EFFECTS OF LOW-HEAD DAMS ON HABITAT STRUCTURE, CARBON AND NITROGEN ALLOCATION, AND MICROBIAL ACTIVITY IN URBAN RIVERS

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By

Lauren E. McGee, B.S.

The Ohio State University
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Approved by

Master's Examination Committee:

Dr. Virginie Bouchard, Adviser ________________________________

Dr. Richard Dick Adviser

Dr. Peter Smiley Environmental Science Graduate Program

ABSTRACT

Humans have negatively altered riverine ecosystems primarily in two ways: (1) indirectly through landscape alteration and (2) directly through modifying the physical structure of river channels themselves. Agriculture and urban development are two popular forms of land use change – with urbanization being unique in that the effects are mostly seen at the hydrological level (e.g., streams tend to show flashy hydrographs). With direct modification of rivers, dam installation predominates, as large high-head dams starve downstream estuaries of sediments and reduce in-stream aquatic biodiversity of upstream reaches. Research, however, is lacking on the effects of small, low-head dams (typically < 5 m in height) to rivers, and in particular, rivers found in urban areas. My study objective was to assess how low-head dams impact urban riverine structure (habitat, carbon and nitrogen allocation in water and sediment) and function (retention or removal of carbon and nitrogen, microbial activity). I hypothesized that low-head dam reservoirs would create conditions that promoted the settling of fine-sized sediments and other materials, therefore making these systems greater sinks for carbon (C) and nitrogen (N) than reference, riverine reaches. I also hypothesized that reservoirs would have greater rates of anaerobic microbial processes in sediments and reduced rates of aerobic processes. I expected that the reduction in water flow in reservoirs would be the primary factor leading to these changes. I conducted this study in five paired reservoir-reference (urban riverine) sites in central Ohio. Sampling occurred during spring, summer, and autumn 2007. Structural components (i.e., water velocity, depth, water physical properties) were measured *in situ*; water and sediment samples were analyzed in the lab for system structure (i.e. sediment particle size) and function (i.e. C and N retention, biological transformations of C and N). A two-factor block ANOVA was used to determine differences between reservoir and reference reaches. Correlation analyses were used to assess linkages amongst studied parameters (e.g., water velocity with other variables). Most hypotheses in this study were confirmed. Low-head dam reservoirs had slower water velocities and greater water depths that reference reaches. Reservoirs also contained more fine-sized sediment, stored more C and N in sediments, and had greater rates of denitrification. However, concentrations of C and N in waters were similar in low-head dam reservoirs and reference reaches, indicating that reservoirs were not strong sinks for C and N. Also, rates of nitrification and methane oxidation were similar between reservoir and references reaches. Although water velocities were reduced in low-head dam reservoirs, water flow was not correlated with storage of C and N in sediments. This lack of a correlation suggests that although reservoirs can attenuate water flows in urban rivers, reductions may not be strong enough to lead to removal of materials (C and N) from overlying waters. Water flow occurred too rapidly in these study sites. Urbanization played a larger role in affecting ecosystem function in the studied urban rivers.

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VITA

March 26, 1984 .... Born – (Los Angeles, CA)

2005 .... B.S. Ag./Earth and Environmental Science
North Carolina Agricultural and Technical
State University

2006-present... Fellow and Graduate Teaching Associate,
The Ohio State University

FIELDS OF STUDY

Major Field: Environmental Science

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CHAPTER 1

INTRODUCTION

Rivers and streams have been referred to as "arteries of the landscape" (Grimm et al. 2005), mostly for their ability to efficiently transport material and energy derived from upstream reaches and the surrounding terrestrial landscape. Unlike arteries, however, these systems do not merely transport materials (Grimm et al. 2005). They can also transform, retain or remove materials that may disrupt the ecological integrity of downstream ecosystems (Postel and Carpenter 1997). The ability for streams and rivers (hereto referred to as rivers) to serve as sinks for materials is an example of an ecosystem service, a service provided by natural ecosystems from which humans benefit (Daily 1997; Palmer et al. 2004; Meyer et al. 2005). Other examples of ecosystem services provided by rivers include: the removal of organic material through respiration (i.e., the degradation of wastewater treatment plant effluent), the production or accumulation of basal resources to support aquatic foodwebs, and the provision of leisure and recreational opportunities for humans (Jansson et al. 2007; Meyer et al. 2005; Palmer et al. 2004). Often, disturbance to riverine structure (e.g., habitat structure, resource allocation) and function (e.g., microbial activity) can disrupt these services provided by rivers to human societies. Such disturbances are frequently caused by physical landscape alteration and direct modification of the river channel itself.

Land uses which alter the structural and functional characteristics of rivers are agriculture, urbanization, logging, mining, and pasture grazing (Bryce et al. 1999; Allan 2004; Poff et al. 1997). Of these, agriculture and urbanization are important. Agriculture comprises approximately 40% of the landscape (Foley 2005; Gordon et al. 2007). It impairs rivers through excessive sediment, nutrient and pesticide inputs (Allan 2004) and through alteration of natural flow regimes (i.e., tile-drainage systems; Arango and Tank 2008). Degradation results in reduced habitat diversity, incised river channels, and increased deposition of fine-sized sediment particles (Allan 2004). Additionally, reduced ecosystem function has also been found in watersheds of predominately agricultural land use (e.g., N removal from waters; Royer et al. 2004; Arango and Tank 2008; Herrman et al. in press). Conversely, urban land use comprises a significantly smaller percentage of the landscape but has far-reaching impacts (Collins et al. 2000; Pickett et al. 2008). Urbanization has been named the primary culprit in imperiling over 130,000 km of rivers in the United States (Paul and Meyer 2001). The "urban stream syndrome" summarizes consequences of urban anthropogenic activities to riverine ecosystems (Meyer et al. 2005; Walsh et al. 2005). General "symptoms" include altered (flashy) hydrographs, channel incision, and excessive loading of nutrients and other pollutants (Paul and Meyer 2001). Of the few studies focusing on impacts to ecosystem function, reduced rates of nutrient removal processes (i.e., ammonium and nitrate from river water using nitrogen spiraling metrics and measures of microbial activity) has been a consistent response (Meyer et al. 2005; Grimm et al. 2005; Arango and Tank 2008). As humans are becoming a more urbanized species (Paul and Meyer 2001), symptoms of the urban stream syndrome may become more prevalent in riverine ecosystems.

In addition to land management practices, humans alter riverine ecosystems through direct channel/habitat alteration, with the most visible of these modifications being dam installation. Large dams are designed to control the flow of water (i.e., dams used for hydroelectric power generation or for flood-control purposes). They block the migration of fish populations (Drinkwater and Frank 1994; Grubbs and Taylor 2004), fragment and isolate aquatic habitats (Winston et al. 1991), alter natural foodwebs (Power et al. 1996), and create reservoirs that generally serve as sources of greenhouse gases (St. Louis et al. 2000) and sinks for nutrients, sediments, and organic matter (Ibanez et al. 1996; Hannan 1979; Kelly 2001). Not all dams, however, are large structures designed to completely control the flow of water. In fact, more than 90 % of dams (or more than two-million dams) on waterways in the United States are low-head dams that are less than five meters in height (Poff and Hart 2002; Shuman 1995) and designed to attenuate floods or store water for industrial purposes (e.g., milling). These low-head dams (LHDs) are unique in the sense that they do not completely impede the flow of water. Yet similar to high-head dams (HHDs), LHDs raise water levels and reduce water velocities (McCully 1996; Poff and Hart 2002). Relatively little research has been done on the ecological effects of these smaller structures to riverine ecosystems. Of the few studies, it has been shown that reservoirs created by LHDs form lentic-like habitats and result in a reduction of fish and macroinvertebrate species richness (Tiemann et al. 2004; Cumming 2004). These structures can also slightly impede the routes of migratory stream biota (e.g., shrimp; Benstead et al. 1999). In addition, LHDs can affect localized water velocity patterns, sediment composition, and particulate organic matter budgets (Pohlen et al. 2007; Magilligan and Nislow 2001; Stanley et al. 2002; Wagner 2003). Pohlon et al. (2007) observed that LHD reservoirs can be beneficial to riverine structure and function. In a channelized river system, LHDs can provide areas of retention which are typically absent or ineffective by creating habitats which promote the deposition of material (e.g., organic matter). However, in urban or agricultural rivers that receive large inputs of nutrients and other materials, will LHD reservoirs be efficient sinks of materials? To assess this question, it is vital to understand both depositional and microbial processes which may act in combination to remove materials from riverine waters.

The objective of my research was to assess how the presence of low-head dam reservoirs could alter ecosystem structure (e.g., habitat features, material allocation in water and sediment) and function (e.g., microbial activity) in urban, channelized rivers. Specifically, I was interested in examining immediate differences in hydrological and sediment properties and in the allocation/transformation of nitrogen (ammonium, nitrate) and carbon (organic matter, methane) between LHD reservoirs and reference reaches lacking LHD reservoirs. I had three main hypotheses:

(H1) Low-head dams reservoirs will alter habitat features of urban rivers by reducing water velocities, increasing water depths, and retaining more fine-sized sediments and moisture within the channel bed.

(H2) The allocation of nitrogen and other materials (e.g. carbon and oxygen) in water and sediments will differ between reservoirs and reference reaches. Specifically, reservoirs will act as sinks for nitrogen and carbon and have higher concentrations of these materials in sediments and lower concentrations in benthic waters (i.e., waters within 5 cm of the water column-sediment interface). Also, dissolved oxygen will be more depleted in benthic waters of reservoirs and thus reservoirs should possess higher concentrations of methane in sediments.

(H3) Microbial processes which require low oxygen conditions (e.g. denitrification) will be enhanced within reservoirs. Processes which require oxygen (e.g. nitrification, methane oxidation) will be reduced within reservoirs.

In addition to the above hypotheses, I expected for each variable to be linked in a conceptual model (Figure 1.1). To evaluate these hypotheses, I conducted a comparative study on two urban river watersheds in Franklin County, Ohio. This area is unique in that approximately twenty low-head dams impound sections of the four major river watersheds in this region.

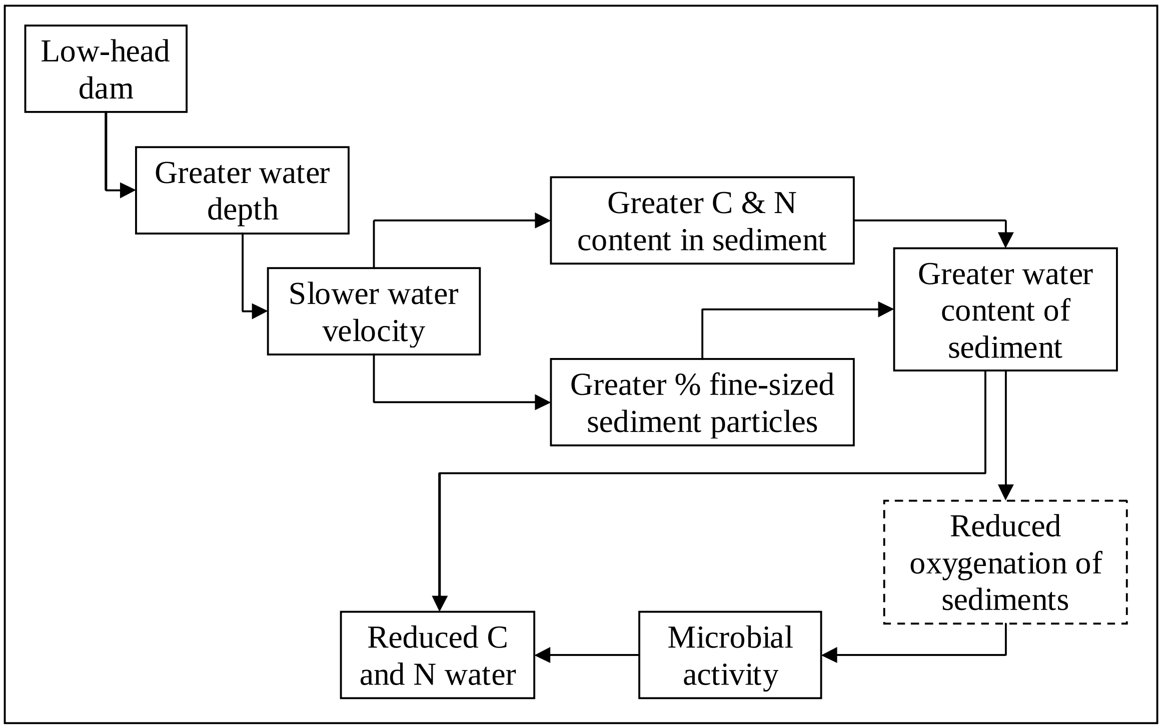

Figure 1.1. Conceptual model of linkages amongst hypothesized results. “Reduced oxygenation of sediments” was not measured in this study.

CHAPTER 2

METHODS

2.1. Study Sites

This study was conducted in two urbanized watersheds dominated by residential and commercial land use in central Ohio, United States (Figure 2.1) – the Olentangy River (watershed area of 1370 km^2) and Alum Creek (watershed area of 490 km^2). Both rivers are controlled by large, high-head dams (Delaware Dam and Alum Creek Dam) that lead to a controlled hydrologic flow regime. Delaware and Alum Creek Dams partition both rivers into two sections, thus the rivers have been given “upper” and “lower” river designations (e.g., upper Alum Creek watershed is above the Alum Creek Dam). All study sites were located in the lower Olentangy River and Alum Creek watersheds and were 6 – 30 km downstream of the high-head dams. Local stormwater runoff was suspected to be a larger contributor to the water flow regime of these lower river networks than the high-head dams. The lower Olentangy River watershed has a drainage area of 445 km^2; Alum Creek watershed has a drainage area of 230 km^2. Urban/impervious surfaces account for 26 % of land area in the Olentangy River watershed and 44 % of land area in the Alum Creek watershed (FLOW 2003, FACT

2002). Low-head dams used in this study were built for a variety of reasons including water supply and possibly upstream (reservoir) use for recreation (Table 2.1).

2.2. Experimental Design

A block design with five replicate "LHD reservoir-reference riverine" blocks (i.e., ten reaches) was used to assess treatment effects (presence of LHD reservoir on urban river vs. its absence) and seasonal effects (spring, summer, autumn) (Figure 2.1). The heights of all low-head dams varied between 2 – 4 m and lengths between 29 – 130 m (Table 2.1). A 30 m reach was delineated at both reservoir and reference reaches, with reservoir boundaries beginning ~15 m upstream of low-head dams. Reference reaches were selected based upon upstream distance from reservoirs (0.5 – 3.5 km), channel width (narrower than paired reservoir), and mean water depth (shallower than paired reservoir). All reference reaches contained adjacent urban land use, but did not contain low-head dams and were not directly impacted by them.

2.3. Field Methods and Sample Handling

Water and sediment samples were collected from three locations within each reach during spring (late May to early June), summer (mid to late August), and autumn (mid Oct to early Nov) of 2007. Sediments were collected with an Ekman dredge (Rickly Hydrological, Columbus, OH). Samples were placed in plastic zip lock bags stored at 4°C until laboratory analysis. Water column physico-chemical measurements were also completed, dependent on water depth of each substation (i.e., a depth > 1 m: surface, mid and benthic depth measurements were made; depth of 0.5 to 1 m: surface and benthic depth measurements were made; depth < 0.5 m: mid depth measurement made). Subsurface (i.e., mid and benthos) water samples were collected with a van Dorn alpha water grab sampler (Wildco, Buffalo, NY). Surface water samples were collected in Nalgene bottles.

2.4. Habitat Characterization

In the field, water depth was recorded with a calibrated pole, and surface water velocity was measured with a mechanical Geopacks water flow meter (London, England). Sediment water content (gravimetric) was determined by drying sediments at 120°C for 24 hrs. Sediment particle size distribution was determined by wet sieving after dispersion of samples in 3% metaphosphate solution and concentrated hydrogen perioxide solution. Three sediment size classes were measured: small gravel (2 – 4 mm), sand (0.067 – 2 mm), and fines (< 0.067 mm). Sediment particles > 4 mm were not included in this analysis. Dissolved oxygen (DO; mg L^{-1}), oxygen (O_2) saturation (%), conductance (mS cm^{-1}), and pH of water were measured with a YSI 600 sonde (YSI, Yellow Springs, OH).

2.5. Allocation of Material Resources

Water samples were analyzed for total and organic carbon (TC, TOC) with a Rosemount-Dohrmann High Temperature TOC Analyzer, Model DC-190 (Diversified Equipment, Lorton, VA). For nitrogen analysis, water samples were filtered with 0.45 μm filters (Millipore, Billerica, MA), acidified with concentrated sulfuric acid, and frozen until analysis. Nitrate (NO_3) concentrations were determined using cadmium reduction, and ammonium (NH_4) concentrations were determined using phenolate methods. Both N species were measured on a QuikChem 8500 (Lachat Instruments, Loveland, CO).

Sediments were analyzed for organic matter by loss on ignition. Sediments used for total carbon (%) and total nitrogen (%) analysis (C and N content) were dried at 50°C for 48 hrs, ground with a pestle and mortar, and sieved through 1.0 mm-sized mesh sieve. C and N content of sediments were measured through dry combustion on a Thermo Quest, NC 2100 (CE Instruments, Lakewood, NJ). Concentrations of NO_3 and NH_4 in sediment were determined using 2 M KCl extractions (Keeney and Nelson 1987). Approximately 4 g of moist sediment was added to a 50 mL Erlenmeyer flask filled with 20 mL of 2 M KCl solution. Samples were shaken on a box shaker at moderate speed for 1 hr, filtered through Whatman size 42 filter paper and frozen until analysis. Concentrations of NO_3 were determined using cadmium reduction, and NH_4 concentrations were determined using salicylate methods on a QuikChem 8500 (Lachat Instruments, Loveland, CO). Concentrations of methane (CH_4) in sediment were determined using methods of Roy and Knowles (1994). Five mL of moist sediment and 5 mL of deionized water were added to 30 mL glass vials capped with septa. Samples were shaken at moderate speed on a box shaker for 2 hrs. Gas sample storage and analysis is described in *Section 2.6.3*.

2.6. Microbial Activity

2.6.1. Denitrification. Denitrification enzyme activity (DEA) was assessed using the acetylene block inhibition method (Tiedje 1994). Approximately 25 g of moist sediment was added to 150 mL glass bottles containing substrates (1.0 mM KNO_3, 1.0 mM dextrose) and 0.0121g of chloramphenicol. Chloramphenicol was added to depress *de novo* enzyme synthesis and produce measurements that estimate rates of denitrification without additional bacterial growth (Murray and Knowles 1999). In addition, chloramphenicol allows for the measurement of linear DEA rates during short-term assays (Tiedje 1994). Although chloramphenicol can have inhibitory effects on rates of potential denitrification (Pell et al. 1996), studies have shown that impacts are minimal when it is used at concentrations of ~ 0.15 g/L (Murray and Knowles 1999; Dendooven et al. 1994). For DEA assays, 0.16 g/L (or 0.5 mM) of chloramphenicol was added to each sample bottle. Samples were filled to 75 mL with deionized water and capped with open-screw-cap fitted with butyl septum. Samples were flushed with pure helium at a rate of 20 mL/min to produce low oxygen conditions and filled with 10% acetylene. Acetylene blocks the full conversion of NO_3 to nitrogen gas (N_2) in denitrification and allows for the measurement of nitrous oxide (N_2O), therefore allowing for the detection of DEA rates. Samples were incubated at 25ºC in the dark in an Echo Therm chilling incubator (Cole Palmer, Vernon Hills, IL) for ~ 2 hrs. Gas samples were taken at 40, 60, 80, and 100 minutes, inserted into 3 mL evacuated vacutainers (Kendall Monoject, Mansfield, MA), and stored at 4ºC until analyses. Within ten days, gas samples were analyzed for N_2O using a Shimadzu Gas Chromatograph 14A (Shimadzu Scientific Instruments, Columbia, MD) equipped with an ECD detector. Carrier gas (95% argon, 5% methane) flow was set at ~ 25 mL/min. Rates of DEA were determined through linear regression of N_2O over time ($r^2 > 0.8$) and were converted to a per dry mass of sediment unit using sediment water content values from **Section 2.4**.

2.6.2. Nitrification. Potential rates of gross nitrification (NIT; measures change in NH_4 over time) were determined using a short-term shaken slurry procedure (Hart et al. 1994). Approximately 15 g of moist sediment and 100 mL solution of 0.15 mM NH_4 and 0.1 mM PO_4 were added to 250 mL Erlenmeyer flasks capped with foam stoppers. Flasks were shaken at room temperature at moderate speed on a box shaker for 24 hrs. Fifteen mL aliquots were taken at 2, 4, 22, and 24 hrs, filtered with a 0.45 μm filter (Millipore, Billerica, MA), acidified with concentrated sulfuric acid, and frozen until analyses. NO_3 concentrations were determined using cadmium reduction and NH_4 concentrations were determined using phenolate methods on a QuikChem 8500 (Lachat Instruments, Loveland, CO). Gross rates of potential nitrification were determined by linear regression of NH_4 values. Rates were corrected using equation (1).

$$NIT = R * (0.15/D) + W \quad (1)$$

NIT corrected rate of gross nitrification (mg N gDM^{-1} h^{-1})

R measured rate of nitrification, determined by linear regression of NH_4 over time (mgN h^{-1})

W water mass of sediment, determined by multiplying sediment water content (%) by mass of field-moist sediment (g)

D mass of solids in sediment (g)

Note – W and D are determined using water content values from **Section 2.4**

(Hart et al. 1994)

2.6.3. Methane Oxidation. Rates of potential methane oxidation (CH_4ox) were assessed using methods of Bodelier et al. (2000). Approximately 25 g of moist sediment was added to 150 mL glass bottles. Bottles were filled to 75 mL with deionized water and capped with open-screw-cap fitted with butyl septum. One percent methane (CH_4) was added to each bottle. Samples were shaken at room temperature at a moderate speed for ~ 24 hrs. Four gas samples were taken between 12 – 24 hrs, inserted into 3 mL evacuated vacutainers (Kendall Monoject, Mansfield, MA), and stored at 4ºC until analyses. Within 30 days, samples were analyzed for CH_4 using a Shimadzu Gas Chromatograph 14A (Shimadzu Scientific Instruments, Columbia, MD) equipped with FID detector, using helium as a carrier gas. Rates of CH_4ox were determined through linear regression of CH_4 over time ($r^2 > 0.8$) and transformed to a rate per dry mass of sediment using water content values from **Section 2.4**.

2.7. Statistical Analyses

A randomized block two-factor ANOVA was used to assess differences amongst parameters. Blocks (n = 5) were random variables. Each block consisted of a low-head dam reservoir and reference reach. Fixed factors were the treatments within each block (reservoir vs. reference) and season (spring, summer, and autumn). Tukey post hoc analyses were used to assess differences between seasons and to assess interactions between treatment x season present within the ANOVAs. Pearson pairwise correlation analyses were used to assess relationships amongst parameters in the conceptual model

(Figure 1.1) according to all seasons (to improve the power of the analyses) and on a seasonal basis (to address strong seasonal effects). P-value of significance was $p < 0.05$.

All statistical analyses were completed in MINITAB statistical software (State College, PA). Response variables were either natural log or arcsine transformed to satisfy conditions of normality and homoscadacity. All sub-replicates (3 per reach) were averaged so that the level of replication was each block (reservoir-reference pair). Of water column values (replicates and depths), sub-replicates of benthic parameters were averaged to give one value per reach.

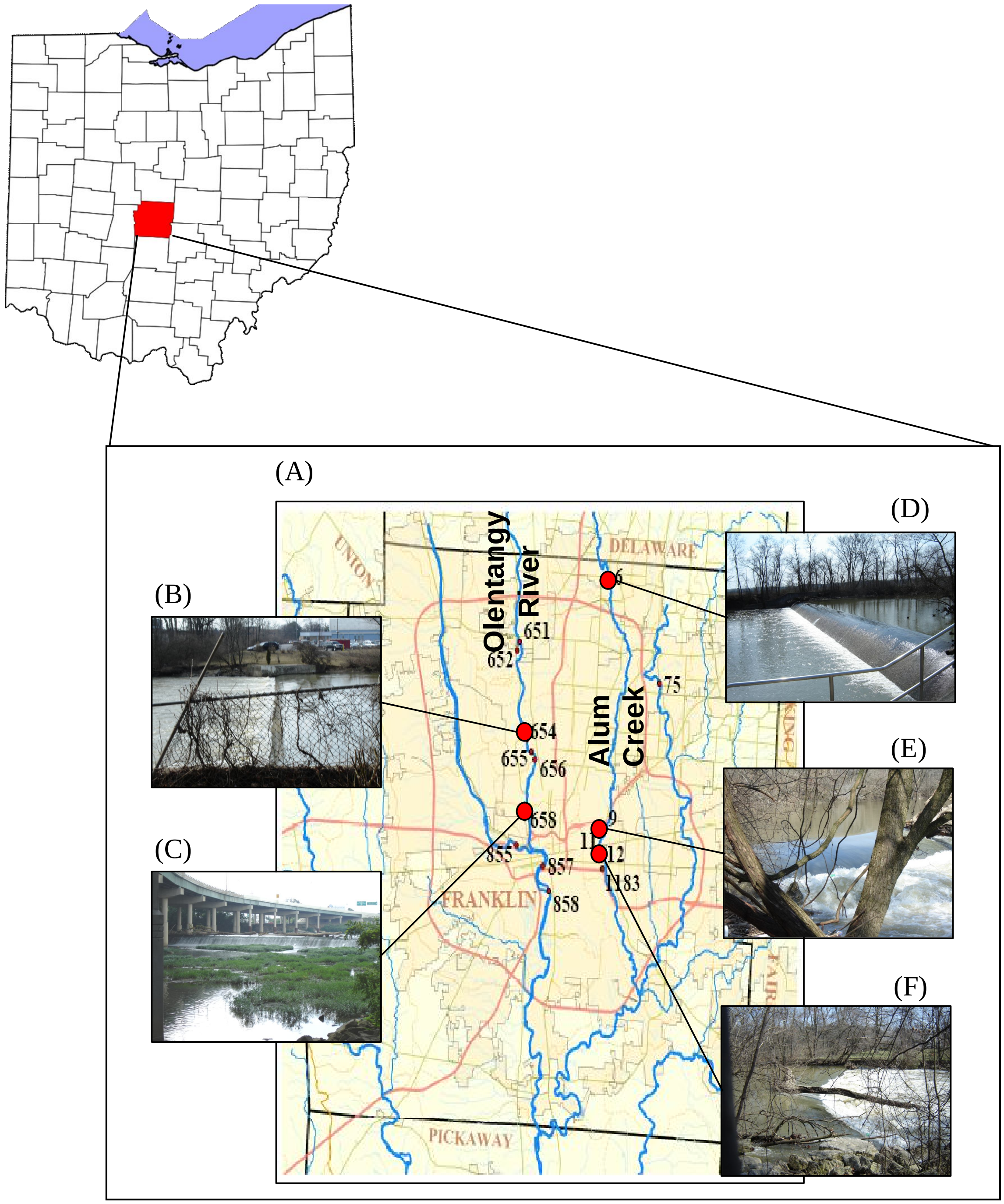

Figure 2.1. Map of study sites. Five paired reservoir-reference blocks (i.e., 10 reaches) were selected in Franklin County, OH. (A) Inventory map of all dams in Franklin County; blocks indicated by large circles. (B) *Dam 654*: Clinton-Como Park (ID: CC). (C) *Dam 658*: 5th Ave. (ID: 5^{th}). (D) *Dam 6*: Cherrington Park (ID: West). (E) *Dam 9*: Nelson Park (ID: Nels). (F) *Dam 12*: Wolfe Park (ID: Wolf).

	5th	CC	Nels	West	Wolf
Latitude	39.99	40.03	39.98	40.12	39.96
Longitude	-83.02	-83.02	-82.95	-82.94	-82.95
Year built	1935	unknown	1940	1935	1920s - 1930s
Purpose	cooling water source for power plant operation	unknown - currently provides sanitary sewer crossings	unknown - possibly for recreation	water supply for city of Westerville	unknown - possibly for recreation
Dam height (m)	2.5	1.9	1.2	> 2	1.2
Dam width (m)	130	55	29	59	32
Dam slated for removal?	YES	NO	YES	NO	YES

Table 2.1. Description of selected low-head dams. IDs correspond with those indicated in **Figure 2.1**

CHAPTER 3

RESULTS

3.1. Habitat Characterization

Reference reaches had faster water velocities and shallower water depths than low-head dam reservoirs (Table 3.1). On average, water velocities were approximately 7-fold faster in references, and reservoirs were almost twice as deep. There were no differences amongst seasons for these two variables (Table 3.2). Physical and chemical conditions of benthic waters (i.e., waters within 5 cm of the water column-sediment interface) were similar between stations for O_2 saturation, dissolved oxygen (mg L^{-1}), pH, and conductivity (Table 3.1). There were seasonal differences with O_2 saturation, dissolved oxygen (mg L^{-1}), and conductivity (Table 3.2). For the two measures of oxygen, concentrations were greatest in spring as compared with summer and autumn. The following seasonal trend occurred with conductivity: spring > autumn > summer.

Sediments from both reach types were composed of mostly sand. Only fine-sized sediments differed between reach types and were greater in reservoirs than in reference reaches (Figure 3.1; Table 3.1). Sediments in reservoirs held more water than reference reaches (Tables 3.1). Lower water content values were present during the spring as compared to the summer and fall seasons (Table 3.2).

3.2. Materials in water and sediments

In benthic waters (i.e., waters within 5 cm of the water column-sediment interface), concentrations of carbon and nitrogen were similar in the two types of reaches (e.g., reservoir and reference). With seasons, total carbon in benthic waters was twice as great during the spring than summer and autumn seasons (Table 3.2). Total organic carbon in benthic waters was greatest during the summer and lowest during the autumn. Dissolved NO_3 was similar amongst seasons. With dissolved NH_4, a singe factor effects ANOVA could not be used due to an interaction between reach type and seasonal effects. Through conducting a Tukey post hoc analysis on the interactions, there were no significant differences between reservoir and reference reaches with respect to seasons ($p < 0.05$).

Total carbon, percentage organic matter, and concentrations of NH_4 were higher in sediments of reservoirs than in reference reaches (Table 3.1). In particular, average concentrations of NH_4 in sediments were approximately 3-times greater in reservoirs. Concentrations of CH_4 and NO_3 in sediments were similar in reservoirs and references reaches. Seasons strongly influenced concentrations of C and N in sediments. Values were lowest in spring for total carbon, ammonium, and nitrate in sediments and greatest during the spring. With percentage organic matter, values were greatest during the spring (Table 3.2). CH_4 concentrations in sediments were lowest during the summer.

3.3. Microbial Activity

Potential rates of nitrification (NIT) and methane oxidation (CH_4ox) were similar in reservoir and reference reaches (Table 3.1). Rates of nitrification during the spring were greater than the autumn season, while rates of methane oxidation were similar amongst seasons (Table 3.2). Denitrification (DEA) was the only measure of microbial activity which differed between the two kinds of reaches. Potential rates of denitrification were approximately 3-times greater in reservoirs (Table 3.1). DEA was lowest during the spring and greatest during the autumn (Table 3.2).

3.4. Linkages amongst parameters in the hypothesized conceptual model

Water velocity was not correlated with indicators of C and N in sediment and the percentage of fine-sized sediment (Table 3.3). Instead, a stronger relationship between water depth, C and N in sediment, and percentage of fine-sized sediment existed (Table 3.4). Water content in sediments was also positively correlated with C and N in sediment and the percentage of fine-sized sediment (Table 3.3). During the autumn, denitrification was positively correlated water content. Although relationships between the other microbial processes and water content in sediment did exist, correlations will not be discussed since rates of nitrification and methane oxidation were similar between reservoirs and reference reaches.

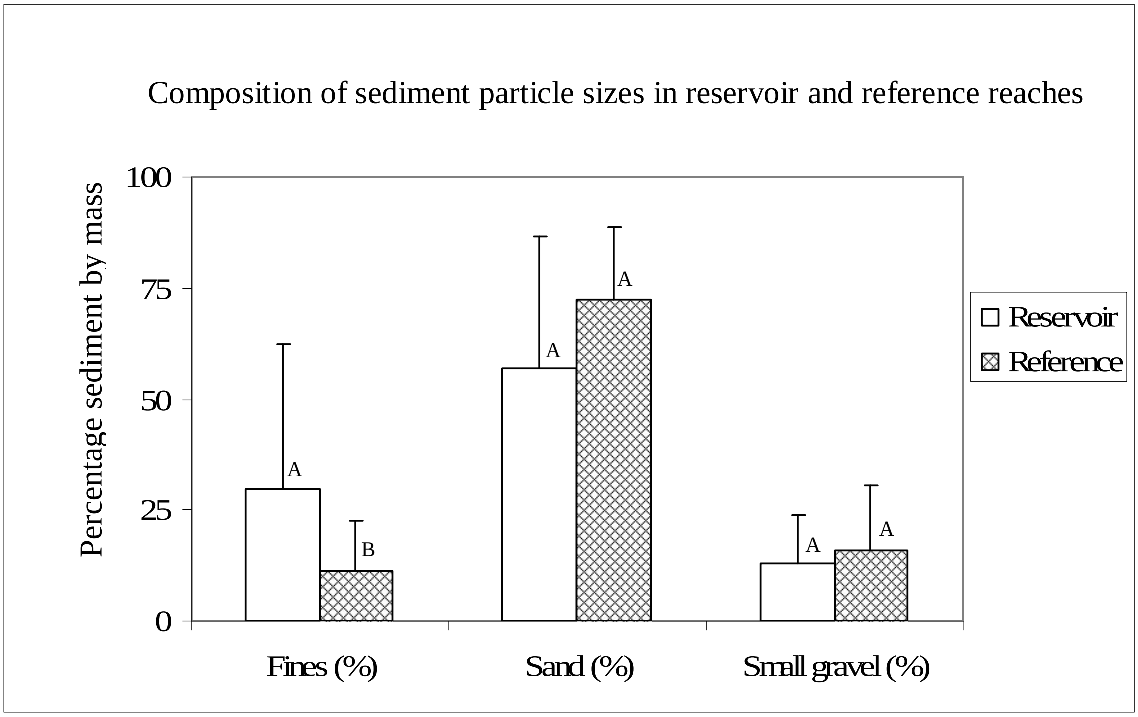

Figure 3.1. Composition of sediment particle sizes in reservoir and reference reaches. A Tukey post hoc test was used to determine differences in percentage fines, sand, and small gravel between reservoir and reference reaches. Differing letters over a category indicate a significant difference between reach types.

	Low-head dam reservoirs	Reference reaches	*p*
Habitat structure			
benthic water			
Water velocity (cm s^{-1})*	0.97 (1.32)	7.14 (10.56)	0.004
Water depth (cm)*	148.90 (40.50)	78.23 (33.40)	< 0.001
O_2 saturation (%)*	63.42 (21.06)	71.37 (28.15)	0.265
Dis. oxygen (mg L^{-1})*	5.66 (1.73)	6.40 (2.29)	0.219
pH*	7.76 (0.11)	7.67 (0.20)	0.323
Conductivity (mS cm^{-1})*	0.605 (0.137)	0.594 (0.120)	0.821
sediment			
Fines (%)**	29.80 (32.29)	11.36 (11.22)	0.012
Sand (%)**	57.00 (29.80)	72.57 (16.02)	0.083
Small gravel (%)*	13.06 (10.82)	16.08 (14.66)	0.998
Water content (%)*	27.38 (6.96)	19.10 (4.94)	< 0.001
Material resources			
benthic water			
Total C (mgC L^{-1})*	31.90 (8.52)	33.04 (8.32)	0.474
Total organic C (mgC L^{-1})**	6.86 (2.17)	7.53 (2.76)	0.41
Dissolved NH_4 (mgN L^{-1})*	0.10 (0.09)	0.18 (0.31)	
Dissolved NO_3 (mgN L^{-1})*	1.63 (1.60)	1.44 (1.41)	0.729
sediment			
Total C (%)**	3.272 (0.681)	2.717 (0.843)	0.005
Organic matter (%)**	6.25 (4.06)	4.685 (2.557)	0.011
CH_4 (μgCH_4 L^{-1})**	151.08 (11.99)	147.75 (11.68)	0.389
NH_4 (μgN g^{-1})*	12.17 (9.95)	4.435 (3.559)	0.001
NO_3 (μgN g^{-1})*	0.60 (1.17)	0.40 (0.60)	0.811

Continued

Table 3.1. Measures of habitat structure, materials in water and sediment, and microbial activity in low-head dam reservoirs and reference reaches. Values given in mean (standard deviation). N/A: interaction between treatment and season present, single factor ANOVA was not used. * values were natural log transformed. ** values were arcsine transformed.

Table 3.1. Continued.

	Low-head dam reservoirs	Reference reaches	*p*
Microbial activity			
DEA (μgN gDM^{-1} h^{-1})*	1.238 (0.896)	0.438 (0.504)	0.013
NIT (μgN gDM^{-1} h^{-1})**	0.110 (0.062)	0.181 (0.225)	0.283
CH_4ox (μgCH_4 gDM^{-1} h^{-1})**	1.276 (0.592)	0.936 (0.099)	0.634

	Spring	Summer	Autumn	p
Habitat structure				
benthic water				
Water velocity ($cm\ s^{-1}$)*	1.70 (1.12) a	4.97 (9.71) a	ND	0.563
Water depth (cm)*	112.2 (55.4) a	115.4 (44.1) a	113.1 (58.5) a	0.854
O_2 saturation (%)*	89.79 (27.12) a	61.59 (13.00) b	50.80 (13.06) b	0.001
Dis. Oxygen ($mg\ L^{-1}$)*	7.82 (2.19) a	5.07 (1.16) b	5.19 (1.34) b	0.003
Conductivity ($mS\ cm^{-1}$)*	0.681 (0.131) a	0.521 (0.087) b	0.581 (0.108) c	< 0.001
pH*	7.69 (0.13) a	7.73 (0.20) a	ND	0.512
sediment				
Water content (%)*	19.10 (6.42) a	25.40 (7.76) b	24.43 (7.06) b	0.014
Material resources				
benthic water				
Total C ($mgC\ L^{-1}$)*	40.78 (4.73) a	27.29 (4.90) b	29.34 (7.60) b	0.001
Total org. C ($mgC\ L^{-1}$)**	6.92 (0.91) a	9.49 (2.36) b	5.17 (1.63) c	< 0.001
Dis. NH_4 ($mgN\ L^{-1}$)*#	0.253 (0.309)	0.047 (0.040)	ND	
Dis. NO_3 ($mgN\ L^{-1}$)*	1.66 (0.83) a	1.22 (0.52) a	1.72 (2.36) a	0.626
sediment				
Total C (%)**	2.57 (0.49) a	3.04 (0.64) ab	3.37 (1.04) b	0.037
Org. matter (%)**	8.81 (4.09) a	3.96 (1.37) b	3.64 (0.74) b	< 0.001
CH_4 ($\mu gCH_4\ L^{-1}$)**	ND	142.34 (10.42) a	156.49 (8.17) b	0.002
NH_4 ($\mu gN\ g^{-1}$)*	3.66 (2.88) a	12.14 (10.01) b	8.18 (8.09) ab	0.026
NO_3 ($\mu gN\ g^{-1}$)*	1.48 (1.26) a	0.06 (0.09) b	0.15 (0.29) b	< 0.001

Continued

Table 3.2. Seasonal measures of habitat structure, material in water and sediment, and microbial activity. Values given in mean (standard deviation). ND: no data. ^{1} $\mu gN\ gDM^{-1}\ h^{-1}$. ^{2}: $\mu gCH_4\ gDM^{-1}\ h^{-1}$. * values were natural log transformed. ** values were arcsine transformed. #: interaction between treatment and season present, single factor ANOVA could not be used.

Table 3.2. Continued.

	Spring	**Summer**	**Autumn**	*p*
Microbial activity				
DEA[1]*	0.327 (0.267) a	1.005 (0.995) ab	1.818 (0.814) b	0.024
NIT[1]**	0.168 (0.056) a	0.198 (0.261) ab	0.075 (0.051) b	0.009
CH_4ox[2]**	0.992 (0.367) a	1.113 (0.360) a	1.215 (0.759) a	0.634

Pairs	Spring	Summer	Autumn	All Seasons
water depth - water velocity	-	-	ND	-
water velocity - % fine sediment	-	-	ND	-
water velocity - sed. TC	-	-	ND	$\underline{-0.622}$
water velocity - sed. OM	-	-	ND	-
water velocity - sed. CH_4	-	-	ND	-
water velocity - sed. NH_4	-	-	ND	-
water velocity - sed. NO_3	-	$\underline{0.879}$	ND	-
% fine sediment - sed. water content	0.615	$\underline{0.715}$	$\underline{0.641}$	$\underline{0.607}$
sed. TC - sed. water content	$\underline{0.722}$	-	-	$\underline{0.470}$
sed. OM - sed. water content	-	$\underline{0.737}$	0.627	-
sed. CH_4 - sed. water content	-	-	0.573	-
sed. NH_4 - sed. water content	-	$\underline{0.790}$	$\underline{0.740}$	$\underline{0.698}$
sed. NO_3 - sed. water content	-	$\underline{-0.684}$	-	$\underline{-0.489}$
sed. water content - ben. water TC	-	-	-	-
sed. water content - ben. water TOC	-	-	-	-
sed. water content - ben. water NH_4	$\underline{-0.838}$	-	-	-
sed. water content - ben. water NO_3	-	-	-	-
sed. water content - NIT	$\underline{-0.810}$	-	-	$\underline{-0.512}$
sed. water content - DEA	-	-	$\underline{0.669}$	0.355
sed. water content - CH_4ox	$\underline{0.652}$	-	-	$\underline{0.402}$
NIT - ben. water TC	-	-	-	$\underline{0.375}$
NIT - ben. water TOC	-	$\underline{0.643}$	-	$\underline{0.474}$
NIT - ben. water NH_4	-	-	-	-
NIT - ben. water NO_3	-	-	-	-
DEA - ben. water TC	-	-0.565	-	$\underline{-0.381}$
DEA - ben. water TOC	-	$\underline{-0.680}$	-	-
DEA - ben. water NH_4	-	-	-	-
DEA - ben. water NO_3	-	-	-	-
CH4ox - ben. water TC	-	-	-	-
CH4ox - ben. water TOC	-	-	-	-
CH4ox - ben. water NH_4	-	-	-	-
CH4ox - ben. water NO_3	-	-	-	-

Table 3.3. Pearson pairwise correlation coefficients of conceptual model variables. Underlined values indicate a statistically significant correlation ($p < 0.05$). All correlations were completed in transformed units: * ln transformation, ** arcsine transformation. "-" and values that are not underlined indicate a non-significant correlation. ND: data unavailable.

Pairs	Spring	Summer	Autumn	All Seasons
water depth - % fine sediment	0.796	0.750	0.782	0.767
water depth - sed. TC	-	-	-	-
water depth - sed. OM	-	0.625	0.622	0.334
water depth - sed. CH4	ND	-	0.568	-
water depth - sed. NH4	-	0.827	0.845	0.697
water depth - sed. NO3	-	-	-	-
water depth - sed. water content	0.764	0.804	0.886	0.742
water depth - NIT	-0.919	-	-0.635	-0.365
water depth - DEA	-	-	0.671	-
water depth - CH4ox	-	-	-0.625	0.448
% fine sediment - sed. TC	-	-	-	-
% fine sediment - sed. OM	-	-	-	-
% fine sediment - sed. CH4	ND	-0.723	-	-
% fine sediment - sed. NH4	-	0.778	0.571	0.611
% fine sediment - sed. NO3	-	-	-	-0.463
NIT - CH4ox	-0.629	-	-0.766	-0.348
NIT - sed. TC	-	-0.578	0.606	-
NIT - sed. OM	-	-	-	-
NIT - sed. NH4	-	-	-0.646	-0.328
NIT - sed. NO3	-	0.700	-	0.370
CH4ox - sed. TC	0.620	-	-	-
CH4ox - sed. OM	-	-	-	-
CH4ox - sed. NH4	-	-	0.552	0.321
CH4ox - sed. NO3	-	-	-	-
sed. TC - ben. water TC	-	-	-	-
sed. OM - ben. water TOC	0.607	-	-	-
sed. NH4 - ben. water NH4	-	-	ND	-0.410
sed. NO3 - ben. water NO3	-	-	0.701	0.387

Table 3.4. Pearson pairwise correlation coefficients of interactions not originally considered in conceptual model. Underlined values indicate a statistically significant correlation ($p < 0.05$). All correlations were completed in transformed units: * ln transformation, ** arcsine transformation. "-" and values that are not underlined indicate a non-significant correlation. ND: data unavailable.

CHAPTER 4

DISCUSSION AND CONCLUSIONS

Results of this study confirmed hypothesis H1. Low-head dam reservoirs altered riverine habitat by reducing water velocities, increasing water depth, and retaining fine-sized sediments and moisture within the channel bed. Reservoirs were also retentive of carbon and nitrogen in sediment (e.g., total carbon, organic matter, and ammonium; H2), but were not sinks for carbon and nitrogen in benthic waters. Although reservoirs did not remove significant amounts of carbon and nitrogen from benthic waters, reservoirs were "hot spots" (McClain et al. 2003) for denitrification (H3). This trend did not exist with nitrification and methane oxidation. Reservoir sediments did not contain more methane than reference reaches (H2). These systems also had concentrations of oxygen (O_2 saturation and dissolved oxygen, mg L^{-1}) in benthic waters that were similar to levels seen in reference reaches (Table 4.1).

Although denitrification and storage of carbon and nitrogen in sediments were greater in studied reservoirs than reference reaches, reservoirs did not efficiently remove materials from benthic waters as hypothesized. This was indicated by similar concentrations of NO_3 and other N and C species in benthic waters of reservoirs and reference reaches. Also, a portion of hypothesis H3 (greater rates of nitrification and methane oxidation in reference reaches as compared with reservoirs) was not confirmed.

Reasons for these similarities (i.e., N and C in benthic waters, nitrification, and methane oxidation) between reach types are unknown, but overarching land use impacts and possible interactions between nitrogen and methane cycling processes could shed light upon some of these uncertainties.

4.1. Conceptual model

I originally hypothesized that reduced water velocity would be the primary factor leading to the accumulation of fine-sized sediments and carbon and nitrogen in sediment (Figure 1.1). Results did not confirm this hypothesis (Figure 4.1). Instead, increased water depth appeared to influence these variables. Although water velocity was faster in low-head head dam reservoirs, I propose that the difference was not significant enough to contribute to the retention of materials (e.g., fine-sized sediment, carbon, and nitrogen) in reservoirs. I suspect that the urban stream syndrome, in particular, a flashy hydrograph could explain this lack of a trend. In addition, flashiness in water flows could also explain why reservoirs were not sinks for carbon and nitrogen from benthic waters.

4.2. Impacts low-head dam reservoirs to ecosystem structure and function in urban rivers

Although reservoirs appeared to show signs of the urban stream syndrome in regard to hydrology, they did serve as sinks for fine-sized sediments. Specifically, sediments in low-head dam reservoirs contained 30 % fine-sized sediment, with the average being 20 % greater than reference reaches. The 30 % value falls within the range

20 – 35 % fines that have been found in watersheds dominated by agricultural land use (Ashley et al. 2006; Tiemann et al. 2004); but, this value is much lower than the average of 60 % found in moderate to slightly developed forested and agricultural watersheds (Orr et al. 2006; Ahearn and Dahlgren 2005). The reduced retention of fine-sized sediments could be related to relatively short water residence times in reservoir reaches. Faster water flows reduce deposition rates of fine-sized suspended sediments and are capable of moving larger-sized particles from river banks into the channel itself (Finkenbine et al. 2000). This later reason seems more applicable to these sites as the percentage of small gravel particles between reservoir and reference reaches were similar.

Increased values of water content in sediments were a characteristic of the reservoir reaches. This trait also showed a positive correlation with water depth and percentage fine-sized sediments. In a low-head dam reservoir in Murphy Creek, California, Ahearn and Dahlgren (2005) found a strong relationship between NH_4 values in sediments and the sediment water content values. Fine-sized sediments behind the reservoir in Murphy Creek were highly saturated and showed a positive relationship between NH_4 storage in sediments and allowed for minimal release of NO_3 into overlying waters (Ahearn and Dahlgren 2005). In this study, a similar trend between NH_4 in sediments and sediment water content values was present, particularly during the summer and autumn sampling periods (as indicated by a positive correlation between NH_4 in sediments and sediment water content values). During the spring, water content of sediments had a negative correlation with dissolved NH_4 in benthic waters, which suggests that sediments with greater water content values could serve as sinks for dissolved NH_4 in overlying waters during this time period. NO_3 trends similar to Ahearn and Dahlgren (2005) were not present in this study. No significant correlations between sediment saturation and NO_3 in sediments or benthic waters were present during the spring and autumn sampling periods. In the summer, there was a strong, negative correlation between sediment water content and NO_3 in sediments. This suggests that sediments with greater water content values could be sources of NO_3 during this time period or that they could be experiencing high rates of processes that promote the removal of NO_3 from sediments (e.g. DEA, dissimilatory NO_3 reduction to NH_4).

While reservoirs tended to have concentrations of oxygen in benthic waters that were lower than reference reaches, differences were not biologically nor statistically significant. Thus, the studied low-head dam reservoirs did not provide conditions in benthic waters that were reduced enough to promote anoxic processes (e.g., methanogenesis). Since reservoirs of high-head dams make up ~ 18 % of anthropogenic CH_4 emissions to the atmosphere (St. Louis et al. 2000), this is a positive result. It suggests that low-head reservoirs are not as strong of a source of CH_4 (a potent greenhouse gas) to the atmosphere as high-head dam reservoirs. Although conditions in low-head dam reservoirs were not reduced enough to perform methanogenesis (as indicated by similar concentrations of CH_4 in sediments between reservoir and reference reaches), there must have been some difference (biologically speaking) in oxygen levels to explain why reservoirs had greater rates of denitrification than reference reaches. During autumn, when the greatest difference between rates in reservoir and reference reaches was present, denitrification was positively correlated with sediment moisture content (an indicator of reduced oxygen levels), water depth, and percentage organic matter in sediments – all of which had greater values in reservoirs. Another factor which may explain increased denitrification rates in reservoirs could be that deposited fine-sized sediments created microhabitats within reservoir sediments that were more depleted in oxygen levels.

4.3. Implications of results and conclusions

Since reservoir reaches retained carbon and nitrogen in sediments and had higher rates of denitrification, there is potential for low-head dam reservoirs in an urban setting to serve as sinks for materials in overlying, benthic waters. But the fact that the reservoirs may be overloaded with dissolved N and have relatively short water residence times in urban watersheds could be the primary factors contributing to the inefficiency of N removal from benthic waters (Saunders and Kalff 2001). Trends such as these are not exclusive to my study. Similar results have been exhibited in a low-head dam reservoir in southeastern Pennsylvania (Velinsky et al. 2006) and in a high-head dam reservoir in the southwestern United States (Kelly 2001). This inconsistent response occurring in both reservoirs created by low-head and high-head dams shows the need for more research to examine how hydrology and N loading impact N retention/removal efficiency in reservoirs.

These results have implications which reach beyond the presence of low-head dam reservoirs and can be applicable to instream channel alteration projects designed to restore function in urban rivers. If a flashy hydrograph, incised channel banks and elevated loads of N are characteristics of an urban riverine ecosystem, creating habitats which are presumed to promote N removal (e.g. large woody debris pools; Craig et al. in press) may not be effective in removing N inputs in water. Behavioral and structural changes within the landscape (i.e., reducing the usage of fertilizers, creating more areas for water and N retention within the landscape) may also have to occur for instream retention structures to become effective in removing N from overlying waters.

Lastly, low-head dam removals have become increasing popular over the past decade (Poff and Hart 2002) as a means of restoring ecosystem integrity in streams and rivers. In rivers of less developed watersheds, dam removal could have positive benefits, particularly in restoring riverine habitat to attract native biota (e.g., lotic fishes and other aquatic organisms). However, removing such structures in a highly urbanized area could do more harm than good. If the primary causes of river degradation in urban watersheds are from an altered or flashy hydrology and excessive pollutant inputs, then removing low-head dams will not improve the quality of such imperiled rivers. Since my results showed that low-head dam reservoirs can store materials in sediments (e.g., fine sized particles, carbon, nitrogen), removing these structures could actually cause these systems to become sources of materials.

Parameter	Reach Type		Hypothesis confirmed?
	Reservoir	Reference	
Habitat structure			
<u>*benthic water*</u>			
Water velocity (cm s^{-1}) *	<		YES
Water depth (cm) *	>		YES
O_2 saturation (%) *	=		NO
Dissolved oxygen (mg L^{-1}) *	=		NO
Conductivity (mS cm^{-1}) *	=		NO
pH *	=		NO
<u>*sediment*</u>			
Fines (%) **	>		YES
Sand (%) **	=		NO
Small gravel (%) *	=		NO
Water content (%) *	>		YES
	Reservoir	Reference	
Material resources			
<u>*benthic water*</u>			
Total C (mgC L^{-1}) *	=		NO
Total organic C (mgC L^{-1}) **	=		NO
Dissolved NH_4 (mgN L^{-1}) *	=		NO
Dissolved NO_3 (mgN L^{-1}) *	=		NO
<u>*sediment*</u>			
Total C (%) **	>		YES
Organic matter (%) **	>		YES
CH_4 (μgCH_4 L^{-1}) **	=		NO
NH_4 (μgN g^{-1}) *	>		YES
NO_3 (μgN g^{-1}) *	=		NO
	Reservoir	Reference	
Microbial activity			
DEA (μgN gDM^{-1} h^{-1}) *	>		YES
NIT (μgN gDM^{-1} h^{-1}) **	=		NO
CH_4ox (μgCH_4 gDM^{-1} h^{-1}) **	=		NO

Table 4.1. Summary of results. All analyses were completed in transformed units: * natural log transformation; ** arcsine transformation.

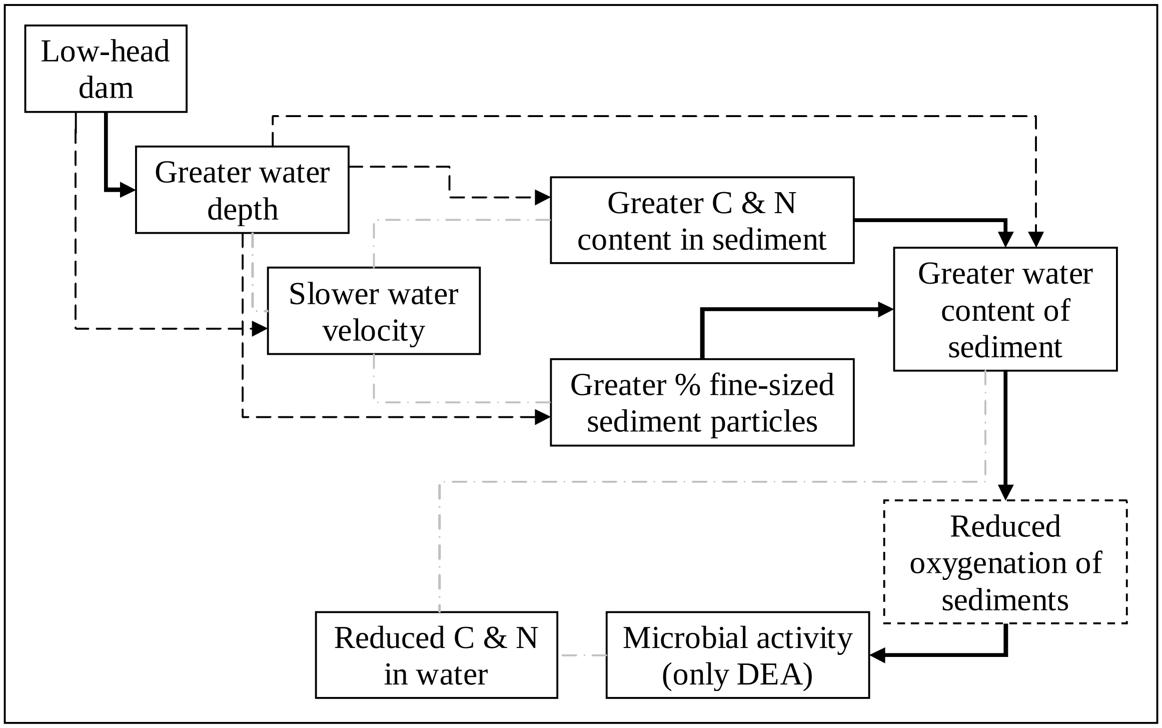

Figure 4.1. Conceptual model of linkages amongst actual results. Bolded solid lines with arrows represent hypothesized linkages that were confirmed; light-colored, dash-dot-dash lines represent hypotheses which were not confirmed. Dashed lines with arrows indicate correlations amongst variables that were not originally considered in the model. “Reduced oxygenation of sediments” was not measured in this study.

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APPENDIX A

HYDROLOGY OF OLENTANGY RIVER AND ALUM CREEK

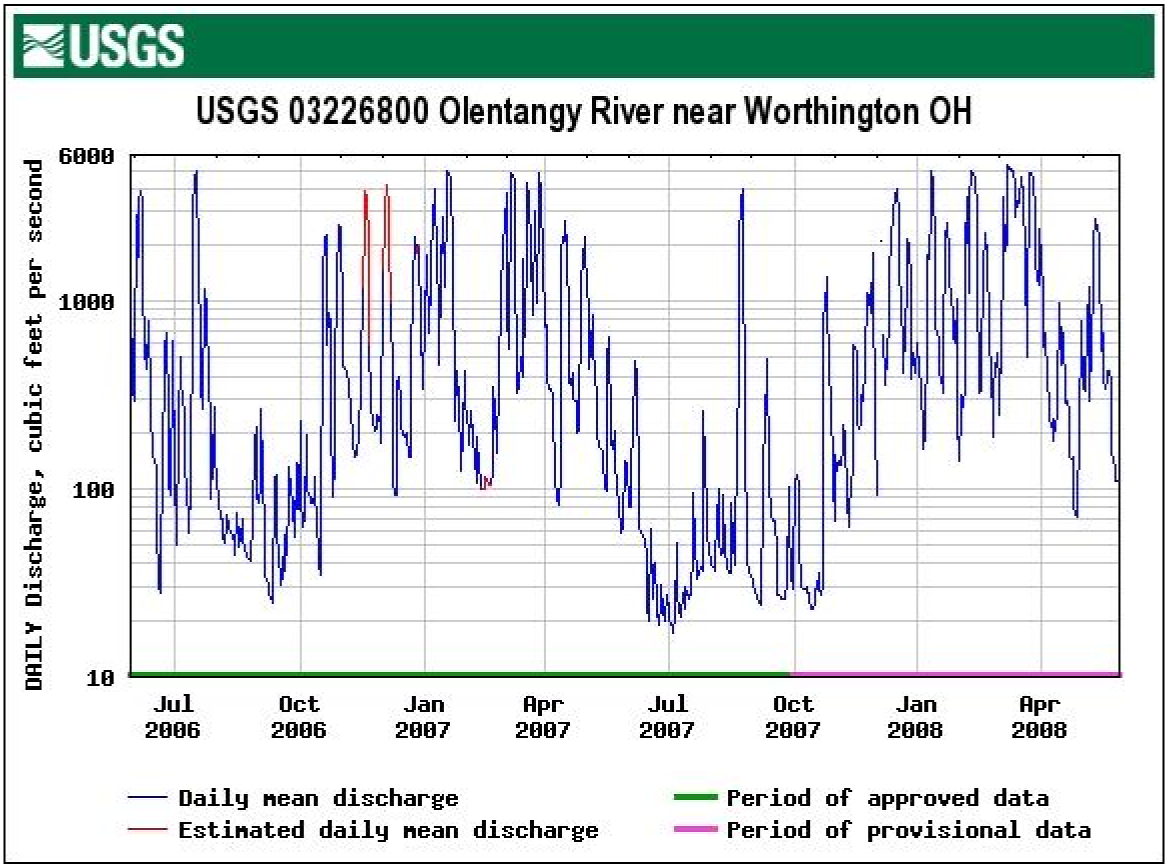

Figure A.1. Daily mean discharge of the Olentangy River from May 2006 – May 2008 (USGS 2008).

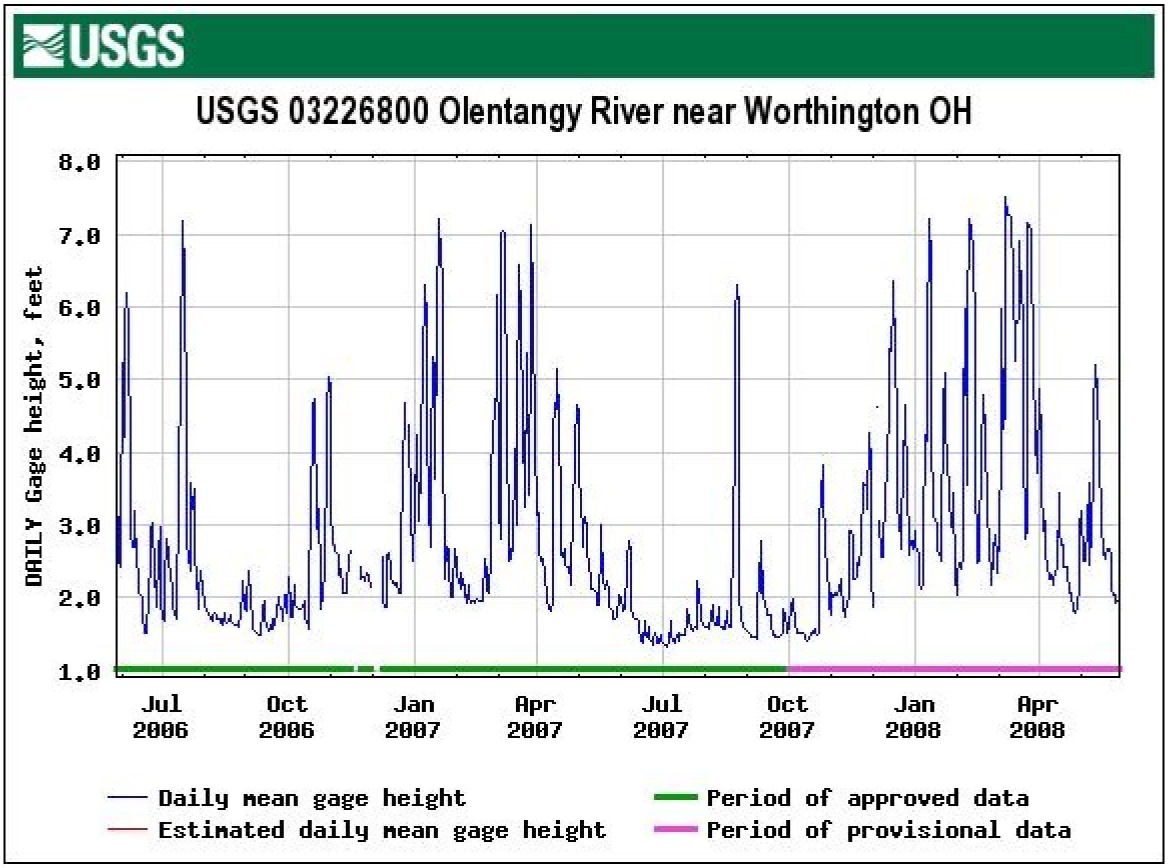

Figure A.2. Daily mean gage height (water depth) of the Olentangy River from May 2006 – May 2008 (USGS 2008).

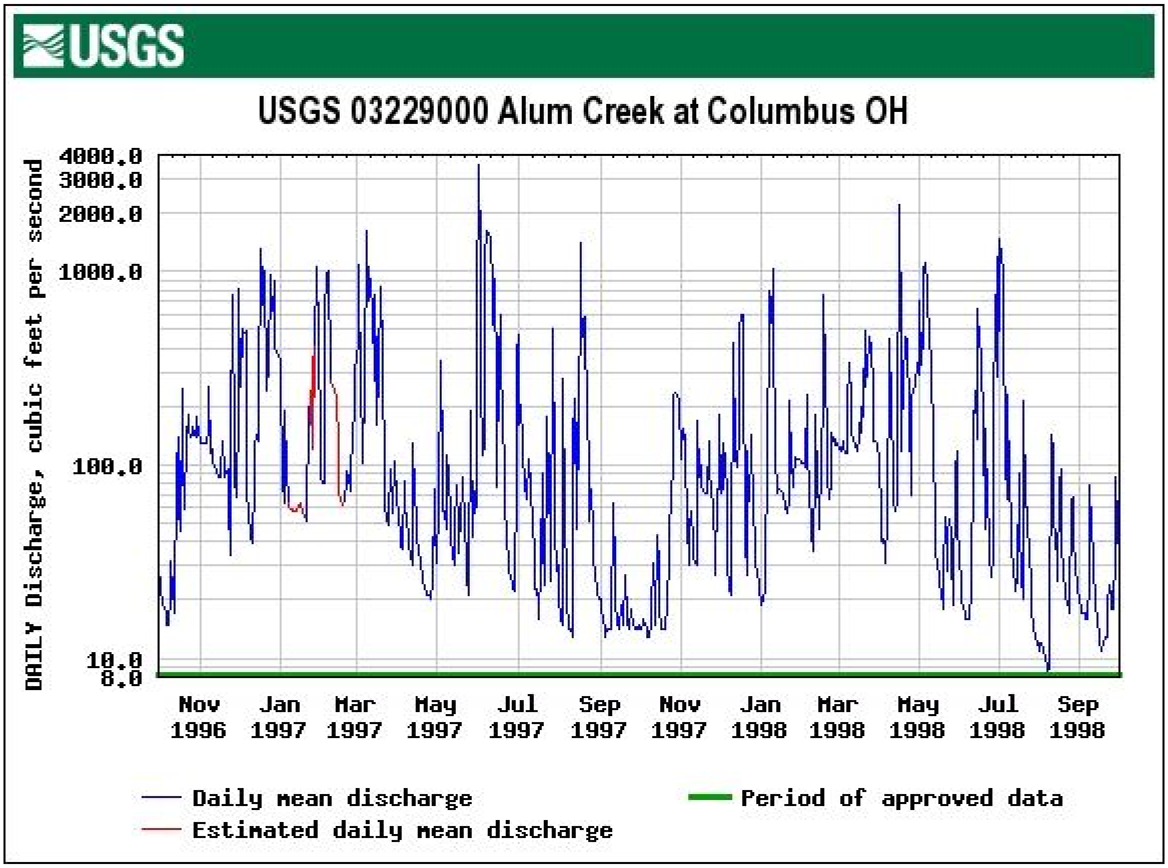

Figure A.3. Daily mean discharge of Alum Creek from Oct 1996 – Oct 1998 (USGS 2008). Hydrograph from sampling period (May 2007 – Nov 2007) was not available.

APPENDIX B

HABITAT DESCRIPTOR DATA

Block	Trmt	Season	Velocity (cm/s)	Water depth (cm)	% Water content (sediment)
West	dam	spring	1.44	229.0	31.15
West	ref	spring	3.04	88.3	14.99
CC	dam	spring	1.06	140.7	16.52
CC	ref	spring	2.05	49.3	17.69
Wolf	dam	spring	0.77	144.0	20.04
Wolf	ref	spring	3.23	51.3	14.12
Nels	dam	spring	0.34	100.7	20.08
Nels	ref	spring	ND	56.7	13.56
5th	dam	spring	ND	131.7	31.29
5th	ref	spring	ND	130.0	19.53
West	dam	summer	0.58	206.0	40.61
West	ref	summer	ND	91.3	19.81
Nels	dam	summer	0.11	115.3	29.91
Nels	ref	summer	5.10	72.2	14.15
Wolf	dam	summer	0.23	151.0	23.61
Wolf	ref	summer	ND	61.7	21.32
CC	dam	summer	4.30	126.7	22.2
CC	ref	summer	28.50	69.7	19.96
5th	dam	summer	0.06	125.0	29.56
5th	ref	summer	0.91	135.3	32.85
West	dam	autumn	BDL	232.0	38.02
West	ref	autumn	BDL	82.3	19.28
Nels	dam	autumn	BDL	117.7	20.34
Nels	ref	autumn	BDL	31.3	16.16
Wolf	dam	autumn	BDL	140.0	29.04
Wolf	ref	autumn	BDL	48.0	20.84
CC	dam	autumn	BDL	130.7	32.65
CC	ref	autumn	BDL	66.7	17.42
5th	dam	autumn	3.44	143.3	25.69
5th	ref	autumn	19.24	139.3	24.86

Table B.1. Seasonal habitat descriptors of hydrology and sediment water content. BDL: below detection limit. ND: no data

Block	Trmt	Water depth	Season	% DO	DO (mg/L)	cond	pH
West	dam	surface	spring	58.3	5.46	0.556	7.72
West	dam	mid	spring	56.3	5.27	0.552	7.77
West	dam	benthos	spring	46.0	4.91	0.534	7.79
West	ref	surface	spring	92.4	8.41	0.479	7.45
West	ref	mid	spring	ND	ND	ND	ND
West	ref	benthos	spring	94.4	8.58	0.479	7.45
CC	dam	surface	spring	108.0	9.45	0.620	7.84
CC	dam	mid	spring	106.3	9.26	0.619	7.76
CC	dam	benthos	spring	107.0	9.34	0.618	7.79
CC	ref	surface	spring	141.4	12.17	0.633	7.59
CC	ref	mid	spring	ND	ND	ND	ND
CC	ref	benthos	spring	ND	ND	ND	ND
Wolf	dam	surface	spring	73.0	6.32	0.835	7.84
Wolf	dam	mid	spring	77.6	6.68	0.831	7.81
Wolf	dam	benthos	spring	74.6	6.49	0.830	7.75
Wolf	ref	surface	spring	114.7	9.57	0.835	7.80
Wolf	ref	mid	spring	ND	ND	ND	ND
Wolf	ref	benthos	spring	ND	ND	ND	ND
Nels	dam	surface	spring	80.5	6.66	0.841	7.75
Nels	dam	mid	spring	73.4	6.20	0.839	7.66
Nels	dam	benthos	spring	79.5	6.80	0.842	7.63
Nels	ref	surface	spring	80.4	7.05	0.765	7.64
Nels	ref	mid	spring	ND	ND	ND	ND
Nels	ref	benthos	spring	ND	ND	ND	ND
5th	dam	surface	spring	119.5	9.71	0.663	8.17
5th	dam	mid	spring	119.1	9.82	0.663	8.00
5th	dam	benthos	spring	95.4	8.02	0.665	7.89
5th	ref	surface	spring	79.0	6.38	0.656	7.76
5th	ref	mid	spring	90.3	7.39	0.653	7.58
5th	ref	benthos	spring	77.9	6.35	0.654	7.73

Continued

Table B.2. Seasonal habitat descriptors of water by depth. ND: no data.

Table B.2. Continued.

Block	Trmt	Water depth	Season	% DO	DO (mg/L)	cond	pH
West	dam	surface	summer	47.8	3.90	0.506	7.84
West	dam	mid	summer	38.8	3.27	0.503	7.74
West	dam	benthos	summer	34.9	2.91	0.502	7.62
West	ref	surface	summer	58.8	4.52	0.459	7.79
West	ref	mid	summer	ND	ND	ND	ND
West	ref	benthos	summer	58.5	4.79	0.455	7.54
CC	dam	surface	summer	59.9	5.08	0.381	8.10
CC	dam	mid	summer	59.9	5.12	0.379	8.24
CC	dam	benthos	summer	57.4	4.91	0.382	7.94
CC	ref	surface	summer	68.6	5.57	0.422	7.79
CC	ref	mid	summer	ND	ND	ND	ND
CC	ref	benthos	summer	52.3	4.35	0.420	7.73
Wolf	dam	surface	summer	57.4	4.50	0.648	8.17
Wolf	dam	mid	summer	51.9	4.17	0.649	8.00
Wolf	dam	benthos	summer	48.7	3.95	0.648	7.87
Wolf	ref	surface	summer	38.4	5.58	0.669	7.77
Wolf	ref	mid	summer	ND	ND	ND	ND
Wolf	ref	benthos	summer	ND	ND	ND	ND
Nels	dam	surface	summer	72.4	6.03	0.487	7.87
Nels	dam	mid	summer	ND	ND	ND	ND
Nels	dam	benthos	summer	72.0	6.01	0.499	7.81
Nels	ref	surface	summer	89.6	7.36	0.568	7.77
Nels	ref	mid	summer	ND	ND	ND	ND
Nels	ref	benthos	summer	ND	ND	ND	ND
5th	dam	surface	summer	71.5	5.60	ND	7.71
5th	dam	mid	summer	50.9	4.12	ND	7.30
5th	dam	benthos	summer	44.7	3.68	ND	7.60
5th	ref	surface	summer	93.8	7.49	ND	7.68
5th	ref	mid	summer	102.5	8.16	ND	7.61
5th	ref	benthos	summer	84.0	6.81	ND	7.59
West	dam	surface	autumn	54.0	5.86	0.503	ND
West	dam	mid	autumn	49.6	5.46	0.507	ND

Continued

Table B.2. Continued.

Block	Trmt	Water depth	Season	% DO	DO (mg/L)	cond	pH
West	dam	benthos	autumn	42.9	4.76	0.507	ND
West	ref	surface	autumn	51.8	5.66	0.458	ND
West	ref	mid	autumn	ND	ND	ND	ND
West	ref	benthos	autumn	46.9	4.86	0.460	ND
CC	dam	surface	autumn	55.3	5.03	0.623	ND
CC	dam	mid	autumn	48.2	4.47	0.621	ND
CC	ref	surface	autumn	52.3	4.84	0.628	ND
CC	ref	mid	autumn	ND	ND	ND	ND
CC	ref	benthos	autumn	47.6	4.39	0.629	ND
CC	dam	benthos	autumn	47.4	4.41	0.602	ND
Wolf	dam	surface	autumn	30.9	2.73	0.700	ND
Wolf	dam	mid	autumn	29.8	2.67	0.701	ND
Wolf	dam	benthos	autumn	29.1	2.67	0.701	ND
Wolf	ref	surface	autumn	47.8	4.57	0.701	ND
Wolf	ref	mid	autumn	ND	ND	ND	ND
Wolf	ref	benthos	autumn	ND	ND	ND	ND
Nels	dam	surface	autumn	62.3	6.14	0.579	ND
Nels	dam	mid	autumn	ND	ND	ND	ND
Nels	dam	benthos	autumn	54.0	5.38	0.688	ND
Nels	ref	surface	autumn	61.2	6.27	0.693	ND
Nels	ref	mid	autumn	ND	ND	ND	ND
Nels	ref	benthos	autumn	ND	ND	ND	ND
5th	dam	surface	autumn	61.7	5.99	0.417	ND
5th	dam	mid	autumn	60.0	5.96	0.418	ND
5th	dam	benthos	autumn	43.9	5.99	0.417	ND
5th	ref	surface	autumn	82.9	8.24	0.437	ND
5th	ref	mid	autumn	80.7	8.03	0.437	ND
5th	ref	benthos	autumn	79.2	7.87	0.437	ND

Block	Trmt	Fines Mass (g)	Sand Mass (g)	Sm gravel Mass (g)	Total particle mass (g)	% fines	% sand	% sm gravel
5th	dam	11.2	48.2	5.5	64.9	17.2	74.3	8.5
5th	ref	8.0	44.8	1.4	54.2	14.7	82.6	2.7
CC	dam	8.3	27.3	16.0	51.6	16.0	53.0	31.0
CC	ref	2.7	39.1	15.5	57.3	4.7	68.3	27.0
Nels	dam	12.9	49.3	11.2	73.4	17.6	67.2	15.2
Nels	ref	2.6	38.9	21.9	63.4	4.1	61.4	34.5
West	dam	42.1	3.6	2.4	48.1	87.5	7.5	5.0
West	ref	20.3	39.4	10.3	70.0	29.0	56.3	14.7
Wolf	dam	8.7	63.9	4.4	77.0	11.3	83.1	5.7
Wolf	ref	2.8	75.3	1.2	79.4	3.6	94.9	1.5

Table B.3. Habitat descriptors of sediment grain size particles.

APPENDIX C

MATERIALS IN WATER AND SEDIMENT DATA

Block	Trmt	Water depth	Season	TC (mgC/L)	TOC (mgC/L)	NH4 (mg N/L)
West	dam	surface	spring	37.86	6.17	ND
West	dam	mid	spring	38.84	7.54	ND
West	dam	benthos	spring	37.49	6.76	ND
West	ref	surface	spring	34.79	5.71	ND
West	ref	mid	spring	ND	ND	ND
West	ref	benthos	spring	36.11	6.90	ND
CC	dam	surface	spring	36.96	6.96	0.14
CC	dam	mid	spring	37.71	7.01	0.10
CC	dam	benthos	spring	36.24	7.06	0.07
CC	ref	surface	spring	38.43	7.78	0.09
CC	ref	mid	spring	ND	ND	ND
CC	ref	benthos	spring	ND	ND	ND
Wolf	dam	surface	spring	40.40	6.20	0.22
Wolf	dam	mid	spring	38.54	5.75	0.51
Wolf	dam	benthos	spring	40.17	5.92	0.26
Wolf	ref	surface	spring	38.51	6.13	0.32
Wolf	ref	mid	spring	ND	ND	ND
Wolf	ref	benthos	spring	ND	ND	ND
Nels	dam	surface	spring	38.31	5.61	0.18
Nels	dam	mid	spring	38.98	6.36	0.21
Nels	dam	benthos	spring	38.64	5.92	0.22
Nels	ref	surface	spring	47.13	8.05	0.97
Nels	ref	mid	spring	ND	ND	ND
Nels	ref	benthos	spring	ND	ND	ND
5th	dam	surface	spring	47.95	7.36	BDL
5th	dam	mid	spring	48.81	7.72	0.04
5th	dam	benthos	spring	47.88	8.46	0.01
5th	ref	surface	spring	46.32	6.86	0.04
5th	ref	mid	spring	47.94	7.07	0.11
5th	ref	benthos	spring	47.22	6.26	0.09

Continued

Table C.1. Seasonal concentration of materials in waters by depth. BDL: below detection limit. ND: no data.

Table C.1. Continued.

Block	Trmt	Water depth	Season	TC (mgC/L)	TOC (mgC/L)	NH4 (mg N/L)
West	dam	surface	summer	35.22	12.62	0.03
West	dam	mid	summer	35.63	11.74	0.10
West	dam	benthos	summer	33.69	9.13	0.10
West	ref	surface	summer	34.21	10.69	0.01
West	ref	mid	summer	ND	ND	ND
West	ref	benthos	summer	40.35	19.09	BDL
CC	dam	surface	summer	25.61	10.24	BDL
CC	dam	mid	summer	18.00	7.43	0.04
CC	dam	benthos	summer	22.69	7.54	0.04
CC	ref	surface	summer	26.62	11.85	BDL
CC	ref	mid	summer	ND	ND	ND
CC	ref	benthos	summer	25.57	7.57	0.01
Wolf	dam	surface	summer	30.10	7.59	0.06
Wolf	dam	mid	summer	25.53	11.54	0.06
Wolf	dam	benthos	summer	27.95	10.89	0.03
Wolf	ref	surface	summer	27.15	9.40	0.11
Wolf	ref	mid	summer	ND	ND	ND
Wolf	ref	benthos	summer	ND	ND	ND
Nels	dam	surface	summer	27.20	11.99	0.02
Nels	dam	mid	summer	ND	ND	ND
Nels	dam	benthos	summer	29.24	9.89	0.09
Nels	ref	surface	summer	22.69	8.29	0.05
Nels	ref	mid	summer	ND	ND	ND
Nels	ref	benthos	summer	ND	ND	ND
5th	dam	surface	summer	24.39	6.93	0.00
5th	dam	mid	summer	35.28	10.97	0.07
5th	dam	benthos	summer	21.57	7.74	0.03
5th	ref	surface	summer	33.22	6.30	0.01
5th	ref	mid	summer	17.15	2.66	0.03
5th	ref	benthos	summer	24.09	5.91	0.03

Continued

Table C.1. Continued.

Block	Trmt	Water depth	Season	TC (mgC/L)	TOC (mgC/L)	NH4 (mg N/L)
West	dam	surface	autumn	21.96	3.92	BDL
West	dam	mid	autumn	24.32	1.55	BDL
West	dam	benthos	autumn	25.66	2.93	0.11
West	ref	surface	autumn	25.82	4.28	BDL
West	ref	mid	autumn	ND	ND	ND
West	ref	benthos	autumn	24.45	2.41	BDL
CC	dam	surface	autumn	23.68	4.88	BDL
CC	dam	mid	autumn	23.03	4.05	BDL
CC	dam	benthos	autumn	23.42	3.80	BDL
CC	ref	surface	autumn	26.77	4.63	BDL
CC	ref	mid	autumn	ND	ND	ND
CC	ref	benthos	autumn	26.92	4.83	BDL
Wolf	dam	surface	autumn	42.28	8.00	BDL
Wolf	dam	mid	autumn	41.71	7.72	BDL
Wolf	dam	benthos	autumn	40.28	6.13	BDL
Wolf	ref	surface	autumn	41.01	8.31	0.01
Wolf	ref	mid	autumn	ND	ND	ND
Wolf	ref	benthos	autumn	ND	ND	ND
Nels	dam	surface	autumn	37.64	7.22	BDL
Nels	dam	mid	autumn	38.50	8.10	BDL
Nels	dam	benthos	autumn	35.37	5.46	0.01
Nels	ref	surface	autumn	32.66	6.09	0.00
Nels	ref	mid	autumn	ND	ND	ND
Nels	ref	benthos	autumn	ND	ND	ND
5th	dam	surface	autumn	23.50	6.01	0.02
5th	dam	mid	autumn	23.11	5.62	BDL
5th	dam	benthos	autumn	18.27	5.30	BDL
5th	ref	surface	autumn	22.39	4.91	BDL
5th	ref	mid	autumn	24.00	5.37	BDL
5th	ref	benthos	autumn	24.81	5.91	BDL

Block	Trmt	Season	% C	% OM	CH4	% N	NH4	NO3
5th	dam	summer	3.45	4.45	141.88	0.12	18.13	0.00
5th	ref	summer	3.78	6.26	148.03	0.11	14.27	0.00
CC	dam	summer	3.17	5.09	142.57	0.13	6.65	0.13
CC	ref	summer	2.28	3.93	146.25	BDL	7.39	0.12
Nels	dam	summer	4.08	4.92	144.05	0.13	15.80	0.00
Nels	ref	summer	2.68	2.04	141.27	BDL	4.78	0.14
West	dam	summer	3.00	4.68	128.61	0.19	36.69	0.00
West	ref	summer	2.04	2.59	121.53	BDL	7.56	0.24
Wolf	dam	summer	3.25	3.23	155.73	BDL	8.08	0.00
Wolf	ref	summer	2.68	2.36	153.51	BDL	2.08	0.00
5th	dam	autumn	3.17	4.25	149.73	BDL	7.02	0.00
5th	ref	autumn	2.55	3.93	153.73	BDL	3.34	0.03
CC	dam	autumn	5.09	5.04	165.49	0.25	7.62	0.05
CC	ref	autumn	2.35	2.79	167.76	BDL	5.34	0.14
Nels	dam	autumn	3.31	3.61	156.59	BDL	5.73	0.19
Nels	ref	autumn	5.21	3.50	144.63	BDL	0.74	0.00
West	dam	autumn	2.43	3.93	167.14	0.18	19.77	0.04
West	ref	autumn	2.71	3.08	149.04	BDL	5.00	0.04
Wolf	dam	autumn	3.77	3.78	159.04	0.13	25.66	0.95
Wolf	ref	autumn	3.12	2.48	151.75	BDL	1.55	0.09
5th	dam	spring	3.49	13.70	ND	0.15	10.25	0.52
5th	ref	spring	2.54	4.93	ND	BDL	1.97	0.38
CC	dam	spring	2.39	8.26	ND	BDL	2.26	1.81
CC	ref	spring	2.34	6.44	ND	BDL	1.42	1.69
Nels	dam	spring	2.84	4.72	ND	BDL	4.53	0.43
Nels	ref	spring	2.03	10.30	ND	BDL	4.28	1.31
West	dam	spring	2.84	17.52	ND	0.19	ND	ND
West	ref	spring	1.72	9.30	ND	BDL	ND	ND
Wolf	dam	spring	2.80	6.60	ND	BDL	2.18	4.24
Wolf	ref	spring	2.75	6.34	ND	BDL	2.37	1.45

Table C.2. Seasonal concentrations of materials in sediment. BLD: below detection limit. ND: no data

APPENDIX D

MICROBIAL ACTIVITY DATA

Block	Trmt	Season	DEA (N mg/DM kg/h)	gross NIT (N mg/DM kg/h)	ABS(CH4OX, CH4 mg/DM kg/h)
West	dam	spring	0.027	ND	1.154
West	ref	spring	0.131	ND	0.169
CC	dam	spring	0.153	0.150	0.927
CC	ref	spring	0.260	0.203	1.307
Wolf	dam	spring	0.549	0.116	1.374
Wolf	ref	spring	0.369	0.275	0.693
Nels	dam	spring	0.602	0.138	0.909
Nels	ref	spring	0.194	0.212	0.865
5th	dam	spring	0.866	0.122	1.306
5th	ref	spring	0.121	0.132	1.214
West	dam	summer	0.649	0.099	0.716
West	ref	summer	0.191	0.918	1.167
CC	dam	summer	2.994	0.247	1.105
CC	ref	summer	0.091	0.122	1.111
Wolf	dam	summer	1.428	0.186	1.733
Wolf	ref	summer	0.415	0.076	1.003
Nels	dam	summer	0.124	0.124	1.328
Nels	ref	summer	0.361	0.143	0.401
5th	dam	summer	1.860	0.045	1.241
5th	ref	summer	1.941	0.020	1.321
West	dam	autumn	1.715	0.047	1.634
West	ref	autumn	0.277	0.088	1.056
CC	dam	autumn	1.914	0.109	0.235
CC	ref	autumn	0.083	0.065	1.236
Wolf	dam	autumn	1.616	0.018	1.382
Wolf	ref	autumn	0.676	0.076	0.707
Nels	dam	autumn	2.472	0.109	1.147
Nels	ref	autumn	0.267	0.182	0.399
5th	dam	autumn	1.596	0.034	2.957
5th	ref	autumn	1.196	0.017	1.394

Table D.1. Seasonal rates of microbial activity in sediment