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ASSESSING STREAM ECOSYSTEM STRUCTURE AND FUNCTION IN AN

URBAN CANAL AND LOGAN RIVER IN LOGAN, UTAH

by

Ellie Smith-Eskridge

A thesis submitted in partial fulfillment

of the requirements for the degree

of

MASTER OF SCIENCE

in

Ecology

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UTAH STATE UNIVERSITY
Logan, Utah

2023

ABSTRACT

Assessing Stream Ecosystem Structure and Function in an Urban Canal and the Logan

River, in Logan, Utah

by

Ellie Smith-Eskridge, Master of Science

Utah State University, 2023

Major Professor: Dr. Michelle A. Baker
Department: Biology

Humans have been creating artificial aquatic ecosystems for thousands of years. Some of these aquatic ecosystems are highly managed, especially in the semi-arid, Intermountain West. Here, humans have constructed extensive conveyance systems to support agriculture, to mitigate flooding, and to discharge stormwater. Despite their regional prevalence, the ecological structure and functioning of these conveyance systems remains largely unknown. To address this gap, I addressed the following questions: 1) How do water quality, freshwater invertebrate assemblages, and leaf decomposition compare between the Northwest Field Canal and its water source, an urbanized reach of the Logan River? 2) How do these measures change longitudinally in both waterways as they traverse Logan City? and 3) Which of the physical, chemical, and biological factors I measured most strongly influence leaf decomposition in these waterways?   I collected water quality and freshwater invertebrate samples, and I measured leaf decomposition at twenty sites along the Logan River and an urban canal. I used Spearman's correlation coefficients to evaluate the associations between physical,

chemical, and biological factors and leaf decomposition. Water quality was similar between waterways, except for the most downstream site of the Logan River, which had elevated concentrations of nutrients and metals, and lower richness and abundance of invertebrates. Leaf decomposition occurred faster in the canal, and the canal had higher biomass of shredders compared to the Logan River. Facultative shredders were associated with the decay rate in the canal, suggesting that these shredders are associated with leaf decomposition. Leaf decomposition was faster at downstream sites in both waterways relative to the upstream sites, due to an abundance of facultative shredders in the canal and elevated nutrients at the most downstream site in the Logan River. Water velocity was associated with leaf decomposition in both waterways, and total phosphorus was positively associated with biomass of shredders and leaf decomposition, the latter of which is likely due to enhanced microbial activity.

(81 pages)

PUBLIC ABSTRACT

Assessing Stream Ecosystem Structure and Function in an Urban Canal and the Logan

River, in Logan, Utah

Ellie Smith-Eskridge

Humans have constructed canals to support agriculture, to mitigate flooding, and to discharge stormwater, especially in the Intermountain West. These canals are common in Cache Valley, where they receive flows from the Logan River during summer months. However, the ecological structure (e.g., water quality, freshwater invertebrates) and function (e.g., leaf decomposition) of these canals remains largely unknown. Studying ecosystem structure and function of these urban waterways is important because it can inform us of the health of these waterways.

My research had three objectives. First, I compared water chemistry, invertebrate assemblages, and leaf decomposition in an urban canal and the Logan River in Logan, Utah. Next, I compared these variables along a longitudinal urban gradient (i.e., from upstream reaches to downstream reaches of the waterways). Last, I examined which of the various environmental factors I measured was correlated with shredders (i.e., leaf-shredding invertebrates) and leaf decomposition in both waterways.

Water quality was similar between the Logan River and the canal, except for the most downstream site of the Logan River which had a higher concentration of nutrients and heavy metals. The canal had faster leaf decomposition, and facultative shredders were abundant in the canal. Facultative shredders increased with the decay rate in the canal, suggesting that these shredders are playing a critical role aiding in leaf

decomposition in the canal. The most downstream sites in the canal may have had faster leaf decomposition due to facultative shredders whereas the most downstream sites in the Logan River may have had faster decomposition because of elevated concentrations of nutrients. I found that physical forces from the flow of streams (e.g., water velocity) and shredder biomass were correlated with leaf decomposition in both waterways. Total phosphorus was also correlated with shredder biomass and leaf decomposition, the latter of which is likely due to enhanced microbial activity on leaf litter.

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Finally, I can't thank these folks enough for being there for me through the ups and downs of my degree: my partner, John, for being willing to blindly move with me to Logan, Utah, from Nebraska, my parents for their words of encouragement, my Dad for

being a shoulder to lean on when I want to discuss my research, and my dear friends from Logan and all over the country.

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INTRODUCTION

Humans have been creating artificial aquatic ecosystems by altering Earth's landscape for 5000 years (Hooke, 2000; Clifford & Heffernan, 2018). These artificial aquatic ecosystems are now ubiquitous because humans have altered a large proportion of Earth's surface (Clifford & Heffernan, 2018; Ellis & Ramankutty, 2008), which has affected the flow and accumulation of water (Clifford & Heffernan, 2018). Humans have impounded rivers, dug ditches, dug sand pit lakes and ponds, dredged lakes, and straightened and channelized rivers for the purposes of irrigation, navigation, drainage, and recreation (Clifford & Heffernan, 2018; Hooke, 2000; Lin et al., 2020).

Despite being constructed by humans, artificial aquatic ecosystems can provide beneficial ecosystem services (Clifford & Heffernan, 2018; Lin et al., 2020). In particular, water diverted from reservoirs and canals can provide water for cities and agriculture (Pickett et al., 2016). Practices of green infrastructure such as constructed wetlands can improve water quality in urban watersheds by reducing the discharge of pollutants into water bodies (Passeport et al., 2013; Reisinger et al., 2019). Additionally, constructed wetlands, roadside ditches, and canals can sustain biodiversity by providing a habitat for aquatic communities (Kennedy & Mayer, 2002; Pitcher & Yee, 2018; Vermonden et al., 2009).

Yet, artificial aquatic ecosystems also can provide disservices. For example, they can contribute to the urban stream syndrome, where there are excess nutrients, warmer water temperatures, a flashier hydrograph, and reduced diversity of aquatic and riparian communities (Walsh et al., 2005; Wenger et al., 2009). Additionally, novel man-made canals between river basins can lead to adverse ecological consequences. Invasive species

and exotic pathogens can be more easily dispersed via these novel canals, and migration of endangered fish species via new connections between river basins can result in lower genetic diversity (Leuven et al., 2009; Lin et al., 2020; Muñoz-Ramírez et al., 2015). Additionally, drought and diversions of fresh water can result in ecological traps, where fish are transported via diversions of water, and they are not able to return to their original population (Lin et al., 2020; Zeug & Cavallo, 2014). Finally, artificial aquatic ecosystems can contribute to greenhouse gas emissions (Clifford & Heffernan, 2018; Palta et al., 2017).

In spite of these services and disservices, less is known about the ecological condition of artificial aquatic ecosystems compared to their natural counterparts. This knowledge gap may be in part because scientists and managers tend not to focus on studying the ecological conditions of these man-made ecosystems (Clifford & Heffernan, 2018; Ellis & Ramankutty, 2008).

The study of urban ecosystems has value in the context of the ecology of and for cities (Pickett et al., 2016). Nearly all cities were established near a water body (Foreman, 2014), often a river, for the provision of water, food, and transportation. Further, urban water systems are highly managed, especially in the semi-arid western US (Pataki et al., 2011). Here, humans have constructed extensive conveyance systems to support agriculture, to mitigate flooding, and to discharge stormwater (Fiege, 1999; Melcher, 2019). In the 1800s, settlers in Utah created canals to divert water from streams and rivers to supply the Cache Valley irrigation canal system for agriculture and the milling industry (Lavoie & Sleipness, 2018). Today, residents of Cache Valley also use these urban canals to water their lawns and gardens during the growing season when rainfall is

infrequent (Melcher, 2019; Mihalevich, 2017). Despite the prevalence of urban canal systems throughout much of the western U.S., the ecological structure and function of these urban and novel ecosystems remains largely unknown.

Ecosystem structure and function are used to evaluate the ecological condition of aquatic ecosystems. Ecosystem structure includes the physical, chemical, and biological components of an aquatic ecosystem (Young et al., 2008). Ecosystem function refers to the ecological changes and processes occurring in an aquatic ecosystem (Young et al., 2008). In this study, macroinvertebrate assemblages and water quality are important components of ecosystem structure, whereas leaf decomposition is a key component of ecosystem function. Freshwater invertebrates are critical to stream food webs and underlie many ecosystem functions (Cummins, 1974; Hynes, 1970). Ecologists have studied leaf decomposition in streams for decades, because leaves can be a dominant source of energy in streams, and they are important to energy flow (Webster & Benfield, 1986).

During the process of leaf decomposition, leaves fall into the streambed and accumulate to form "leaf packs" (Petersen & Cummins, 1974). Leaching then occurs where soluble compounds are removed from the leaves and flow downstream (Webster & Benfield, 1986). The stream community (i.e., macroinvertebrates, algae, fungi, bacteria) then begins to process the leaf litter, and physical forces from flowing water also begin to fragment the leaves (Webster & Benfield, 1986; Graça, 2001). A particular group of macroinvertebrates called shredders play a role in leaf decomposition (Cummins & Klug, 1979; Tank et al., 2010). Shredders tear up leaves with their mouthparts into smaller pieces, which increases leaf decomposition rates (Chadwick et al., 2006; Cummins &

Klug, 1979; Tank et al., 2010). While shredders feed on the leaves, microbial

communities on the leaves provide nutrition to the shredders (Graça, 2001). Studies have

also assessed how biodiversity relates to ecosystem function such as leaf decomposition

(Woodward, 2009). One mechanism, the sampling effect, hypothesizes that assemblages

with higher diversity lead to a higher likelihood of dominant taxa, where dominant taxa

contribute the most to ecosystem function (e.g., leaf decomposition) (Woodward, 2009).

Young et al. (2008) reviewed multiple studies that assessed the other drivers of

leaf decomposition. For example, water temperature is one driver, with moderately

warmer temperatures increasing microbial activity, thereby increasing the rate of leaf

decomposition, such that decomposition rates differ among climates and stream habitats

(Webster & Benfield, 1986; Young et al., 2008). Freshwater invertebrates require a

specific water temperature range to survive, which can influence the rate of leaf

decomposition (Bonacina et al., 2022). Additionally, shredder abundance is associated

with leaf decomposition rate (Young et al., 2008). Shredders have been found to be most

abundant in smaller streams surrounded by riparian vegetation, leading to a faster leaf

decomposition rate in such streams (Young et al., 2008).

Environmental stressors from human disturbance can also affect leaf

decomposition in stream ecosystems (Young et al., 2008). For example, an increase in

fine sediment has been shown to decrease leaf decomposition rate (Young et al., 2008).

Moreover, toxic chemicals such as heavy metals can slow leaf decomposition rate by

reducing communities of microbes and invertebrates that directly aid in leaf breakdown

(Young et al., 2008). In contrast, an increase in nutrients can increase leaf decomposition

rate, by stimulating microbial colonization on the leaves (Rosemond et al., 2015; Webster

& Benfield, 1986; Young et al., 2008). Other studies have found that as the amount of impervious surfaces from urbanization in watersheds increases, macroinvertebrate functional feeding groups such as shredders disappear, which associates with a slower leaf decomposition process (Chadwick et al., 2006; Classen-Rodríguez et al., 2019). In addition, environmental stressors may interact with one another, resulting in varying patterns of leaf breakdown responses, especially in urban streams. For instance, a negative effect on leaf decomposition from increased heavy metal concentrations may be counteracted by a positive effect on leaf decomposition from an increase in nutrients (Young et al., 2008).

The physical force of flowing water is another factor that may influence leaf decomposition (Webster and Benfield, 1986). We might assume that a flashier hydrograph in urban streams, one symptom of the urban stream syndrome, will lead to an increase in physical fragmentation, and thus an increase in leaf breakdown rate (Paul et al., 2006). Water velocity is one metric that can be used to evaluate the physical forces of flowing water on leaf breakdown. In particular, water velocity can affect leaf breakdown by reducing the deposition of sediment on the surfaces of leaf litter, enhancing microbial activity, further increasing the process of leaf breakdown (Canton & Martinson, 1990). Water velocity can also drive leaf breakdown by controlling the retention and transport of leaf litter particles (Bastias et al., 2020).

The Logan River and the Northwest Field Canal (the canal) provide a unique opportunity to compare ecosystem structure and function of natural and artificial aquatic ecosystems that are in close proximity and share a common water source. The Logan River, a natural stream with a typical snowmelt-dominated hydrograph, and one of its

diversions, the Northwest Field Canal, both flow through mixed land uses (urban residential and commercial and suburban/agricultural) in Logan, Utah. The canal is a novel artificial aquatic ecosystem because as a conveyance structure, it is a non-perennial waterway that only receives regular flows during the growing season (May through October) when water is diverted from the Logan River (Melcher, 2019). The canal also receives stormwater runoff throughout the year from various urban land uses such as residential, commercial, and mixed-use neighborhoods (Melcher, 2019).

Despite their regional prevalence, the ecological structure and functioning of canals remains largely unknown. To address this gap, I asked the following questions: 1) How do water quality, freshwater invertebrate assemblages, and leaf decomposition compare between the Northwest Field Canal and its water source, an urbanized reach of the Logan River? 2) How do these measures change longitudinally in both waterways as they traverse Logan City? and 3) Which of the physical, chemical, and biological factors I measured most strongly associate with leaf decomposition in these waterways? Figure 1 presents a conceptual model of various factors known to influence leaf decomposition in freshwaters. These linkages guided my hypotheses for how the canal and the Logan River may respond in an urbanized watershed.

I hypothesized that water quality would be poorer in the canal relative to the Logan River because the canal receives proportionately more urban stormwater inputs and residential irrigation return flows (Melcher, 2019; Mihalevich, 2017). Accordingly, I predicted the canal to have higher concentrations of nutrients and metals, warmer water temperature, and higher concentrations of total suspended solids (TSS) compared to the Logan River. Because of a combination of water quality and intermittent hydrology, I

predicted that the canal would be lower in shredder biomass compared to the Logan River and would have lower taxa richness and density of invertebrates compared to the Logan River.

I hypothesized that invertebrates would be one driver of leaf decomposition (Webster & Benfield, 1986). I expected that leaf decomposition would be less when invertebrates were excluded. I expected a lower biomass of shredders to result in slower leaf decomposition in the canal relative to the Logan River. Finally, I expected that invertebrate richness would be higher in the Logan River relative to the canal, which would associate with a faster rate of leaf decomposition via an underlying mechanism, the sampling effect.

I hypothesized that both waterways would exhibit symptoms of the urban stream syndrome as they traversed the urban land uses in Logan City because urban pollutants accumulate as waterways flow through urban areas (Kaushal & Belt, 2012; Walsh et al., 2005). I expected higher concentrations of nutrients, metals, TSS, and a warmer stream temperature further downstream in both waterways. I also predicted a lower density of shredders and lower invertebrate richness and density further downstream in both waterways. I expected a slower leaf decomposition rate further downstream in both waterways due to poorer water quality and fewer invertebrates feeding on leaf litter.

Last, I aimed to assess the effects of physical, chemical, and biological factors on leaf decomposition in both water bodies because less is known about the effects of these factors on leaf decomposition in artificial aquatic ecosystems, like the canal. Factors known to affect leaf decomposition are presented in a conceptual model (Figure 1). I predicted that water velocity and biomass of shredders would negatively associate with

the dry mass of leaves remaining, because these factors directly aid in leaf breakdown

(Webster & Benfield, 1986). I predicted that metals and TSS would negatively associate

with biomass of shredders, because these factors can impair invertebrates (Clements et

al., 2000; Walsh et al., 2005)

METHODS

Overall Study Approach

To answer these questions, I established 10 sites each in the canal and the Logan

River (i.e., twenty sites total). At each site, I measured dissolved and total nutrients,

dissolved metals, total suspended solids, water temperature, invertebrate metrics, and leaf

decomposition over the course of approximately 3 weeks during summer 2020. Leaf

decomposition was assessed at each study site using the leaf pack method (Lamberti and

Hauer, 2017). Some leaf bags were coarse-mesh (0.5-1 cm aperture) to allow for

colonization by macroinvertebrates. Other leaf bags were fine-mesh (≤1 mm) to exclude

invertebrates and to adjust for the effects of invertebrates on leaf decomposition. Field

data were collected on days 4, 7, 11, 14, and 18 following site establishment. Another set

of coarse- and fine-mesh leaf bags were placed in troughs with flowing water in Dr.

Charles Hawkins' lab to verify that invertebrates play a role in leaf decomposition. To

address my first research question, data were organized by site to make comparisons of

water quality, attributes of invertebrates (i.e., richness, density, and biomass of

shredders), and leaf decomposition between the canal and the Logan River using

Wilcoxon's rank sum tests and Welch's t-tests. Data were organized by site in each

waterway to qualitatively evaluate longitudinal patterns. Finally, I used Spearman's correlation coefficients to assess relationships between the physical, chemical, and biological factors that I measured and leaf decomposition. Detailed methods are presented below.

Study Sites

I conducted the study in the Logan River and Northwest Field Canal in Logan, UT. The Logan River flows southwest from Logan Canyon in the Bear River Mountains into the city of Logan and is diverted into four agricultural irrigation canals, one of which is the Northwest Field Canal. The canals flow north through Logan and then west, emptying into Cutler Reservoir. The canals receive flows from Logan River during summer months when water is diverted for irrigation (Melcher, 2019). Additionally, stormwater runoff from various land uses such as residential, commercial, and mixed-use neighborhoods drains into the canal (Melcher, 2019). The Northwest Field Canal was selected for this study because past research on stormwater runoff had been conducted on the canal (Melcher, 2019; Mihalevich, 2017), but the ecosystem structure and function of the canal had not yet been studied.

I deployed bags filled with leaf litter (i.e., leaf bags), and I collected freshwater invertebrate and water chemistry samples along the Logan River and the canal (Figure 2). The most upstream site in the Logan River was located in Logan Canyon outside of city limits, approximately 31 kilometers above Site 2. The most downstream site was located near Mendon, UT, approximately 36 kilometers downstream of the most upstream site. The rest of the sites along the Logan River were within the city of Logan, UT. All ten

sites along the Northwest Field Canal were located within Logan, UT, city limits. The

most upstream site in the canal was located approximately 4.4 kilometers upstream of the

most downstream site in the canal. I selected sites that were easily accessible from public

roads, parking lots, city parks, and the Utah Water Research Laboratory. Some sites were

locations of previous research including canal sites (Melcher, 2019; Mihalevich, 2017)

and stations that are part of the Logan River Observatory (Jones et al., 2017; Neilson et

al., 2021).

Sampling Design

Water Chemistry

During each day of leaf bag retrieval, I collected water samples at each of the 20

sites for analysis of nutrients including total nitrogen (TN), total phosphorus (TP), nitrate-

N (NO_3), ammonium-N (NH_4), and soluble reactive phosphorus (SRP); total suspended

solids (TSS); and dissolved metals. I filtered water samples in the field with pre-ashed

glass fiber filters (Whatman GF/F, Maidstone, United Kingdom). Filtered samples were

analyzed for NO_3, SRP, and NH_4, and dissolved metals, and unfiltered samples were

analyzed for TN and TP. Water quality parameters such as stream temperature, dissolved

oxygen, pH, and conductivity were measured with a YSI multiparameter hand-held

meter. I stored water chemistry samples in a cooler while they were transported back to

the lab. Water chemistry samples were stored frozen up to 3 months until nutrient

analyses at the Aquatic Biogeochemistry Lab at Utah State University. Dissolved

nutrients were measured using micro-segmented flow analysis on an Astoria Analyzer

(Astoria-Pacific, Klackamas, OR). NO_3 was quantified using cadmium reduction

(USEPA 1993a), NH$_4$ was quantified using the phenol-sodium nitroprusside method

(USEPA 1993b), and SRP was measured using the molybdenum blue method (USEPA

1993c). Total N and P were measured using a persulfate digestion (Ameel et al. 1993)

followed by micro-segmented flow analysis as NO$_3$ or SRP as above. Quality control

was assessed using reagent blanks, field blanks, spikes, and check standards. I sent

samples to the Environmental Analytical Laboratory at Brigham Young University to be

analyzed for metals using ICP-OES (iCAP 7400, Thermo Scientific, Waltham, MA,

USA). All concentrations were in mg/L. Prior to data analysis of the water chemistry

samples, concentrations flagged as below detection limit were assigned a value at half of

the detection limit. The detection limit for lead was 0.0045 mg/L and the detection limit

for iron was 0.0008 mg/L.

Water Velocity

For each site and day of leaf bag retrieval, I measured water velocity to estimate

physical forces from the water current. I used a Marsh-McBirney Flow-Mate 2000 to

measure water velocity (meters/second) at 60% of the water depth at three points along

the upstream side of the leaf bags and at three points along the downstream side of the

leaf bags.

Cumulative Criterion Unit

I used the cumulative criterion unit (CCU) to represent the toxicity of heavy

metals (e.g., lead and iron) as one variable because heavy metals are assumed to have an

additive effect on freshwater invertebrates at chronic concentrations via aqueous

exposure (Clements et al., 2000; Rainbow, 2002). I calculated the CCU as the ratio of

metal concentrations measured over the U.S. EPA criterion value, summed for lead and

iron, for each site,

$$CCU = \sum m_i/c_i$$

where m_i is the measured metal concentration for the ith metal and c_i is criterion value for

the ith metal (Clements et al., 2000). Heavy metal concentrations that are above the U.S.

EPA criterion value are considered harmful to aquatic life (US EPA, 2015). Prior to

calculating the CCU, I summarized the data by calculating the mean of iron and lead by

site and waterway, so there were 20 observations of each metal. The criterion value for

lead was calculated based on a water hardness of 200 mg/L, which is typical of the Logan

River (Rupp & Adams, 1981).

Freshwater Invertebrates

I collected freshwater invertebrates with two different sampling methods 1) from

the coarse-mesh leaf bags (discussed below) and 2) using a Surber sampler. Invertebrates

from the leaf bags were assumed to be associated with leaf decomposition, whereas the

invertebrates from the Surber sampler were collected to obtain site-level information on

assemblage composition. I placed coarse-mesh bags into a labeled zip lock bag

immediately after retrieval to prevent invertebrates from falling out. In the lab, I rinsed

leaves with tap water onto a 250-µm sieve to separate macroinvertebrates which were

then stored in separate bottles with 90 percent ethanol (Classen-Rodríguez et al., 2019). I

collected invertebrates with a Surber sampler during the first, third, and fifth leaf bag

retrievals (Days 4, 11, and 18) by placing the sampler next to the leaf bags at each site,

and dislodged invertebrates by hand from the benthic substrates. I transferred the material

from the net of the Surber sampler into a 250 µm sieve. Each sample in the sieve was

then transferred into a 1-L Nalgene bottle and topped off with 90 percent ethanol into the

jars to preserve the samples. The Surber samples from each date were then pooled for

each site, so that there were a total of 20 Surber samples. Invertebrates from the Surber

samples were sent to the BLM/USU National Aquatic Monitoring Center (NAMC) at

Utah State University for processing. There, each Surber sample was processed by

subsampling to at least 600 individuals if present, where each sample was split in half and

processed repeatedly until at least 600 individuals were counted and removed. Each

subsample was processed in its entirety. NAMC used a dissecting microscope to separate

invertebrates from organic matter. Large and rare individuals were identified and counted

from the unsorted material from each sample. Invertebrates were identified to genus, if

possible. Invertebrates from the leaf bags also were identified to genus with the assistance

of NAMC, if possible, except for the family Chironomidae, which were identified to

Tanypodinae or non-Tanypodinae to separate Chironomidae into functional feeding

groups, predators (e.g., Tanypodinae) and collector-gatherers (e.g., non-Tanypodinae).

The BLM/USU NAMC estimated densities of invertebrates per square meter collected

with the Surber samples by using the equation as follows:

$$\left[Split\ Count\ x\ \frac{100}{Lab\ Split} \right] + \left[Big\ Rare\ Count\ x\ \frac{100}{Field\ Split}\ x\ \frac{1}{Area\ Sampled} \right]$$

where *Split Count* is the number of individuals randomly subsampled for identification,

Lab Split is the percent of sampled processed to obtain 600 individuals if present, *Big*

Rare Count is the number of "big and rare" organisms selected non-randomly for

identification from the entire sample, *Field Split* is the percentage of sample submitted

for processing, and *Area Sampled* is the area of the Surber sampler. Additionally, I

counted individuals from coarse-mesh bags and measured the length of each individual to

the nearest one millimeter using a dissecting microscope. Invertebrate biomass in each coarse-mesh bag was estimated using length-to-mass relationships (Baumgärtner & Rothhaupt, 2003; Benke et al., 1999; Dumont et al., 1975; Edwards et al., 2009; Tellez et al., 2008). Macroinvertebrates from both types of samples were classified into their respective functional feeding groups (Merritt et al., 2008). I categorized *Hyalella sp.* and *Gammarus sp.* as facultative shredders because previous literature has found that these amphipods can feed on coarse particulate matter such as leaves (Cook & Hoellein, 2016; Scriber, 2013; Strong, 1972).

I calculated the rarefied richness for each site and type of invertebrate sample using the vegan package in R to compare diversity of invertebrates between waterways and between sites. Taxa richness was defined as the expected number of taxa for a given number of randomly sampled individuals (McCabe & Gotelli, 2000). Richness was rarefied based on the minimum number of individuals across all sites. The minimum number of individuals was 70 individuals in the Surber samples and 540 individuals in the leaf bags across all sites.

Leaf Decomposition

Leaf decomposition was assessed at each study site using the leaf pack method (Lamberti and Hauer, 2017) with fine-mesh and coarse-mesh leaf bags. Briefly, I collected green leaves from boxelder (*Acer negundo*) trees near the storage buildings of the USU Ecology Center near Green Canyon in May 2020. These were air dried for 5-8 days until they reached a constant dry weight, then the leaves were separated, weighed, and placed into their respective fine- and coarse-mesh leaf bags. Each bag contained 6±0.02 grams of leaves. A total of 10 bags of each type were prepared for deployment at

each site (Figure 3). When leaf bags were ready for deployment, I transported leaves in a sturdy cooler to avoid leaf breakage (Lamberti & Hauer, 2017). I prepared extra sets of leaf bags of both mesh sizes to account for the mass of organic matter that was lost during processing, transporting, and placing bags into the stream. These handling loss bags were deployed at two sites in the Logan River and one site in the canal and then immediately retrieved.

At each site along the canal and Logan River, leaf bags were anchored to the stream bottom using poultry wire and bricks (Classen-Rodríguez et al., 2019; Lamberti & Hauer, 2017). Two of the ten replicates of each bag type were retrieved on each of the five sampling days (Figure 3). If possible, I placed leaf bags near shallow riffles in Logan River, so invertebrates were coming from consistent habitat types, and so it was shallow enough to easily access the leaf bags (Young et al., 2008). We had originally planned to leave leaf bags deployed for 2-16 weeks but leaves began to decompose more quickly than anticipated, so I collected leaf bags on days 4, 7, 11, 14, and 18 from each site starting on July 31, 2020, and ending August 18, 2020. In the field, each leaf bag was placed into a zip lock bag as it was removed from the poultry wire to prevent material from falling out of the bag. Sample information was recorded on the outside of the bag and on weatherproof paper inside of the bag. I placed leaf bags in a cooler to transport to the lab where they were kept frozen until further processing.

Another set of course- and fine-mesh leaf bags (10 coarse-mesh bags, 10 fine-mesh bags) were placed in troughs with flowing water in Dr. Charles Hawkins' lab to verify that invertebrates play a role in leaf decomposition and to assess whether the type of bag (i.e., fine vs. coarse) was a confounding factor affecting leaf decomposition. The

experimental set of leaf bags was in a lab-controlled setting for the same amount of time as in Logan River and the canal, and replicate leaf bags were collected on Days 4, 7, 11, 14, and 18. Each time I retrieved leaf bags in the lab, I measured water temperature with a handheld YSI meter, and I measured water velocity at 6 locations around the leaf bags with a Marsh-McBirney Flow-Mate 2000. The water temperature and water velocity stayed consistent throughout the lab experiment, with a mean and standard error of water velocity at 0.006±0.003 meters per second and water temperature at 11.9±0.01degrees Celsius. The experimental set of leaf bags were also transported in a cooler to the lab. Handling loss was accounted for as described above.

In the lab, I processed the leaves from each leaf bag to calculate ash-free dry mass (AFDM). I rinsed the leaf bags with tap water to remove sediments and to separate macroinvertebrates. Leaves were then oven-dried at 60°C for 24 hours to a constant dry mass and dry mass was recorded. I ensured there was a constant dry mass at 24 hours by re-weighing the dry mass of each sample from the leaf bags at 48 hours. A subsample of leaf material from each leaf bag was combusted at 450°C in a muffle furnace and weighed for computation of AFDM.

Leaf decay rates were calculated by fitting a negative exponential decay model as follows:

$$M_t = M_0 e^{-kt},$$

where M_t is the AFDM at time t and M_0 is the initial AFDM. The decay rate coefficient (expressed as k) is the slope of the linear regression of AFDM remaining (log transformed) on time (days) as follows:

$$\log(M_t) = \log(M_0) - kt$$

The decay rate coefficient was calculated for each site. Percent of AFDM was calculated by dividing AFDM by the initial dry weight of leaves. The initial dry weight of leaves was calculated by subtracting the mass that was lost when handling the leaf bags from the dry weight of leaves before being placing leaf bags in the waterways (Lamberti & Hauer, 2017). Two coarse-mesh bags from two sites on Days 7 and 11 (i.e., two observations of AFDM remaining) were removed from the models because they were damaged.

Statistical Analyses

For the first research question, I compared the canal and the Logan River regarding leaf decomposition, water quality, and freshwater invertebrate attributes by running Welch's *t*-tests and Wilcoxon's rank-sum tests. Prior to running statistical tests, I organized the data by computing the means of each variable by site and waterway averaged over five sampling dates, so that there was a total of 20 observations (i.e., 10 observations per waterway). Depending on the variable, I used Welch's *t*-tests to adjust for unequal variances assuming normality, and I used the Wilcoxon's ranks-sum test to accommodate for non-normality, assuming variances were equal. I used the functions wilcox_test() and t_test() from the R package RStatix (Kassambra, 2021) to run the Welch's t-tests and Wilcoxon's rank sum tests. I used the function ggqqplot() from the ggpubr R Package (Kassambra, 2020) to determine if the residuals were normally distributed. I used the function levene_test() to test for equal variance (R version 4.2.0, R Package RStatix). I log-transformed measures of soluble reactive phosphorus, total suspended solids, and the biomass of shredders to better meet the normal distribution assumptions. I compared leaf decomposition between waterways by running Wilcoxon's

rank-sum tests and Welch's *t*-tests to compare the medians and means of the decay rates

and to compare the medians and means of percent of ash-free dry mass remaining

between waterways for fine- and coarse-mesh bags. I compared water quality between

waterways by running Wilcoxon's rank-sum tests and Welch's *t*-tests to compare the

medians and means of water temperature, nutrients, metals, TSS, and water velocity

between waterways. I compared freshwater invertebrate metrics between waterways by

running Wilcoxon's rank-sum tests and Welch's *t*-tests to compare the medians and

means of invertebrate richness and densities between waterways for both types of

invertebrate samples. I also compared the medians of shredder biomass between

waterways by running Wilcoxon's rank sum tests. I estimated effect sizes when statistical

differences were detected. I used the function cohens_d() to estimate the effect size of

Cohen's *d* for the Welch's test (R Package RStatix), and I used the function

wilcox_effsize() to estimate the effect size of the Wilcoxon's rank sum tests (R Package

coin).

I ran two ANCOVAs to 1) compare the relationship of invertebrate richness and

decay rate between waterways and 2) compare the relationship of shredder biomass and

decay rate between waterways. In particular, I ran an ANCOVA to compare the effect of

shredder biomass from leaf bags (log-transformed and centered) on decay rate between

waterways. The waterway (i.e., the canal or the Logan River) was the treatment, and

shredder biomass was the covariate. I organized the data by computing the means of

biomass of shredders by site and waterway (20 observations). Similarly, I also ran an

ANCOVA to compare the relationship between invertebrate richness and decay rate with

centered invertebrate richness as the covariate, and waterway as the treatment. Both

analyses were conducted separately for each type of invertebrate sample. I organized the data by computing invertebrate richness by site and waterway (20 observations). Analysis of variance tables were obtained using the function Anova() from the R Package car (Fox & Sanford, 2018) for Type III hypothesis tests. I used these hypothesis tests because I used an interaction term to test for the equality of slopes.

I also ran an ANCOVA 1) to compare the effect of time(days) on AFDM (log-scale) between the canal, the Logan River, and the lab experiment, 2) to compare the effect time on AFDM (log-scale) between fine- and coarse-mesh bags in the lab experiment, and 3) to compare the effect of time on AFDM (log-scale) between the two waterways. Analysis of variance tables were obtained using the function Anova() from the R Package car (Fox & Sanford, 2018) for Type III hypothesis tests.

For the second research question, I qualitatively evaluated leaf decomposition, water quality, and freshwater invertebrate metrics longitudinally in both waterways by constructing box plots of variables I measured. I organized the box plots from upstream to downstream sites.

For the third research question, I used Spearman's correlation coefficients to evaluate relationships between leaf decomposition, water quality, and biomass of shredders from the hypothesized conceptual model in Figure 1. Prior to assessing the relationships using correlation coefficients, I organized the data by computing the mean of each variable by site and waterway, resulting in 20 observations (i.e., 10 observations per waterway). Additionally, I imputed values for total nitrogen, lead, and iron for three missing water samples. I imputed the values by calculating the mean concentration by day and waterway that each water sample was collected. The dry mass remaining (g

AFDM) variable included only data from the coarse-mesh leaf bags on Day 18 (i.e., the last day of the leaf bag retrieval). The variable for metals was represented as the cumulative criterion unit for lead and iron. I used the Spearman's correlation coefficients because the data were not normally distributed. The threshold for statistical significance of the correlation coefficients was α = 0.05. Water temperature was not included in the analysis of correlation coefficients because 1) there was limited data on water temperature with one observation every 3-4 days and so temperature measurements were not representative of water temperature over a 3-4 day period, and 2) the water temperature data was not consistent with time of day that samples were collected. TSS was not included in this analysis because it showed little variability.

RESULTS

How Comparable are the Canal and the Logan River?

Water Quality

I hypothesized that the canal would have poorer water quality relative to the Logan River, where I expected the canal to have higher concentrations of nutrients, metals, and TSS compared to the Logan River. My hypothesis and predictions were not supported. Logan River and the canal differed significantly for a few physicochemical parameters (Table 1). Logan River had higher mean concentrations of ammonium (0.023±0.0013 mg/L) compared to the canal (0.016±0.0005 mg/L) (p=0.002, effect size = 0.677). The canal had higher mean concentrations of total phosphorus (0.022±0.0005 mg/L) compared to the Logan River (0.018±0.0006) (p=0.0341, effect size = 0.482).

Additionally, the canal had higher mean concentrations of TSS (0.032±0.010 mg/L)

relative to the Logan River (0.008±0.0027 mg/L) (p=0.002, effect size = 1.59).

Freshwater Invertebrates

The canal had an order of magnitude more shredder biomass (log-scale) than the

Logan River (p = 0.002, effect size = 0.693, mean shredder biomass (original scale) in

the canal = 29.1 mg, mean shredder biomass in the Logan River = 2.02 mg, Wilcoxon's

rank sum test), which was opposite of what I expected (Figure 4). The median shredder

biomass (on the original scale) in the coarse-mesh leaf bags in the canal was 18.71 mg,

whereas the median shredder biomass in the coarse-mesh leaf bags for Logan River was

0.0575 mg.

The canal contained the shredder taxa, *Hyalella sp.* and *Gammarus sp.*, whereas

the Logan River contained four shredder taxa of caddisflies, one shredder taxa of

stoneflies (*Malenka sp.)* in addition to *Hyalella sp.* and *Gammarus sp.* (Table 2).

Additionally, *Hyalella sp.* and *Gammarus sp.* were the majority of the biomass of

shredders in the canal (Table 2). However, the majority of the shredder biomass in the

leaf bags retrieved from the Logan River was the caddisfly *Onocosmoecus sp.* The top

five most abundant taxa by waterway and type of sample are shown in Table 3.

I expected the canal to have reduced richness and density of invertebrates relative

to the Logan River, but my prediction was not supported. Richness did not differ

between waterways from the Surber samples (p=0.732, mean richness in the canal = 9.55,

mean richness in the Logan River = 9.99, Welch's *t*-test, Figure 5a) or from the leaf bags

(p= 0.684, mean richness in the canal = 10.2, mean richness in the Logan River = 10.5,

Wilcoxon's rank sum test, Figure 5b). Additionally, the density of invertebrates from the

Logan River was three times higher compared to the canal; however, this finding was not statistically significant (p = 0.063, mean density in the canal =3046, mean density in the Logan River =16,989 , Wilcoxon's rank sum test, Figure 5c) or from the leaf bags (p=0.739, mean density in the canal =3079, mean density in the Logan River = 3532, Wilcoxon's rank sum test, Figure 5d).

My prediction that the relationship between decay rate and shredder biomass would differ between the canal and the Logan River was not supported (p=0.577) (Figure 7), nor was there enough evidence to suggest that decay rate increased with increased shredder biomass (p=0.171). The slope of the relationship between shredder biomass and decay rate was 0.016±0.011.

My prediction that the relationship between decay rate and invertebrate richness would differ between the canal and the Logan River was not supported. For the Surber samples, the slope of the regression of decay rate versus invertebrate richness was not different between waterways (p=0.460) (Figure 8a). Additionally, there was no evidence to suggest that decay rate changed with invertebrate richness in the Surber samples (p=0.491). The slope of the effect of richness on decay rate was -0.0024±0.0034. The same was true for the leaf bags, where the slope of the regression of decay rate on invertebrate richness did not differ between waterways (p= 0.256) (Figure 8b). Nor did decay rate increase with invertebrate richness in the leaf bags (p=0.572). The slope of the effect of richness on decay rate was 0.00015±0.0026.

Leaf Decomposition

I predicted the canal to have slower leaf decomposition compared to the Logan River; however, my prediction was not supported. The canal had faster leaf

decomposition relative to the Logan River. Coarse-mesh bags incubated in the canal had a lower median % AFDM remaining (18.1%) compared to the coarse-mesh bags in the Logan River (24.8%) (p=0.0185, effect size = 0.524, Wilcoxon's rank sum test, Figure 6a). The median % AFDM remaining in the fine-mesh bags did not differ between the canal (33.5%) and the Logan River (34.6%) (Wilcoxon's rank sum test, p=0.218, Figure 6b). The median decay rates did not differ between the canal (0.132/day) and the Logan River (0.114/day) for the coarse-mesh bags (Welch's t-test, p=0.336, Figure 6c). Additionally, the median decay rates did not differ between the canal (0.053/day) and the Logan River (0.047/day) for the fine-mesh bags (Wilcoxon's rank sum test, p=0.247, Figure 6d). Decay rate coefficients for each site and waterway are found in the Appendix in Table A1.

Leaf Decomposition in the Absence and Presence of Freshwater Invertebrates

In the absence of freshwater invertebrates in lab-controlled settings, no difference existed between coarse- and fine-mesh bags in terms of leaf loss (ANCOVA, $b_{coarse-}$ = 0.044/day, b_{fine-} = 0.041/day, p=0.678, Appendix, Figure A 1). The leaves from the coarse-mesh bags decayed more slowly in the lab experiment than the leaves from the coarse-mesh bags incubated in both waterways in the field (ANCOVA, p=0.041, Appendix, Figure A 2a). The decay rate coefficient from the coarse-mesh bags for the canal was 0.13/day, 0.12/day for the Logan River, and 0.044/day for the lab experiment. In contrast, the decay rate coefficient from the fine-mesh bags did not differ among the lab experiment or both waterways (ANCOVA, p=0.239, Appendix, Figure A 2b). The decay rate coefficient from the fine-mesh bags for the canal was 0.059/day, 0.049/day for the Logan River, and 0.041/day for the lab experiment.

How Do Water Quality, Freshwater Invertebrates, and Leaf Decomposition Change Along an Urban Gradient?

I hypothesized that both waterways would exhibit symptoms of the urban stream syndrome as they crossed urban land uses because urban pollutants accumulate as flowing waters traverse urban areas. In particular, I predicted that downstream sites of both waterways would have higher concentrations of nutrients, heavy metals, TSS, reduced abundance of shredders, a lower richness and density of invertebrates, and faster leaf decomposition.

Water Quality

The most downstream site of the Logan River generally had markedly higher median concentrations of nutrients and iron than other sites, whereas concentrations of nutrients and metals were variable in the canal, showing no distinct longitudinal patterns (Figures 9-11). For example, the median concentrations of total nitrogen, nitrate, and ammonium had a higher median concentration at the most downstream site in the Logan River with median concentrations of 0.53 mg/L, 0.422 mg/L, and 0.049 mg/L, respectively (Figure 9). The two highest median concentrations of SRP occurred at the most downstream site (0.015 mg/L) and at Site 4 (0.014 mg/L) of the Logan River (Figure 10d). Concentrations of total phosphorus ranged in the canal from 0.005 mg/L to 0.033 mg/L (Figure 10 a) and in the Logan River from 0.011 mg/L to 0.048 mg/L (Figure 10b). Most concentrations of lead were above 0.0053 mg/L, which is considered to be chronically harmful to aquatic life at a water hardness of 200 mg/L according to the National Recommended Water Quality Criteria (US EPA, 2015) (Figures 11c and 11d). Iron in both waterways falls under the concentration (1 mg/L) that is considered to be

harmful to aquatic life (US EPA, 2015) (Figures 11a and 11b). Concentrations of TSS ranged between 0 mg/L and 0.581 mg/L with most concentrations of TSS being close to 0 mg/L (Figure 12). Water temperature was not reported because there was limited data, with one observation at each site every 3-4 days.

Freshwater Invertebrates

Functional feeding groups in both the canal and the Logan River generally consisted of collector-gatherers, shredders (i.e., mainly *Hyalella sp.,* and *Gammarus sp.)*, predators, and scrapers (Figure 13). The predators were mainly *Arctopsyche sp.*, Trombidiformes, and *Rhyacophila* in the Logan River and Dytiscidae, *Helobdella stagnalis*, Trombidiformes, and Tanypodinae in the canal. Collector-gatherers were Chironomidae (i.e., Non-Tanypodinae), Oligochaeta, and *Baetis sp.* in the canal, and Chironomidae (i.e., Non-Tanypodinae) in the Logan River. Scrapers mainly consisted of *Oligophlebodes sp.* at the most upstream site in the Logan River in the Surber samples. Additionally, scrapers mainly consisted of *Gyraulus sp., Lymnaea sp.,* and *Physa sp.* at an upstream site in the canal in the leaf bags. Collector-filterers were mainly in the Simuliidae family in the Logan River and the canal, except for the most downstream site in the Logan River. Collector-filterers at the most downstream site in the Logan River were mainly Cladocera in the leaf bags.

Invertebrate richness and density varied across sites in the canal and the Logan River. Invertebrate densities from the leaf bags ranged from 540 individuals to 6944 individuals in the Logan River, and from 932 individuals to 6221 individuals in the canal (Figures 14c and 14d). Invertebrate densities from the Surber samples ranged from 251 to

78,555 individuals/m^2 in the Logan River and from 806 to 9289 individuals in the canal (Figures 14a and 14b).

Leaf Decomposition

The most downstream sites had faster leaf decomposition compared to the most upstream sites in both waterways, measured as both % mass remaining (Figures 15a and 15b) and as decay rate (Figures 15c and 15d). Across all sites, the coarse-mesh bags had less mass of leaves remaining and faster decay rates relative to the fine-mesh bags (Figures 15a-15d).

Associations Between Leaf Decomposition and Physical, Chemical, and Biological Factors

Some of the physical, chemical, and biological factors I measured were associated with leaf decomposition as hypothesized in Figure 1 (Table 4). Water velocity was moderately and negatively associated with dry mass of leaves remaining ($r = -0.63$, $p = 0.003$). Shredder biomass was not correlated with dry mass of leaves remaining ($r = -0.4$, $p = 0.081$). Interestingly, total phosphorus was strongly and positively associated with biomass of shredders ($r = 0.79$, $p < 0.01$), and moderately and negatively associated with dry mass of leaves remaining ($r = -0.62$, $p = 0.003$).

DISCUSSION

How Comparable are the Canal and the Logan River?

Water Quality

Water quality was similar between the canal and Logan River, likely due to being derived from a common water source. Both waterways have similar hydrologic modifications that can affect water quality. For example, the canal receives urban stormwater runoff from several outlets along the canal (Melcher, 2019; Mihalevich, 2017), and the urban reaches of the Logan River receive ungaged inflows which carry urban and agricultural surface water and groundwater (Tennant et al., 2021).

Freshwater Invertebrates

There were few differences in the invertebrate assemblages in the canal and the Logan River, with the exception of the high abundance and biomass of facultative shredders (*Hyalella sp.* and *Gammarus sp.*) found in the canal, despite the canal receiving flows only in the summer. These taxa may have been abundant in the canal due to the lack of predatory fish (Covich et al., 2010; Simon & Travis, 2011; Thorp & Rogers, 2011). Moreover, the time of year that I sampled may have influenced the abundance of *Hyalella sp.* in the canal. This genus can produce multiple broods each breeding season, which is typically during spring and summer (Covich et al., 2010; Thorp & Rogers, 2011).

The amphipods *Hyalella sp.* and *Gammarus sp.* have also been found in other man-made waterways. *Hyalella sp.* were abundant in modified streams with intermittent flows, which were developed for agricultural drainage in southwestern Minnesota, and *Gammarus sp.* were abundant in an agricultural ditch (Marsh & Waters, 1980). Future

research should continue to assess why these amphipods are prevalent in artificial water bodies, like the canal.

Leaf Decomposition

The canal generally had faster leaf decomposition compared to the Logan River, possibly due to the invertebrate assemblages. For instance, the canal had a high biomass and abundance of the facultative shredders *Hyalella sp.* and *Gammarus sp.*, which may have led to faster leaf decomposition. Shredder biomass increased with the absolute value of the decay rate in the canal (p=0.0088, $b_{\text{shredder biomass}}$=0.012, simple linear regression), suggesting that the canal had faster leaf decomposition due to the facultative shredders feeding on the leaves. This idea aligns with previous studies that have found amphipods to influence leaf decomposition in urban streams (Cook & Hoellein, 2016; Dangles & Malmqvist, 2004). Some studies have also found that a few dominant taxa, rather than richness of taxa, drive leaf decomposition, (Dangles & Malmqvist, 2004; Tolkkinen et al., 2013). It is possible that a few key taxa of amphipods like *Hyalella sp.* and *Gammarus sp.* are driving leaf decomposition in the canal. Future studies should evaluate how patterns of dominant taxa influence ecosystem processes in urban water bodies.

Leaf Decomposition in the Absence and Presence of Freshwater Invertebrates

Freshwater invertebrates and the physical forces from the water current both likely play a role in leaf decomposition. In the absence of freshwater invertebrates and high-water current, the leaves in the lab experiment decayed more slowly than the leaves incubated in both waterways in the field. The physical forces from the water current could have influenced leaf decomposition because the mean water velocity was only

0.006 m/s in the lab experiment, as opposed to 0.315 m/s in the canal and 0.198 m/s in

the Logan River.

How Do Water Quality, Freshwater Invertebrates, and Leaf Decomposition Change Along an Urban Gradient?

Water Quality

Neither waterway showed an accumulation of nutrients and metals through the

city of Logan, but higher concentrations of nutrients and metals occurred in the most

downstream site of the Logan River. The more downstream site along the Logan River

may have had higher concentrations of nutrients and metals as a function of watershed

size (Likens & Buso, 2006). For instance, Blacksmith Fork River and Spring Creek feed

into the Logan River upstream of the site I sampled near Mendon, where there is a

transition between urban and agricultural areas (Neilson et al., 2021; *The National Map -*

Advanced Viewer, n.d.). Despite these tributary inputs, the lowest reach of the Logan

River may have had higher concentrations of nutrients from a variety of nonpoint and

point sources. Sources of pollution at this site in Logan River could include animal

manure and chemical fertilizers from agricultural fields, urban stormwater runoff, leaking

septic tanks, and wastewater from sanitary sewers (Bernhardt et al., 2008; Delesantro et

al., 2022; Puckett, 1995). Hall et al. (2015) observed higher ^{15}N signatures in riparian

plant tissue in the lower reaches of the Logan River, suggesting nutrient pollution from

animal manure and septic tank leakage.

There are several possible sources for the heavy metals, iron and lead, in the canal

and Logan River For example, just upstream of the most downstream site of the Logan

River, there are old cars lining the Logan River, which were placed for the purpose of stabilizing the banks (Howe, 2021) and which could leach heavy metals, such as iron. Arentsen et al. (2004) found that some canals in Cache Valley are polluted with heavy metals from stormwater runoff. However, concentrations of lead from most sites were variable and only slightly higher than the average ambient concentrations (3 mg/L) of lead in groundwater (American Public Health Association, 2012), suggesting a natural source.

Freshwater Invertebrates

Multiple functional feeding groups were supported from upstream to downstream sites in both the canal and the Logan River. Facultative shredders (*Hyalella sp.* and *Gammarus sp.*) and collector-gatherers, which were mainly Chironomidae, were abundant at the most downstream sites of the canal, whereas all sites in the Logan River had mainly collector-gatherers (Chironomidae) which feed on fine- particulate organic matter. It is not surprising that collector-gatherers were abundant in the Logan River given the position of the urbanized Logan River in the river continuum, (Vannote et al., 1980). Additionally, the facultative shredders can feed on leaf litter in addition to having a generalist feeding habit (Covich et al., 2010; Strong, 1972; Thorp & Rogers, 2011).

Assemblages of freshwater invertebrates occurred throughout the canal and the Logan River. The lowest reach of the Logan River had lower density and richness compared to the canal and upstream sites of the Logan River, potentially due to poorer water quality as indicated by elevated concentrations of nutrients and metals. This explanation aligns with other studies that have found reduced invertebrate richness and abundance in polluted streams (Clements et al., 2000; Walsh et al., 2005).

Leaf Decomposition

Higher leaf breakdown rates occurred at downstream sites in both waterways, relative to the upstream sites, potentially due to water quality and shredder assemblages. The most downstream sites of the Logan River had faster breakdown rates, likely due to elevated concentrations of nutrients, as elevated concentrations of nutrients accelerate leaf breakdown by stimulating the growth of the microbial activity on leaf litter (Pascoal et al., 2003; Tant et al., 2015; Webster & Benfield, 1986). Moreover, the most downstream sites of the canal had faster leaf decomposition relative to all other sites in the canal and the Logan River, possibly due to the high biomass of the shredders. This idea aligns with other literature that has found amphipods, like *Hyalella sp.* and *Gammarus sp.,* to influence leaf decomposition (Cook & Hoellein, 2016).

Associations Between Leaf Decomposition and Physical, Chemical, and Biological Factors

It is possible that water velocity and other physical forces contribute to leaf breakdown (Bastias et al., 2020) because water velocity was negatively associated with leaf decomposition. Similar to water velocity, shear velocity is another physical factor that could be associated with leaf breakdown. Shear velocity is related to the bed shear stress moving along the surface of the stream bed and its ability to transport sediment (Garcia, 2008). High water velocities can increase shear stress at the stream bed, which can increase the transport of leaves from a specific location (Bastias et al., 2020; Cordova et al., 2008), which could potentially influence leaf breakdown. To evaluate the association between local shear stress at the bed and the dry mass of leaves remaining (i.e., leaf breakdown), I calculated shear velocity or the local shear stress at the bed by

using the Stickler-Manning flow resistance formula (Garcia, 2008). To calculate shear

velocity using this formula, I used estimates of grain size and local measurements of

water depth and water velocity that were taken near the leaf bags. After calculating shear

velocity for each site, I found that shear velocity was negatively associated with dry mass

of leaves remaining (Spearman's correlation coefficient = -0.64, p=0.002), which was

similar to the negative association between water velocity and dry mass remaining

(Spearman's correlation coefficient = -0.63, p=0.003). It is possible that both shear

velocity and water velocity are associated with leaf breakdown similarly, where the

physical forces from the water current and shear stress at the bed are pushing leaf litter

out and from the leaf bags. However, I did not measure the grain size of streambed

substrates surrounding the leaf bags. This information is needed to accurately assess the

association between shear velocity and leaf breakdown (Cordova et al., 2008; Larrañaga

et al., 2003).

I speculate that total phosphorus was positively associated with biomass of

shredders and dry mass of leaves remaining because this limiting nutrient was driving the

colonization of microbes on leaves. For example, microbes first assimilate the limited

nutrient (e.g., total phosphorus), which leads to more palatable leaves for invertebrates,

and helps to provide nutrients to shredders (Cummins, 1974; Graça, 2001; Tant et al.,

2015). In this case, total phosphorus was a limiting nutrient, because the ratio (by mass)

of total nitrogen to total phosphorus was greater than 8:1 across all sites. Numerous

studies have found that limiting nutrients drive microbial activity, which then can

accelerate leaf decomposition, especially in landscapes modified by humans (Kominoski

et al., 2015; Usher et al., 2020; Webster & Benfield, 1986). Additionally, one study found

that nutrients increase the biomass of both microbes and shredders, further increasing the process of leaf decomposition (Tant et al., 2015).

Limitations

There were several limitations in this study. The sampling timeframe for this study was much shorter than anticipated because leaf litter from the bags completely decomposed within three to four weeks. Most studies of leaf decomposition have evaluated the process of leaf decomposition by deploying and retrieving leaf litter bags over a time frame of 2-4 months (Lamberti & Hauer, 2017), instead of three to four weeks, making it difficult to compare this study to other leaf breakdown studies. I speculate that the leaves may have decomposed more quickly than anticipated because the leaves were fresh, and not yet senesced. One study found fresh leaves to have a generally fast decay rate relative to senesced leaves, possibly due to a higher nutrient content in fresh leaves (Maloney & Lamberti, 1995). Additionally, sites along the Logan River and the canal may not be representative of the waterways. I only chose sites along the Logan River and the canal that were accessible via public property such as a city road or park. I was not able to interpret the stream temperature data in both waterways. I was only able to collect one stream temperature measurement at each site every 3-4 days, and these measurements were not collected at consistent times of the day, so I could not discern whether temperature was changing longitudinally or temporally.

Future Research

Future research should continue to study artificial aquatic ecosystems, like the canal, because their ecological condition is less known than their natural counterparts (Clifford & Heffernan, 2018). One should consider following this study in this canal and others in Cache Valley over a longer period of time to better understand water quality, invertebrate communities, and leaf decomposition of these novel ecosystems. For example, there could be a more complete explanation as to why facultative shredders (*Hyalella sp.* and *Gammarus sp.*) were abundant at downstream sites in the canal and whether or not these amphipods are driving leaf decomposition. One should also measure water temperature continuously to evaluate how stream temperature affects leaf decomposition, and to further assess how a warming climate may affect the ecological structure and function of these man-made conveyance systems. Future research in these canals should consider how other factors like shear stress, microbial activity, and management activities of the canal (e.g., chemical weeding and mowing of vegetation along the canal, regulation of water depth) might also be shaping ecosystem structure and function in these urban waterways (Clifford & Heffernan, 2018; Lin et al., 2020).

Conclusions

Overall, I gained a better understanding of water quality, freshwater invertebrate assemblages, and leaf decomposition of one artificial water body, an urban canal, relative to its main source and a natural stream, the Logan River. For example, the water quality of both waterways was comparable likely due to sharing a common water source. Both waterways also supported freshwater invertebrates, where facultative shredders (*Hyalella*

sp. and *Gammarus sp.*) were abundant in the canal relative to the urbanized reach of the Logan River. The canal overall had faster leaf decomposition than the Logan River, where facultative shredders (*Hyalella sp.* and *Gammarus sp.*) might play a role in influencing leaf decomposition, particularly in the canal. Moreover, I have a better understanding of water quality, freshwater invertebrates, and leaf decomposition along an urban gradient in a part of the urban watershed in Cache Valley, Utah, particularly during the summer months. The most downstream site of the Logan River had poorer water quality, reduced richness, and invertebrate density compared to the canal and the upstream sites of the Logan River. Additionally, the most downstream sites had faster leaf decomposition in both waterways, possibly due to an abundance of shredder assemblages in the canal, and elevated nutrients at the most downstream site in the Logan River. Finally, some physical, chemical, and biological factors were associated with the process of leaf decomposition in both waterways. Water velocity was associated with leaf decomposition. Additionally, total phosphorus was associated with leaf decomposition and shredder biomass, possibly via enhancement of microbial activity.

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TABLES AND FIGURES

Table 1

Means and standard errors of physicochemical variables including stream temperature, total nitrogen, ammonium, nitrate, soluble reactive phosphorus, total phosphorus, water velocity, TSS, and metals in each waterway

Waterway	Canal		Logan River		
Variable	Mean +/- SE	Range	Mean +/- SE	Range	p-value
NH4 (mg/L)	0.0159±0.0005	0.002-0.024	0.0231±0.0013	0.002-0.078	0.002*
NO3 (mg/L)	0.1118±0.005	0.067-0.163	0.16552±0.011	0.018-0.441	0.089
TN (mg/L)	0.2085±0.004	0.163-0.324	0.26018±0.015	0.126-0.86	0.481
TP (mg/L)	0.02202±0.0005	0.005-0.033	0.01884±0.0006	0.011-0.048	0.034*
SRP (mg/L)	0.00588±0.0005	0.001-0.026	0.0076±0.0006	0.001-0.026	0.194
Stream temperature (Celsius)	14.336±0.099	12.7-17.4	14.912±0.14	12.6-18.3	0.143
TSS (mg/L)	0.032±0.010	0-0.581	0.00836±0.0027	0-0.185	0.002*
Pb (mg/L)	0.007±0.0003	0.00225-0.012	0.00737±0.00029	0.00225-0.017	0.548
Fe (mg/L)	0.00222±0.0001	0.0004-0.006	0.002972±0.00032	0.0004-0.014	0.423
Water velocity (m/s)	0.315±0.022	0.055-0.7083	0.198±0.021	0-0.543	0.0826

Note. The * indicates a significant difference between waterways. The range provides the

minimum and maximum value for each variable.

Table 2

Total biomass (mg) and total abundance of shredder taxa for each waterway taken from the coarse-mesh leaf bags

	Canal		Logan River	
Shredder Taxa	Total Biomass	Total Abundance	Total Biomass	Total Abundance
Hyalella sp.	2395.78	8430	22.44	114
Gammarus sp.	511.43	40	10.13	9
Lepidostoma sp.	0	0	3.68	33
Malenka sp.	0	0	1.16	11
Onocosmoecus sp.	0	0	69.25	2
Onocosmoecus unicolor	0	0	72.77	2
Psychoglypha sp.	0	0	22.52	1

Table 3

Top five most abundant taxa by waterway and type of sample

Canal		Logan River	
Surber samples	Leaf bags	Surber samples	Leaf bags
Non-Tanypodinae	Non-Tanypodinae	Non-Tanypodinae	Non-Tanypodinae
Hyalella sp.	*Hyalella sp.*	Oligochaeta	*Simulium sp.*
Oligochaeta	*Baetis sp.*	*Oligophlebodes sp.*	*Baetis sp.*
Baetis sp.	*Lymnaea sp.*	*Simulium sp.*	Cladocera
Simulium sp.	*Simulium sp.*	*Baetis sp.*	Trombidiformes

Note. Non-Tanypodinae is a subset of the family Chironomidae.

Table 4

Spearman's correlation coefficients to evaluate relationships between leaf decomposition and total nitrogen, total phosphorus, water velocity, CCU, and shredder biomass based on the linkages in Figure 1

Variable	Shredder Biomass (mg)	Dry Mass Remaining (g)
Water Velocity (m/s)		-0.63*
TN (mg/L)	0.02	0.01
TP (mg/L)	0.79*	-0.62*
CCU (iron and lead)	-0.13	0.3
Shredder Biomass (mg)		-0.4

Note. The symbol "*" represents a p-value of less than 0.01.

Figure 1

Conceptual model depicting known effects of physical, chemical, and biological factors on leaf decomposition
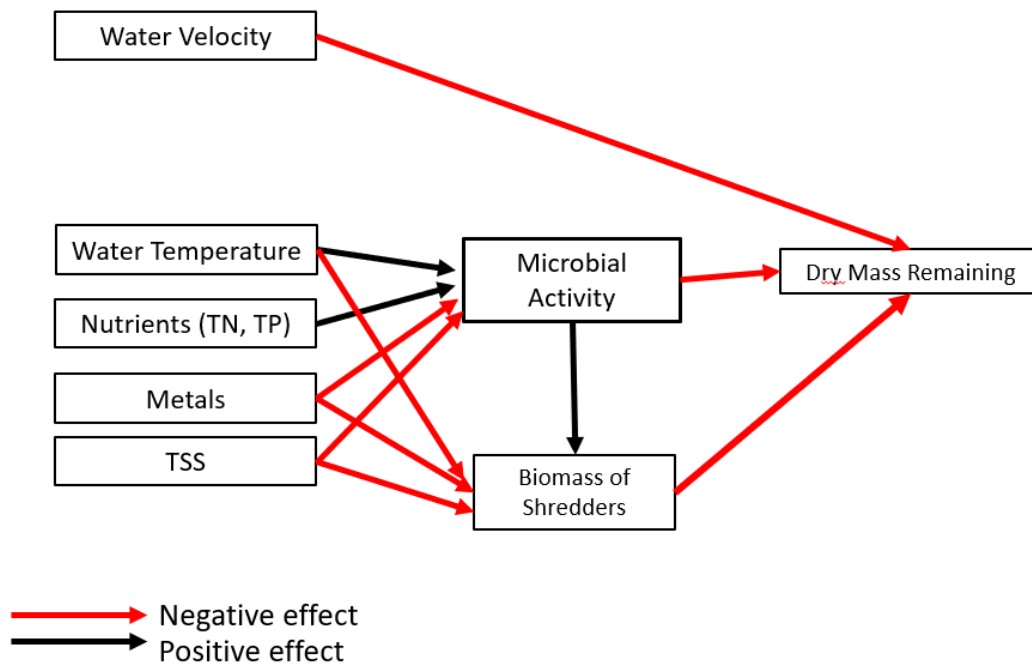
Note. Dry mass remaining represents the mass of leaves remaining after they have decomposed.

Figure 2

Map of sites along Logan River and Northwest Field Canal
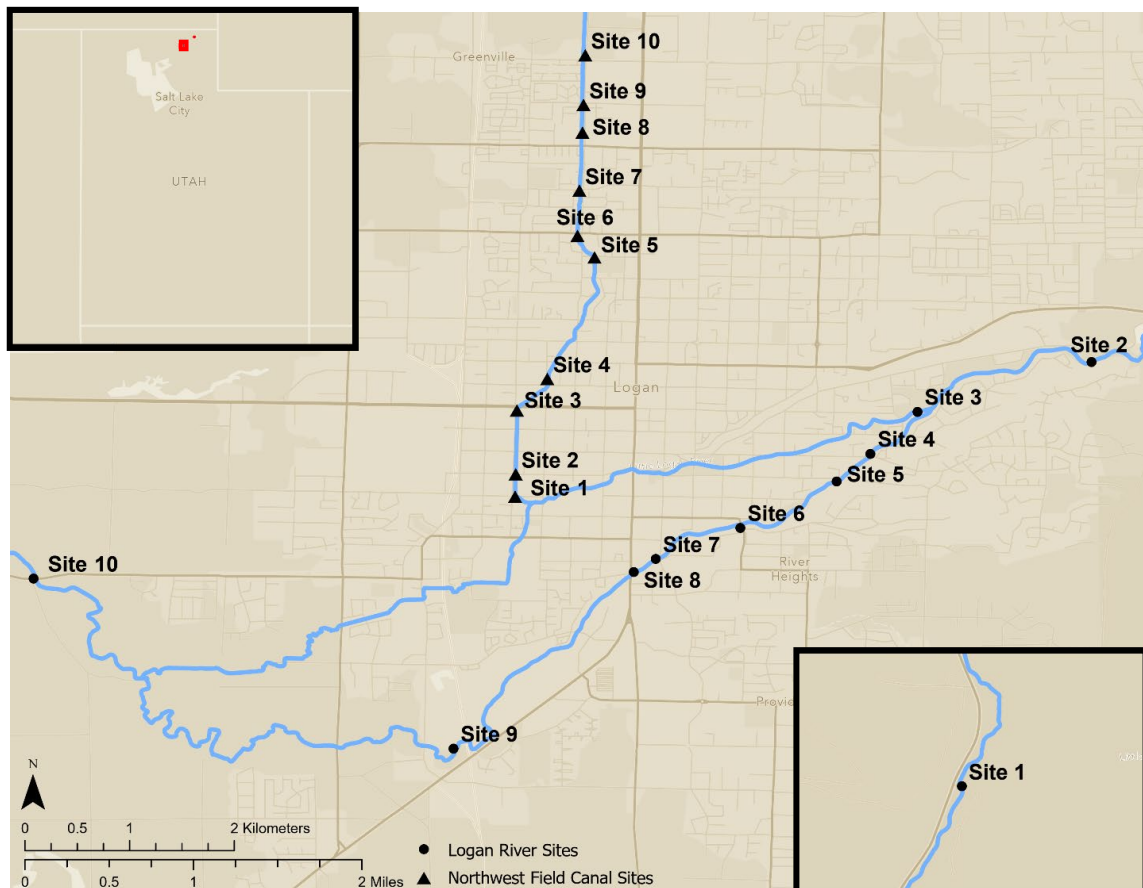
Note. Site 1 in the Logan River and canal are the most upstream sites. Site 10 in the canal and the Logan River are the most downstream sites. Site 1 in the Logan River is located approximately 31 kilometers above Site 2 in the Logan River.

Figure 3

Sampling design of leaf litter bags at each site
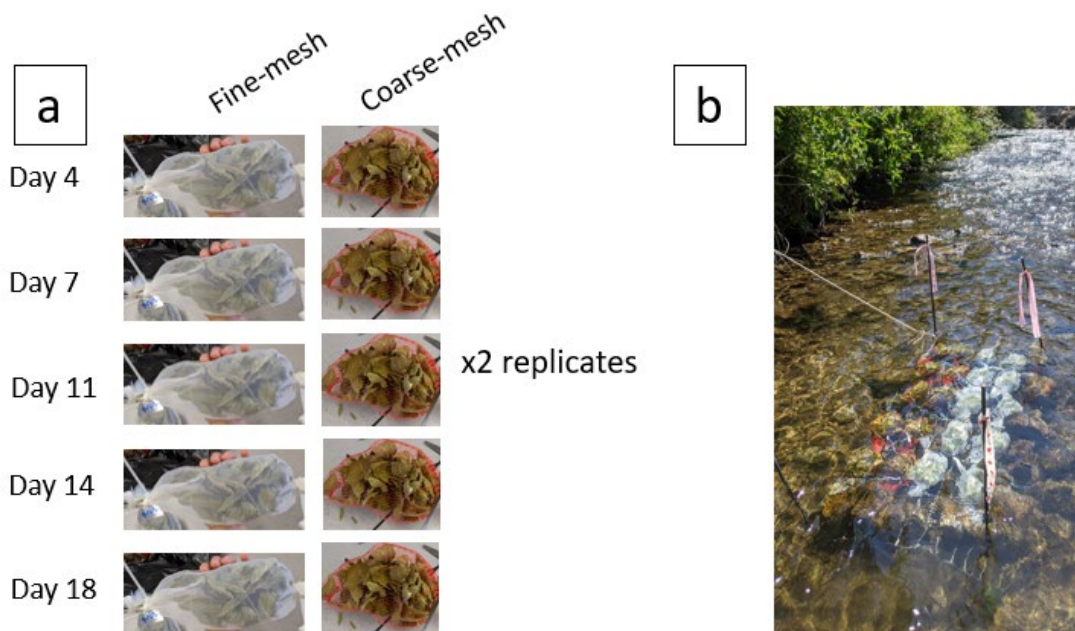
Note. a) design of fine-mesh and coarse-mesh bags that were collected on Days 4, 7, 11, 14, and 18 at each site, and b) the setup of the leaf litter bags at one site in the Logan River.

Figure 4

Boxplot of mean shredder biomass in milligrams per coarse-mesh leaf bag across sites for the canal and Logan River
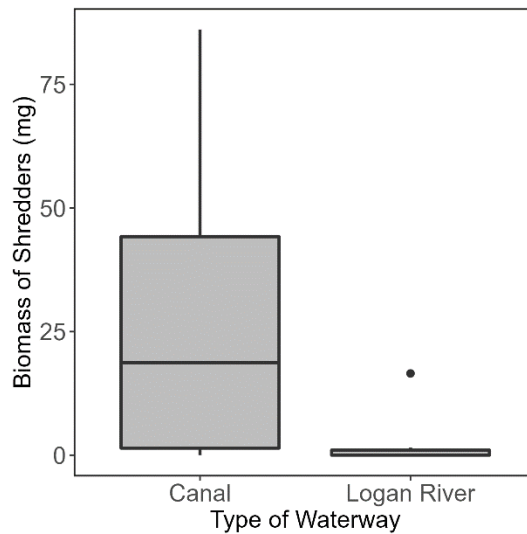
Note. The canal had higher biomass of shredders (p<0.05). Black lines indicate the

median biomass. The end of the whiskers represents the maximum and minimum of

biomass, and the edge of the boxes indicate the first and third quartiles. Outliers are

represented as black dots.

Figure 5

Boxplots of rarefied richness and total invertebrate count for each waterway and type of invertebrate sample
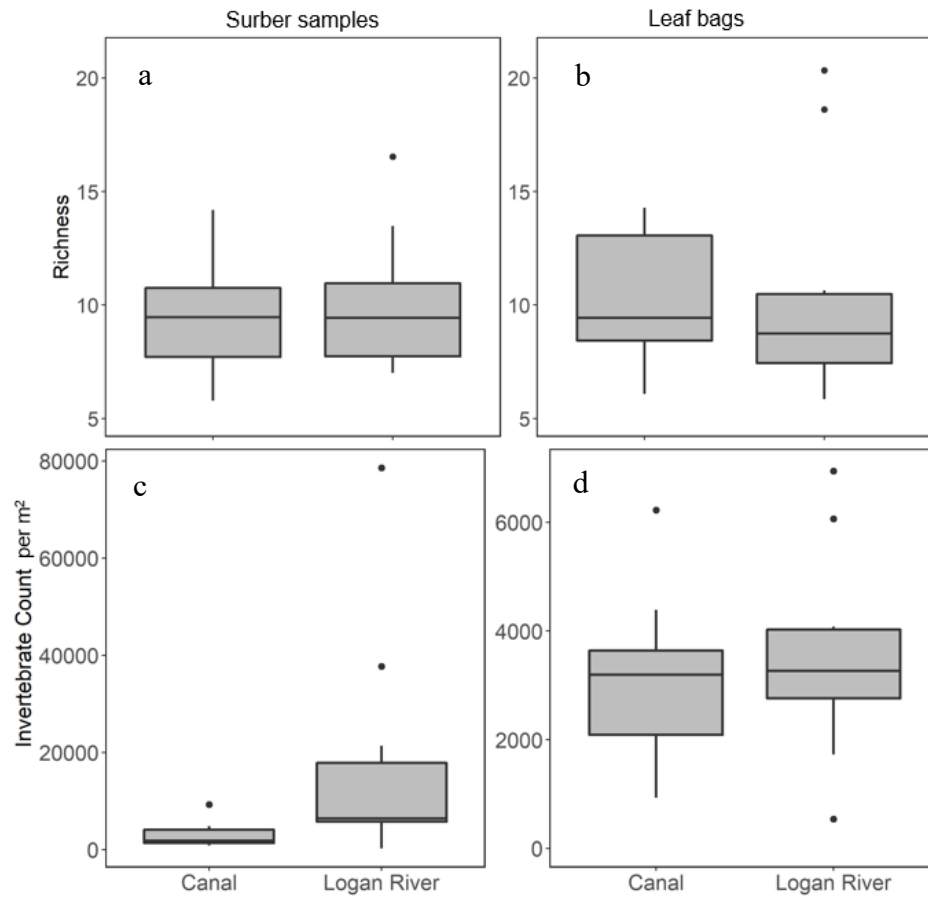
Note. a) rarefied richness from the Surber samples, b) rarefied richness from the leaf bags, c) estimated count of invertebrates per square meter from the Surber samples, d) invertebrate count from the leaf bags. Note that black lines indicate the median of count and richness for each site. The end of the whiskers represents the maximum and minimum values, and the edge of the boxes indicate the first and third quartiles. Outliers are represented as black dots. Note that there are different scales in the y-axis between panels c and d.

Figure 6

Box plots of percent of AFDM remaining and the absolute value of the decay rates for the canal and Logan River
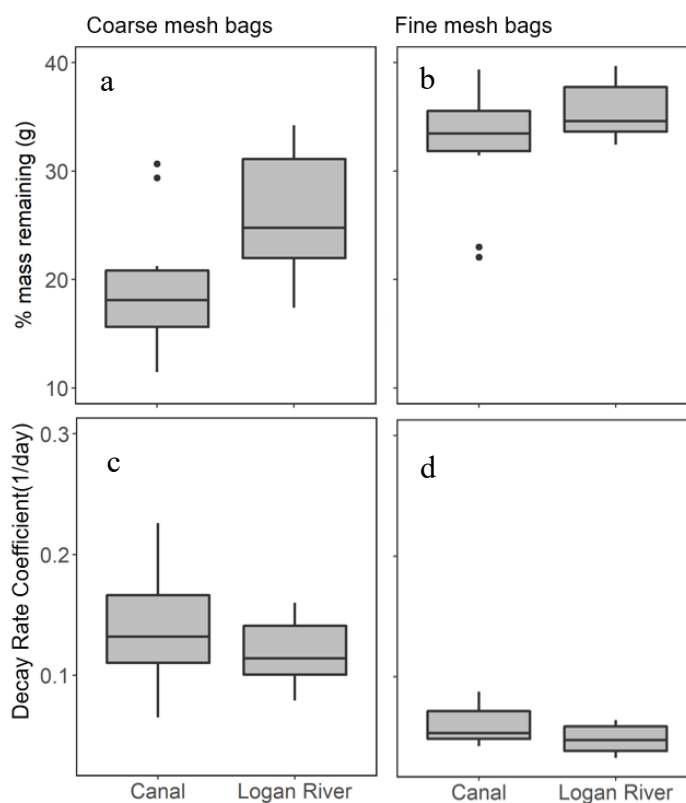
Note. a) the percent of mass remaining in the coarse-mesh bags, b) the percent of mass remaining in the fine-mesh bags, c) the decay rate in the coarse-mesh-bags, and d) the decay rate in the fine-mesh bags. Decay rates with a larger value indicate faster leaf breakdown. Also note that the black bolded lines indicate the median. The end of the whiskers represents the maximum and minimum values, and the edge of the boxes indicate the first and third quartiles. Outliers are represented as black dots.

Figure 7

The relationship between biomass of shredders (log-scale) and the absolute value of decay rate
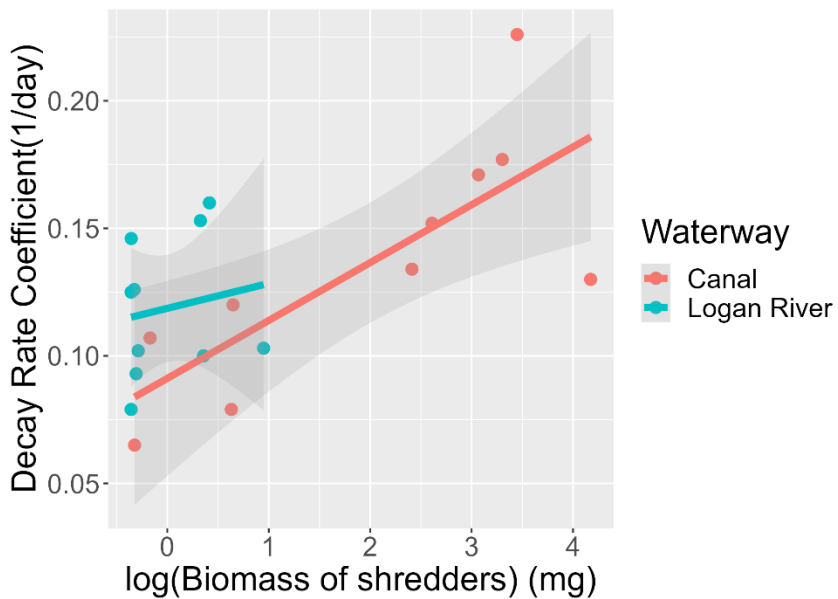
Note. The red line represents the predicted values of the canal, and the blue line represents the predicted values of the Logan River. The shaded areas around the lines and data points represent 95 percent confidence intervals. Decay rates with a larger absolute value indicate faster leaf breakdown.
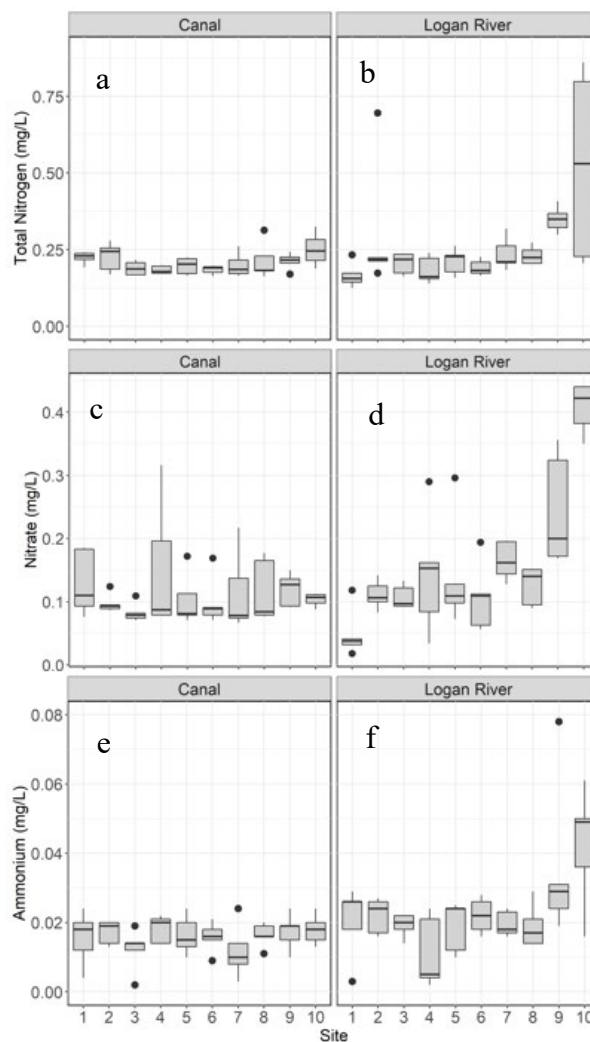
Figure 8

The relationship between decay rate and rarefied invertebrate richness by site from a)
the Surber samples, and b) the leaf bags
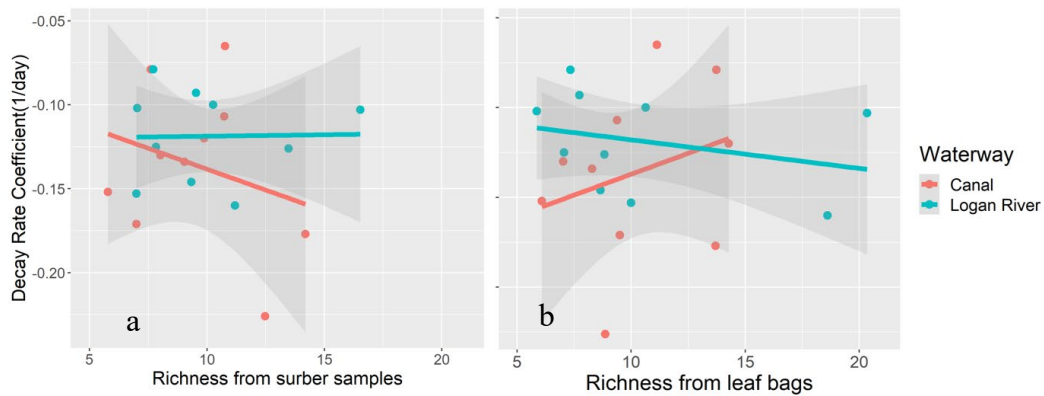
Note. The shaded area around the lines represent 95 percent confidence intervals.

Figure 9

Longitudinal patterns of concentrations of total nitrogen, nitrate, and ammonium


Note. a) total nitrogen in the canal, b) total nitrogen in the Logan River, c) nitrate in the canal, d) nitrate in the Logan River, e) ammonium in the canal, and f) ammonium in the Logan River. Sites are ordered by distance downstream. Site 10 is the most downstream site, and site 1 is the most upstream site. Bolded black lines in the boxplots indicate the median concentration for each site. The end of the whiskers represents the maximum and minimum concentrations, and the edge of the boxes indicate the first and third quartiles. Outliers are represented as black dots.

Figure 10

*Longitudinal patterns of concentrations of total phosphorus and SRP*
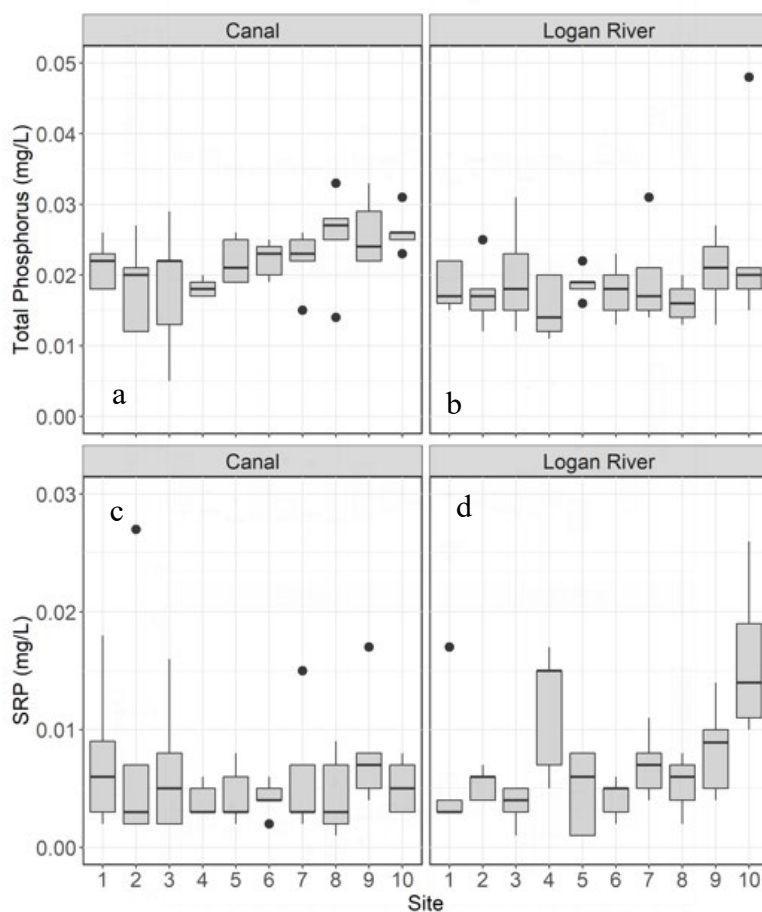
Note. a) longitudinal patterns of total phosphorus in the canal, b) longitudinal patterns of

total phosphorus in the Logan River, c) longitudinal patterns of SRP in the canal, and d)

longitudinal patterns of SRP in the Logan River. Sites are ordered by distance

downstream. Site 10 is the most downstream site, and site 1 is the most upstream site.

Bolded black lines in the boxplots indicate the median concentration for each site. The

end of the whiskers represents the maximum and minimum concentrations, and the edge

of the boxes indicate the first and third quartiles. Outliers are represented as black dots.

Figure 11

*Longitudinal patterns of concentrations of iron and lead*
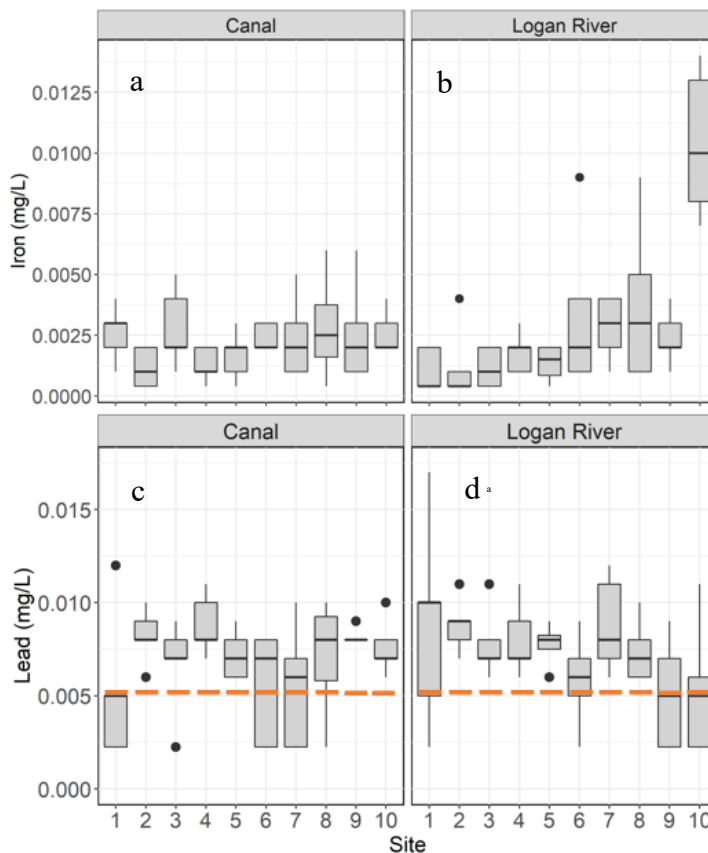
Note. a) longitudinal patterns of iron in the canal, b) longitudinal patterns of iron in the

Logan River, c) longitudinal patterns of lead in the canal, and d) longitudinal patterns of

lead in the Logan River. Sites are ordered by distance downstream. Site 10 is the most

downstream site, and site 1 is the most upstream site. The orange lines denote the

criterion for lead (0.0053 mg/L) from the National Recommended Water Quality Criteria

(US EPA, 2015). Bolded black lines in the boxplots indicate the median concentration

for each site. The end of the whiskers represents the maximum and minimum

concentrations, and the edge of the boxes indicate the first and third quartiles. Outliers are

represented as black dots.

Figure 12

Longitudinal patterns of TSS
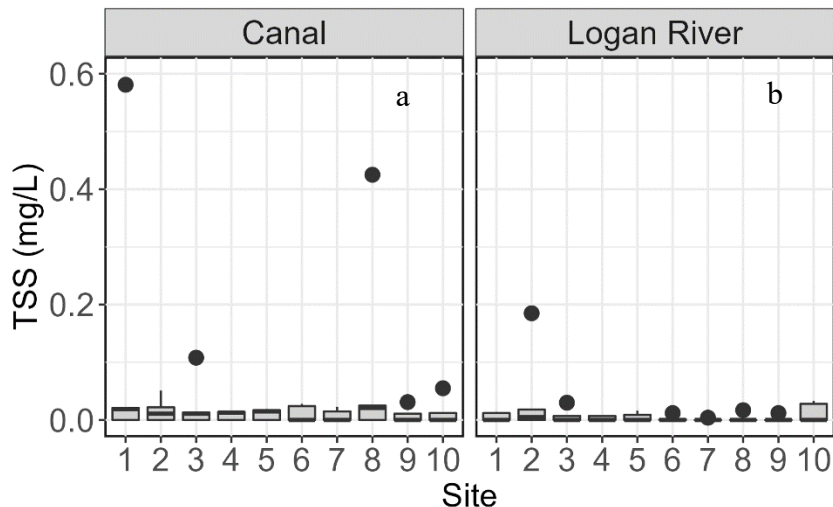
Note. a) longitudinal patterns of TSS in the canal, and b) longitudinal patterns of TSS in the Logan River. Sites are ordered by distance downstream. Site 10 is the most downstream site, and site 1 is the most upstream site. Bolded black lines of the boxplots indicate the median concentration for each site. The end of the whiskers represents the maximum and minimum concentrations, and the edge of the boxes indicate the first and third quartiles. Outliers are represented as black dots.

Figure 13

Longitudinal patterns of FFGs from all days of leaf bag retrieval
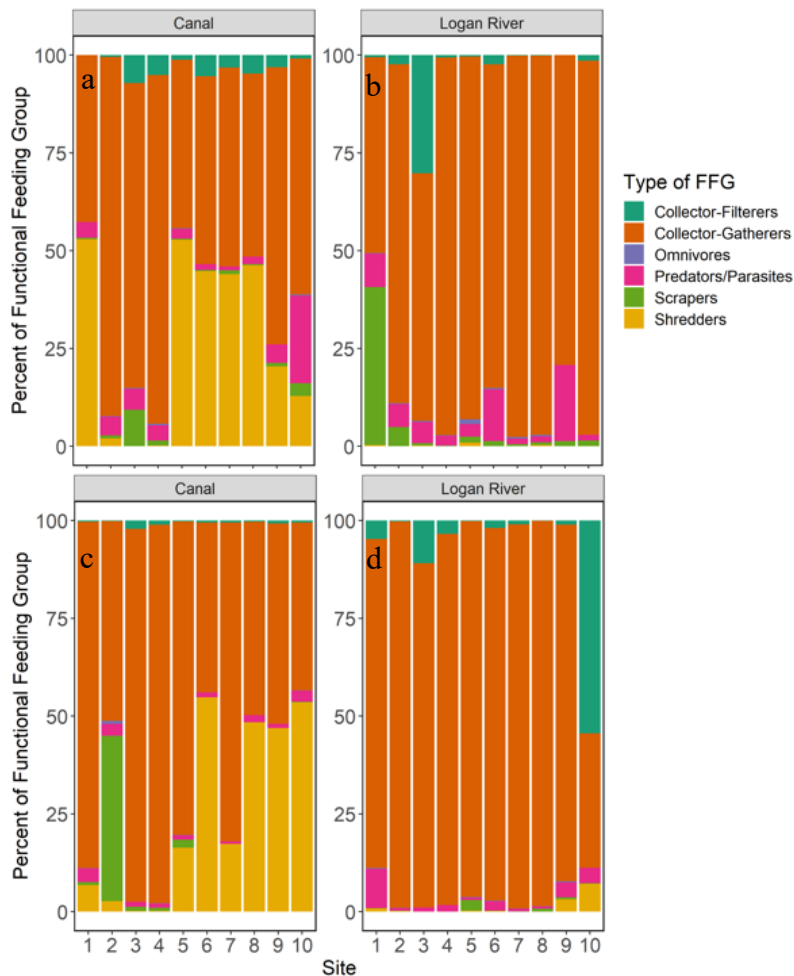
Note. a) FFGs in the Surber samples from the canal, b) FFGs in the Surber samples from the Logan River, c) FFGs in the coarse-mesh leaf bags from the canal, and d) FFGs in the coarse-mesh leaf bags from the Logan River. Site 1 is from the most upstream location, and Site 10 is located at the most downstream location in both waterways.

Figure 14

Longitudinal patterns of rarefied invertebrate richness and density for both leaf bags and Surber samples
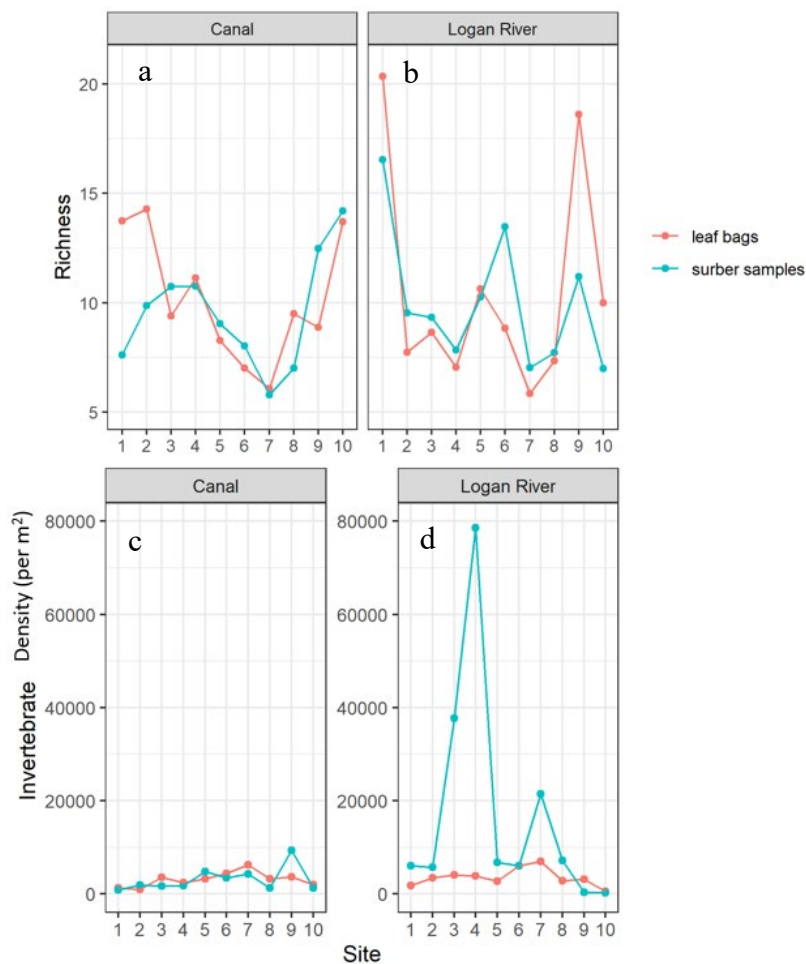
Note. The USU Bug Lab estimated the density of invertebrates per square meter from the Surber samples, whereas invertebrate density from the leaf bags was calculated as the total number of invertebrates from all leaf bags collected from each site. Note that the y-axis for invertebrate count and the x-axis for distance downstream are vastly different between the Logan River and the canal. Site 1 is from the most upstream location, and Site 10 is located at the most downstream location in both waterways.

Figure 15

Longitudinal patterns of the percent of AFDM remaining (g) from Day 18 and the absolute value of decay rate
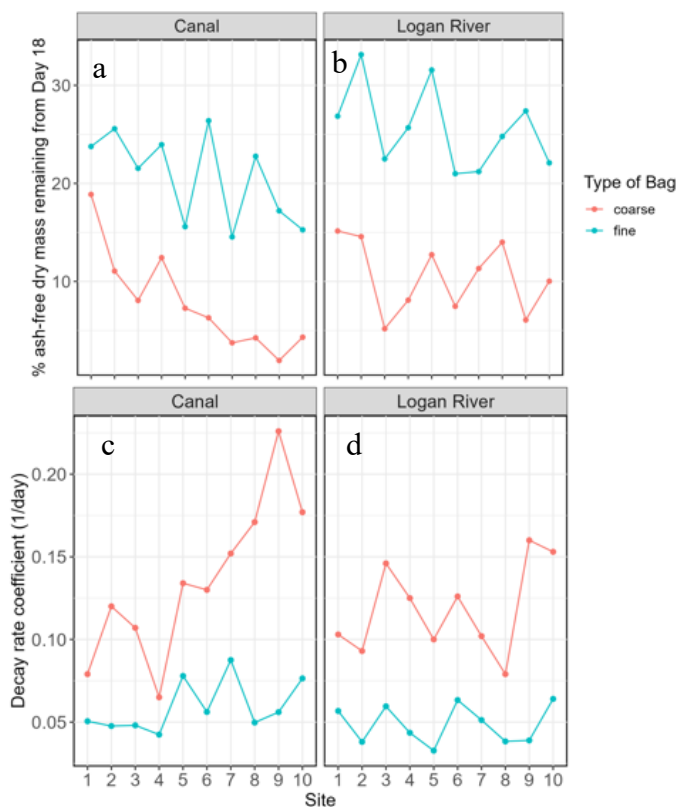
Note. a) Percent of AFDM remaining in the canal, b) percent of AFDM remaining in the Logan River, c) decay rate coefficient (1/day) in the canal, and d) decay rate coefficient in the Logan River. Each color of line represents the type of bag. Decay rates with a larger absolute value indicate a faster leaf breakdown process. Leaf bag replicates from Day 18 were averaged by waterway and site for % AFDM remaining. Site 1 is from the most upstream location, and Site 10 is located at the most downstream location in both waterways.

APPENDIX

Table A 1

*Decay rates across all sites for the Logan River and the canal*

	Decay Rate (1/day)			
	Canal		Logan River	
Site	fine-mesh	coarse-mesh	fine-mesh	coarse-mesh
1	-0.051	-0.079	-0.057	-0.103
2	-0.048	-0.120	-0.038	-0.093
3	-0.048	-0.107	-0.060	-0.146
4	-0.042	-0.065	-0.044	-0.125
5	-0.078	-0.134	-0.033	-0.100
6	-0.056	-0.130	-0.063	-0.126
7	-0.088	-0.152	-0.051	-0.102
8	-0.050	-0.171	-0.038	-0.079
9	-0.056	-0.226	-0.039	-0.160
10	-0.076	-0.177	-0.064	-0.153

Note. Site 1 in both waterways are the most upstream location, whereas site 10 in both waterways is the most downstream location.
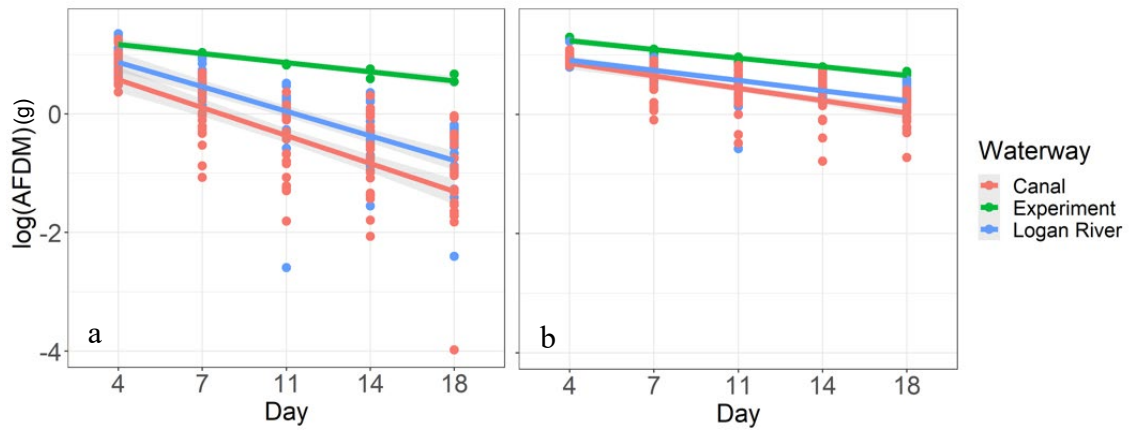
(g)

Figure A 1

AFDM in grams (log-scale) over time (days) for fine-mesh and coarse-mesh bags from the lab experiment in the lab
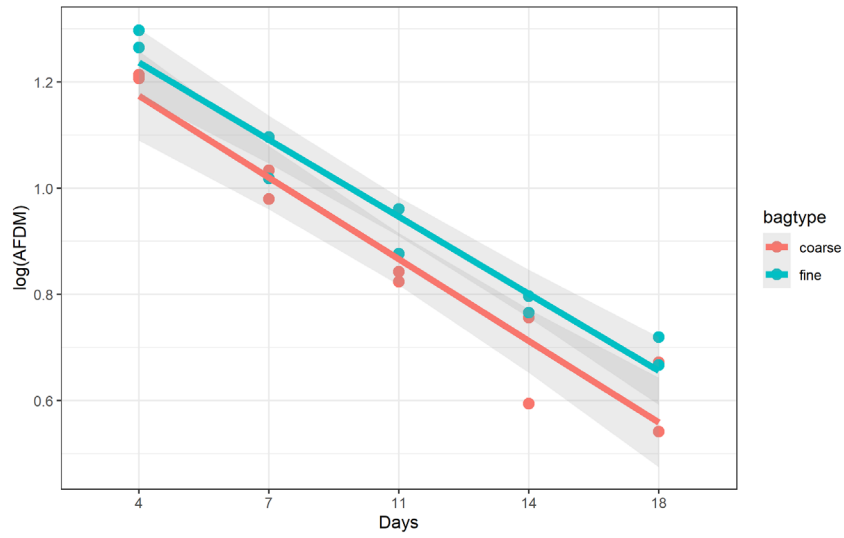
Note. The shaded area represents 95 percent confidence intervals

Figure A 2

AFDM in grams (log-scale) over time (days) for the lab experiment, canal, and Logan River for a) coarse-mesh bags and b) fine-mesh bags

Note. The green lines and data points represent the experiment, the blue lines and data points represent the Logan River, and the red lines and data points represent the canal. The shaded area represents 95 percent confidence intervals.