
PAN_{REF} vs. BAT_{HARV}	Shannon's Diversity Index	2016	**No** (two-factor ANOVA, p=0.76)	**No** (two-factor ANOVA, p=0.28)	**Yes** (two-factor ANOVA, p=0.0007: higher downstream than upstream for PAN (Tukey's Multiple Comparisons, p=0.010) and

					middle-reach and downstream than upstream (p=0.041, 0.005)
KER A_{REF} vs. KER B_{HARV}	Shannon's Diversity Index	2016	**No** (two-factor ANOVA, p=0.87)	**No** (two-factor ANOVA, p=0.10)	**No** (two-factor ANOVA, p=0.13)
PAN_{REF} vs. BAT_{HARV}	Shannon's Diversity Index	2017	**No** (two-factor ANOVA, p=0.83)	**No** (two-factor ANOVA, p=0.25)	**No** (two-factor ANOVA, p=0.28)
KER A_{REF} vs. KER B_{HARV}	Shannon's Diversity Index	2017	**Yes** (two-factor ANOVA, p=0.037)	**No** (multiple t-tests, p>0.05)	**Yes** – KER A only (one-factor ANOVA, p<0.0001: higher upstream than middle-reach and downstream (Tukey's Multiple Comparisons, p=0.0001, 0.0003)

Appendix Table 28: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for % Ephemeroptera, Plecoptera, and Trichoptera (EPT) in 2016 and 2017, as determined via AIC_c model selection ($\Delta AIC_c < 7$). The conditional R^2 in 2016 and 2017 were 0.73 and 0.33, respectively. Statistically significant EVs are bolded.

Response Variable		Explanatory Variables							
		Treatment	DOM_PC2	Road	% Organic	% EVSA	Site	DOM_PC1	Treat:Site
% EPT 2016	Slope Coefficient	0.32	0.16	0.15	0.09	-0.05	-0.05		
	95% Confidence Interval	-0.07 to 0.72	-0.13 to 0.46	-0.16 to 0.46	-0.25 to 0.43	-0.35 to 0.24	-0.33 to 0.23		
	Relative Variable Importance	0.26	0.10	0.09	0.06	0.05	0.05		
% EPT 2017	Slope Coefficient	**0.36**	0.20	-0.15		-0.27	0.28	-0.09	0.22
	95% Confidence Interval	**0.06 to 0.66**	-0.09 to 0.50	--0.50 to 0.20		-0.57 to 0.03	-0.002 to 0.56	-0.51 to 0.33	-0.054 to 0.50
	Relative Variable Importance	**0.33**	0.15	0.09		0.23	0.33	0.08	0.01

[1]Treatment = % sub-catchment area harvested within the last 5 years; DOM_PC2 = scores for DOM PCA axis 2; Road = road density (m per ha sub-catchment); % Organic = % organic of fine sediment; % EVSA = % effective variable source area compared to sub-catchment area; Site = distance (km) to farthest downstream site within catchment multiplied by -1; DOM_PC1 = scores for DOM PCA axis 1; Treat:Site = interaction between Treatment and Site.

[2]Site in 2016 and 2017 highly correlated (r=0.87, 0.85) with catchment size (not included in full models); Treatment in 2016 and 2017 highly correlated (r=0.77, 0.78) with WC_PC1 (not included in full models); DOM_PC2 in 2016 was highly correlated (r=0.75) with WC_PC1 (not included in full model); % EVSA in 2017 highly correlated (r=-0.70) with Flow (not included in full model); Dep and DOM_PC1 in 2017 highly correlated (r=0.87, 0.75) with WC_PC1 (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4) in 2016 and 2017, so % Dec removed from full models; Road and DOM_PC1 had high VIFs (>4) in 2016, so DOM_PC1 removed from full models; Road and Dep had high VIFs (>4) in 2017, so Road removed from full model.

Appendix Table 29: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for Margalef's Richness in 2016 and 2017, as determined via AIC_c model selection ($\Delta AIC_c < 7$). The conditional R^2 in 2016 and 2017 were 0.66 and 0.68, respectively. Statistically significant EVs are bolded.

Response Variable		Explanatory Variables							
		% EVSA	Site	Treatment	DOM_PC2	Road	DOM_PC1	Treat:Site	**% Organic**
Margalef's Richness 2016	Slope Coefficient	**0.51**	0.44	-0.33	-0.26	-0.16	0.04	0.01	
	95% Confidence Interval	**-1.01 to -0.006**	-0.08 to 0.10	-0.82 to 0.15	-0.76 to 0.23	-0.73 to 0.40	-0.53 to 0.60	-0.54 to 0.56	
	Relative Variable Importance	**0.21**	0.34	0.25	0.18	0.13	0.11	<0.01	
Margalef's Richness 2017	Slope Coefficient	-0.30	0.35	-0.19	-0.19	-0.16			**0.36**
	95% Confidence Interval	-0.63 to 0.024	-0.001 to 0.71	-0.64 to 0.27	-0.53 to 0.15	-0.57 to 0.25			**0.10 to 0.63**
	Relative Variable Importance	0.29	0.33	0.10	0.12	0.08			**0.54**

[1]% EVSA = % effective variable source area compared to sub-catchment area; Site = distance (km) to farthest downstream site within catchment multiplied by -1; Treatment = % sub-catchment area harvested within the last 5 years; DOM_PC2 = scores for DOM PCA axis 2; Road = road density (m per ha sub-catchment); DOM_PC1 = scores for DOM PCA axis 1; Treat:Site = Interaction between Treatment and Site; % Organic = % organic of fine sediment.

[2]Site in 2016 and 2017 highly correlated (r=0.87, 0.85) with catchment size (not included in full models); Treatment in 2016 and 2017 highly correlated (r=0.77, 0.78) with WC_PC1 (not included in full models); DOM_PC2 in 2016 was highly correlated (r=0.75) with WC_PC1 (not included in full model); % EVSA in 2017 highly correlated (r=-0.70) with Flow (not included in full model); Dep and DOM_PC1 in 2017 highly correlated (r=0.87, 0.75) with WC_PC1 (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4) in 2016 and 2017, so % Dec removed from full models; Road and DOM_PC1 had high VIFs (>4) in 2016 and 2017, so DOM_PC1 removed from full models; Interaction between Treatment and Site, Road, and Dep had high VIFs (>4) in 2017, so Interaction and Dep removed from full model.

Appendix Table 30: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for Shannon's Diversity Index in 2016 and 2017, as determined via AIC_c model selection ($\Delta AIC_c < 7$). The conditional R^2 in 2016 and 2017 were 0.68 and 0.44, respectively. Statistically significant EVs are bolded.

Response Variable		Explanatory Variables						
		% EVSA	**Site**	Treatment	Road	DOM_PC1	DOM_PC2	% Organic
Shannon's Diversity Index 2016	Slope Coefficient	**-0.24**	**0.22**	-0.15	-0.13	0.13	-0.03	
	95% Confidence Interval	**-0.42 to -0.05**	**0.02 to 0.42**	-0.35 to 0.05	-0.35 to 0.08	-0.10 to 0.36	-0.25 to 0.18	
	Relative Variable Importance	**0.34**	**0.28**	0.07	0.08	0.08	0.04	
Shannon's Diversity Index 2017	Slope Coefficient	**-0.17**	**0.15**	0.10	-0.04	-0.11	0.06	0.11
	95% Confidence Interval	**-0.27 to -0.06**	**0.04 to 0.25**	-0.03 to 0.22	-0.14 to 0.06	-0.25 to 0.03	-0.06 to 0.19	-0.0002 to 0.22
	Relative Variable Importance	**0.39**	**0.37**	0.05	0.02	0.04	0.02	0.13

[1]% EVSA = % effective variable source area compared to sub-catchment area; Site = distance (km) to farthest downstream site within catchment multiplied by -1; Treatment = % sub-catchment area harvested within the last 5 years; Road = road density (m per ha sub-catchment); DOM_PC1 = scores for DOM PCA axis 1; DOM_PC2 = scores for DOM PCA axis 2; % Organic = % organic of fine sediment.
[2]Site in 2016 and 2017 highly correlated (r=0.87, 0.85) with catchment size (not included in full models); Treatment in 2016 and 2017 highly correlated (r=0.77, 0.78) with WC_PC1 (not included in full models); DOM_PC2 in 2016 was highly correlated (r=0.75) with WC_PC1 (not included in full model); % EVSA in 2017 highly correlated (r=-0.70) with Flow (not included in full model); Dep and DOM_PC1 in 2017 highly correlated (r=0.87, 0.75) with WC_PC1 (not included in full model); % Dec (% of deciduous tree species) and % EVSA had high VIFs (>4) in 2016 and 2017, so % Dec removed from full models; Road and DOM_PC1 had high VIFs (>4) in 2016, so DOM_PC1 removed from full model; Interaction between Treatment and Site, Road, and Dep had high VIFs (>4) in 2017, so Interaction and Dep removed from full model.

Appendix Table 31: List of "rare" taxa (i.e., present at less than 10% of sites) excluded from NMDS analysis of leaf litter invertebrate community structure in 2016 (total possible N=90) and 2017 (total possible N=94). See Tables A22 to A23 above for complete list of taxa and invertebrate counts.

Rare Taxa 2016	Rare Taxa 2017
• *Baetis* (early instar) • *Rhithrogena* • *Ephemera* • *Pteronarcys* • *Oecetis* • *Hydroptila* • *Nyctophylax* • *Hesperophylax* • *Diplectrona* • *Ptilostomis* • Chironomidae (pupae) • *Clinocera* • *Oreogeton* • Empididae (other) • Simuliidae (pupae) • *Tabanidae* • *Dolichopodidae* • *Antocha* • *Molophilus* • *Holorusia* • *Pedicia* • *Pilaria* • Gomphidae • Hirudinea • Dysticidae • *Sialis*	• *Baetis* (early instar) • *Baetis brunniecolour* • *Caenis* • *Epeorus* • *Ephemera* • *Pteronarcys* • *Acroneuria* • Chloroperlidae (broken) • Perlodidae (broken/immature) • *Oecetis* • *Hydroptila* • *Nyctiophylax* • *Hesperophylax* • *Pseudostenophylax* • *Phylocentropus* • Hydropsychidae (other) • *Hydropsyche* • *Diplectrona* • *Rhyacophila fuscula* • *Clinocera* • *Oreogeton* • Simuliidae (pupae) • Tabanidae • *Dolichopodidae* • *Antocha* • *Hexatoma* • *Holorusia* • *Pedicia* • *Pilaria* • Gomphidae • *Cordulegaster* • Hirudinea • Dystiscidae • *Sialis* • *Gammarus* • Coenagrionidae
Total # Excluded: 26 **Total # Included: 64**	**Total # Excluded: 36** **Total # Included: 58**

A.8. Hydrogen Stable Isotopes

Appendix Table 32: Average (± SD, N=1-3) hydrogen isotope values (δ^2H, ‰) for stream water, Hydropsychids (H), leaves, biofilm, algae (calculated – not measured), coarse sediment (CS), and fine sediment (FS) in October 2016.

Site Name	Avg Water δ^2H (± SD, ‰)	Avg H δ^2H (± SD, ‰)	Avg Leaf δ^2H (± SD, ‰)	Avg Biofilm δ^2H (± SD, ‰)	Avg Algae δ^2H (± SD, ‰)	Avg CS δ^2H (± SD, ‰)	Avg FS δ^2H (± SD, ‰)
PAN 3	-62.85 (0.25)	No data	-106.68 (2.44)	-111.67 (2.08)	-232.71 (0.25)	-92.25 (5.16)	-89.34 (2.52)
PAN 2	-72.65 (0.30)	-110.77 (8.79)	-115.02 (2.04)	-142.17 (2.91)	-242.74 (0.30)	-58.26 (6.35)	-76.44 (2.75)
PAN 1	-69.85 (0.26)	-104.94 (1.52)	-118.27 (2.90)	-103.91 (1.07)	-239.89 (0.26)	-71.03 (2.37)	-74.40 (1.47)
BAT 3	-75.24 (0.32)	-98.17 (10.35)	-119.16 (3.38)	-114.75 (0.52)	-245.29 (0.32)	-95.56 (8.33)	-100.09 (2.97)
BAT 2	-85.79 (0.25)	-133.90 (8.76)	-127.58 (5.10)	-139.44 (3.81)	-255.90 (0.25)	-104.14 (6.96)	-101.60 (0.77)
BAT 1	-83.29 (0.94)	-117.10 (8.72)	-112.40 (3.48)	-120.73 (6.89)	-253.06 (0.94)	-80.91 (15.28)	-65.43 (10.44)
KER 3	-68.71 (0.58)	-114.84 (10.90)	-109.49 (1.05)	-138.12 (1.82)	-239.02 (0.58)	-76.01 (5.42)	-83.45 (2.89)
KER 2	-69.79 (0.41)	-107.99 (6.43)	-117.08 (3.42)	-134.45 (0.60)	239.84 (0.41)	-73.62 (1.48)	-79.69 (2.03)
KER 1	-69.72 (0.68)	-138.19 (5.21)	-122.16 (1.33)	-119.15 (3.05)	-239.67 (0.68)	-76.74 (11.49)	-77.29 (1.61)
KER 6	-63.39 (0.26)	-118.67 (11.46)	-119.28 (0.73)	-135.93 (1.44)	-233.76 (0.26)	-73.53 (5.82)	-78.10 (2.91)
KER 5	-66.15 (0.17)	-108.31 (7.53)	-119.07 (2.50)	-117.92 (1.92)	-236.25 (0.17)	-71.49 (11.73)	-88.64 (2.83)
KER 4	-67.49 (0.36)	-129.52 (9.44)	-127.29 (1.33)	-110.85 (1.33)	-237.69 (0.36)	-78.65 (11.39)	-89.70 (0.55)

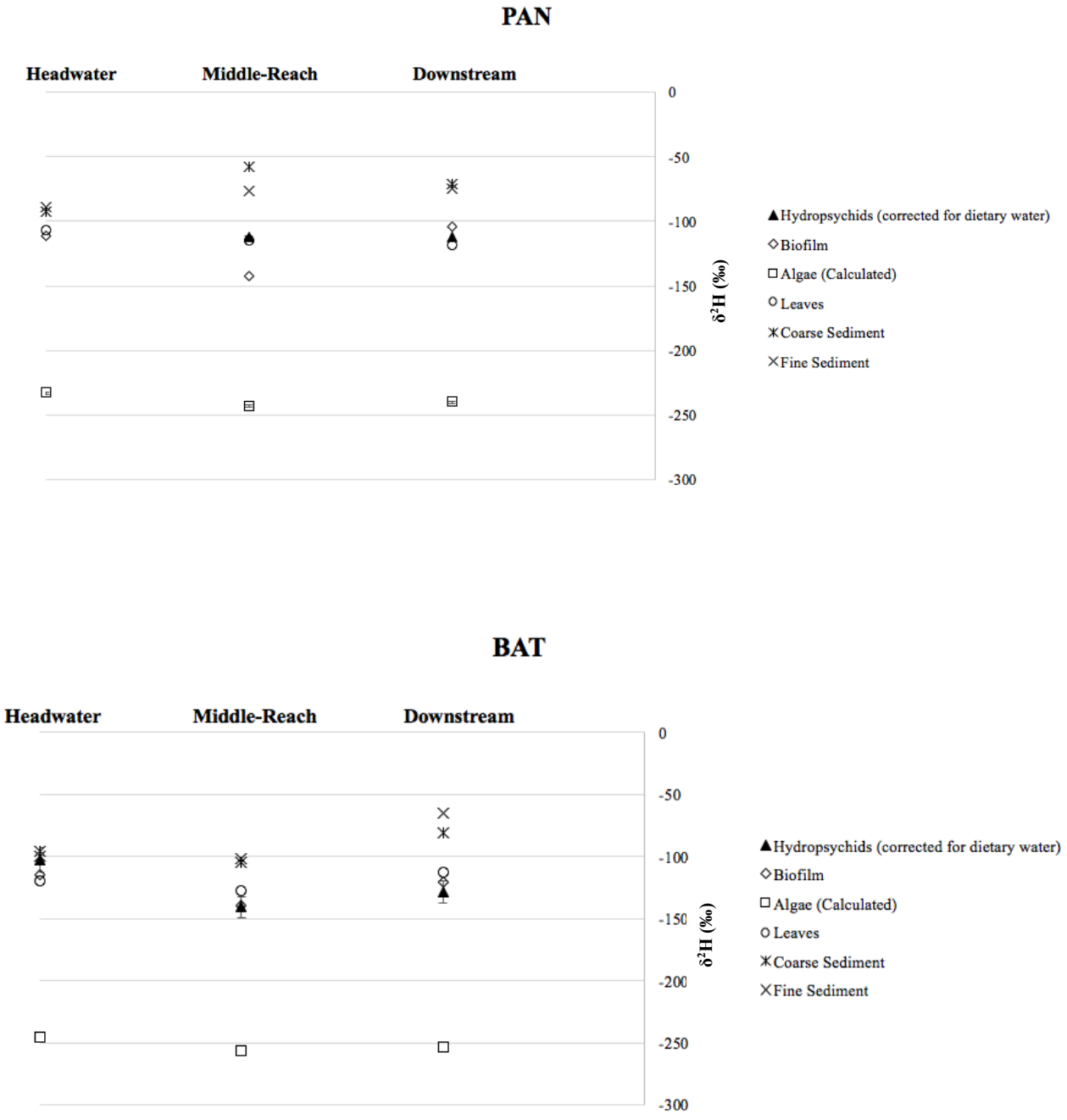
Appendix Table 33: Average (± SD, N=1-3) hydrogen isotope values (δ^2H, ‰) for stream water, Hydropsychids (H), leaves, biofilm, and algae (calculated – not measured) in August, September, and October 2017.

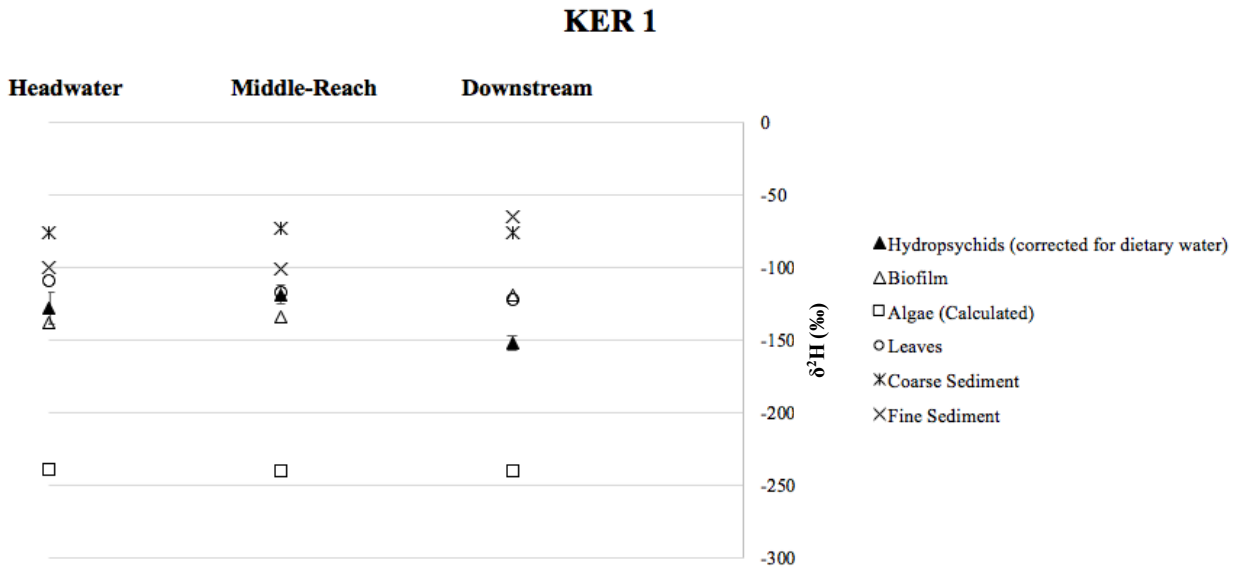
Site Name	Avg Water δ^2H (± SD, ‰)	Avg H δ^2H (± SD, ‰)	Avg Leaf δ^2H (± SD, ‰)	Avg Biofilm δ^2H (± SD, ‰)	Avg Algae δ^2H (± SD, ‰)
August 2017					
PAN 3	-69.48 (0.10)	-115.64 (1.38)	-111.34 (2.94)	-115.64 (1.38)	-239.48 (0.10)
PAN 2	-74.09 (0.30)	-116.28 (0.59)	-112.54 (2.45)	-131.21	-244.09 (0.30)
PAN 1	-74.44 (0.46)	-120.46	-102.76 (1.23)	-104.56	-244.44 (0.46)
BAT 3	-80.97 (0.69)	-106.79 (3.95)	-111.55 (2.14)		-250.97 (0.69)
BAT 2	-83.81 (0.48)	-149.40	-117.12 (5.96)	-129.27 (7.17)	-253.81 (0.48)
BAT 2new*	-84.38 (0.45)	-134.07 (0.68)	-117.67 (3.29)	-144.95 (3.88)	-254.38 (0.45)
BAT 1	-84.55 (0.64)	-115.75 (0.57)	-112.22 (1.97)	-148.99	-254.55 (0.64)
KER 3	-68.47 (0.06)	-109.15 (1.14)	-115.74 (5.43)	-117.24 (0.27)	-238.47 (0.06)
KER 2	-73.71 (0.42)	-107.18 (2.95)	-117.12 (2.91)	-119.43 (3.89)	-243.71 (0.42)
KER 1	-75.04 (0.24)	-126.14 (1.47)	-118.87 (2.11)	-150.18 (1.94)	-245.04 (0.24)
KER 6	-70.18 (0.27)	-112.71 (5.93)	-108.87 (2.11)	-128.80 (0.46)	-240.18 (0.27)
KER 5	-73.31 (0.84)	-115.23 (3.07)	-108.07 (1.23)	-116.81 (0.58)	-243.31 (0.84)
KER 4	-73.70 (0.54)	-116.40 (1.70)	-114.63 (0.98	-120.45 (1.74)	-243.70 (0.54)

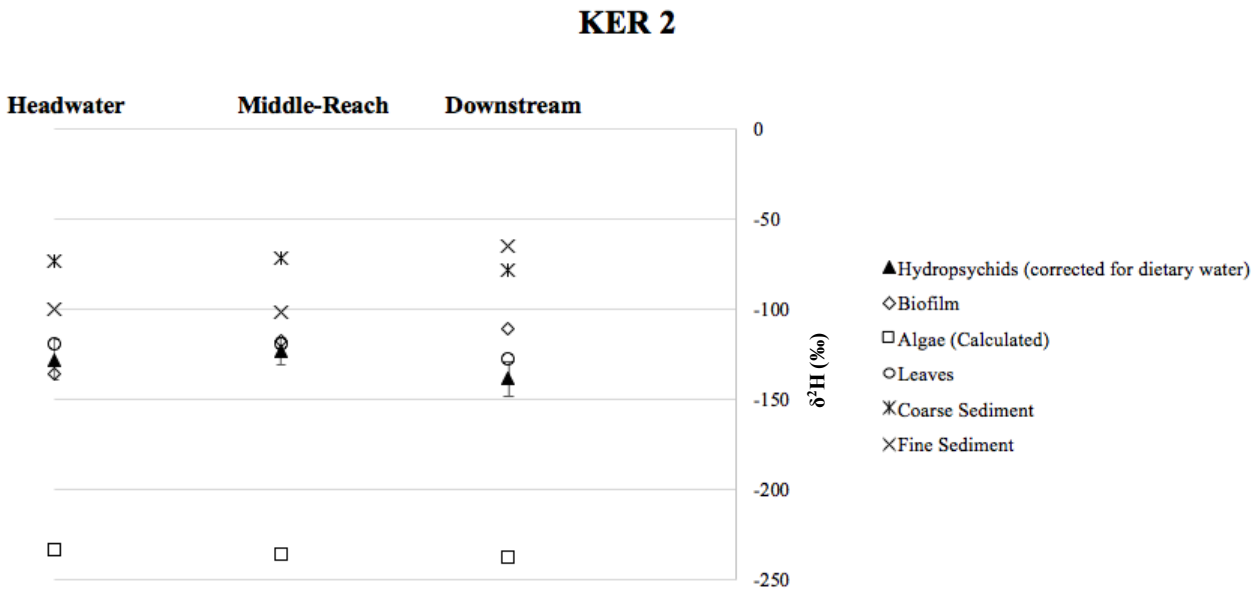
September 2017					
PAN 3	-73.04 (0.45)	-102.72	-111.03 (2.38)	-123.11	-243.04 (0.45)
PAN 2	-74.52 (0.34)	-109.00	-103.96 (1.89)	-134.42 (1.28)	-244.52 (0.34)
PAN 1	-76.84 (0.39)	-117.98 (6.89)	-115.69 (6.06)	-127.92 (2.52)	-246.84 (0.39)
BAT 3	-82.34 (0.17)	-98.27 (0.84)	-112.64 (2.97)	-119.57 (0.26)	-252.34 (0.17)
BAT 2	-87.33 (0.54)	-131.71 (13.45)	-119.99 (0.68)	-130.33 (2.94)	-257.33 (0.54)
BAT 2new*	-84.17 (0.09)	-145.77 (10.65)	-137.06 (2.62)	-147.06 (1.09)	-254.17 (0.09)
BAT 1	-84.53 (0.14)	-123.05 (4.10)	-108.81 (2.02)	-137.48 (0.94)	-254.53 (0.14)
KER 3	-68.12 (0.77)	-105.48 (3.66)	-106.28 (2.29)	-127.95 (1.19)	-238.12 (0.77)
KER 2	-73.04 (0.46)	-113.19 (3.41)	-114.83 (1.38)	-118.17 (2.21)	-243.04 (0.46)
KER 1	-73.88 (0.37)	-121.75 (2.79)	-118.21 (2.19)	-123.85 (2.34)	-243.88 (0.37)
KER 6	-70.01 (0.15)	-108.60 (4.22)	-99.83 (1.82)	-123.48 (1.17)	-240.01 (0.15)
KER 5	-71.89 (0.18)	-123.16 (4.91)	-106.17 (3.47)	-104.48 (0.80)	-241.89 (0.18)
KER 4	-74.93 (0.63)	-110.74 (2.92)	-117.59 (0.70)	-115.08 (3.02)	-244.93 (0.63)
October 2017					
PAN 3	-66.43 (3.34)	-104.27 (6.12)	-120.52 (1.33)	-117.07 (0.69)	-236.43 (3.34)
PAN 2	-69.95 (0.25)	-108.33 (3.44)	-109.12 (5.09)	-117.02	-239.95 (0.25)
PAN 1	-67.87 (2.54)	-106.62 (1.62)	-117.84 (4.62)	-114.15 (2.10)	-237.87 (2.54)
BAT 3	-71.42 (0.29)	-91.06 (7.79)	-116.96 (2.06)	-120.35 (2.87)	-241.42 (0.29)

BAT 2	-79.03 (0.40)	-151.89 (1.80)	-121.15 (1.39)	-149.72 (5.09)	-249.03 (0.40)
BAT 2new*	-77.14 (0.39)	-139.75 (4.79)	-122.90 (2.53)	-141.05 (3.66)	-247.14 (0.39)
BAT 1	-76.53 (0.35)	-121.93 (0.27)	-124.69 (2.06)	-120.77 (1.61)	-246.53 (0.35)
KER 3	-61.27 (12.17)	-102.33 (7.40)	-113.92 (0.52)	-106.96	-231.27 (12.17)
KER 2	-66.35 (0.30)	-110.73 (0.95)	-113.34 (2.36)	-116.53 (0.78)	-236.35 (0.30)
KER 1	-67.20 (0.50)	-118.63	-121.45 (3.09)	-107.59 (0.14)	-237.20 (0.50)
KER 6	-67.45 (0.35)	-108.14 (8.68)	-113.14 (3.99)	108.57 (2.75)	-237.45 (0.35)
KER 5	-67.43 (0.66)	-109.89 (5.67)	-123.89 (1.27)	-96.65 (0.62)	-237.43 (0.66)
KER 4	-67.37 (0.18)	-120.34 (3.08)	-131.71 (1.95)	-104.10 (0.64)	-237.37 (0.18)

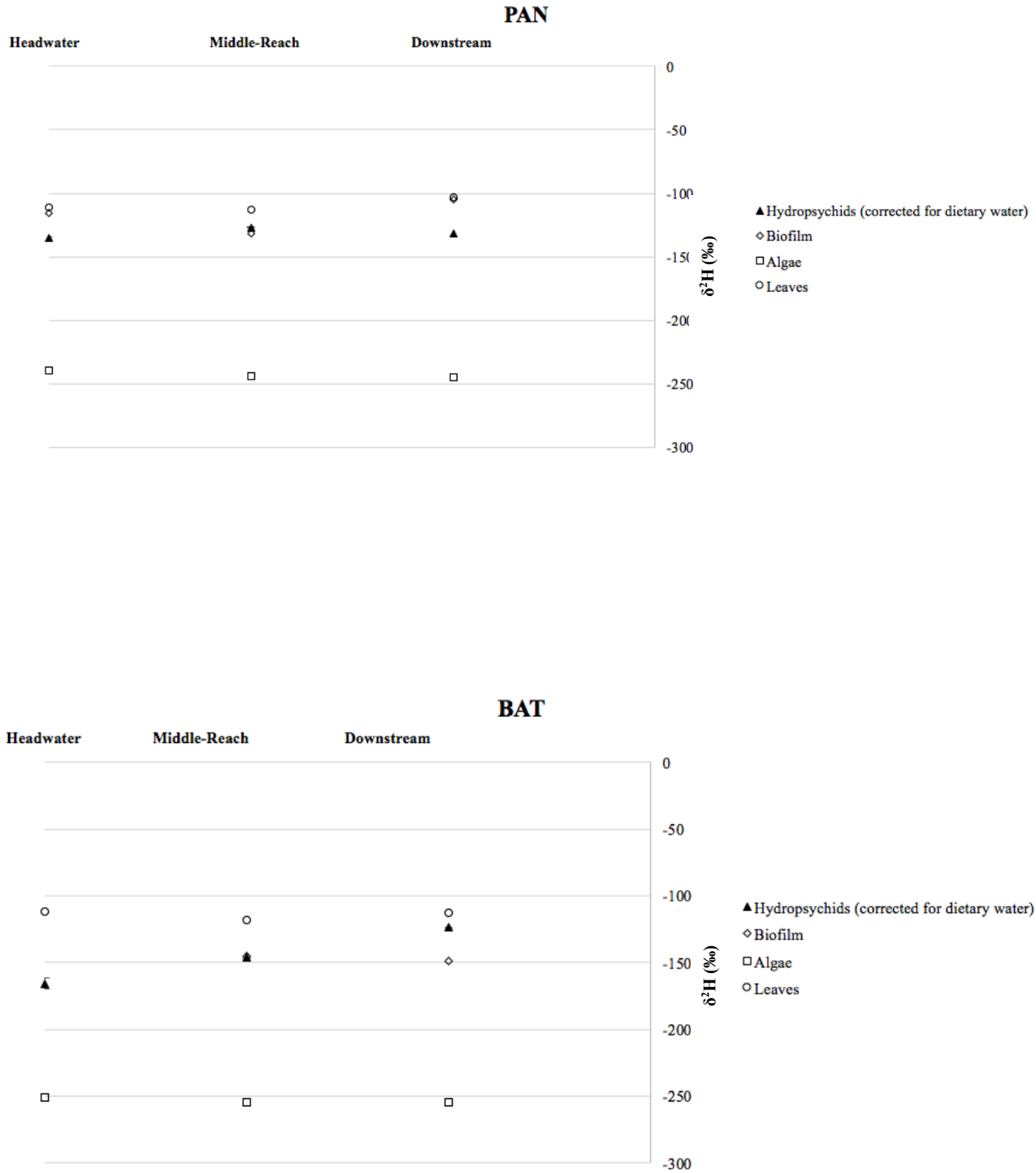
Appendix Figure 4: Average (± SD) hydrogen isotope values (δ^2H, ‰) for consumer (Hydropsychids) and all possible food sources in each catchment in October 2016.

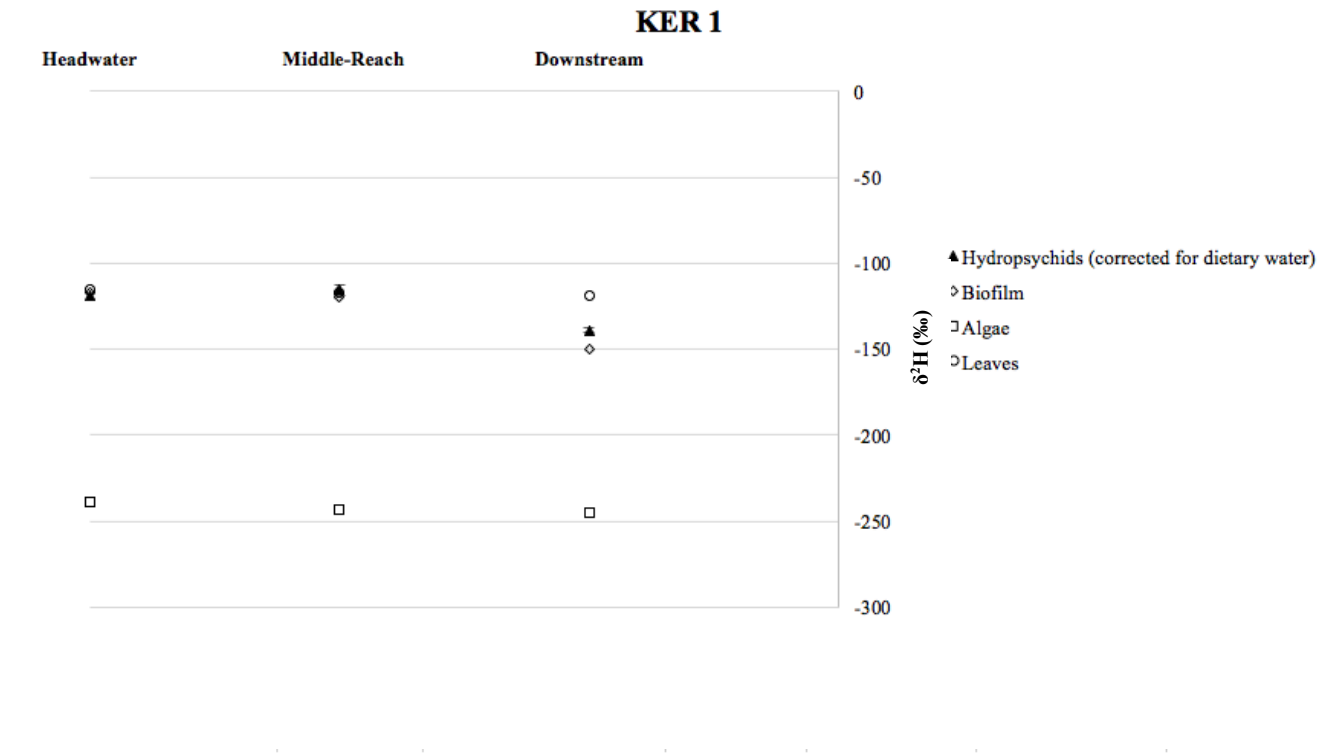


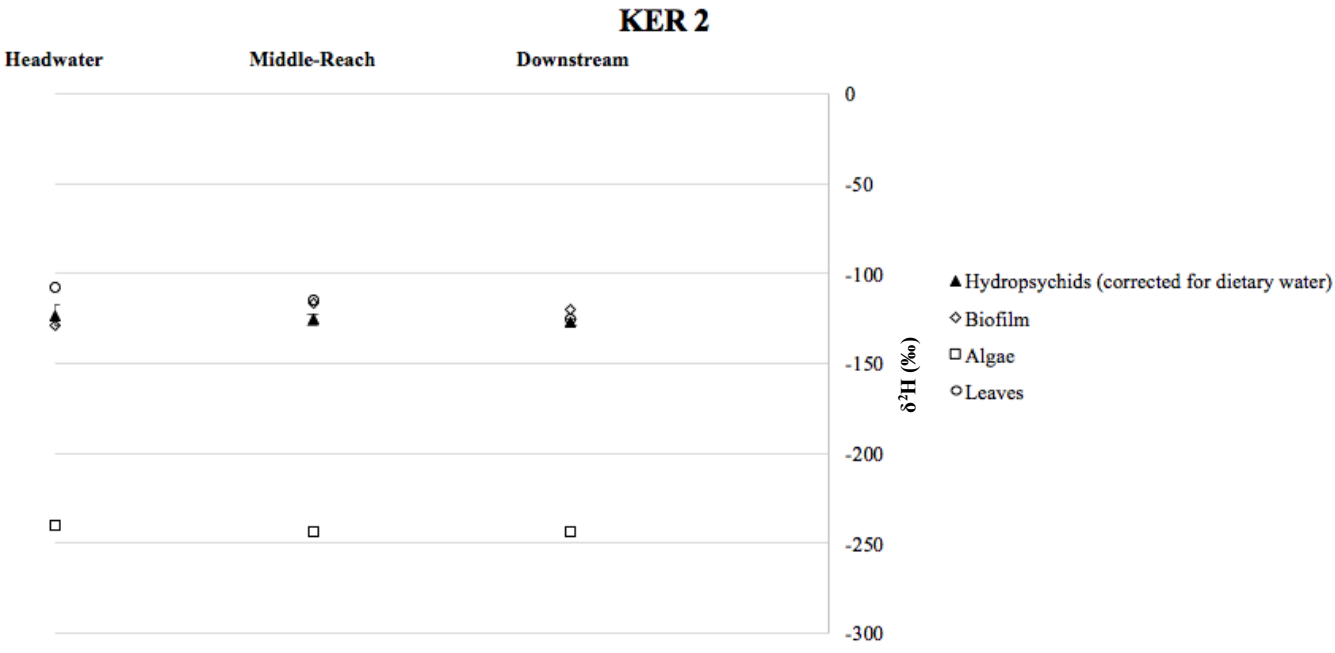

KER 1
Headwater
Middle-Reach
Downstream
0
-50
-100
-150
-200
-250
-300
δ²H (‰)
▲Hydropsychids (corrected for dietary water)
△Biofilm
□Algae (Calculated)
○Leaves
✳Coarse Sediment
×Fine Sediment


KER 2
Headwater
Middle-Reach
Downstream
0
-50
-100
-150
-200
-250
δ²H (‰)
▲Hydropsychids (corrected for dietary water)
◇Biofilm
□Algae (Calculated)
○Leaves
✳Coarse Sediment
×Fine Sediment

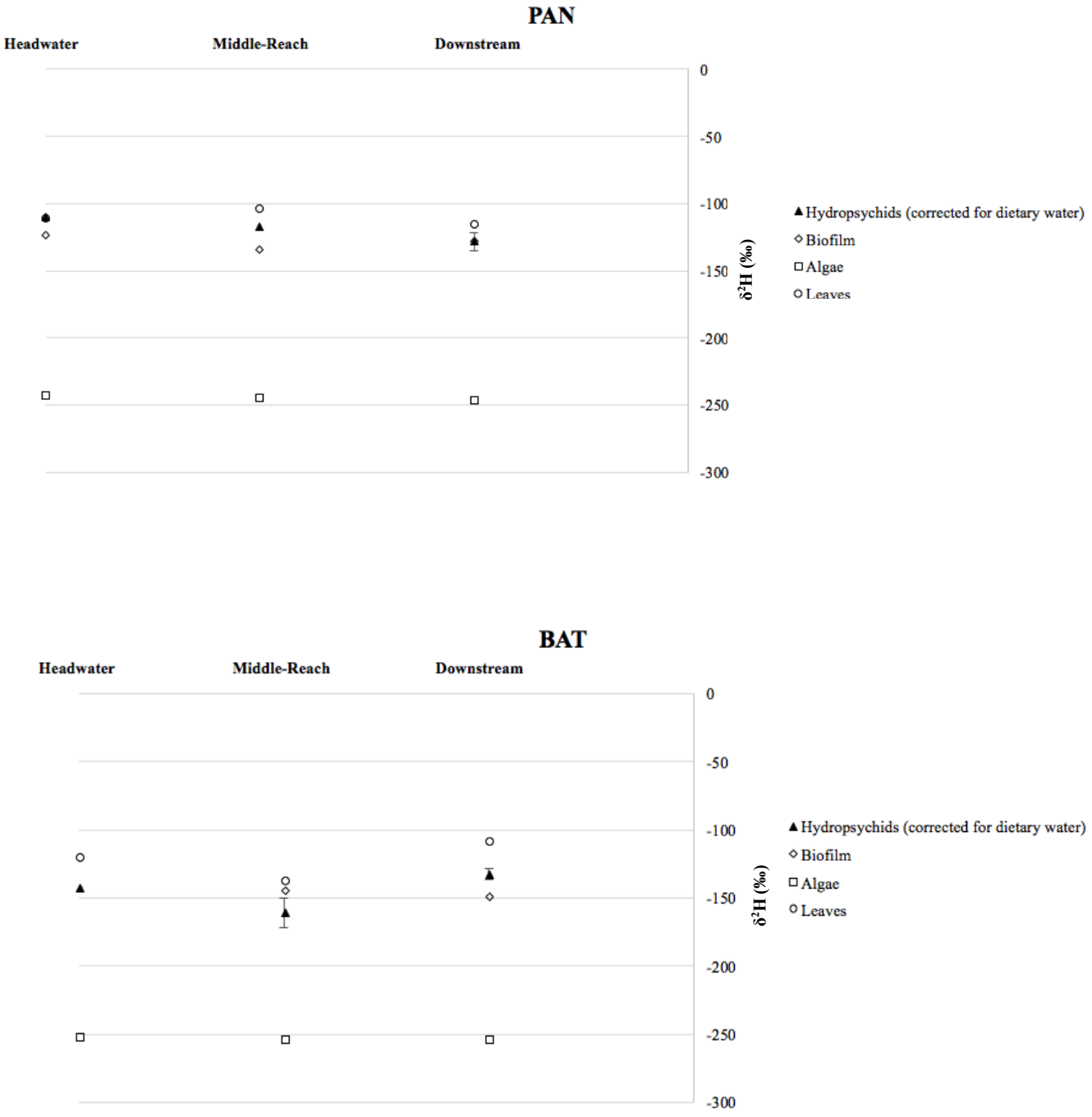
Appendix Figure 5: Average (± SD) hydrogen isotope values (dD, ‰) for consumer (Hydropsychids) and all possible food sources in each catchment in August 2017.

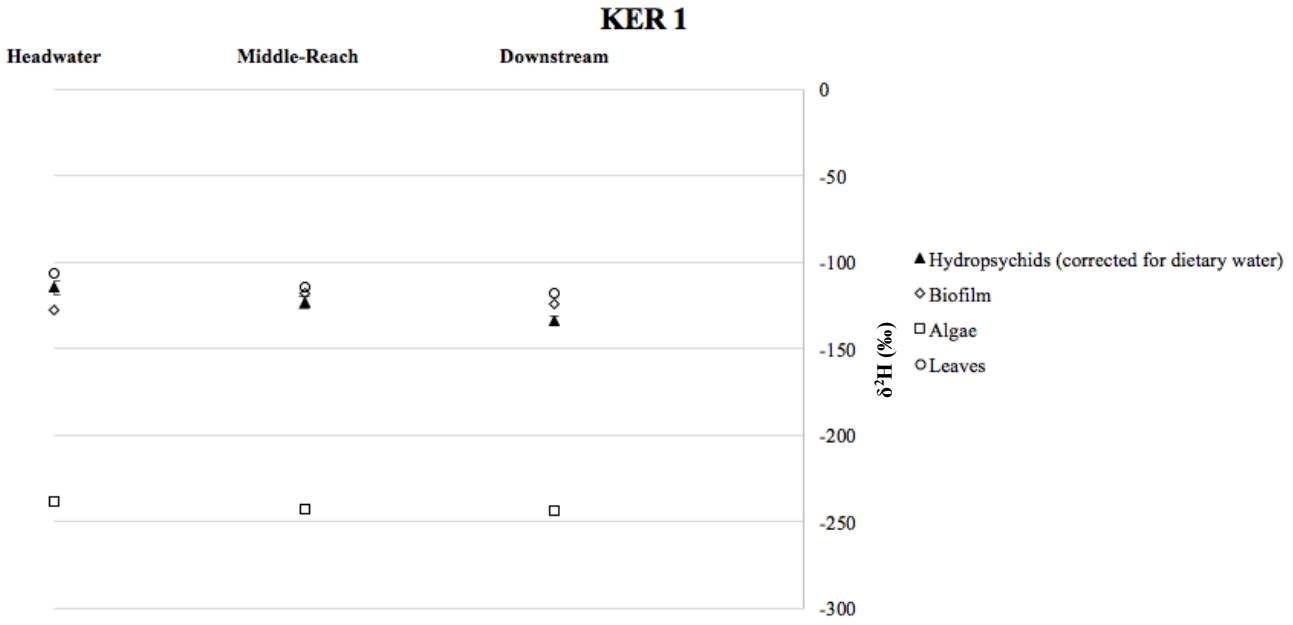


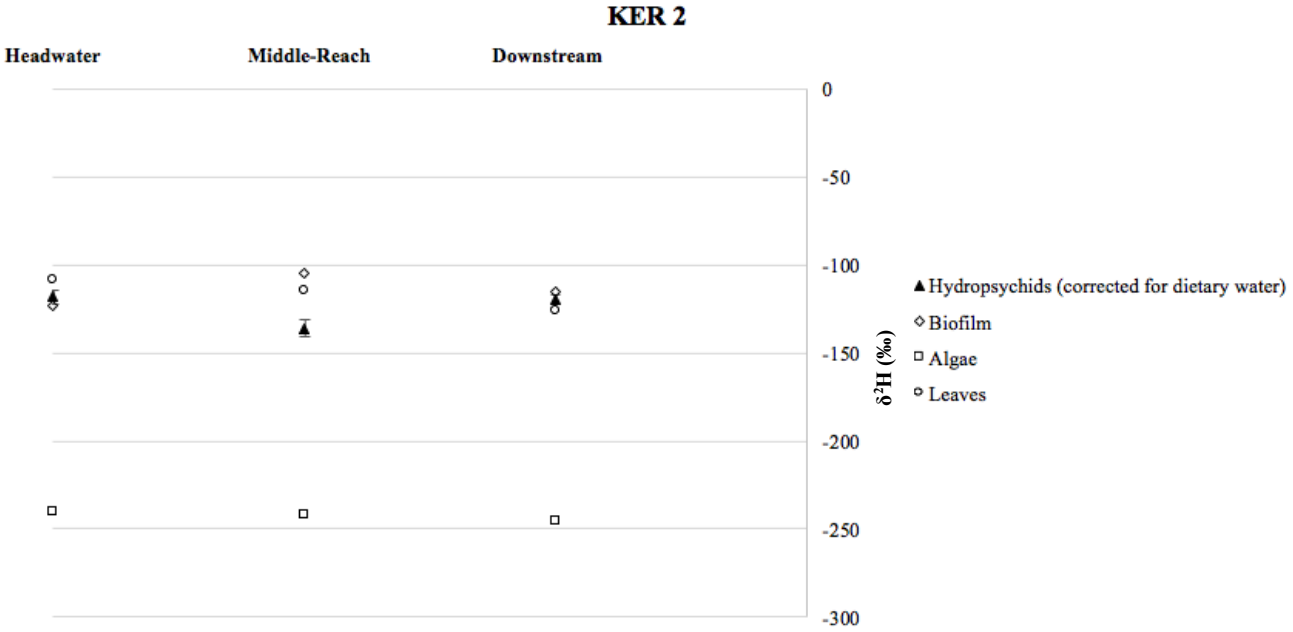

KER 1
Headwater
Middle-Reach
Downstream
0
-50
-100
-150
-200
-250
-300
δ²H (‰)
Hydropsychids (corrected for dietary water)
Biofilm
Algae
Leaves


KER 2
Headwater
Middle-Reach
Downstream
0
-50
-100
-150
-200
-250
-300
δ²H (‰)
Hydropsychids (corrected for dietary water)
Biofilm
Algae
Leaves

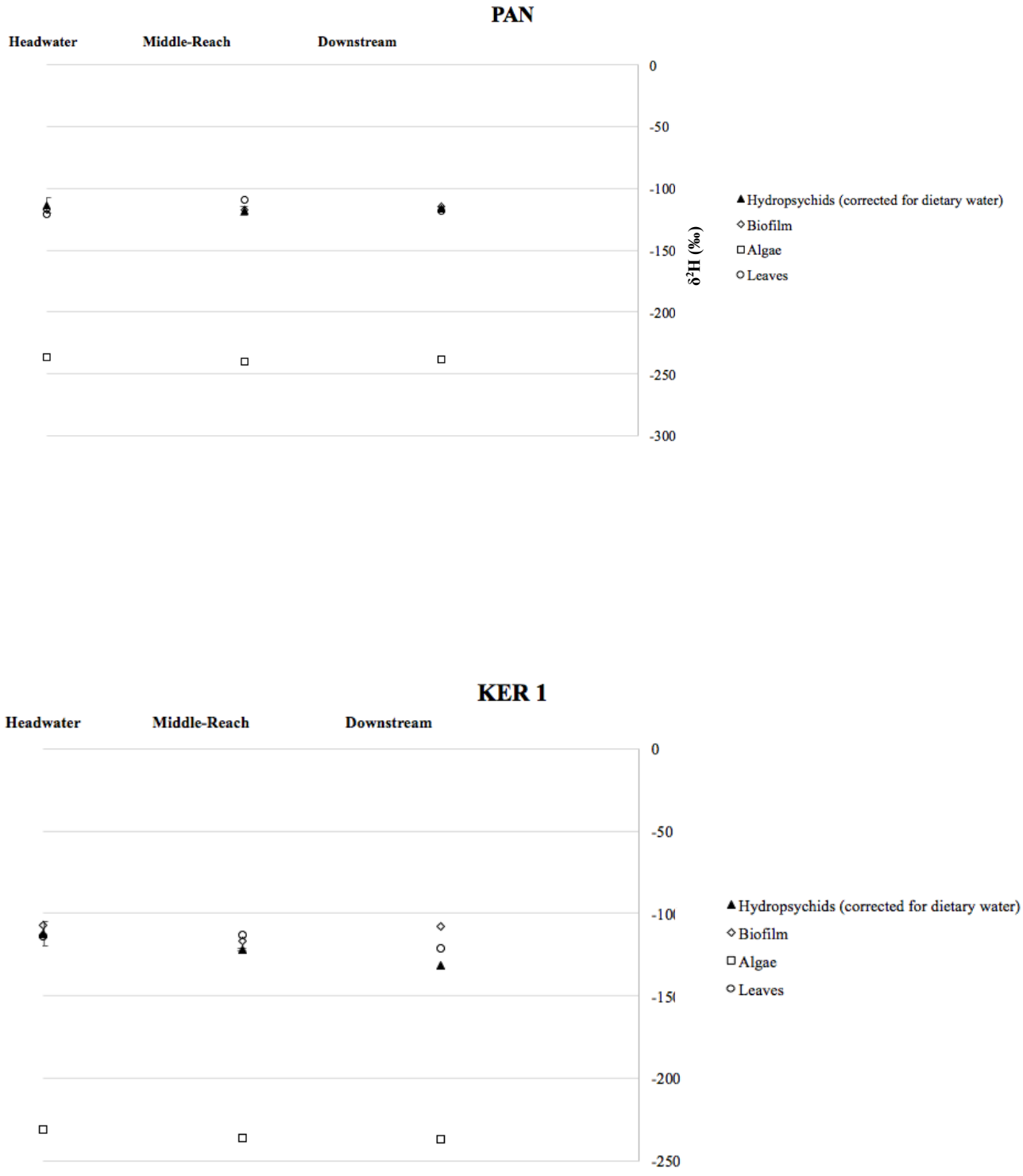
Appendix Figure 6: Average (±SD) hydrogen isotope values (dD, ‰) for consumer (Hydropsychids) and all possible food sources in each catchment in September 2017.

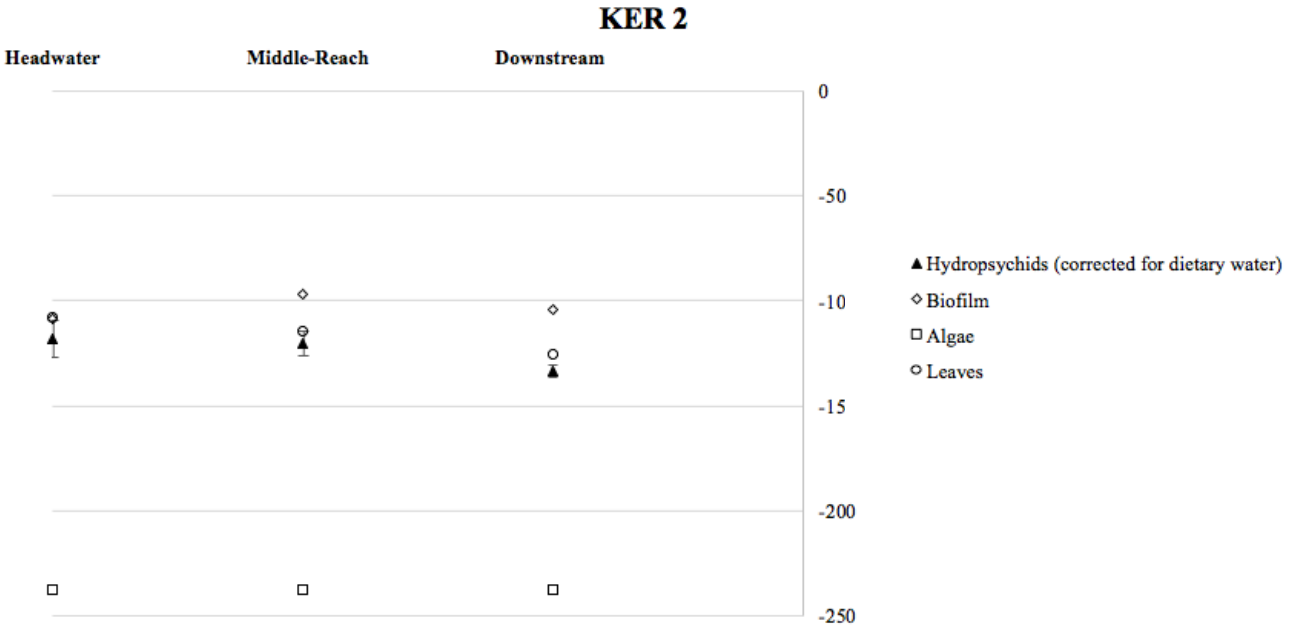



KER 1
Headwater
Middle-Reach
Downstream
0
-50
-100
-150
-200
-250
-300
δ²H (‰)
Hydropsychids (corrected for dietary water)
Biofilm
Algae
Leaves


KER 2
Headwater
Middle-Reach
Downstream
0
-50
-100
-150
-200
-250
-300
δ²H (‰)
Hydropsychids (corrected for dietary water)
Biofilm
Algae
Leaves

Appendix Figure 7: Average (± SD) hydrogen isotope values (dD, ‰) for consumer (Hydropsychids) and all possible food sources in each catchment in October 2017.




KER 2
Headwater
Middle-Reach
Downstream
0
-50
-10
-15
-200
-250
Hydropsychids (corrected for dietary water)
Biofilm
Algae
Leaves

Appendix Table 34: Test results for significance for interaction between treatment and site, effect of treatment, and effect of site for leaf litter %algae excluded from results section. All variables were transformed when necessary to meet the assumption of normal distribution.

Paired-Catchment Comparison	Response Variable	Year	Interaction?	Effect of Treatment?	Effect of Site?
PAN_{REF} vs. BAT_{HARV}	% Algae	Aug 2017	**Yes** (two-factor ANOVA, p=0.0003)	**Yes** (higher for PAN than BAT at upstream (multiple t-tests, p=0.0066))	**Yes** (one-factor ANOVA, p=0.0018, 0.020: higher upstream and downstream than middle-reach for PAN (Tukey's Multiple Comparisons, p=0.0029, 0.0030) and for middle-reach than upstream for BAT (p=0.019))
KER A_{REF} vs. KER B_{HARV}	% Algae	Aug 2017	**Yes** (two-factor ANOVA, p<0.0001)	**Yes** (higher for KER B than KER A at middle-reach (multiple t-tests, p=0.00061), for KER A than KER B at downstream (p<0.0001))	**Yes** (one-factor ANOVA, p<0.0001, 0.026: higher downstream than upstream and middle-reach and upstream than middle-reach for KER A (Tukey's Multiple Comparisons, p<0.0001, <0.0001, 0.0038) and upstream than downstream for KER B (p=0.030))
PAN_{REF} vs. BAT_{HARV}	% Algae	Oct 2017	**Yes** (two-factor ANOVA, p=0.0006)	**Yes** (higher for BAT than PAN at middle-reach and downstream (multiple t-tests, p=0.011, 0.0058))	**Yes** – BAT only (one-factor ANOVA, p=0.0010: higher middle-reach and downstream than upstream and middle-reach than downstream (Tukey's Multiple Comparisons, p=0.0008, 0.012, 0.018))
KER A_{REF} vs. KER B_{HARV}	% Algae	Oct 2017	**No**	**No** (two-factor ANOVA, p-0.12)	**No** (two-factor ANOVA, p=0.98)

(two-factor
ANOVA,
p=0.38)

A.9. Mercury

Appendix Table 35: Average (± SD, N=3) concentration of MeHg in unfiltered (UF) and filtered (F, 0.45 µm) water, average (± SD, N=3) MeHg and Hg(II) in Hydropscyhids (H, µg/kg dw), %MeHg in Hydropsychids, and bioaccumulation factor for MeHg in Hydropscyhids (BAF) in October 2016.

Site Name	Avg UF Water MeHg (± SD, ng/L)	Avg F Water MeHg (± SD, ng/L)	Avg H MeHg (± SD, µg/kg)	Avg H Hg(II) (± SD, µg/kg)	H %MeHg	BAF
PAN 3	0.099 (0.029)	0.10 (0.014)	34.24 (1.47)	80.89 (43.90)	32.56	342327
PAN 2	0.081 (0.011)	0.12 (0.0028)	93.01 (23.66)	60.60 (28.43)	36.81	754104
PAN 1	0.15 (0.0026)	0.13 (0.013)	65.19 (24.13)	84.59 (14.31)	42.65	493817
BAT 3	0.16 (0.0032)	0.17 (0.011)	136.09 (8.21)	31.45 (6.27)	81.20	794059
BAT 2	0.13 (0.0051)	0.12 (0.025)	65.61 (3.51)	157.20 (18.95)	29.59	549583
BAT 1	0.15 (0.036)	0.10 (0.0081)	119.20 (15.81)	151.12 (22.28)	44.13	No Data
KER 3	0.096 (0.027)	0.095 (0.0066)	29.68 (4.98)	41.04 (6.45)	42.08	1147108
KER 2	0.27 (0.13)	0.36 (0.22)	69.80 (2.61)	125.73 (18.12)	35.92	311271
KER 1	0.12 (0.059)	0.19 (0.015)	83.78 (6.15)	108.98 (4.76)	43.45	194470
KER 6	0.40 (0.15)	0.30 (0.11)	100.90 (6.51)	117.74 (15.93)	46.28	448381
KER 5	0.20 (0.047)	0.24 (0.087)	132.14 (6.40)	131.66 (9.69)	50.12	338988
KER 4	0.18 (0.012)	0.16 (0.025)	87.81 (11.07)	82.50 (18.09)	51.80	550115

Appendix Table 36: Number of measurements for average seston Hg in 2017.

Site Name	N (number of measurements included in average for all sites)	Details of N
PAN 3	3	Aug, Sept, Oct 2017
PAN 2	2	Aug, Oct 2017
PAN 1	1	Aug 2017
BAT 3	3	Aug, Sept, Oct 2017
BAT 2	1	Aug 2017
BAT 2new*	2	Aug, Oct 2017
BAT 1	2	Aug, Oct 2017
KER 3	3	Aug, Sept, Oct 2017
KER 2	3	Aug, Sept, Oct 2017
KER 1	1	Aug 2017
KER 6	2	Aug, Oct 2017
KER 5	2	Aug, Oct 2017
KER 4	1	Oct 2017

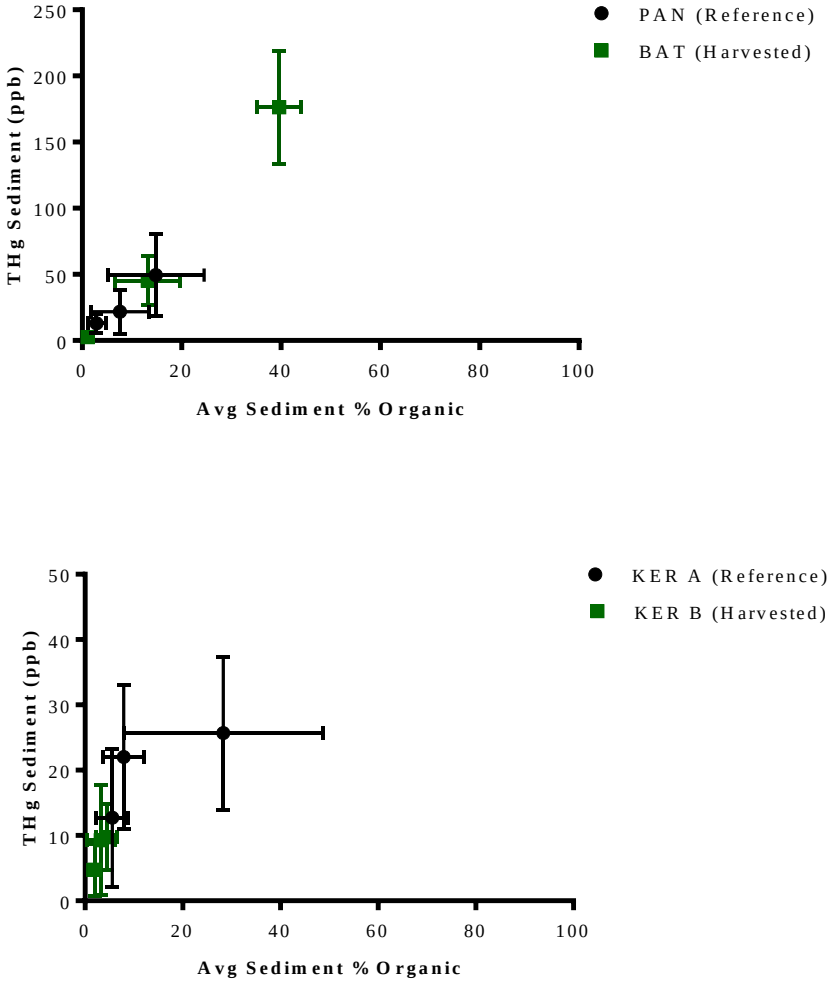
Appendix Table 37: Average (± SD, N=1-3) concentration of MeHg in unfiltered (UF) and filtered (F, 0.45 µm) water, average (± SD, N=1-3) MeHg and Hg(II) in Hydropscyhids (H, µg/kg dw) and seston (S, µg/kg dw), % MeHg in Hydropsychids and seston, THg in sediment (Sed, ppm), bioaccumulation factor for MeHg in filtered water and Hydropsychids (BAF), and biomagnification factor for MeHg in seston and Hydropsychids in August, September, and October 2017.

Site Name	**Avg UF Water MeHg** (± SD, ng/L)	**Avg F Water MeHg** (± SD, ng/L)	**Avg H MeHg** (± SD, µg/kg)	**Avg H Hg(II)** (± SD, µg/kg)	**H % MeHg**	**Avg S MeHg**	**Avg S Hg(II)** (± SD, µg/kg)	**S % MeHg** (± SD, µg/kg)	**THg Sed** (ppm)	**BAF**	**BMF**
August 2017											
PAN 3	0.074 (0.0060)	0.060 (0.0059)	45.71 (7.41)	93.02 (20.90)	32.95	3.26	239.72	1.34	66.79	761110	19.74
PAN 2	0.18	0.14	194.20 (2.64)	238.90 (86.13)	44.84	4.43	794.17	0.55	40.22	1355598	55.01
PAN 1	0.31	0.30	179.74 (24.16)	150.53 (31.02)	54.42	9.39	263.20	3.45	18.17	595493	19.13
BAT 3	0.16 (0.0077)	0.14 (0.0096)	191.37 (12.60)	67.95 (8.31)	73.80	5.43 (0.19)	196.04 (2.073)	2.69	48.01	1375229	39.85
BAT 2	0.18	0.084	137.86	111.83	55.21	4.93	205.88	2.34	24.49	1638599	27.97
BAT 2new*	0.13	0.11	170.11 (10.02)	180.53 (25.62)	48.52	9.87	318.14	3.01	173.62	1535650	26.67
BAT 1	0.12	0.095	185.67 (15.56)	158.11 (13.40)	54.01	7.79	185.20	4.04	4.04	1961018	33.11
KER 3	0.066 (0.011)	0.038	56.17 (4.32)	68.42 (11.88)	45.09	12.05 (11.34)	403.89 (314.56)	2.90	No data	1465995	8.94
KER 2	0.081	0.065	123.30 (9.77)	107.86 (9.11)	53.34	32.90	858.32	3.69	34.63	1885684	8.88
KER 1	0.079 (0.0057)	0.062	98.39 (10.37)	99.44 (37.47)	49.73	2.98	94.47	3.06	5.45	1577532	33.01
KER 6	0.18 (0.0061)	0.19	136.67 (10.03)	112.50 (9.60)	54.85	2.84	92.96	2.96	3.64	718059	51.04
KER 5	0.15	0.18	229.99 (2.29)	156.90 (40.57)	59.44	1.87	49.34	3.65	1.90	1291511	80.22
KER 4	0.16 (0.013)	0.13	137.05 (6.68)	89.50 (5.43)	60.50	No data	No data	No data	13.80	1084133	52.79

September 2017											
PAN 3	0.014 (0.0013)	0.024 (0.0028)	58.22 (2.83)	38.66 (7.39)	60.09	2.93	169.11	1.70	68.17	2398583	25.14
PAN 2	0.12	0.12	192.23 (32.09)	162.57 (58.28)	54.18	No data	No data	No data	16.28	1561675	54.45
PAN 1	0.30	0.27	175.07 (21.93)	138.96 (30.91)	55.75	No data	No data	No data	16.14	655708	18.64
BAT 3	0.40 (0.034)	0.13 (0.0075)	133.72 (7.19)	45.99 (1.40)	74.41	3.98	156.66	2.48	25.57	1035638	27.85
BAT 2	0.10	0.093	131.09 (20.61)	154.07 (26.79)	45.97	No data	No data	No data	43.81	1409239	26.59
BAT 2new*	0.15	0.14	150.84 (22.32)	213.58 (35.92)	41.39	No data	No data	No data	135.05	1069845	23.65
BAT 1	0.11	0.11	153.06 (10.68)	187.17 (28.66)	44.99	No data	No data	No data	1.81	1451381	27.30
KER 3	0.14 (0.085)	0.074 (0.0043)	77.57 (21.04)	126.24 (76.66)	29.14	3.46	149.43	2.26	34.00	1042641	12.35
KER 2	0.14	0.11	109.99 (11.82)	181.75 (50.90)	37.70	6.65	149.60	4.26	14.53	960481	7.93
KER 1	0.10 (0.0089)	0.091 (0.012)	93.17 (2.21)	145.53 (2.26)	39.03	No data	No data	No data	24.81	1025812	31.26
KER 6	0.17 (0.0065)	0.16 (0.0061)	105.70 (2.91)	164.01 (30.52)	39.19	No data	No data	No data	9.20	676814	39.47
KER 5	0.14	0.13	160.49 (10.49)	184.78 (27.68)	46.48	No data	No data	No data	18.47	1195546	55.98
KER 4	0.095 (0.0096)	0.086 (0.0096)	85.35 (2.58)	107.68 (6.43)	44.22	No data	No data	No data	11.33	992816	32.88
October 2017											
PAN 3	0.050 (0.024)	0.088 (0.0088)	36.33 (1.43)	58.74 (4.80)	38.21	0.76	61.49	1.22	13.21	413245	15.69
PAN 2	0.11	0.097	88.50 (3.49)	152.08 (29.44)	36.79	2.63	194.28	1.34	8.57	911319	25.07
PAN 1	0.30	0.26	93.38 (8.42)	163.64 (41.07)	36.33	No data	No data	No data	4.61	363772	9.94
BAT 3	0.13 (0.026)	0.079 (0.0038)	No data	No data	No data	5.00	187.76	2.59	62.13	No data	No data

BAT 2	0.066	0.051	107.45	305.27	26.04	No data	No data	No data	45.24	2101841	21.80
BAT 2new*	0.081	0.063	No data	No data	No data	2.89	146.02	1.94	220.53	No data	No data
BAT 1	0.082	0.076	122.85	708.96	14.77	3.42	115.43	115.43	2.88	1.84	1623078
KER 3	0.043 (0.0034)	0.027 (0.0039)	72.54 (3.92)	118.03 (16.84)	38.06	3.33	104.07	104.07	3.10	17.40	2674355
KER 2	0.10	0.076	68.59 (2.15)	151.71 (11.31)	31.13	2.09	78.19	78.19	2.60	16.94	899835
KER 1	0.083 (0.00068)	0.069 (0.0047)	72.85 (7.01)	146.66 (30.73)	33.19	No data	No data	No data	No data	7.90	1053443
KER 6	0.18 (0.0088)	0.16 (0.0054)	54.51 (9.26)	115.81 (11.06)	32.01	2.52	72.37	72.37	3.36	1.53	331382
KER 5	0.14	0.13	112.83 (2.72)	159.58 (20.48)	41.42	3.87	118.18	118.18	3.17	7.39	850140
KER 4	0.13 (0.0096)	0.11 (0.0041)	95.65 (6.28)	125.75 (30.15)	43.20	2.60	97.21	97.21	2.60	4.01	900090

Appendix Figure 8: Average (± SD, N=3) THg (ppb) in sediment (averaged for three collections in August, September, and October 2017) plotted against the average (± SD, N=3) %organic of that sediment (i.e., loss on ignition (LOI), averaged for 3 collections in August, September, and October in 2017).



Appendix Table 38: Test results for significance for interaction between treatment and site, effect of treatment, and effect of site for mercury response variables excluded from results section. All variables were transformed when necessary to meet the assumption of normal distribution.

Paired-Catchment Comparison	Response Variable	Year	Interaction?	Effect of Treatment?	Effect of Site?
PAN_{REF} vs. BAT_{HARV}	MeHg Unfiltered Water	2016	**Yes** (two-factor ANOVA, p=0.023)	**Yes** (higher for BAT than PAN at middle-reach (multiple t-tests, p=0.010))	**Yes** – PAN only (one-factor ANOVA, p=0.014: higher downstream than middle-reach (Tukey's Multiple Comparisons, p=0.014))
KER A_{REF} vs. KER B_{HARV}	MeHg Unfiltered Water	2016	**Yes** (two-factor ANOVA, p=0.0033)	**Yes** (higher for KER B than KER A at upstream (multiple t-tests, p=0.018))	**Yes** (one-factor ANOVA, p=0.045, 0.020: higher at middle-reach than upstream for KER A (Tukey's Multiple Comparisons, p=0.047), at upstream than middle-reach and downstream for KER B (p=0.043, 0.023)
PAN_{REF} vs. BAT_{HARV}	MeHg Unfiltered Water	2017	**Yes** (two-factor ANOVA, p=0.0008)	**Yes** (higher PAN than BAT at downstream (multiple t-tests, p=0.0002)	**Yes** – PAN only (one-factor ANOVA, p<0.0001: higher at middle-reach and downstream than upstream, and downstream than middle-reach (Tukey's Multiple Comparisons, p=0.0130, <0.0001, 0.0006))
KER A_{REF} vs. KER B_{HARV}	MeHg Unfiltered Water	2017	**No** (two-factor ANOVA, p=0.18)	**Yes** (two-factor ANOVA, p=0.0010: higher KER B than KER A upstream (Sidak's Multiple Comparisons, p=0.0042)	**No** (two-factor ANOVA, p=0.37)

PAN_{REF} vs. BAT_{HARV}	MeHg Hydropsychids	Aug 2017	**Yes** (two-factor ANOVA, p<0.0001)	**Yes** (higher BAT than PAN at upstream (multiple t-tests, p=0.0002) and PAN than BAT middle-reach (p=0.0310))	**Yes** – PAN only (one-factor ANOVA, p=0.0450: higher middle-reach and downstream than upstream (Tukey's Multiple Comparisons, p=<0.0001, <0.0001))
KER A_{REF} vs. KER B_{HARV}	MeHg Hydropsychids	Aug 2017	**Yes** (two-factor ANOVA, p<0.0001)	**Yes** (higher KER B than KER A at all sites (multiple t-tests p=0.0002, 0.00005, 0.00560))	**Yes** (one-factor ANOVA, p=0.0002, <0.0001: higher middle-reach than upstream and downstream and downstream than upstream for KER A (Tukey's Multiple Comparisons, p=0.0002, 0.0280, 0.0023) and higher middle-reach than upstream and downstream for KER B (p=<0.0001, <0.0001))
PAN_{REF} vs. BAT_{HARV}	MeHg Hydropsychids	Oct 2017	**N/A** (missing data in BAT)	**N/A** (missing data in BAT)	**N/A** (missing data in BAT)
KER A_{REF} vs. KER B_{HARV}	MeHg Hydropsychids	Oct 2017	**Yes** (two-factor ANOVA, p<0.0001)	**Yes** (higher KER A than KER B at upstream, higher KER A than KER B at middle-reach and downstream (multiple t-tests, p=0.03600, 0.00008, 0.02700))	**Yes –** KER B only (one-factor ANOVA, p=0.0001: higher middle-reach than upstream and downstream, higher downstream than upstream (Tukey's Multiple Comparisons, p<0.0001, =0.0007, 0.044))
PAN_{REF} vs. BAT_{HARV}	Hg(II) Hydropsychids	Aug 2017	**No** (two-factor ANOVA, p=0.39)	**No** (two-factor ANOVA, p=0.21)	**Yes** (two-factor ANOVA, p=0.0005: higher middle-reach than upstream and downstream for PAN (Tukey's

					Multiple Comparisons, p=0.0022, 0.048), higher middle-reach and downstream than upstream for BAT (p=0.013, 0.044) **Yes** – KER B only (two-factor ANOVA, p=0.019: higher middle-reach than downstream (Tukey's Multiple Comparisons, p=0.012)
PAN_{REF} vs. BAT_{HARV}	Hg(II) Hydropsychids	Oct 2017	**N/A** (missing data in BAT)	**N/A** (missing data in BAT)	**N/A** (missing data in BAT)
KER A_{REF} vs. KER B_{HARV}	Hg(II) Hydropsychids	Oct 2017	**No** (two-factor ANOVA, p=0.52)	**No** (two-factor ANOVA, p=0.63)	**No** (Tukey's Multiple Comparisons, p>0.05)
PAN_{REF} vs. BAT_{HARV}	% MeHg Hydropsychids	2016	**N/A** (not an average)	**No** (two-factor ANOVA, p=0.50)	**No** (two-factor ANOVA, p=0.62)
KER A_{REF} vs. KER B_{HARV}	% MeHg Hydropsychids	2016	**N/A** (not an average)	**No** (two-factor ANOVA, p=0.091)	**No** (two-factor ANOVA, p=0.52)
PAN_{REF} vs. BAT_{HARV}	% MeHg Hydropsychids	Aug 2017	**N/A** (not an average)	**No** (two-factor ANOVA, p=0.38)	**No** (two-factor ANOVA, p=0.88)
KER A_{REF} vs. KER B_{HARV}	% MeHg Hydropsychids	Aug 2017	**N/A** (not an average)	**No** (Sidak's Multiple Comparisons, p>0.05)	**No** (two-factor ANOVA, p=0.37)
PAN_{REF} vs. BAT_{HARV}	% MeHg Hydropsychids	Sept 2017	**N/A** (not an average)	**No** (two-factor ANOVA, p=0.76)	**No** (two-factor ANOVA, p=0.34)
KER A_{REF} vs. KER B_{HARV}	% MeHg Hydropsychids	Sept 2017	**N/A** (not an average)	**No** (Sidak's Multiple Comparisons, p>0.05)	**No** (two-factor ANOVA, p=0.074)
PAN_{REF} vs. BAT_{HARV}	% MeHg Hydropsychids	Oct 2017	**N/A** (missing data in BAT)	**N/A** (missing data in BAT)	**N/A** (missing data in BAT)
KER A_{REF} vs. KER B_{HARV}	% MeHg Hydropsychids	Oct 2017	**N/A** (not an average)	**No** (two-factor ANOVA, p=0.47)	**No** (two-factor ANOVA, p=0.90)
PAN_{REF} vs. BAT_{HARV}	THg Sediment	2017	**Yes** (two-factor ANOVA, p=0.0010)	**Yes** (higher BAT than PAN at middle-	**Yes – BAT only** (one-factor ANOVA, p<0.0001: higher

				reach (multiple t-tests, *p*-0.025)	middle-reach than upstream and downstream, and upstream than downstream (Tukey's Multiple Comparisons *p*=0.012, <0.0001, 0.0003))
KER A_{REF} vs. KER B_{HARV}	THg Sediment	2017	**No** (two-factor ANOVA, *p*=0.20)	**Yes** (two-factor ANOVA *p*=0.013: higher KER A than KER B at upstream site (Sidak's Multiple Comparisons, *p*=0.35))	**No** (two-factor ANOVA, *p*=0.89)

Appendix Table 39: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for MeHg in unfiltered stream water, as determined via AIC_c model selection ($\Delta AIC_c < 7$). The conditional R^2 was 0.50. Statistically significant EVs are bolded.

Response Variable		Explanatory Variables				
		Year	**DOM_ PC1**	% EVSA	Flow	% Organic
MeHg in Unfiltered Water	Slope Coefficient	**-0.03**	**0.05**	-0.02	0.02	-0.03
	95% Confidence Interval	**-0.05 to -0.005**	**0.02 to 0.08**	-0.07 to 0.02	-0.001 to 0.04	-0.06 to -0.007
	Relative Variable Importance	**0.29**	**0.23**	0.01	0.02	0.04

[1]Year = year of sampling; DOM_PC1 = scores for DOM PCA axis 1; % EVSA = % effective variable source area compared to sub-catchment area; Flow = stream velocity (m/s); % Organic = % organic of fine sediment.
[2]Site highly correlated (*r*=0.70) with catchment size (not included in full model); Treatment highly correlated (*r*=0.75) with Road (not included in full model); Treatment highly correlated (*r*=0.78) with WC_PC2 (not included in full model); DOM_PC1 and Dep highly correlated (*r*=-0.73, -0.73) with WC_PC1 (not included in full model); DOM_PC1 and Dep had high VIFs (>4), so Dep removed from full model; Catchment (as a fixed effect) and % Dec has high VIFs (>4), so Catchment removed from full model.

Appendix Table 40: Slope coefficient estimates, 95% confidence intervals, and relative variable importance for explanatory variables (EVs) included in the average model for % MeHg in Hydropsychids, as determined via AIC_c model selection ($\Delta AIC_c < 7$). The conditional R^2 was 0.79. Statistically significant EVs are bolded.

Response Variable		Explanatory Variables											
		MeHg F Water	**Year**	**% EVSA**	Treat ment	Month	Site	DOM_ PC1	MeHg Seston	DOM_ PC2	Dep	**% Organic**	**Treat: Year**
% MeHg in Hydrops ychids	Slope Coefficient	-0.05	**-0.40**	**0.26**	0.14	0.04	-0.07	0.14	-0.07	-0.18	0.02	**-0.13**	**-0.14**
	95% Confidence Interval	-0.13 to 0.04	**-0.50 to -0.31**	**0.06 to 0.46**	-0.08 to 0.37	-0.02 to 0.11	-0.36 to 0.22	-0.08 to 0.35	-0.33 to 0.19	-0.41 to 0.05	-0.13 to 0.17	**-0.24 to -0.02**	**-0.20 to -0.07**
	Relative Variable Importance	0.02	**0.82**	**0.44**	0.81	0.03	0.06	0.10	0.05	0.15	0.02	**0.25**	**0.81**

[1]MeHg F water = concentration of MeHg (ng/L) in filtered water; Year = year of sampling; % EVSA = % effective variable source area compared to sub-catchment area; Treatment = % sub-catchment area harvested within the last 5 years; Month = month of sampling (1=August, 2=September, 3=October); Site = distance (km) to farthest downstream site within catchment multiplied by -1; DOM_PC1 = scores for DOM PCA axis 1; MeHg Seston = concentration of MeHg (µg/kg) in seston; DOM_PC2 = scores for DOM PCA axis 2; % Organic = % organic of fine sediment; Treat:Year = interaction between Treatment and Year.
[2]Site highly correlated (*r*=0.85) with catchment size (not included in full model); Treatment highly correlated (*r*=0.73) with WC_PC2 (not included in full model); DOM_PC1 and Dep highly correlated (*r*=-0.75, -0.74) with WC_PC1 (not included in full model); MeHg in filtered water highly correlated (*r*=0.70) with MeHg in unfiltered water (not included in full model); % Dec and % EVSA had high VIFs (>4), so % Dec removed from full models; Road and Treat:Site had high VIFs (>4), so Road removed from full model.

Curriculum Vitae

Kelli Leona Charbonneau

Bachelor of Science in Environmental Toxicology, Queen's University, 2016

Publications:

Adams, J., K. Charbonneau, D. Tuori, R.S. Brown, and P.V. Hodson. 2017. Review of Methods for Measuring the Toxicity to Aquatic Organisms of the Water Accommodated Fraction (WAF) and Chemically-Enhanced Water Accommodated Fraction (CEWAF) of petroleum. DFO Can. Si. Advis. Sec. Res. Doc. 2017/nnn. Vi + xx p.

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Charbonneau, K., K. Kidd, M. Gray, D. Kreutzweiser, E. Emilson, P. Sibley, and N. O'Driscoll. Poster presentation. "Do impacts from forest harvesting accumulate spatially in streams? A multi-indicator study." Society for Freshwater Science Annual Meeting. May 20-24, 2018. Detroit, MI.

Charbonneau, K., K. Kidd, M. Gray, D. Kreutzweiser, E. Emilson, P. Sibley, and N. O'Driscoll Poster presentation. "Do harvesting-induced disturbances accumulate in forest streams? A multi-indicator study." McMaster Water Network Student Research Showcase. Nov 3, 2017. Hamilton, ON.

Charbonneau, K., K. Kidd, D. Kreutzweiser, P. Sibley, and N. O'Driscoll. Poster presentation. "Are there spatially cumulative impacts downstream of forest harvesting operations in northern hardwood catchments?" 13th International Conference on Mercury as a Global Pollutant. July 16-21, 2017. Providence, RI.

Charbonneau, K., K. Kidd, D. Kreutzweiser, P. Sibley, and N. O'Driscoll. Poster and oral presentations. "Are there spatially cumulative impacts downstream of forest harvesting operations in northern hardwood catchments?" Canadian Network for Aquatic Ecosystem Services General Meeting. April 26-28, 2017. Toronto, ON.

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